APPENDIX D

POPULATION VIABILITY ANALYSIS FOR PACIFIC COAST WESTERN SNOWY PLOVERS

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Introduction

In 1993 the Pacific coast population of the western snowy plover (*Charadrius alexandrinus nivosus*) (western snowy plover) was designated as threatened by the U.S. Fish and Wildlife Service under the Endangered Species Act of 1973, as amended (16 USC 1531 *et seq.*). To aid the Western Snowy Plover Recovery Team in developing recovery criteria, the authors developed this population viability analysis for the Pacific coast population of the western snowy plovers.

Population viability analysis is used increasingly as a tool for developing conservation, management or restoration strategies for threatened, endangered, or potentially threatened species. The method is reviewed by Boyce (1992), Burgman *et al*. (1993), Beissinger and Westphal (1998) and Nur and Sydeman (1999). Examples of its use include Haig *et al*. (1993), Maguire *et al*. (1995), Akçakaya *et al*. (1995), and Bustamante (1996). In particular, population viability analyses have been developed for the congener piping plover *Charadrius melodus* (Great Plains population: Ryan *et al.* 1993; Atlantic coast population: Melvin and Gibbs 1996).

General Features of the Population Viability Analysis Model

The model is stochastic. Stochasticity is one of the defining features of Population Viability Analyses in general (Burgman *et al*. 1993). Two types of random variation are incorporated: unpredictable variation in the environment and "demographic stochasticity." Demographic stochasticity can be thought of as follows: even if all relevant features of the environment (including predators, competitors, abiotic factors, *etc.*) impinging on western snowy plovers are known, and even though, on average, survival or reproductive success can be related to these environmental features, there will still be an element of unpredictability regarding the precise number of young or adults that survive or the number of fledglings produced in any time period.

For the population viability analysis, we have used a metapopulation model with six subpopulations linked by dispersal of individuals. A metapopulation is a set of subpopulations among which there is restricted dispersal (Harrison 1994, Nur and Sydeman *in press*). In this population viability analysis, we have incorporated into the metapopulation model the best available estimates on dispersal. However, using the same model structure, one can easily alter the parameter values of dispersal, and, indeed, we do so. An alternative approach would be to treat Pacific coast birds as a single population, with unrestricted mating among all individuals, regardless of location. The latter model assumes that a bird from, say, Oregon is as likely to mate with a bird from San Diego as with a bird from Oregon. Such an assumption is exceedingly unrealistic; hence, we have adopted a metapopulation model. Another virtue of the metapopulation approach is that survival and/or fecundity can be allowed to vary among subpopulations, rather than being assumed homogeneous throughout the species' range. Note that the Atlantic coast piping plover population viability analysis assumed a single, panmictic population instead of a metapopulation (U.S. Fish and Wildlife Service 1996).

The population viability analysis is carried out using the RAMAS/GIS program which is commercially available (Akçakaya 1997) and has been widely used for population viability analyses. Use of an off-the-shelf program makes modeling convenient and reproducible, but there are attendant limitations regarding input and output. For example, RAMAS/GIS allows one to specify the degree of stochastic variability in survival and reproductive success, but not dispersal. Other limitations are mentioned in the "Discussion." The Western Snowy Plover Recovery Team determined that the cost of developing a specially written program to carry out the population viability analysis was not justified.

The type of model that can be generated using RAMAS/GIS does not incorporate the production and elimination of genetic variation brought about by sexual reproduction (Caswell 1989, Beissinger and Westphal 1998). As a simplification, only one sex is modeled. We have used males because their demographic parameters can be estimated with greater certainty than for females. In addition, there is reason to consider that the availability of males is limiting reproductive success because they are responsible for post-hatching parental care and females can lay clutches for more than one male (Warriner *et al*. 1986).

The western snowy plover population viability analysis projects into the future up to 100 years. Although, there is considerable uncertainty in projecting 100 years, this time-horizon is commonly used and is recommended by Mace and Lande (1991). This time horizon was also used for the Atlantic coast Piping Plover Recovery Plan. We also depict population trajectories for shorter time-horizons.

The population viability analysis indicates trends and quantifies the risk that the total population goes extinct or falls below a **specified threshold**. We used a specified threshold of 50 individuals, but the population viability analysis could be modified by choosing any other threshold value.

The population viability analysis includes different scenarios pertaining to changes in reproductive success resulting from predator management and could be used to model other changes in management practices or the environment, affecting any of the other demographic parameters.

Subpopulations

The Western Snowy Plover Recovery Team has identified six subpopulations of western snowy plovers, each corresponding to a region of the U S. Pacific coast. The population viability analysis assumes restricted dispersal among subpopulations, but unrestricted access to mates within subpopulations. The six subpopulations, with their two-letter or three-letter designations, and estimated population sizes are:

1. Oregon and Washington coast (OR) estimated at 134 plovers;
2. Northern California coast (NC; Del Norte, Humboldt and Mendocino counties) with 50 plovers;
3. San Francisco Bay (SFB; primarily South Bay) with 264 plovers;
4. Monterey Bay (MB; coast of Sonoma, Marin, San Mateo, Santa Cruz and Monterey counties) with 300 plovers;
5. coast of San Luis Obispo, Santa Barbara and Ventura counties (SLO) with 886 plovers;
6. San Diego area (SD; Los Angeles, Orange and San Diego counties) with 316 plovers.

For the OR, MB, and SD subpopulations, intensive monitoring of color-banded individuals was carried out in 1997, and population size was estimated on that basis. For the NC, SFB and SLO subpopulations, information is less complete. Instead, we relied on "window surveys" conducted in 1995, 1991, and 1995, respectively. To account for birds missed during the window surveys we applied a correction factor to the survey numbers for the NC, SFB and SLO subpopulations. Where window surveys were conducted at locations with color banded birds, the number of marked birds known to be at the location was underestimated by about 22 percent. This takes into account both birds known to be present but missed and birds that were double counted. The correction factor used is 1/(1-.222) = 1.286. For the NC and SLO subpopulations, the correction factor was applied to the number of birds counted on window surveys in 1995.

However, for the SFB subpopulation, no window survey has been carried out since 1991. Uncertainty about population trends since 1991 compounds uncertainty about current abundance. We therefore considered there to be an upper bound of 310 individuals (219 individuals observed on the window survey in 1991 x 1.286 x 1.1, to account for modest population growth since 1991) and a lower bound of 219 individuals (population decline since 1991, equal in magnitude to the undercounting during the window survey). For modeling, we used the mean of those two estimates (= 264 individuals).

Conceptual Framework of the Model

The key demographic parameters in the model are: (1) adult survival, (2) juvenile survival, (3) reproductive success, and (4) dispersal. All individuals 1 year or older are considered to be adult, and assumed to breed (see below). The demographic parameters are linked in the population model in the following manner, ignoring dispersal among subpopulations (detailed later) and ignoring any stochastic effects.

The model keeps track of the abundance of each age class (1-year-old, 2-year-old, *etc.*, up to 20-year-old individuals) in each subpopulation. This enumeration by the model is carried out at the onset of the breeding season; this is referred to as a pre-breeding census. In the model, the number of 2-year-olds in year t+1, symbolized $N(2)_{t+1}$ is equal to the number of 1-year-olds in year t, symbolized $N(1)_t$, times the annual survival rate of 1-year-olds, symbolized S_1. Note that S_1 is not constant, but varies stochastically from year to year, and differs among subpopulations. Similar calculations are performed for the number of 3-year-olds, *i.e.*, $N(3)_{t+1} = N(2)_t*S_2$, 4-year-olds, *etc*. In the model, adult survival is assumed to be the same for all ages, *i.e.*, $S_1 = S_2 = ... = S_{19}$, but no adult lives beyond 20 years of age, which is considered maximum age for this species.

The number of 1-year-olds in a given year is equal to the number of fledged chicks produced the year before times the probability that a fledged chick will survive to reach the age of 1 year. If the total number of adults the year before is written $N(A)_t = N(1)_t + N(2)_t + ... + N(20)_t$, then the number of 1-year-olds in year t+1, symbolized $N(1)_{t+1}$, is equal to the product $N(A)_t*F*S_0$, where F is the number of male fledglings produced per male adult in each year, and S_0 is the probability a fledgling survives to 1 year (12 months) of age. Since the sex ratio of fledglings is unknown, we assume a 1:1 ratio. Any non-breeding among adults would act to reduce F; however, all adults are assumed to breed (see below). In the model, F and S_0 also vary among subpopulations and vary randomly among years, with a specified mean and standard deviation.

Parameter Estimates

Adult survival - The best estimates for adult survival came from capture/recapture analyses of Monterey Bay color-banded plovers, a major study population (henceforth Monterey Bay) situated within the MB subpopulation. Additional data for analyses came from color-banded study populations on Oregon beaches (Oregon) and San Diego beaches (San Diego). Note that we distinguish between study areas (Monterey Bay, Oregon and San Diego) and their respective, more inclusive subpopulations (MB, OR, SD). Analyses of survival were carried out using the program SURGE (Lebreton *et al*. 1992, Cooch *et al*. 1996) and for Monterey Bay were based on 777 adults

(361 males, 416 females) followed over 14 years. Sample sizes for Oregon were 108 males and 70 females, followed over 8 years, and for San Diego 91 males and 137 females, followed over 4 years. Since male survival significantly exceeded female survival at Monterey Bay and only males were modeled, we present only estimates for male adults, for the Monterey Bay, Oregon and San Diego study populations.

We fit a two-age class model for male adult survival, in which the first age class covers the first year after first capture, and the second age class covers all subsequent years. Estimates of survival for the first age class can be biased due to behavioral responses to trapping and banding, lower site-fidelity among some first-time captures, and other methodological difficulties. These biases do not apply to survival after the first year of banding (Pradel *et al*. 1997). For this reason, several studies have used only survival estimates from the second age class (*e.g.*, Gaston 1992, Johnston *et al*. 1997); we adopted the same practice.

A potential shortcoming of capture/recapture analyses of survival is that they cannot allow for permanent emigration, though they can allow for temporary emigration (Lebreton *et al*. 1992). A bird which moves permanently out of the study area cannot be distinguished from one that has died. The problem of permanent emigration can be overcome somewhat by enlarging the study area. In our analyses we compare survival estimates from three nested data sets, which differ only in the spatial and temporal extent of resightings. The most restricted data set included only resightings from birds seen during the breeding season in the same study area. In the next, more comprehensive data set, resightings of color-banded birds at other study areas were also included. In the most extensive data set, resightings during the entire year were included, as well as resightings at other study areas. The extent to which survival estimates differ among the three data sets provides insight into the magnitude of the problem of dispersal (permanent emigration).

Male survival estimates for Monterey Bay, for 2nd-year and older adults, were 74, 74, and 75 percent for the three data sets (Table D-1A). In other words, survival estimates differed slightly depending on the spatial extent of coverage and whether winter observations were included. Increasing the study area for Monterey Bay birds (either spatially or through observations outside the breeding season), increased the survival estimates by up to 1 percent. This implies that 1 percent of the individuals, inferred to be dead if observations are only from one study area and only during the breeding season, are inferred to be alive using the data from the enlarged study area. These results suggest that amount of dispersal out of the original study area is not negligible but it is also not great. Since not all breeding areas of Pacific coast western snowy plovers are adequately surveyed for color-banded birds, we assume that there was additional, **undetected** dispersal out of the study area on the order of 1 percent. If so, then the true adult survival rate is 76 percent.

For the Oregon study population, male survival values were 74 to 75 percent, *i.e.*, nearly identical to those from Monterey Bay (Table D-1A). Estimates for San Diego are somewhat lower, at 71 percent, but the difference between the San Diego estimates and those from Monterey Bay is no greater than the standard error of these estimates (Table D-1A). Among all three sites, survival estimates did not differ to a statistically significant degree. In the population viability analysis, we assume a survival rate of 76 percent for all subpopulations, but also model population trajectories with an adult survival of 75 percent and 77 percent, for all subpopulations. Capture/recapture analyses of Atlantic coast piping plovers resulted in a survival estimate of 74 percent (Melvin and Gibbs 1996). Paton (1994) analyzed survival for Great Salt Lake western snowy plovers over a 3-year period. Survival rates were pooled over the two sexes (unlike our analyses), and differed among years, ranging from 58 percent to 88 percent, with median survival = 73 percent. Thus, survival values from other plover studies are consistent with the survival values used here.

Finally, the year to year variation in male survival for Monterey Bay was estimated to be 5.65 percent (standard deviation). We used this parameter value in our simulations, for all six subpopulations. Note that "catastrophic mortality" (see below), represents additional temporal variation.

Juvenile survival - Table D-1B shows survival estimates for first year birds (from fledging to 12 months of age), by study population and data set. Sample sizes were 1069 fledged young at Monterey Bay, 207 at Oregon and 102 at San Diego. Results were very similar at Monterey Bay and San Diego; Oregon values were somewhat higher but not statistically different from Monterey Bay. We, therefore, used juvenile survival estimates for Monterey Bay for all subpopulations. The different estimates for Monterey Bay, depending on the data set, were 39 percent, 44 percent and 45 percent. Note that for Monterey Bay as we expand the data from just 1 study site to a large network of sites, the survival estimate increases by 5 to 6 percent in absolute terms, and by 15 percent in relative terms. Compare this to the increase in adult survival estimates by 1 percent for the same series of nested data sets (see above). Thus, it is clear that there is quite a bit of dispersal among first-year birds. Undoubtedly, we are still underestimating survival because of permanent emigration. Therefore, we increased the survival estimate to 50 percent. This would imply that among 100 fledged young, 50 survive to age 1, but of these only 39 are inferred to survive based on observations at the single study population, with 11 out of 50 surviving juveniles (or 22 percent) dispersing out of the single study population. This estimate of dispersal is consistent with that directly observed and included in the population viability analysis (see below). Annual variation in juvenile survival (obtained from Monterey Bay) is also shown in Table D-1B.

Reproductive Success - Here we had empirical data for three study areas, corresponding to three subpopulations (Table D-1C). For Monterey Bay, reproductive success was 0.849 fledged young reared per breeding male in years without predator control and without any exclosures, versus 1.105 fledged young per male in years with predator control and with exclosures. Reproductive success was similar but slightly lower (= 1.04 chicks per male) in Oregon, where intensive management has occurred in all years for which we had data; estimates for Oregon and Monterey Bay are not statistically significantly different for years in which predators were managed. Reproductive success at the San Diego study area, where some (indirect) management activities are thought to have some protective effect on breeding western snowy plovers, is a little more than that observed at Monterey Bay without any management activity, but substantially, and significantly, lower than that observed at Monterey Bay and Oregon with management activity.

Simulations assuming that protective management continues in MB and OR, used the respective, current reproductive success values of 1.105 and 1.04 fledglings per male. For SD we did not use the observed reproductive success of 0.917 chicks per male, because this would have produced a subpopulation that (in the absence of net immigration) would have declined at 1.8 percent per year. Such a decline would have been inconsistent with observations and window surveys, which indicate a relatively stable or perhaps increasing SD subpopulation since 1995. Therefore, for the SD subpopulation, we assume that with current management practices continuing, reproductive success is 0.988 chicks per male, a value that produces a numerically stable subpopulation in the long-term (given the other demographic parameter estimates and assumptions). Reproductive success estimates for San Diego were based on only 3 years of data, and the overall mean of 0.917 may have underestimated the long-term, expected reproductive success.

In the scenarios below we use Monterey Bay past reproductive success (in the absence of intervention) for NC and SFB; *i.e.*, we use that as a best estimate for reproductive success in the absence of predator control/exclosures. We also assume that if management activities cease in MB, OR, and SD regions then reproductive success will be at 0.849 fledged young per male, as well.

For the SLO subpopulation there was considerable uncertainty regarding the appropriate reproductive success value to use. Window surveys indicate that western snowy plover numbers have fluctuated over time, with no clear trend discernible, except that, whatever the trend, it is not increasing. At best, the SLO subpopulation might be considered stable; at worst the subpopulation is declining. On that basis, we considered there to be an "optimistic" and a "pessimistic" reproductive success value. The optimistic value is that level of reproductive success which would produce a stable, self-sustaining population (given all other assumptions); that value is 0.988 (the same value used for the SD subpopulation). The pessimistic value is 0.849 chicks per male, the same as used for NC and SFB subpopulations. A third possibility is to use an intermediate value (the mean of the optimistic and pessimistic values = 0.919 chicks per male). In our simulations, we consider all three possibilities, to demonstrate the sensitivity of model results to assumptions about SLO reproductive success. However, in all but two series of simulations, we use the intermediate reproductive success value of 0.919 fledged chicks per male, which in the long-term (given other parameter estimates and assumptions) would produce a population decline of 1.8 percent per year.

For annual variation in reproductive success we used a value of 0.157 (standard deviation.), which is the variation observed in reproductive success at Monterey Bay from 1992-1997. We also note that annual variation in reproductive success among the three sites showed weak but not significant correlations. In the scenarios below we assume that all demographic parameters show weak positive correlations (r = + 0.10 between pairs of subpopulations).

RAMAS/METAPOP allows one to add "catastrophic mortality" over and above "regular mortality." Catastrophic mortality can include both reproductive failure and changes in survival of juveniles and adults. It is not clear that western snowy plovers suffer from catastrophic mortality (none was apparent in the data sets analyzed), yet we should not rule it out. On the basis of recommendations of the Western Snowy Plover Recovery Team our simulations include additional mortality due to reproductive failure (see below). We also compare simulations with and without this additional catastrophic mortality.

Dispersal - There are qualitative data indicating dispersal, especially of first-year birds, to/from all three intensively studied areas (Monterey Bay, Oregon, and San Diego). The only extensive quantitative data are from Monterey Bay. These data indicated that 21 percent of individuals hatched in Monterey Bay and later observed breeding, were known to breed in areas other than at Monterey Bay. Results from the SURGE analyses of juvenile survival implied a similar dispersal rate of 22 percent among surviving juveniles (see above). Individuals observed dispersing were seen as far north as Washington and Oregon, and as far south as SLO, but none in the sample were observed going to SD. However, there have been additional observations of Monterey Bay individuals dispersing to SD. Meanwhile, dispersal from SD (43 individuals born at San Diego), indicated a small percentage going to SLO. Using these results, we assumed the following: a general dispersal rate of 25 percent for first-year males; adult males are assumed not to disperse. In other words, we assumed that the total number of birds dispersing exceeded the number known to have dispersed; *i.e.*, some birds dispersed but were undetected. The exception to these assumed dispersal rates was for the most northern subpopulation (OR, which includes Washington) and the most southern, SD. For these, dispersal rates were assumed to be 20 percent, allowing for reduced dispersal from subpopulations, located on the edge of the metapopulation.

We also assumed dispersal was constant, in the absence of information to the contrary. Thus, dispersal did not increase or decrease as subpopulation size increased or decreased. There is little information on dispersal rates in relation to population characteristics for other, similar species (Nur and Sydeman *in press*). For example, a study of Roseate Terns (*Sterna dougallii*; Spendelow *et al*. 1995) found no relationship of dispersal rates to colony size (either colony of origin or colony of destination). RAMAS/GIS does not allow for stochastic variation in dispersal rates among years. Note also, that the metapopulation model does not include dispersal to or from Baja California. This is equivalent to assuming that the number of immigrants from Baja California to the metapopulation equals the number of emigrants dispersing to Baja California. This assumption of balanced dispersal to and from Baja California may be unrealistic, but we had no data on which to develop a metapopulation model which incorporates Baja California.

To demonstrate the impact of a change (or uncertainty) in dispersal rates, we also carry out simulations in which dispersal rates are reduced by 50 percent and by 100 percent.

Additional Assumptions

Density Dependence - Not much is known about this, for any bird species. Following input from Western Snowy Plover Recovery Team members, we assume a limit on availability of beach habitat, *i.e.*, that there is a region-specific limit on adequate nesting sites. Based on information provided by the recovery team, we estimate the limit, or ceiling, of breeding western snowy plovers to be:

Subpopulation	Ceiling size
OR	300
NC	200
SFB	500
MB	500
SLO	1600
SD	550

These ceilings are about 80 percent greater than current numbers, and are similar to, or slightly in excess of, estimates of target population size, obtained by Western Snowy Plover Recovery Team biologists, on a site by site basis (see Appendix B). A realistic assumption is that ceilings represent the maximum number of individuals that can successfully breed for each subpopulation. Under such an assumption, individuals in excess of the ceiling are still alive but cannot breed successfully in the current year. However, such an assumption cannot be implemented by RAMAS/GIS 2.0. Therefore, we made a more restrictive (and admittedly less realistic) assumption: individuals in excess of ceiling numbers do not survive the current year. This imposes a hard limit on maximum number of individuals in each subpopulation. Note that the metapopulation only reaches ceiling levels under Scenarios 17-19; in the other Scenarios, the metapopulation declines and/or is well below ceiling levels. Note also that there is no decrement in survival until the breeding population size **exceeds** the ceiling for that subpopulation.

Catastrophic Mortality - There is at present no evidence of catastrophic mortality in western snowy plovers, but the 1998 El Niño may prove otherwise[1]. Though it may seem desirable to include catastrophic mortality, the problem is that we have no idea of its magnitude or frequency of occurrence. Thus any quantitative results (when this is included) depend entirely on the assumptions made. On the basis of input from Western Snowy Plover Recovery Team members we assume catastrophic mortality in the form of "reproductive failure." We assume that catastrophes occur, on average, once every 20 years (*i.e.*, in each year with 5 percent probability), and that in a catastrophe year reproductive success is reduced to 50 percent of what it "normally" would have

[1] It is believed that western snowy plovers suffered unusually high winter mortality in the 1998 El Niño and the subsequent La Niña. Point Reyes Bird Observatory plans to examine this issue when appropriate data have been incorporated into the survivorship database (Gary Page, Point Reyes Bird Observatory, pers. comm. 2001).

been. Note that model results are identical whether reproductive success itself is impacted, as part of catastrophic mortality, or whether juvenile survival is impacted.   Catastrophes were assumed to occur independently of one another (*i.e.*, the reproductive failure is specific to a subpopulation). We also consider a scenario with no catastrophic mortality and one in which catastrophic mortality includes reduction in adult survival (50 percent reduction compared to "normal" levels of survival, with a 5 percent probability per year) in addition to catastrophic reproductive failure.

All one-year-olds breed - This may be an overestimate but not likely by much; available field data (PRBO, unpubl.) indicate that the actual percent of males breeding is close to 100 percent. If we allow for less than 100 percent breeding among 1-year-olds (or even among older adults), then results presented would be more pessimistic.

Weak, positive environmental correlations among subpopulations - This is a compromise between assuming strong correlations (for which there is no evidence) and assuming no correlation (which at least for survival would seem **unlikely**). Empirical data on reproductive success supports the assumption of weak, positive correlation among subpopulations.

Extinction Threshold

The Atlantic coast Piping Plover Recovery Plan had an objective of keeping the probability of extinction below 5 percent for the entire (meta)population in the next 100 years (U.S. Fish and Wildlife Service 1996). A scenario in which Pacific coast western snowy plovers fall to a few individuals should not, in our opinion, be considered acceptable. Therefore, we consider the endpoint of "quasi-extinction," defined here as 50 individuals, rather than extinction itself (Burgman *et al*. 1993). This follows recommendations of Beissinger and Westphal (1998) and others. If there were as few as 50 individuals we expect that extreme measures would be undertaken to prevent extinction, such as captive breeding (as was the case for the California Condor). Also, an effective population size (N_e) of 50 individuals is considered close to the threshold number below which genetic and demographic forces combine, in the absence of intervention, to produce an "extinction vortex" (Gilpin and Soule 1986). It is difficult to determine what is the actual population size that corresponds to an effective population size of 50; for simplicity, in the results we present the probability that actual population size decreases below 50 individuals, but we recognize that N_e is always less than actual population size.

Results

Deterministic Results

With 0.76 adult survival, 0.50 juvenile survival, and fecundity = 1.105 (see above), the geometric rate of population growth (lambda) is 1.036, or 3.6 percent increase per year. All results in this section assume no stochastic effects (which are treated below) and in particular no catastrophic mortality. With 0.75 adult survival, and all other values the same, the growth rate decreases to .026 per year (lambda = 1.026). To produce a population growth rate of 1.0, requires 0.964 fledged young/male assuming .076 adult survival and .050 juvenile survival; if adult survival is 0.75, 1.003 fledged young/male are required. Note that increasing fecundity by 0.037 chicks per male has an effect equivalent to increasing adult survival by 0.01 (*i.e.*, decreasing adult mortality by 0.01, or 4 percent in relative terms).

Sensitivity analysis for Deterministic Results

A change in adult survival of 0.01 (0.75 to 0.76), produces a change in lambda of .001. A change in fecundity of 0.08 (in relative terms), *e.g.* from 1.00 to 1.08, changes lambda by 2.24 percent. The same is true for a change in juvenile survival, *e.g.*, increasing juvenile survival from 0.50 to 0.54, changes lambda by 2.24 percent. Clearly, a small difference in adult survival (*e.g.*, 1 percent) can have a substantial impact on population trajectory, especially over a 100-year time period.

Stochastic Results

We present results from 19 different scenarios for the Pacific coast western snowy plover metapopulation. Each scenario differs with respect to one or more demographic parameters, or starting population size, or other assumptions (*e.g.*, catastrophic mortality). In all cases, results from 400 replications of each scenario are shown. Scenario 1 is for **"Status Quo"** conditions: current values for reproductive success, etc., are assumed to continue indefinitely, *i.e.*, management activities continue in OR, MB, and SD. Scenario 1 uses our best estimates for the suite of demographic parameters outlined above. This includes 0.76 adult survival and catastrophic reproductive failure, but no other catastrophic mortality. Results for Scenario 1 are summarized in Tables D-2A and D-2B. The overall trajectory for the metapopulation is shown in Fig. D-1A; shown also are the highest and lowest values obtained in the 400 simulations (depicted with diamonds), the mean outcome and also outcomes that are plus or minus one standard deviation (S.D.). Thus, about 16 percent of outcomes will be above the mean + 1 S.D. level and about 16 percent of outcomes will be below the mean - 1 S.D. level. Furthermore, about 68 percent of outcomes, on average, will be within +/- 1 S.D. of the mean. We also depict two examples of representative population trajectories, out of the total of 400 simulations (Fig. D-1B).

We see that even with continued levels of ongoing management into the future, the prognosis is for a slowly-decreasing metapopulation, one that, on average, declines at 0.92 percent per year (Table D-2A). After 100 years, the metapopulation can be expected to be 39 percent of its original size. The probability that the metapopulation will increase in 100 years is essentially zero (Fig. D-1A). On the other hand, the probability of quasi-extinction (fewer than 50 individuals) is also zero. Fig. D-1C depicts the probability of the metapopulation declining below specified levels. For example, there is a nearly 100 percent chance of declining below 1800 individuals (compared to the estimated 1950 at present), but only a 1 percent chance of declining below 200 individuals. The probability of at least a 50 percent decline after 100 years is 72 percent (Table D-2B). Results for individual subpopulations after 100 years are shown in Fig. D-1D; these show that, in almost all simulations, all six subpopulations are likely to persist for 100 years, but in some cases at very low levels (close to zero).

Sensitivity Analysis of Stochastic Results

In this section, we carry out a sensitivity analysis with respect to demographic parameters. We examine the effect of a change in one parameter (adult survival, juvenile survival, reproductive success, dispersal, or catastrophic mortality) on the future trajectory of the metapopulation, compared to Scenario 1. Such comparisons provide insight into the sensitivity of model outcomes to the assumptions made regarding each parameter, as well as providing insight into the response of the metapopulation to a change in a demographic parameter, either due to environmental alteration or to an anthropogenic effect.

Change in Adult Survival - In Scenario 2 adult survival is assumed to be 75 percent; all other parameter values and assumptions are as in Scenario 1. Compared to Scenario 1, the metapopulation declines at a faster rate - 1.59 percent per year, on average (Fig. D-2, Table D-2). After 100 years, the metapopulation will have declined on average by 80 percent (Table D-2A). The probability of quasi-extinction is 2.8 percent (Table D-2B), with an approximate 95 percent confidence interval about that estimate of 0 to 7.2 percent. There is nearly 100 percent probability that the metapopulation will decline by at least 32 percent after 100 years. The probability of at least a 50 percent decline after 100 years is 96 percent. These results confirm that a small change in adult survival can have potent effects on the long-term metapopulation trajectory. Scenario 3 demonstrates the sensitivity of results to a 1 percent increase in adult survival. The metapopulation is still expected to decline, but at an even shallower rate compared to Scenario 1 - on average 0.46 percent per year, and 37 percent after 100 years (Table D-2A). The chance of any decline at all after 100 years is reduced to 96 percent. It would require a greater increase in adult survival (to above 78 percent) to produce a metapopulation whose long-term trajectory is essentially stable (Results not shown).

Change in Juvenile Survival - We consider two alternative scenarios. In Scenario 4, juvenile survival is reduced by 10 percent in relative terms, *i.e.*, a reduction of .05 in absolute terms, from 0.50 to 0.45 probability of surviving. A difference in survival of 0.05 is not unreasonably large; it is less than the standard error of the most precise estimate available for juvenile survival (Table D-1). 0.05 is also the quantity by which we incremented the Monterey Bay juvenile survival estimate to account for permanent emigration. Results (Fig. D-3A, Table D-2) under this scenario depict a metapopulation that is quickly declining (at 2.8 percent per year, on average) and quickly approaches critical levels. Under Scenario 4, there is a 42 percent chance of quasi-extinction. The probability of a 50 percent decline is essentially 100 percent. In fact, in 50 percent of the simulations, the metapopulation declines by 96 percent or more.

Scenario 4 shows the stark effects of a 10 percent relative change in juvenile survival. But what about the impact of more subtle changes in juvenile survival? To answer that question, in Scenario 5, we consider a 4 percent decrease, in relative terms, of juvenile survival, from 0.50 to 0.48. Note that from the point of view of a change in **mortality** (rather than survival), a change in juvenile survival from 0.50 to 0.48 implies a 4 percent relative increase in mortality, just as does a change in adult survival from 0.76 to 0.75. Results (Table D-2, Fig. D-3B) in this scenario demonstrate a metapopulation that declines with 100 percent probability, with an average decline of 1.5 percent per year, and a 78 percent decline after 100 years. Moreover, in 100 percent of simulations metapopulation size decreased by at least 26 percent. However, the probability of quasi-extinction is low, 3.5 percent (Table D-2B). We conclude that relatively small changes in juvenile survival will have sizeable impacts on long-term population trends, but will not have large effects on quasi-extinction probabilities.

Change in Reproductive Success - In the age-structured model used in the population viability analysis, a change in juvenile survival of *k* percent is exactly equivalent to a change in reproductive success (fledglings per male adult) of *k* percent. This is because only the product of juvenile survival × reproductive success is modeled. Hence, Scenarios 4 and 5 (discussed above) demonstrate the effects of a 10 percent and 4 percent change, respectively, *in reproductive success*, just as they do for a change in juvenile survival. We also consider sensitivity of model results to assumptions about reproductive success of just the SLO subpopulation. In Scenarios 1-5 above, an intermediate value of reproductive success was assumed for the SLO subpopulation (0.919 fledged young per male). Scenario 6, instead, assumes an optimistic value of 0.988 fledged chicks per male; *i.e.*, that value of reproductive success which would produce a stable, self-sustaining population in the absence of immigration and emigration. Scenario 7, instead, assumes a pessimistic value of 0.849 fledged chicks per male; *i.e.*, the same reproductive success as assumed for NC and SFB and as observed in Monterey Bay in the absence of intensive management. Results are summarized in Tables D-2A and D-2B. The effect of a 7.5 percent relative change in SLO reproductive success, either an increase (Scenario 6) or a decrease (Scenario 7), is fairly minor. For example, comparing Scenarios 1 and 6, lambda for the metapopulation increases slightly from 0.9908 to 0.9926, a difference of less than 0.2 percent (Table D-2A). The chance of a 50 percent decline for the metapopulation decreases from 72 percent (Scenario 1) to 59 percent (Scenario 6) (Table D-2B). Similarly, comparisons of Scenarios 7 and 1, indicate only minor differences (Table D-2). We conclude that, though reproductive success for SLO cannot be estimated with great certainty, results of the population viability analysis are not very sensitive to assumptions made regarding this parameter, providing they are within a reasonable range (bounded by the optimistic and pessimistic values considered).

Change in Catastrophe - Scenario 8 assumes no catastrophic reproductive failure at all. Compared to Scenario 1, the effect of eliminating catastrophic reproductive failure is to increase lambda slightly, by 0.3 percent (0.9938 instead of 0.9908; Table D-2A). However, the absence of catastrophic failure results in a substantial reduction in risk of metapopulation decline, from 72 percent chance of a 50 percent decline to a 42 percent probability in Scenario 8 (Table D-2B). An even larger impact on the risk of metapopulation decline is observed in Scenario 9, in which catastrophic mortality of adults is added to catastrophic reproductive failure in years of catastrophe. In Scenario 9, lambda decreases substantially, to 0.9763 (Table D-2A). Under this scenario, we expect, on average, a 91 percent decline in metapopulation size. In addition, the risk of quasi-extinction is 29 percent, with a 99 percent probability that the metapopulation decreases by at least 50 percent after 100 years (Table D-2B). These results demonstrate that a relatively rare catastrophic event (5 percent probability per year) can have a large long-term effect on population growth and risk, if it entails a substantial increase in adult (and possibly juvenile) mortality. If catastrophes are as common as is assumed in Scenario 9, then the risk of metapopulation decline will be severely underestimated by any model which does not incorporate catastrophes.

Change in Dispersal - Here we consider the impact of a 50 percent and a 100 percent decrease in dispersal rates (Scenarios 10 and 11, respectively). That is, in Scenario 10 all dispersal rates were reduced by 1/2, and in Scenario 11, we assumed no dispersal whatsoever among subpopulations. The dynamics of the metapopulation as a whole were not much affected by even large changes in dispersal rates (Tables D-2A and D-2B). With a 50 percent reduction in dispersal (Scenario 10), the population growth rate increased slightly to lambda = 0.9914, that is, the metapopulation declined at an average of 0.86 percent per year instead of 0.92 percent (Scenario 1). The probability of quasi-extinction remained essentially zero, and the probability of a 50 percent decline after 100 years was little changed (71 percent instead of 72 percent for Scenario 1). Even when dispersal was eliminated the dynamics were not altered greatly. In the latter case, lambda decreased to 0.9906, almost identical to that observed in Scenario 1. The probability of a 50 percent decline after 100 years increased somewhat, from 72 percent in Scenario 1 to 79 percent in Scenario 11.

A 50 percent reduction in dispersal rates, also had only minor effects on the expected sizes of the six subpopulations after 100 years (Fig. D-4A; cf. Fig. D-1D). The most notable difference is an increased size of the MB subpopulation with reduced dispersal. With the elimination of dispersal, two subpopulations could be expected to go completely extinct with more than 50 percent probability, NC and SFB (Fig. D-4B). We conclude that within the likely range of dispersal rates, model results are not very sensitive to the exact parameter values used.

Changes in Management

We consider the impact of changes in management practice that may increase or decrease reproductive success. It is possible for changes in management practice to impact other demographic parameters, but we consider that possibility less likely.

Scenario 12 assumes "**No Management**". We assume cessation of management in OR, MB, and SD and that the other subpopulations continue as in the present (*i.e.*, as in Status Quo, Scenario 1). In Scenario 12, reproductive success is assumed to be 0.849 chicks per male for OR, MB, and SD, just as it is for NC and SFB. All other parameter values are as in Scenario 1. The expected outcome under this Scenario is for the metapopulation to show a strongly declining trend (Fig. D-5A, Table D-2A). Likelihood of decrease below specified population levels (for the entire metapopulation) is shown in Fig. D-5B. The probability that the metapopulation will decline by at least 50 percent after 100 years is 100 percent. In fact, there is a 100 percent probability of at least a 77 percent decline (Fig. D-5B). The probability of quasi-extinction is 51 percent (Table D-2B).

Clearly, the abandonment of management that protects western snowy plovers is an unpalatable alternative.

Scenario 13 is a modification of Scenario 12. In Scenario 13, metapopulation size is assumed to begin with 3500 individuals, close to, and slightly in excess of, the number of individuals for which there is at present available beach habitat. One can imagine that intensive management resulted in an increase in western snowy plover numbers until a population size of 3500 was reached, but that once reached, management activities ceased. In other words, Scenario 13 differs from Scenario 12 only with respect to starting population sizes. It is also assumed that with a metapopulation size of 3500, all ceiling values are increased by 10 percent (*i.e.*, to 3850 breeding individuals). As expected, the metapopulation shows the same steep population decline as in Scenario 12 (Table D-2A). In one sense, all Scenario 13 does (compared to Scenario 12) is to buy some time for the metapopulation. After 21 years, the metapopulation has decreased from 3500 individuals to about 1950, the starting level for Scenario 5. After 100 years, the probability that the metapopulation has fallen below 50 individuals is 35 percent (cf. to 51 percent for Scenario 5). There is a 100 percent probability that the population will decline at least 85 percent. These results demonstrate that simply increasing population size is not a viable solution for the western snowy plover metapopulation.

We next considered scenarios in which reproductive success is enhanced. In the next four scenarios we assumed that management continues in OR, MB, and SD, as it has, and that, therefore, fecundity and other parameter values continue as at present. In the first of these (Scenario 14), we assume that management activities in SLO (the largest subpopulation) results in an increase in fecundity to that obtained in MB now (*i.e.*, 1.105 chicks fledged per breeding male). Results are shown in Fig. D-6, indicating that, on average, the population declines, albeit at a very slight rate (0.3 percent decline per year; Table D-2A). There is an 85 percent chance of at least some decline, and a 19 percent chance of a 50 percent decline (Table D-2B). The probability of quasi-extinction is zero.

In the next scenario (Scenario 15), it is assumed that management activities at SLO are not quite as effective, and that reproductive success can only be increased to 1.0 fledged chicks per male. In this case, population growth rate declines at, on average, 0.7 percent per year (Table D-2A). As a result, there is a 51 percent probability of at least a 50 percent decline, over 100 years. While, this result is an improvement over the results of the Status Quo scenario (Scenario 1), it would still not be considered a desirable outcome.

An alternative scenario (Scenario 16) is for management action to increase reproductive success in NC and SFB, with SLO remaining as it is now. Results of Scenario 16 are a slight decline, just as in Scenario 14 (0.3 percent decline per year; Table D-2A). However, results from this scenario indicate less variability of outcome (Fig. D-7) compared to Scenario 14, in which SLO reproductive success was enhanced. As a result, the probability of a 50 percent decline is only 6 percent (Table D-2B). The probability of quasi-extinction is zero.

Comparison of results from Scenarios 14 and 16 indicate that increases in reproductive success of either SLO or SFB and NC would be effective in stabilizing western snowy plover numbers, and reducing the risk of substantial population decline in the future.

None of the scenarios presented above result in likely population increase. We therefore considered three additional metapopulation scenarios (Scenarios 17-19). In Scenario 17, management at SLO, NC, and SFB are such that all three subpopulations achieve fecundity of 1.105 chicks reared per breeding male (with the other three subpopulations as assumed above). Under this scenario the metapopulation does show an increase, but a surprisingly shallow increase: lambda = 1.0013 (Table D-2A), an annual growth rate of 0.13 percent per year. At the end of 100 years, the metapopulation is expected to grow by a total of 14.4 percent, on average. The relatively flat trajectory is surprising because we expected numbers to show an increase to close to ceiling levels, an 87 percent increase if all ceiling levels were attained. It turns out that some subpopulations achieved ceiling levels while others did not (Fig. D-8). Fig. D-8 demonstrates that (under assumptions of the model), OR, NC, SFB, and MB, were on average close to their ceiling levels, but SLO and SD are not. SLO and SD numbers would increase much further if excess individuals at other subpopulations (above ceiling levels) were to disperse to SLO and SD; however, such selective dispersal was not incorporated into the simulations, nor is it possible to do so using the RAMAS/GIS 2.0 program. Therefore, we consider the results from Scenario 17 to be somewhat unrealistic, since they incorporate unrealistic assumptions about dispersal when subpopulation size is at or near ceiling levels. A more sophisticated modeling program is required to incorporate assumptions about the dependence of dispersal on population size relative to population ceiling size.

Finally, we considered two scenarios in which population increase can be expected to reach 3000 western snowy plovers within a 25 year period. In the first of these (Scenario 18), reproductive success is assumed to be 1.3 chicks per male for all subpopulations. This level of reproductive success is high, but attainable; in 1998, western snowy plovers in the Monterey Bay study area achieved this level of reproductive success. This scenario assumes that with sufficiently intensive management, all subpopulations will be able to achieve this level of reproductive success at some time in the future. Under this scenario, there is an 82 percent chance of the population reaching 3000 or more birds at the end of 25 years (see Table D-3). At first the size of the metapopulation increases rapidly, but the rate of growth slows down beyond year 10 (Fig. D-9), and then shows very slow growth beyond year 15.

The last scenario (Scenario 19) assumes that reproductive success of 1.2 chicks fledged per male is achieved for all subpopulations. Under this scenario, there is a 57 percent chance that the metapopulation will contain 3000 or more individuals after 25 years. The median outcome after 25 years is 3110 individuals, which is only 540 less than the overall maximum allowed for the metapopulation. Scenarios 18 and 19 demonstrate that there is a reasonably high probability of achieving at least 3000 birds within 25 years, provided that reproductive success averages 1.2 or more chicks per male over all subpopulations.

Discussion

In all modeling exercises, the results are sensitive to the assumptions. In this case we have tried to make assumptions explicit and we have examined the influence of the assumptions (or assumed values) on model results. The strength of the current analysis is that demographic estimates were based on data gathered from study populations within the Pacific coast metapopulation. An important feature of the population viability analysis is the use of a metapopulation structure that allows estimates for parameters to vary among subpopulations. We consider it highly desirable for population viability analyses to incorporate such flexibility.

Reproductive Parameters

That we could allow for subpopulation-specific parameters is a boon, yet the lack of available estimates for several of the subpopulations constitutes a drawback to the population viability analysis. In particular, no demographic parameter estimates are available for the SLO subpopulation, which is estimated to contain 45 percent of the entire metapopulation. Obtaining fecundity estimates for this subpopulation, as well as for NC and SFB, should be a priority. Even when we assumed that reproductive success in SLO was sufficiently high to produce a self-sustaining population, the metapopulation, on average, showed a decline at 0.74 percent per year, under the Status Quo conditions ("optimistic" scenario, Scenario 6). On the other hand, if reproductive success in SLO is as low as 0.849 chicks per breeding male ("pessimistic" Scenario,

Scenario 7) then the metapopulation would be expected to decline at a faster rate, at 1.1 percent per year. Though it would be desirable to obtain estimates from the SLO subpopulation itself, the sensitivity analyses demonstrated that results were not unduly sensitive to the estimate of reproductive success for this subpopulation, if SLO reproductive success was within the range of values modeled.

Dispersal

Theoretical studies have demonstrated that dispersal among subpopulations will reduce the chance of extinction of the metapopulation (Burgman et al. 1993, Harrison 1994), compared to a set of isolated subpopulations. In this case, we had reasonably good empirical data from the Monterey Bay study population, indicating dispersal rates of 20 percent to 25 percent among first-year birds. An area of uncertainty was whether dispersal rates varied with density (Beissinger and Westphal 1998). Recent observations of western snowy plovers indicate that dispersal occurs at high and low densities, and therefore we did not include density-dependent dispersal in the modeling. However, there may be a threshold effect: once a breeding area (*e.g.*, beach) is saturated, dispersal from that area may be enhanced. Future modeling could address this possibility, and its implications. Though our knowledge of dispersal was incomplete, it did not appear that model results were very sensitive to assumed dispersal rates. In particular, a 50 percent relative reduction in dispersal had almost no discernible effect on the metapopulation trajectory, persistence, or on subpopulation composition. This provides us with some confidence in model results despite the acknowledged uncertainty in dispersal rates.

Adult and Juvenile Survival

The sensitivity analysis (Scenarios 2-11) demonstrated a strong effect of inclusion of catastrophic mortality of adults. It is possible that the El Niño of 1998 will demonstrate such catastrophic mortality, but such a phenomenon cannot be demonstrated until completion of the 1999 breeding season, at the earliest. The sensitivity analysis also confirmed the sensitivity of metapopulation trajectory to moderately large changes in reproductive success and/or juvenile survival. We did not examine the sensitivity of results to a moderately large long-term change in adult survival, but even a small change (1 percent change in absolute survival) had a noticeable effect on metapopulation trajectory. Nevertheless, the probability of quasi-extinction was low whether adult survival was 0.75 (Scenario 2), 0.76 (Scenario 1), or 0.77 (Scenario 3). We conclude that, in general, the results shown are applicable, assuming that adult survival was between 0.75 and 0.77. We consider it unlikely that adult survival was much lower than 0.75. At the same time, there is no support for assuming that adult survival was greater than 0.77. Adult survival would have to be greater than

0.78 (Results not shown) to produce a metapopulation that is likely to grow, and even then it would only be growing slowly.

In most Scenarios, we assumed 0.50 juvenile survival. Though juvenile survival was surely at least 0.45, it is debatable just how much greater it is than 0.45. Thus, our results could be considered a bit liberal, or optimistic. If juvenile survival was actually lower than 0.50 (as in Scenarios 4 and 5) population trends would be more pessimistic.

Limitations to the Population Viability Analysis

There are several limitations to the population viability analysis. First, we did not include risk to the metapopulation due to genetic factors. Such a simplification (ignoring genetic factors) is consistent with recommendations of Beissinger and Westphal (1998). Genetics would become much more important to consider if metapopulation size would likely decrease to low levels, that is, 50 or fewer. However, population viability analysis results here indicate decrease to such low levels unlikely.

Second, we did not take into account an "Allee effect," which is a decrease in survival or reproductive success with a decrease in population size, usually due to social factors. For example, Allee effects can arise if individuals have difficulty securing mates when density is low. However, we believe that as long as metapopulation size remains at 50 or more (see above), Allee effects are not likely important.

The use of a packaged program (RAMAS/GIS) had the advantages of convenience, reproducibility, and general availability. Balancing that were limitations of that particular program. As already mentioned, dispersal was modeled at a constant rate and does not vary stochastically. Dispersal cannot vary with the size of the target population. Nor can one specify a constant number of dispersers. Thus, for example, one cannot specify balanced dispersal (dispersal from the population exactly equals dispersal to that population). Furthermore, with RAMAS/GIS dispersal cannot be modeled as a threshold phenomenon (e.g., dispersal only for those in excess of carrying capacity). Even if dispersal could be modeled in very sophisticated ways, we are limited by the lack of information regarding dispersal. Other limitations of RAMAS/GIS included the requirement that temporal covariation of population parameters is 100 percent. If it is a very good year for survival, the program assumes it is a very good year for reproductive success. There are many limitations on modeling density dependence with RAMAS/GIS. For example, we could not model a "ceiling effect" on reproductive success (*i.e.*, individuals in excess of the ceiling do not reproduce), and had to assume that excess individuals were dead.

Tentative Conclusions

Results from this population viability analysis highlight the need for increased management of Pacific coast western snowy plovers and their habitats. Under status quo scenarios, even with intensive management in some areas, the population is almost certain to decline. Without question, ceasing current management efforts (area closures, predator exclosures, and predator control) would be disastrous for the Pacific coast population. The Western Snowy Plover Recovery Team, however, has identified population growth as a prerequisite to recovery. The most direct means to increase population size will be to enhance reproductive success throughout the western snowy plover range. The model suggests that productivity of **at least** 1.0 chicks fledged per breeding male per year should result in a stable population, if our estimates of adult and juvenile survivorship are accurate. Productivity of 1.2 or more chicks fledged per breeding male should increase population size at a moderate pace before growth slows as the metapopulation approaches its ceiling. Population growth would be hastened, of course, if survival of adults or juveniles can also be improved. Under this population growth scenario, the metapopulation could increase to 3000 individuals within the relatively short time span of 25 years. Recovery is plausible. It will require, however, short-term intensive management and long-term commitments to maintaining gains.

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Table D-1. Western snowy plover demographic parameter estimates.

A) Percent adult male survival, for males, excluding first-year after banding.

	All Observations (including Winter)	Expanded Area (Breeding Season only)	Single Study Area (Breeding Season only)
Monterey Bay	74.7 ± 1.9	74.3 ± 1.9	73.7 ± 3.6
Oregon	74.5 ± 13	74.3 ± 8.5	73.6 ± 18
San Diego	71.3 ± 9.0	71.3 ± 9.0	71.3 ± 16

Notes: Observed between-year standard deviation in Monterey Bay = 5.65 percent; mean adult male survival used in the population viability analysis is 76 percent (also 75 percent and 77 percent, see text).

B) Percent Juvenile (1st Year) survival, post-fledging.

	All Observations (including Winter)	Expanded Area (Breeding Season only)	Single Study Area (Breeding Season only)
Monterey Bay	45 ± 15	44 ± 6.7	39 ± 12
Oregon	51 ± 40	49 ± 53	44 ± 65
San Diego	45 ± 22	43 ± 15	42 ± 16

Notes: Between-year standard deviation = 6.8 percent for Monterey Bay. Juvenile survival used in population viability analysis = 50 percent (also 48 percent and 45 percent, see text).

C) Fecundity (chicks reared to fledging, per adult male).

Study Population	Years	Mean	Between-year standard deviation
Monterey Bay w/o predator control	1984-1991	0.849	0.173
Monterey Bay w/ predator control	1992-1997	1.105	0.157
Oregon	1993-1997	1.040	---
San Diego	1995-1997	0.917	---

Table D-2. Summary of stochastic results, after 100 years (400 simulations each scenario).

A. Summary of long-term population trajectories.

Scenario No.	Description	Minimum	X - S.D.	Mean	X + S.D.	Lambda	Percent Change
1	Status Quo (SQ)	61	410	771	1131	0.9908	-61
2	SQ but 75 percent adult survival	0	127	391	654	0.9841	-80
3	SQ but 77 percent adult survival	182	817	1232	1647	0.9954	-37
4	Juvenile survival or reproductive success reduced 10 percent	0	5	118	231	0.9723	-94
5	Juvenile survival or reproductive success reduced 4 percent	3	134	437	740	0.9851	-78
6	SQ but optimistic SLO reproductive success estimate	28	511	930	1348	0.9926	-52
7	SQ but pessimistic SLO reproductive success estimate	28	306	639	972	0.9889	-67
8	SQ, no catastrophic mortality	147	669	1044	1419	0.9938	-46
9	Catastrophic mortality includes survival and reproductive failure	0	0	177	362	0.9763	-91
10	Dispersal reduced by 1/2	85	453	825	1196	0.9914	-58
11	No dispersal	7	448	757	1066	0.9906	-62
12	No management	0	5	86	166	0.9692	-96
13	Start with 3500 total; no management	0	16	116	215	0.9722	-94
14	Improve SLO reproductive success to 1.105 chicks	198	934	1445	1957	0.9970	-26
15	Improve SLO reproductive success to 1.0 chicks	80	560	975	1389	0.9931	-50
16	Improve NC and SFB reproductive success to 1.105 chicks	601	1138	1440	1742	0.9970	-26
17	Improve reproductive success at SLO, NC and SFB to 1.105 chicks	1018	1741	2230	2718	1.0013	14.4

Note: The last column shows mean total percent decline after 100 years, except for Scenario 17, for which percent increase is shown.

Table D-2. Summary of Stochastic Results, continued

B. Probability of Quasi-extinction and Probability of Specified Declines during 100 years.

Scenario No.	Description	Probability of Quasi-Extinction, percent[1]	Probability of any decline, as percent	Probability of 50 percent decline, as percent	Median percent decline[2]
1	Status Quo (SQ)	0	100	72	61
2	SQ w/ 75 percent Adult Survival	2.8	100	96	83
3	SQ w/ 77 percent Adult Survival	0	96	27	36
4	Juvenile Survival/reproductive success reduced 10 percent	42	100	100	96
5	Juvenile Survival or reproductive success reduced 4 percent	3.5	100	92	81
6	SQ + optimistic SLO reproductive success estimate	0.3	100	59	54
7	SQ + pessimistic SLO reproductive success estimate	0.3	100	83	69
8	SQ, no catastrophic reproductive failure	0	100	42	46
9	Catastrophic mortality includes survival and reproductive failure	29	100	99	94
10	Dispersal reduced by 1/2	0	100	71	59
11	No dispersal	0.3	100	79	64
12	No management	51	100	100	97
13	Start with 3500; no management	35	100	100	97
14	Improve SLO reproductive success to 1.105 chicks	0	85	19	26
15	Improve SLO reproductive success to 1.0 chicks	0.3	99	51	50
16	Improve NC and SFB reproductive success to 1.105 chicks	0	97	6	25
17	Improve reproductive success at SLO, NC and SFB to 1.105 chicks	0	30	0	12[2]

[1] - Standard error of the estimate of Probability of Quasi-extinction is $\pm$ 2.2 percent in all cases.

[2] - Median percent increase in total population size.

Table D-3. Summary of results for growth scenarios, at the end of 25 years.

Scenario No.	Description	Median outcome after 25 years, N	Probability of 3000+ after 25 years, percent	Population size reached after 25 years with 80 percent probability, N	Percent annual growth rate in first 15 years[1]
18	Improve reproductive success to 1.3 chicks per male in all subpopulations	3341	82	3018	3.35
19	Improve reproductive success to 1.2 chicks per male in all subpopulations	3110	57	2740	2.95

[1] - Annualized growth rate, calculated for first 15 years.

A

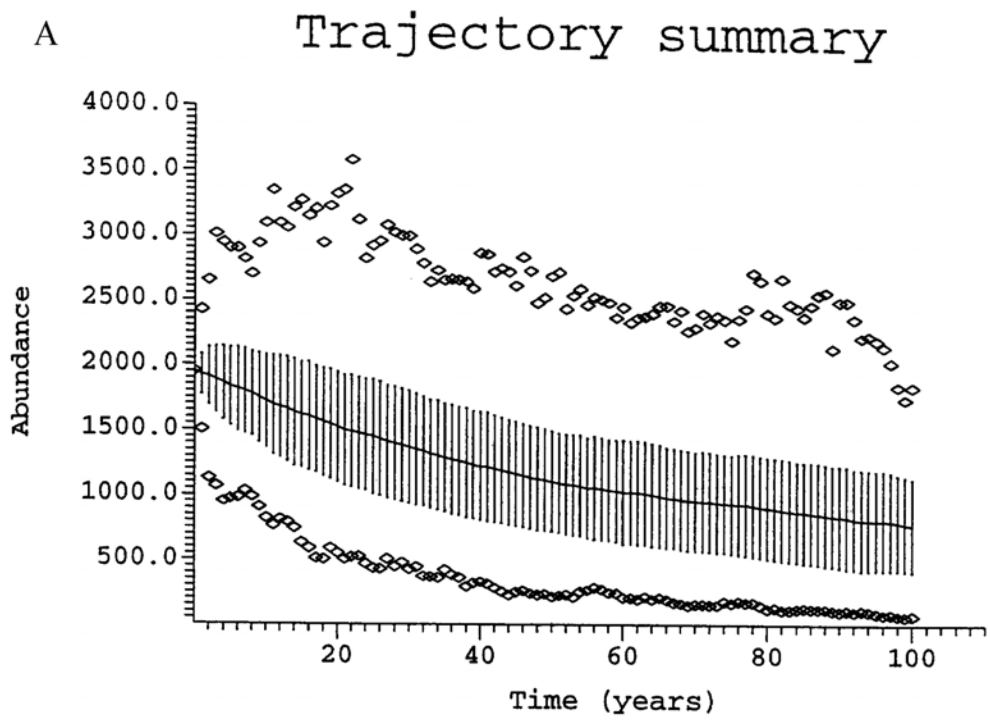
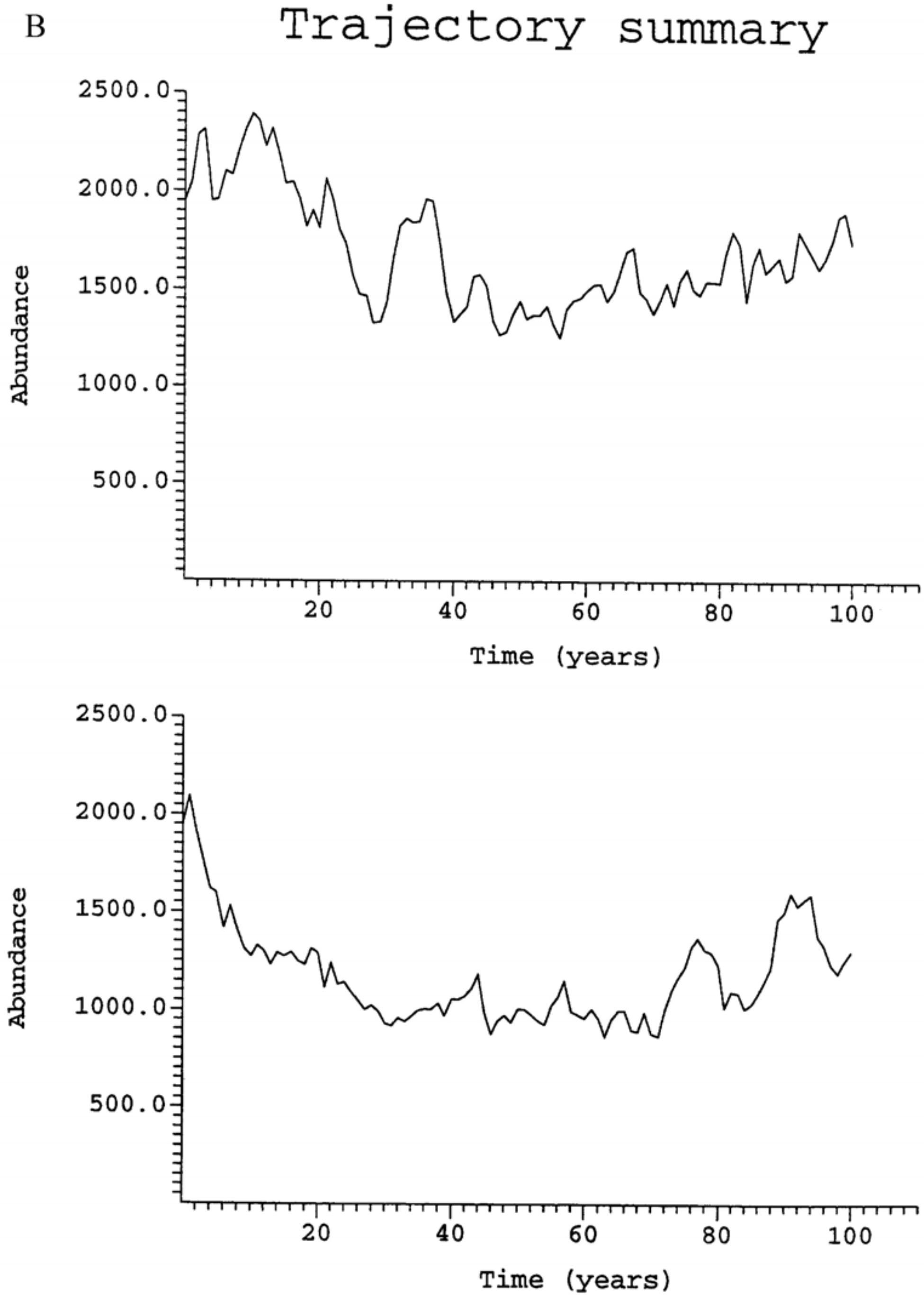


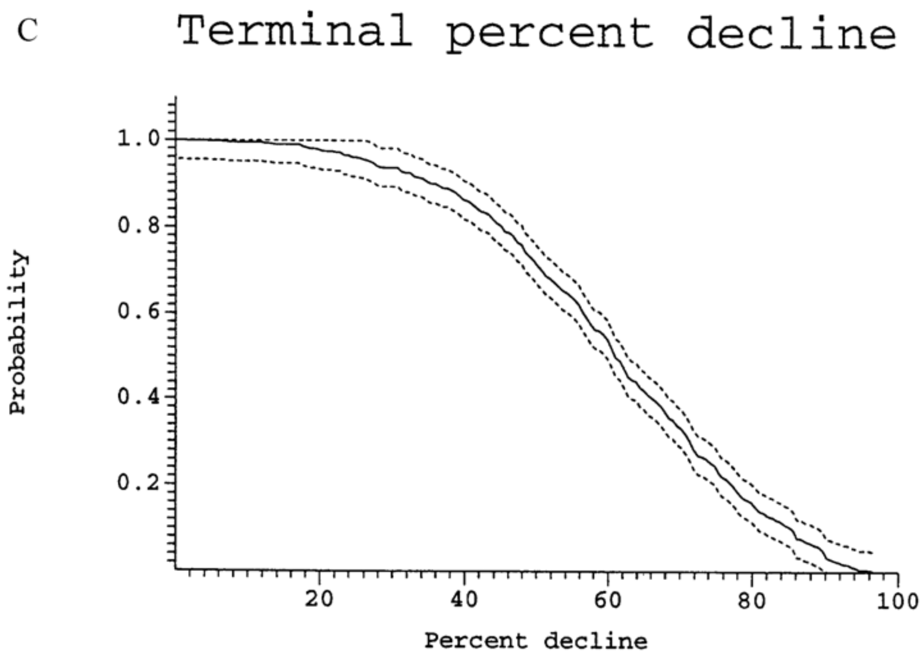
Figure D-1. Scenario 1: Status Quo (see text). A) Population trajectory for the metapopulation. Diamonds indicate maximum and minimum (400 simulations, total). Horizontal line indicates mean trajectory. Vertical lines connect mean +/- 1 standard deviation of outcome. B) Population trajectories for two sample simulations (among 400), under Scenario 1. C) Probability that after 100 years the metapopulation will have declined below specified level. Dotted lines indicate approximate 95 percent confidence interval. D) Abundance for each subpopulation (abbreviated as in text) at the end of 100 years. Bars indicate means, vertical lines with bars indicate + 1 standard deviation. Diamonds show maximum (among 400 simulations).

B

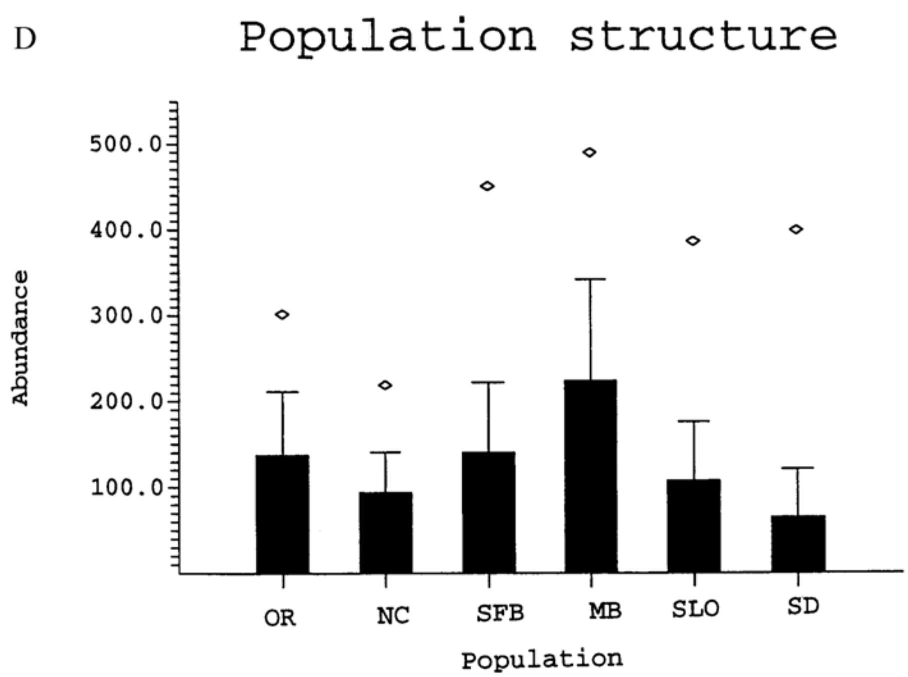


Trajectory summary
Abundance
2500.0
2000.0
1500.0
1000.0
500.0
20
40
60
80
100
Time (years)
Abundance
2500.0
2000.0
1500.0
1000.0
500.0
20
40
60
80
100
Time (years)

C


Terminal percent decline
Probability
1.0
0.8
0.6
0.4
0.2
20
40
60
80
100
Percent decline

D


Population structure
Abundance
500.0
400.0
300.0
200.0
100.0
OR
NC
SFB
MB
SLO
SD
Population

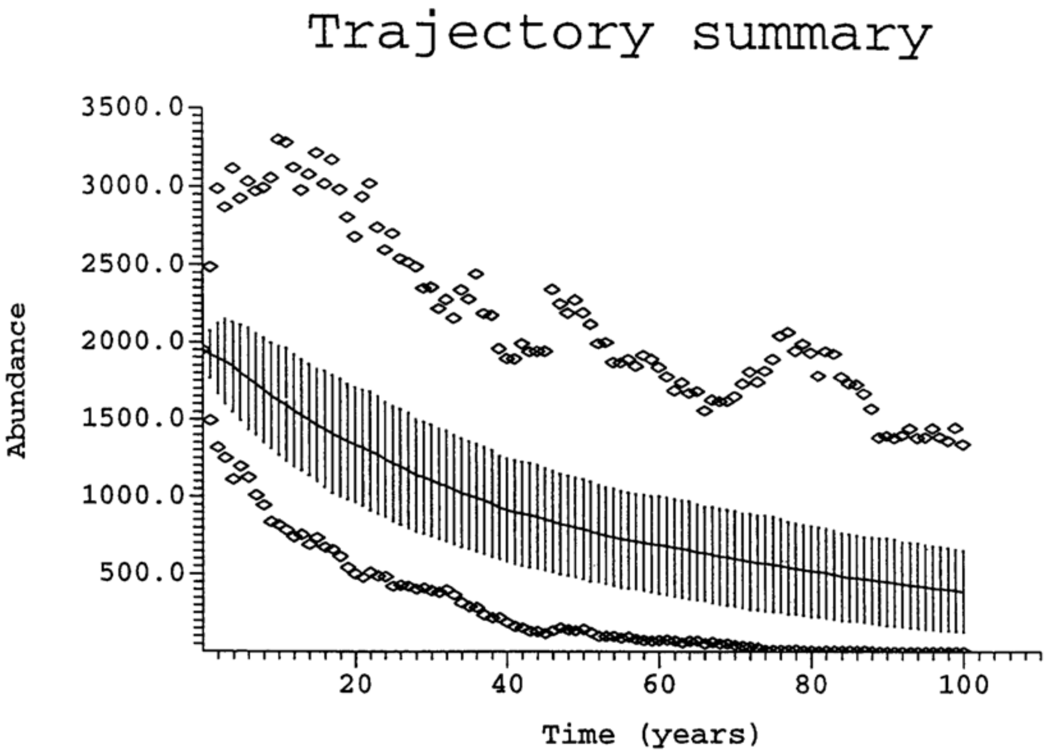


Figure D-2. Scenario 2: Status Quo with 75 percent adult survival instead of 76 percent. Population trajectory for the metapopulation. Diamonds indicate maximum and minimum (400 simulations, total). Horizontal line indicates mean trajectory. Vertical lines connect mean +/- 1 standard deviation of outcome.

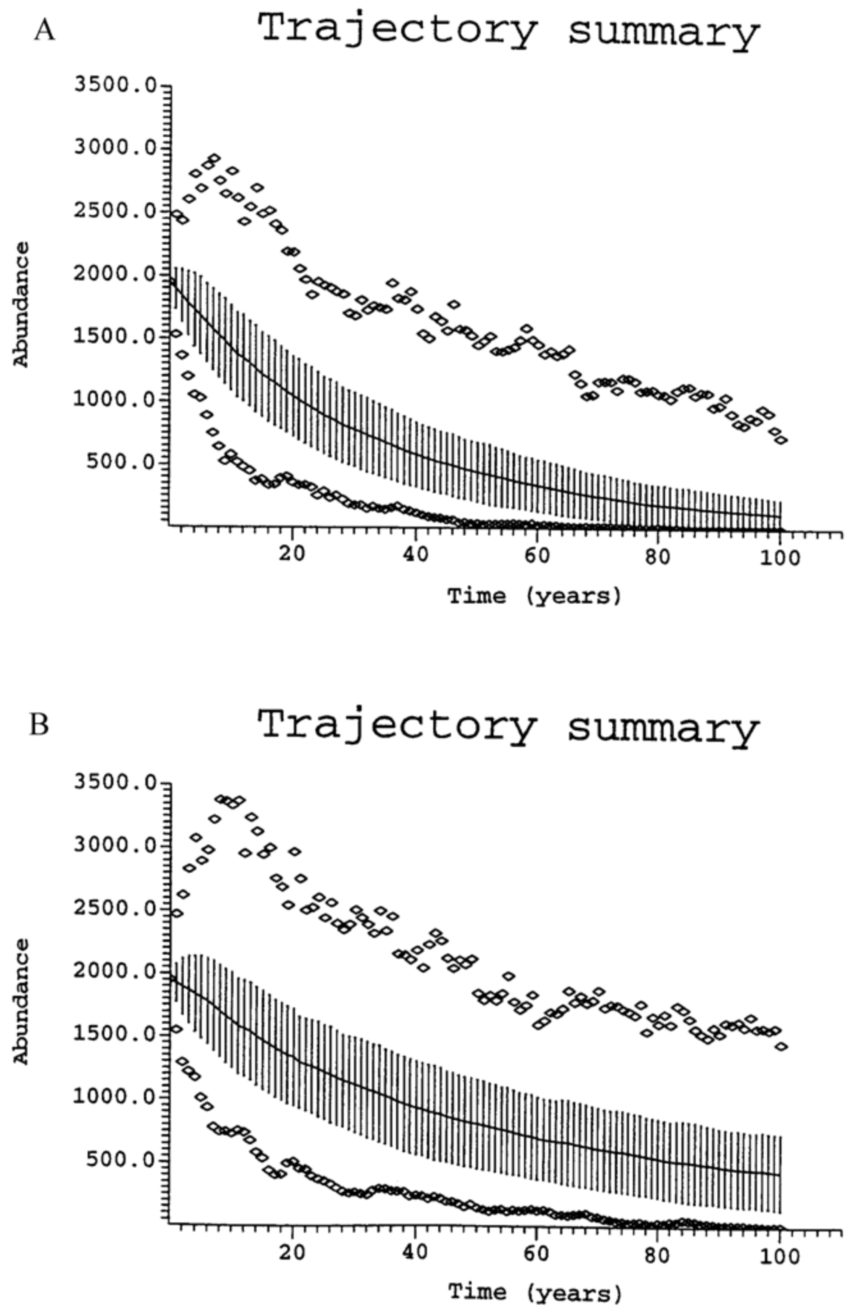


Figure D-3. Scenarios 4 and 5: Status Quo with reduction in juvenile survival (equivalently, reproductive success) by 10 percent (A) and by 4 percent (B). In each Figure panel: Population trajectory for the metapopulation. Diamonds indicate maximum and minimum (400 simulations, total). Horizontal line indicates mean trajectory. Vertical lines connect mean +/- 1 standard deviation of outcome.

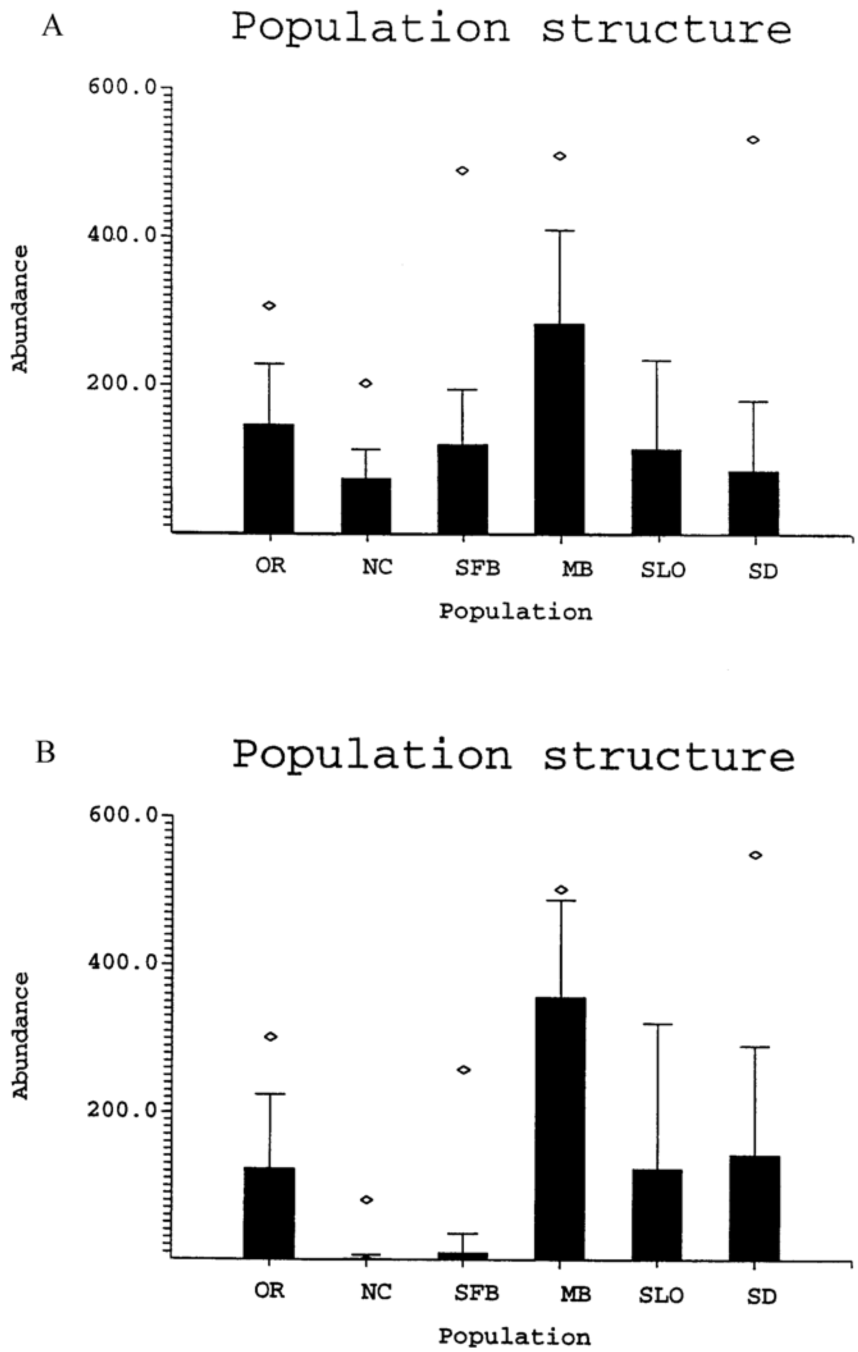
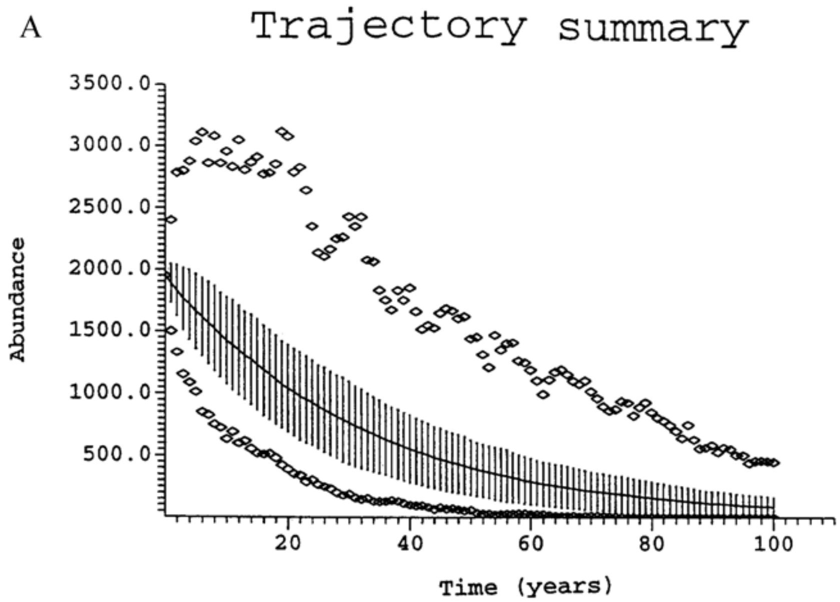


Figure D-4. Scenarios 8 and 9: Status Quo with reduction in dispersal. A) Dispersal reduced by 1/2 (Scenario 8). B) No dispersal (Scenario 9). For each Figure panel: Abundance for each subpopulation at the end of 100 years. Bars indicate means; vertical lines with bar indicate +1 standard deviation. Diamonds show maximum (among 400 simulations).

A



B

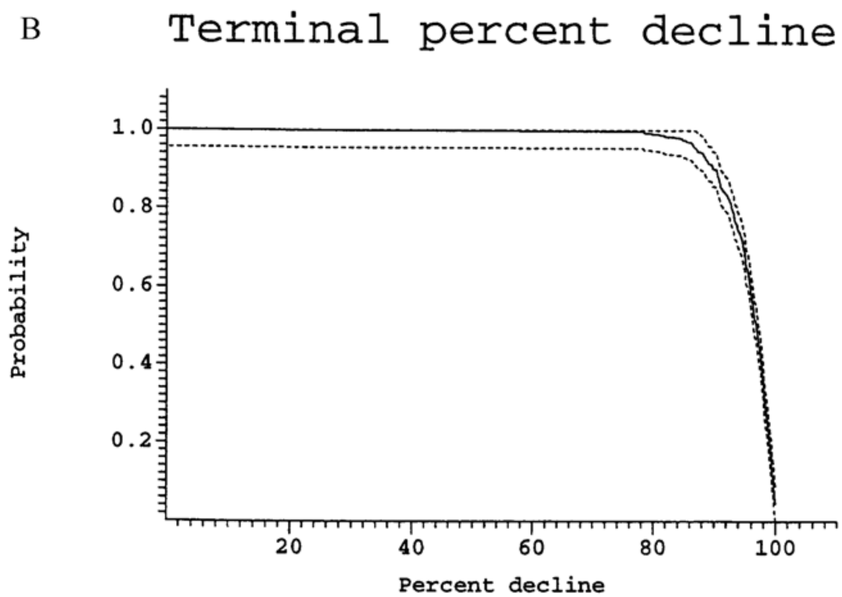


Figure D-5. Scenario 12: No Management. A) Population trajectory for the metapopulation. Diamonds indicate maximum and minimum (400 simulations, total). Horizontal line indicates mean trajectory. Vertical lines connect mean +/- 1 standard deviation of outcome. B) Probability that at the end of 100 years the metapopulation will have declined below specified level. Dotted lines indicate approximate 95 percent confidence interval.

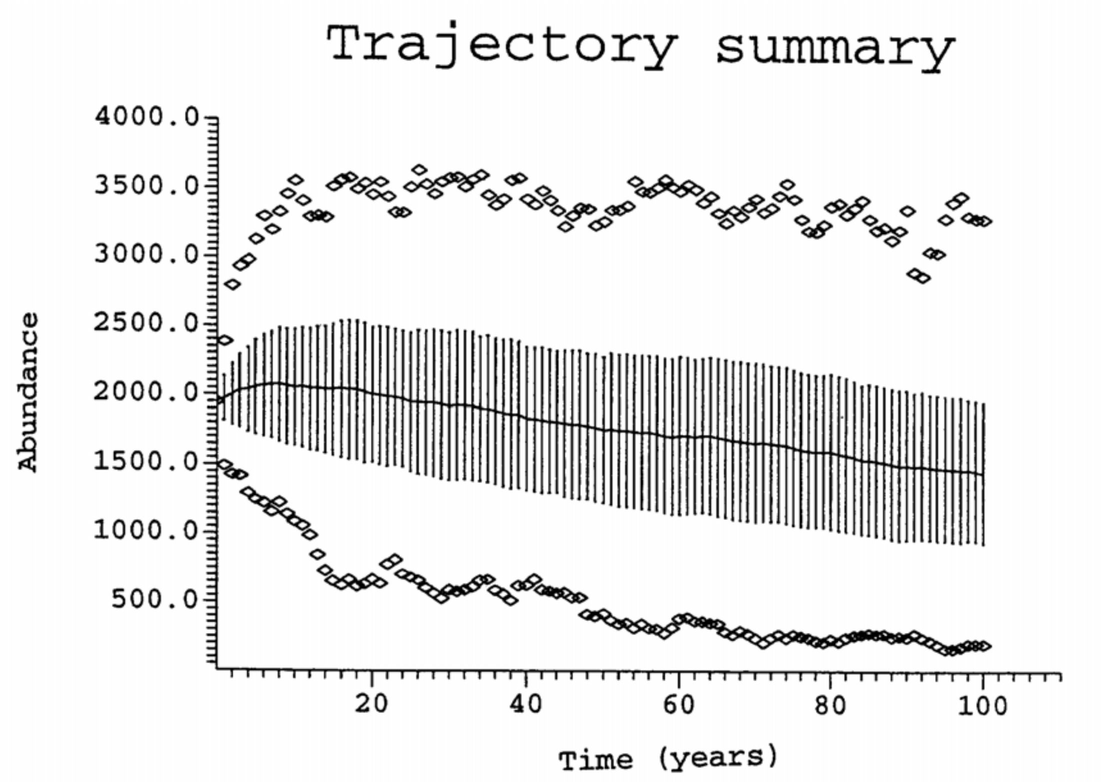


Figure D-6. Scenario 14: Improve reproductive success in San Luis Obispo/Santa Barbara/Ventura subpopulation and Status Quo elsewhere; see text. Population trajectory for the metapopulation. Diamonds indicate maximum and minimum (400 simulations, total). Horizontal line indicates mean trajectory. Vertical lines connect mean +/- 1 standard deviation of outcome.

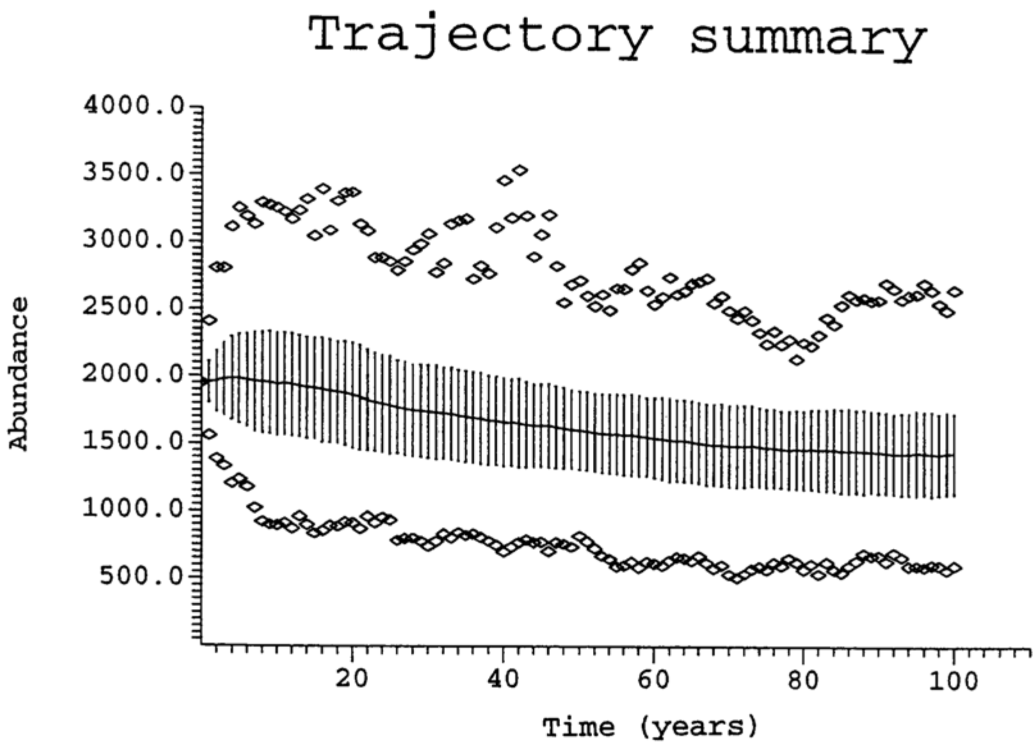


Figure D-7. Scenario 16: Improve reproductive success in San Francisco Bay and Northern California Coast subpopulations, Status Quo elsewhere; see text. Population trajectory for the metapopulation. Diamonds indicate maximum and minimum (400 simulations, total). Horizontal line indicates mean trajectory. Vertical lines connect mean +/- 1 standard deviation of outcome.

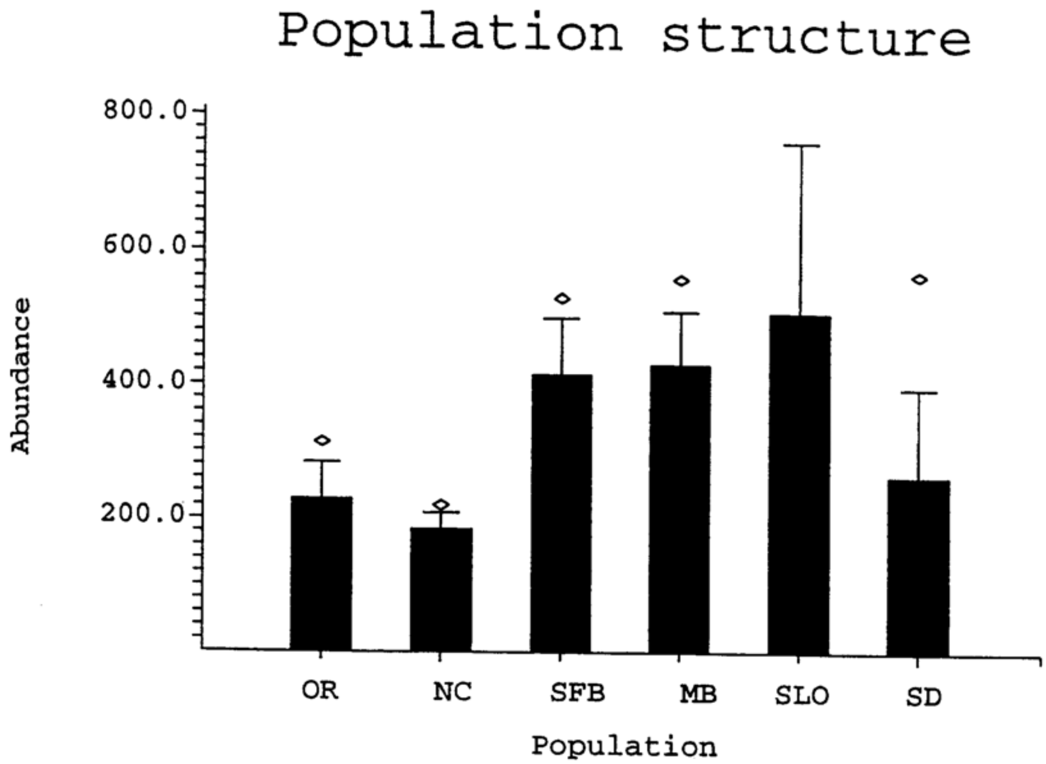


Figure D-8. Scenario 17: Management at all areas (see text). Abundance for each subpopulation at the end of 100 years. Bars indicate means; vertical lines with bars indicate + 1 standard deviation. Diamonds show maximum (among 400 simulations).

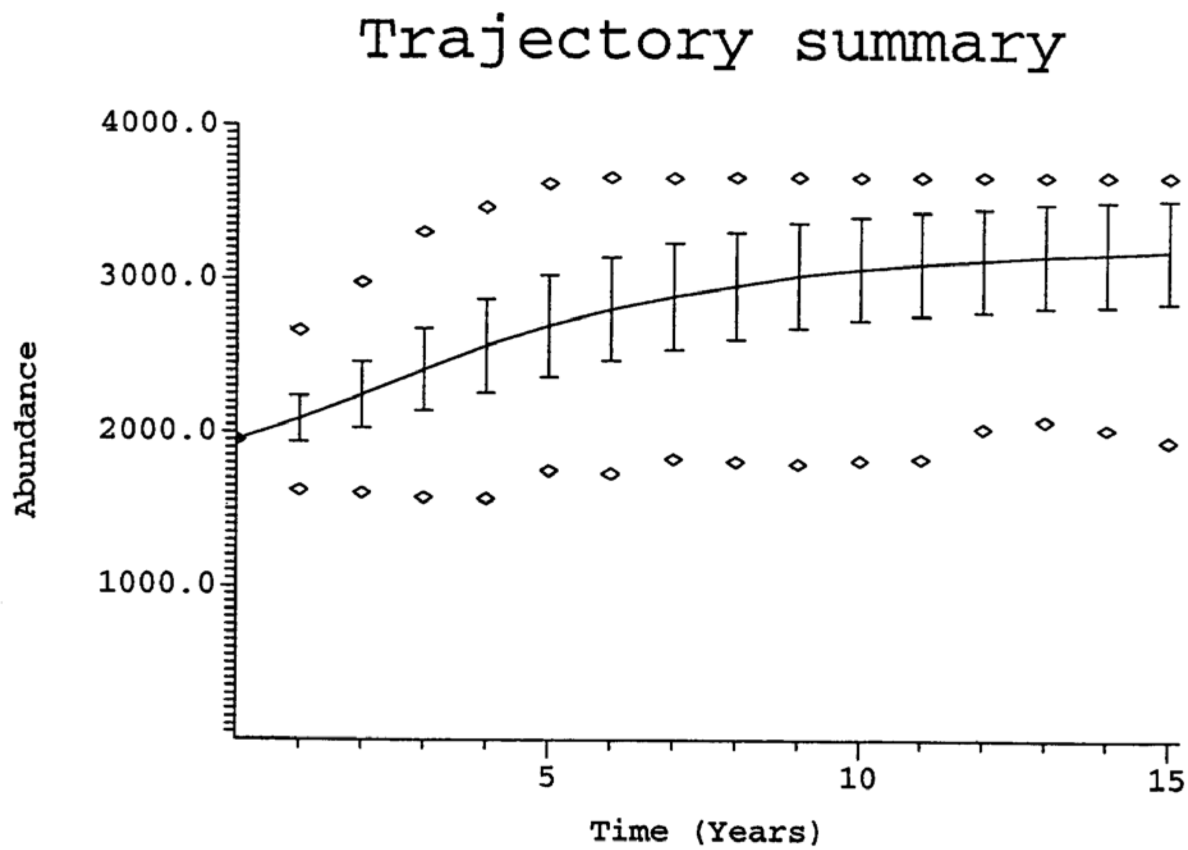


Figure D-9. Scenario 18: Recovery of western snowy plovers assuming 1.3 chicks fledged per male in all subpopulations. Population trajectory for the metapopulation is shown for first 15 years of the scenario. Diamonds indicate maximum and minimum (400 simulations, total). Horizontal line indicates mean trajectory. Vertical lines connect mean +/- 1 standard deviation of outcome.

APPENDIX E

ASSOCIATED SENSITIVE SPECIES OF THE COASTAL BEACH-DUNE ECOSYSTEM AND ADJACENT HABITATS

We, the U.S. Fish and Wildlife Service, are committed to applying an ecosystem approach to conservation to allow for efficient and effective conservation of our nation's biological diversity (U.S. Fish and Wildlife Service 1994*a*). In terms of recovery plans, it is our policy to incorporate ecosystem considerations in the following manner:

(1) Develop and implement recovery plans for communities or ecosystems where multiple listed species, candidates and species of concern occur.

(2) Develop and implement recovery plans for threatened and endangered species in a manner that restores, reconstructs, or rehabilitates the structure, distribution, connectivity, and function upon which those listed species depend. In particular, these recovery plans shall be developed and implemented in a manner that conserves the biotic diversity of the ecosystems upon which the listed species depend.

(3) Expand the scope of recovery plans to address ecosystem conservation by enlisting local jurisdictions, private organizations, and affected individuals in recovery plan development and implementation.

(4) Develop and implement agreements among multiple agencies that allow for sharing of resources and decision making on recovery actions for wide-ranging species (U.S. Fish and Wildlife Service 1994*a*).

Improved habitat conditions for co-occurring species within the coastal beach-dune ecosystem will undoubtedly occur through attainment of western snowy plover recovery objectives. Many listed, proposed, or candidate fish and wildlife species, and federally recognized species of concern occur in habitats within or adjacent to this ecosystem (Table E-1). Some of these species are included in existing or developing recovery plans, and actions to recover the western snowy plover will also contribute to implementation of those recovery plans (*e.g.*, beach layia, Howell's spineflower, Menzies' wallflower, Monterey gilia, Monterey spineflower, Sonoma spineflower, Tidestrom's lupine, Myrtle's silverspot butterfly, Smith's blue butterfly, California least tern, American bald eagle, American peregrine falcon, California brown pelican, Pacific pocket mouse, tidewater goby, coho salmon, and steelhead trout) (Table E-1). Other sensitive species which are not covered by regulatory processes or existing recovery planning efforts should also benefit from implementation of the western snowy plover recovery plan through improvements in coastal beach, dune, and adjacent habitats where their ranges coincide with the western snowy plover (*i.e.*, beach invertebrates and other rare plants included in Table E-1). Marine mammals, which use the coastal beach-dune ecosystem and are protected under the Marine Mammal Protection Act of 1972 (16 USC 1361 *et seq.*), also would benefit from conservation of western snowy plover habitat. However, marine mammals are addressed primarily because of the potential need to manage these species when they usurp western snowy plover nesting habitat (*e.g.*, pinnipeds) or become stranded in western snowy plover breeding areas (*e.g.*, cetaceans). This appendix contains brief species accounts for the sensitive species listed in Table E-1.

Federal Status

Endangered: Any species which is in danger of extinction throughout all or a significant portion of it's range.

Threatened: Any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.

Species of concern: Federally-recognized sensitive species for which further biological research and field study are needed to resolve its conservation status.

Table E-1. Associated sensitive fish, wildlife, and plants.

Taxon (Scientific Name)	Federal Status/State Status
Federally-listed plants	
Beach layia (*Layia carnosa*)	Endangered/Endangered (CA)
Coastal dunes milk vetch (*Astragalus tener* var. *titi*)	Endangered/Endangered (CA)
Hoffman's slender-flowered gilia (*Gilia tenuiflora* var. *hoffmanii*)	Endangered
Howell's spineflower (*Chorizanthe howellii*)	Endangered/Threatened (CA)
Island malacothrix (*Malacothrix squalida*)	Endangered
Menzies' wallflower (*Erysimum menziesii*)	Endangered/Endangered (CA)
Monterey gilia (*Gilia tenuiflora* ssp. *arenaria*)	Endangered/Threatened (CA)
Monterey spineflower (*Chorizanthe pungens* var. *pungens*)	Threatened
Soft-leaved Indian paintbrush (*Castilleja mollis*)	Endangered
Sonoma spineflower (*Chorizanthe valida*)	Endangered/Endangered (CA)
Tidestrom's lupine (*Lupinus tidestromii*)	Endangered/Endangered (CA)
Federally-listed animals	
El Segundo blue butterfly (*Euphilotes battoides allyni*)	Endangered

Taxon (Scientific Name)	Federal Status/State Status
Morro shoulderband snail (*Helminthoglypta walkeriana*)	Endangered
Myrtle's silverspot butterfly (*Speyeria zerene myrtleae*)	Endangered
Smith's blue butterfly (*Euphilotes enoptes smithi*)	Endangered
California brown pelican (*Pelecanus occidentalis californicus*)	Endangered/Endangered (CA)
California least tern (*Sterna antillarum browni*)	Endangered/Endangered (CA)
Pacific pocket mouse *(Perognathus longimembris pacificus*)	Endangered
Tidewater goby (*Eucyclogobius newberryi*)	Endangered
Coho salmon (*Oncorhynchus kisutch*)	Varies by geographic area
Steelhead trout (*Oncorhynchus mykiss*)	Varies by geographic area
Federally-proposed plants	
La Graciosa thistle (*Cirsium loncholepis*)	Proposed Endangered/Threatened (CA)
Nipomo mesa lupine (*Lupinus nipomensis*)	Proposed Endangered/Endangered (CA)
Federal Candidate Animals	
Streaked horned lark (*Eremophila alpestris strigata*)	Candidate
Animals delisted or proposed for delisting	

Taxon (Scientific Name)	Federal Status/State Status
American bald eagle (*Haliaeetus leucocephalus*)	Endangered (1978); Threatened (1995); Delisted (2007)/ Threatened (WA);Endangered (CA)
American peregrine falcon (*Falco peregrinis anatum*)	Delisted (1999)/Endangered (WA, CA)
Plant species of concern	
Northcoast phacelia (*Phacelia insularis* var. *continentis*)	Species of concern
Beach spectacle pod (*Dithyrea maritima*)	Species of concern/Threatened (CA)
Pink sand-verbena (*Abronia umbellata* ssp. *breviflora*)	Species of concern/Endangered (OR)
San Francisco spineflower (*Chorizanthe cuspidata* var. *cuspidata*)	Species of concern
Surf thistle (*Cirsium rhothophilum*)	Species of concern/Threatened (CA)
Animal species of concern	
Barrier beach tiger beetle (*Cicindela latesignata latesignata*)	Species of concern
Belkin’s dune fly (*Brennania belkini*)	Species of concern
Gabb’s tiger beetle (*Cicindela gabbi*)	Species of concern
Globose dune beetle (*Coelus globosus*)	Species of concern
Little bear scarab beetle (*Lichnanthe ursina*)	Species of concern

Taxon (Scientific Name)	Federal Status/State Status
Mimic tryonia snail (*Tyronia imitator*)	Species of concern
Morro blue butterfly (*Icaricia icarioides morroensis*)	Species of concern
Mudflat tiger beetle (*Cicindela trifasciata sigmoidea*)	Species of concern
Oblivious tiger beetle (*Cicindela latesignata obliviosa*)	Species of concern
Oso Flaco flightless moth (*Areniscythris brachypteris*)	Species of concern
Oso Flaco patch butterfly (*Chlosyne leanira*)	Species of concern
Oso Flaco robber fly (*Ablautus schlingeri*)	Species of concern
Point Conception Jerusalum cricket (*Ammopelmatus muwu*)	Species of concern
Point Reyes blue butterfly (*Icaricia icarioides* ssp.)	Species of concern
Rude's longhorn beetle (*Necydalis rudei*)	Species of concern
Salt marsh skipper (*Panoquina erans*)	Species of concern
Sandy beach tiger beetle (*Cicindela hirticollis gravida*)	Species of concern
White sand bear scarab (*Lichnanthe albopilosa*)	Species of concern

Marine Mammals (all protected under the Marine Mammal Protection Act and some protected under the Endangered Species Act)

Pinnipeds:
California sea lion (*Zalophus californianus*)
Guadalupe fur seal (*Arctocephalus townsendi*)
Harbor seal (*Phoca vitulina richardsi*)
Northern elephant seal (*Mirounga angustirostris*)
Northern fur seal (*Callorhinus ursinus*)
Steller sea lion (*Eumetopias jubatus*)

Cetaceans:
Gray whale (*Eschrichtius robustus*)
Sperm whale (*Physeter macrocephalus*)
Blue whale (*Balaenoptera musculus*)
Humpback whale (*Megaptera novaeangliae*)
Minke whale (*Balaenoptera acutorostrata*)
Killer whale (*Orcinus oraca*)

Federally-listed plants

Beach layia *(Layia carnosa)* is a small succulent plant in the sunflower family (Asteraceae). Until recent surveys, 17 California occurrences of *Layia carnosa* located in 8 dune systems from Santa Barbara County to Humboldt County had been found. Currently, 21 populations are known. Although the species range is relatively unchanged, at least five historical occurrences are thought to be extirpated. The species is restricted to coastal sand dunes. In northern California, it occurs in the northern dune scrub community; in Monterey County, the species occurs in the central dune scrub community. It generally occurs behind the northern foredune community, occupying sparsely vegetated open areas on semi-stabilized dunes. The species also will occur in open areas, such as along trails and roads. The largest populations are in Humboldt County. Three of the historic Humboldt County occurrences were on the Samoa Peninsula in the Humboldt dune system, and two have been extirpated (U.S. Fish and Wildlife Service 1998*a*). In 1995, a small population was rediscovered on Vandenberg Air Force Base (D. Keil pers. comm. 1995 *in* U.S. Fish and Wildlife Service 1998*a*). The threats to *Layia carnosa* include displacement by invasive, non-native vegetation, recreational uses such as off-road vehicles and pedestrians, and development.

<u>Beach spectacle pod</u> *(Dithyrea maritima)* is a low-growing dune perennial in the mustard family (Brassicaceae or Cruciferae). *Dithyrea maritima* grows in the active foredune habitat of coastal sand dune systems, mainly at the base of the small transverse dunes. The range of the species has been greatly reduced from its historic distribution (Morey 1989). Historically, *Dithyrea maritima* was found just north of the Palos Verdes Peninsula along the coastal dune strip including Hermosa and Redondo Beaches, Los Angeles County. The current mainland distribution is patchy, occurring from Surf, in western Santa Barbara County, north to the Morro Bay sand spit, San Luis Obispo County. Approximately 14 populations are known to still exist. A small Channel Islands population is known from San Miguel Island and scattered locations of the plant occur on the west end of San Nicolas Island. A single location in Baja California, Mexico, just south of San Quintin was documented for this species in 1886. The Los Angeles populations have been extirpated since the early 1930's, and the species has not been seen in Mexico for over 100 years (Rollins 1979). The largest known extant population is on Vandenberg Air Force Base in Santa Barbara County. It occurs intermittently along the coast from Shuman Creek to Purisima Point. *Dithyrea maritima* is extremely vulnerable to physical damage and habitat deterioration caused by foot traffic and off-road vehicle activities. Foot traffic is a continuing threat at Surf Beach on Vandenberg Air Force Base, and occasional errant off-road vehicles from the Nipomo Dunes State Vehicular Recreation Area continue to degrade habitat of the species as does the continued operation of oil fields. Within the Nipomo Dunes State Vehicular Recreation Area all but one small population of *Dithyrea maritima* has been eliminated by off-road vehicle activity. This remaining population is in an unrestricted area subjected to off-road vehicle use and is consequently threatened by habitat degradation (Morey 1989).

<u>Coastal dunes milk vetch</u> *(Astragalus tener* var. *titi)* is a diminutive annual herb of the pea family (Fabaceae). Colonies of the milk-vetch occur on a relatively flat coastal terrace within 30 meters (100 feet) of the ocean beach and 8 meters (25 feet) above sea level. Two historical locations from Los Angeles County (Hyde

Park in Inglewood and Santa Monica) and two from San Diego County (Silver Strand and Soledad) were annotated by Barneby as *Astragalus tener* var. *titi* (Barneby 1950). The only known extant population of this species occurs along 17-Mile Drive on the western edge of the Monterey Peninsula on land owned by the Pebble Beach Company and the Monterey Peninsula Country Club. It is unlikely that suitable habitat remains at the Los Angeles locations, since they have been heavily urbanized. In San Diego County, the Silver Strand area is owned by the Department of Defense (Miramar Naval Weapons Center), and a portion has been subjected to amphibious vehicle training exercises. Another portion of Silver Strand has been leased by the Navy to the California Department of Parks and Recreation for development of a campground and recreational facilities. Numerous unsuccessful searches for the plant have been made in these locations since 1980 (Ferreira 1995, California Natural Diversity Data Base 1997). This species is currently threatened with alteration of habitat from trampling associated with recreational activities, such as hiking, picnicking, ocean viewing, wildlife photography, equestrian use, and golfing. Due to the fragmented nature of the plants habitat and the human uses that surround it, the species is vulnerable to extinction from random events. The species is also threatened by competition from two non-native plants, fig-marigold (*Carpobrotus edulis*) and cut-leaf plantain (*Plantago coronopus*) (U.S. Fish and Wildlife Service 1998*b*).

<u>Hoffman's slender-flowered gilia</u> *(Gilia tenuiflora* ssp. *hoffmannii)* is a small, erect annual herb in the phlox (Polemoniaceae) family. It has been collected from three extant populations on Santa Rosa Island (C. Rutherford and T. Thomas *in litt.* 1994). One population occurs at the type locality near East Point on Santa Rosa Island, California, where it occurs as a component of dune scrub vegetation (Thomas 1993). A partially-fenced population was found in 1994 on stabilized dunes at Skunk Point, Santa Rosa Island. The third population corresponds reasonably well with a 1941 specimen of Reid Moran which was collected between Ranch and Carrington Point. Threats to *Gilia tenuiflora* ssp. *hoffmannii* are soil loss, habitat alteration, competition from non-native grasses, cattle grazing, and elk and deer browsing (U.S. Fish and Wildlife Service 1999*a*). It is also vulnerable to random extinction by such events as storms, drought, or fire. The small number of populations and limited number of individuals make the species vulnerable to randomly, naturally occurring events.

Howell's spineflower *(Chorizanthe howellii)* is an herbaceous annual in the buckwheat family (Polygonaceae). It occurs in coastal dunes and adjacent sandy soils of coastal prairies at elevations ranging from sea level to 37 meters (120 feet). In coastal dunes, it is associated with yellow sand verbena (*Abronia latifolia*) and Menzies' wallflower (*Erysimum menziesii*) (California Department of Fish and Game 1985). The species occurs in areas of relatively mild maritime climate, characterized by fog and winter rains. *Chorizanthe howellii* is known, both historically and currently, from only one area north of Fort Bragg in Mendocino County, California. Three populations are known in the dune system south of Ten Mile River in that county. One extended population is located in MacKerricher State Park, with a portion of one occurrence extending beyond State park land to include adjacent private property (California Department of Fish and Game, California Natural Diversity Data Base). The other populations occur on private lands. The majority of this species occurs within MacKerricher State Park, where recreational and maintenance activities were described as the main threats to the continued existence of this species (U.S. Fish and Wildlife Service1998*a*). Recreational activities historically included off-road vehicle use and hiker and equestrian traffic that caused habitat degradation. In addition, dune habitat is being invaded by non-native plants such as iceplant (*Carpobrotus edulis*), European beachgrass (*Ammophila arenaria*), and burclover (*Medicago polymorpha*), which can outcompete and displace native species and can be a serious threat to *Chorizanthe howellii*. Conservation measures undertaken for this species have included the elimination of off-road vehicle use, management of invasive, non-native plants including iceplant, European beachgrass, and burclover, and the revegetation of this species and *Erysimum menziesii* in MacKerricher State Park. The Park has redirected an equestrian trail away from occupied habitat. The Park has also developed the MacKerricher State Park Ten Mile Dunes Restoration Plan that describes measures to protect and enhance the habitat for this species within the Park.

Island malacothrix *(Malacothrix squalida)* is an annual herb in the sunflower family (Asteraceae). It has been collected from two locations along the north shore of Santa Cruz Island. Green collected it near Prisoner's Harbor in 1886, but the species was not seen on the island again until Philbrick and Benedict collected it in 1968 near Potato Harbor (Rutherford and Thomas *in litt.* 1994). Two populations are also known from Middle Anacapa Island. Threats to *Malacothrix squalida* are soil loss, habitat alteration resulting from sheep grazing, feral pig rooting, and seabird nesting. The species is also vulnerable to random extinction by such events as storms, drought, or fire. The small numbers of isolated populations and restricted number of individuals also make the species vulnerable to reduced reproductive vigor (U.S. Fish and Wildlife Service 1999*a*).

<u>Menzies' wallflower</u> (*Erysimum menzies*ii) is a member of the mustard family (Brassicaceae or Cruciferae) it may be a biennial or a short-lived perennial depending on the particular population. It is restricted to coastal dunes in Humboldt, Mendocino, and Monterey Counties. The species is recognized to have three subspecies which are geographically distinct, *E. menziesii* ssp. *menziesii, E. menziesii* ssp. *eurekense*, and *E. menziesii* ssp. *yadonii*. This species occurs on coastal sand dunes in Monterey County from Point Pinos south to Cypress Point and in the Marina Dunes; in Mendocino County from Fort Bragg north to Ten Mile River; and in Humboldt County on the Samoa Peninsula (North Spit) of Humboldt Bay from the southern tip of the North Spit to the Lanphere-Christensen Dunes Preserve, and on the South Spit of Humboldt Bay. In Monterey, the species occurs on coastal strand, close to the high tide line, but protected from wave action. The species has high exposure to strong wind, salt spray, and occasional wave action from storms and high tides. Habitat also occurs in recent bluff scrub, and open, sparsely-vegetated dunes. Subspecies *menziesii* is located in Monterey and Mendocino Counties. It occurs in 10 isolated populations along the Monterey Peninsula from Point Pinos to Cypress Point. The Mendocino County populations range from Ten Mile River south to Fort Bragg. Many of the populations are associated with MacKerricher State Park, except for the Pudding Creek population which is near Fort Bragg. Subpecies *eurekense* occurs in Humboldt County from the coastal dunes of the South Spit to the Lanphere-Christensen Dunes Preserve. Extant Humboldt County populations of the subspecies *eurekense* have six recorded occurrences (California Natural Diversity Data Base 2003) in the Lanphere-Christensen Dunes Preserve, northwest of Mad River Slough, north of Manila (Samoa Peninsula), U.S. Coast Guard Station (Samoa Peninsula), and the South Spit (Humboldt Bay). *Erysimum menziesii* ssp. *yadonii* is restricted to six populations in the vicinity of the Marina Dunes, two at Marina State Beach, and the others at the RMC Lonestar Cement Company property approximately 0.8

kilometer (0.5 mile) south of the Salinas River Lagoon, Monterey County, California. California Natural Diversity Data Base occurrences for subspecies *yadonii* are found in the following habitats: coastal dunes, foredunes, and coastal strand; for subspecies *eurekense*, occurrences are in coastal dunes and foredunes; and for subspecies *menziesii,* occurrences are in coastal strand, coastal dunes, central dune scrub, and northern dune scrub. The species is threatened by invasion by non-native plant species, industrial and residential development, and trampling by recreational users such as pedestrians, equestrians, and hang-gliders. Off-road vehicle recreation, which historically degraded habitat for the species, is again threatening the species (U.S. Fish and Wildlife Service 1998*a*). The displacement of subspecies *menziesii* by the invasive non-native iceplant (*Carpobrotus* sp.) is a threat to Monterey County populations and the populations north of Fort Bragg. In Monterey County, additional threats include browsing by deer (attempts to plant seedlings are successful only with caging of the plants), recreational land uses, coastal erosion, sand mining activities, and the deposition of dredged material from adjacent wetlands (U.S. Fish and Wildlife Service 1998*a*).

<u>Monterey gilia</u> *(Gilia tenuiflora* ssp. *arenaria)* is a member of the phlox family (Polemoniaceae). This species grows in sandy soils of dune scrub and maritime chaparral habitat in the coastal dunes of Monterey County. The species occurs most commonly in sites with limited exposure to strong winds, salt spray, and waves. It grows in open areas and wind-sheltered openings in the low-growing dune scrub vegetation or in areas where the sand has experienced some disturbance, such as along trails and roads. The species is usually tolerant of small amounts of drifting sand. Monterey Bay dune populations occur from Moss Landing to Monterey, along coastal and inland dunes. Monterey Peninsula populations occur in the vicinity of Spanish Bay and Asilomar State Beach. One of the largest populations known of this species was recently discovered at Fort Ord in 1993; preliminary estimates indicate that as much as 60 percent of the species may occur at Fort Ord (U.S. Fish and Wildlife Service 1998*a*). The species is threatened by encroachment of invasive, non-native plant species, sand mining trampling by equestrians and pedestrians, and habitat removal for commercial and residential development. Off-road vehicle activities and golf course development have historically degraded habitat for this species (U.S. Fish and Wildlife Service 1998*a*).

Monterey spineflower *(Chorizanthe pungens* var. *pungens)* is an herbaceous annual in the buckwheat family (Polygonaceae). It occurs in areas of relatively mild maritime climate, characterized by fog and winter rains. This species occurs in coastal dunes, coastal scrub, and further inland on sandy soils derived from ancient stabilized dunes, dating to the Ice Age (Pleistocene); it tends to occur on bare sandy patches where there is little vegetative cover (Zoger and Pavlik 1987). Sites on Fort Ord where this species was found included firebreaks, along roadsides, in sandy openings between shrubs, the central portion of the firing range, and areas where military activities resulted in frequent habitat disturbances. It occurs from the Monterey Peninsula (Monterey County) northward along the coast to southern Santa Cruz County, and inland to the Salinas Valley (Reveal and Hardham 1989; Ertter 1990). Early collections by Gambel in 1842 indicated that this species historically occurred as far south as San Simeon near the northern boundary of San Luis Obispo County; however, in recent times this species has not been found south of the Monterey Peninsula (Reveal and Hardham 1989). The species is currently known from seven populations with the largest number of plants occurring at Fort Ord. In 1992, Jones & Stokes Associates found this species in almost all the undeveloped areas on the western half of Fort Ord (U.S. Army Corps of Engineers 1992). Populations of the species also are found on California Department of Parks and Recreation lands at Manressa, Sunset, Salinas River, and Asilomar State Beaches and Fort Ord Dunes State Park (C. Roye *in litt.* 1996). In 1987, a survey of 6 properties in the Marina Dunes found a total of 43 individuals of *Chorizanthe pungens* var. *pungens* occurring on 5 of the 6 properties surveyed: Marina State Beach, Granite Rock Company, Gullwing, RMC Lonestar Cement Company, and Martin properties (Zoger and Pavlik 1987). Habitat loss, conversion from agricultural use, residential development, activities at military institutions, and invasion by non-native plants were identified as the primary threats to this species. Hikers and equestrians may trample these plants at various locations throughout its range. The conversion of the Fort Ord military base to other uses, including educational and scientific research facilities, may pose threats to this species if new buildings are constructed; however, large portions of this plant's habitat on Fort Ord are to be reserved for open space. Populations of this species at Sunset State Beach are threatened by recreational activities and are subject to trampling. Invasive non-native species which were introduced as part of dune stabilization programs (i.e., European beachgrass (*Ammophila arenaria*) and iceplant (*Carpobrotus edulis*)) are also a threat to these populations. This plant at Sunset State Beach may be enhanced by a restoration program established for the removal of non-native species (Ferreira 1989). Restoration of dunes at the Naval Post Graduate School in Monterey where it occurs also may be beneficial. Personnel from Marina State Beach and Asilomar State Park have implemented an aggressive eradication program for invasive, non-native plants, have conducted dune revegetation, and protected dune habitat from recreational uses (*i.e.*, use of raised wooden walkways). The State has installed interpretive signs that educate park visitors on the sensitivity of dune habitat and endangered plant species. Designating large portions of Fort Ord as open space will provide conservation opportunities for this species (U.S. Fish and Wildlife Service 1998*a*).

<u>Soft-leaved Indian paintbrush</u> *(Castilleja mollis)* is a presumably partially parasitic perennial herb in the figwort family (Scrophulariaceae). Two collections of this species were made by F. H. Elmore from Point Bennett on San Miguel Island in 1938 (Heckard *et al.* 1991); despite recent searches, this plant has not been seen on the island since then (S. Junak pers. comm. 1994). *Castilleja mollis* is known from two areas on Santa Rosa Island, Carrington Point in the northeast corner of the island, and west of Jaw Gulch and Orr's Camp (this location also referred to as Pocket Field) along the north shore of the island. At Carrington Point, the plant is associated with stabilized dune scrub vegetation that is dominated by goldenbush (*Isocoma menziesii* var. *sedoides*), lupine (*Lupinus albifrons*), and Pacific ryegrass (*Leymus pacificus*). Goldenbush is likely a host plant to the soft-leaved Indian paintbrush, providing water and nutrients (U.S. Fish and Wildlife Service 1998*a*). At the Pocket Field location, the paintbrush is associated with non-native iceplant (*Carpobrotus* sp. and *Mesembryanthemum* sp.), native milkvetch (*Astragalus miguelensis*), and alien grasses. Threats to *Castilleja mollis* are soil loss, habitat alteration, cattle grazing, deer and elk browsing, deer bedding, and competition with alien plant taxa (S. Chaney pers. comm. 1994). Because of the small numbers of isolated populations and individuals, this species is also vulnerable to random extinction by such events as storms, drought, or fire. Small numbers of populations and individuals also make the species vulnerable to random naturally occurring events (U.S. Fish and Wildlife Service 1998*a*).

Sonoma spineflower *(Chorizanthe valida)* is an herbaceous annual in the buckwheat family (Polygonaceae). The species is found in areas of relatively mild maritime climate, characterized by fog and winter rains. It occurs exclusively in the sandy soil of a coastal prairie near Abbott's Lagoon, at an elevation of approximately 12 meters (40 feet). This site is adjacent to the dune system which stretches about 19 kilometers (12 miles) from Tomales Point to Reyes (Cooper 1967). The only known extant population of *Chorizanthe valida* (California Natural Diversity Data Base) is located in the Lunny pasture adjacent to Abbott's Lagoon in Point Reyes National Seashore (Davis and Sherman 1990). Historically, the plant was more widespread on the peninsula. The population is located in a pasture that has been grazed for over a century. Changes in grazing or trampling could alter the vegetation structure that has allowed the plant to persist. Increased grazing or trampling may increase seedling mortality, and reduced grazing and trampling may allow surrounding vegetation to outcompete *Chorizanthe valida* (U.S. Fish and Wildlife Service 1998*a*).

Tidestrom's lupine *(Lupinus tidestromii*) is a low, creeping perennial member of the pea family (Fabaceae). This species grows in active dune ecosystems and on partially stabilized coastal dunes. With its prostrate habit, it can survive partial burial, providing local dune stabilization. It occurs from sea level to 7.6 meters (25 feet). Several of the occurrences on the Monterey Peninsula are on remnant dunes in the yards of private residences. It occurs in the mild maritime climate of the central California coast, growing in coastal scrub communities in association with Menzies' wallflower (*Erysimum menziesii*) and sand gilia (*Gilia tenuiflora* ssp. *arenaria*). This species occurs from the Monterey Peninsula in Monterey County northward to the south bank of the Russian River near its mouth in Sonoma County. Clark and Fellers (1986) identified three populations of this species in Point Reyes National Seashore, extending from Abbott's Lagoon to Point Reyes Station. The major threats to *Lupinus tidestromii* include loss of habitat due to development, trampling by hikers and equestrians, and livestock grazing. Two populations on the Monterey Peninsula were eliminated by construction of a golf course; mitigation plantings were implemented. Other populations on privately-owned sites in Monterey are potentially threatened by residential and recreational development. At the time of listing, the populations in Asilomar State Park and Point Reyes National Seashore were subject to trampling by hikers, a problem that is now corrected by controlled pedestrian routes. Additionally, cattle grazing on the dune system near Dillon Beach presents a potential threat of trampling to this species. Many sites are also threatened by the invasion of non-native species, such as iceplant *(Carpobrotus* sp.) and European beachgrass (*Ammophila arenaria*) (U.S. Fish and Wildlife Service 1998*c*). Asilomar State Beach has developed a management plan for dune enhancement. This plan proposes restoration of native dune vegetation, control of invasive, non-native species, monitoring and mitigation of human-use impacts, and changing visitor use patterns. Boardwalks have been constructed to direct visitors away from sensitive dune areas and allow beach access while minimizing trampling of dune vegetation (C. Roye *in litt.* 1996).

Federally-listed animals

<u>El Segundo blue butterfly</u>. The El Segundo blue butterfly (*Euphilotes battoides allyni*) is a member of the Order Lepidoptera and Family Lycaenidae. It is endemic to the formerly expansive El Segundo sand dunes near Los Angeles, California. The El Segundo blue butterfly is currently found at only two sites, on about 32 hectares (80 acres) at the west end of the Los Angeles Airport runways, and on an approximately 0.8-hectare (2-acre) lot at the Chevron oil refinery in El Segundo. Adult butterflies can be found from mid-July to early September at both sites. The emergence of adult butterflies occurs with the peak flowering period of its primary food plant, the seacliff buckwheat (*Eriogonum parvifolium* Sm. in Rees (Polygonaceae)). The coastal buckwheat *(Eriogonum cinereum)* is a secondary food plant at the Los Angeles Airport. Both buckwheats are used as larval and adult food plants. Historically, the coastal dunes inhabited by this butterfly were altered by urbanization, industrialization, highway construction, sand mining, and planting of non-native ground covers, especially iceplant. Invasion of non-native plants and insufficient suitable habitat are the primary limiting factors affecting its survival (U.S. Fish and Wildlife Service 1985).

<u>Morro shoulderband snail</u>. The Morro shoulderband snail (*Helminthoglypta walkeriana*), also commonly known as the banded dune snail, belongs to the Class Gastropoda and Family Helminthoglyptidae. It occurs in coastal dune and sage scrub communities. Throughout most of its range, the dominant shrub associated

with the snail's habitat is mock heather (*Ericameria ericoides*). This species is found only in western San Luis Obispo County. At the time of listing, the Morro shoulderband snail was known to be distributed near Morro Bay. Its currently known range now includes areas south of Morro Bay, west of Los Osos Creek, and north of Hazard Canyon. This species has also been reported near San Luis Obispo City and south of Cayucos (Roth 1985). The survival of the Morro shoulderband snail is threatened by the destruction of its habitat (due to increasing development) and degradation of its habitat due to invasion of non-native plant species (*i.e.*, veldt grass), structural senescense of dune vegetation, and unauthorized recreational use (*i.e.*, off-road vehicle activity).

<u>Myrtle's silverspot butterfly</u>. The Myrtle's silverspot butterfly (*Speyeria zerene myrtleae*) is a member of the Order Lepidoptera and Family Nymphalidae. The current distribution of the butterfly is Sonoma and Marin Counties (Launer *et al.* 1992). This butterfly inhabits coastal dunes, coastal prairie, and coastal scrub at elevations ranging from sea level to 300 meters (1,000 feet) (Launer *et al.* 1992). Populations of the Myrtle's silverspot butterfly are seriously threatened by several factors. Urban development has extirpated and is currently threatening populations of Myrtle's silverspot. The spread of non-native iceplant, grasses, and forbs is a competitive threat to the several plant species which either provide nectar sources for the adults or a food source for the larvae. Two populations are currently protected at Point Reyes National Seashore; however, there is no management plan for the conservation of these two populations (U.S. Fish and Wildlife Service 1998*a*).

<u>Smith's blue butterfly</u>. The Smith's blue butterfly (*Euphilotes enoptes smithi*) is a member of the Order Lepidoptera and Family Lycaenidae. It occupies coastal sand dunes, inland sand dunes, serpentine grasslands, and coastal cliffside chaparral communities. The Smith's blue butterfly is currently found in San Mateo, Santa Cruz, and Monterey Counties (Arnold 1991; U.S. Fish and Wildlife Service 1984). At the time of listing, the Smith's blue butterfly was known primarily from the mouth of the Salinas River to Del Rey Creek in California (U.S. Fish and Wildlife Service 1984). Its current range is from southern Santa Cruz County to the Monterey-San Luis Obispo County line and inland to the Salinas Valley (Arnold 1991). It typically occurs in foredunes and rear sand dunes in the Monterey Bay

region (U.S. Fish and Wildlife Service 1998*a*). South of the Carmel River, the species also occurs in grassland and coastal scrub and the interface between these two habitat types (U.S. Fish and Wildlife Service 1998*a*). The Smith's blue butterfly's distribution is limited to the occurrence of its host plants (buckwheat). Non-native plants (*e.g.*, iceplants, Kikuyu grass, genista) are known to invade the habitats where the host plants occur (Norman 1994). The Smith's blue butterfly's habitat is also threatened by heavy foot and off-road vehicle traffic. Landslides, sand mining, and urbanization are also reasons for the decline and threats to the butterfly's survival.

<u>California brown pelican</u>. The California brown pelican (*Pelecanus occidentalis californicus*) is a conspicuous bird along the coasts of California and Baja California, Mexico. It typically has a bright red gular pouch (basal portion) during the breeding season. The breeding distribution of the California brown pelican ranges from the Channel Islands of southern California southward to Islas Isabela and Tres Marias off Nayarit, Mexico. Nesting habitat includes islands with steep, rocky slopes. Between breeding seasons, pelicans migrate along the Pacific Coast, ranging as far north as Vancouver Island. Brown pelicans inhabit Oregon part of the year. They roost on the North Spit of Coos Bay, Oregon, and on estuaries along the Oregon Coast (E.Y. Zielinski and R.W. Williams *in litt.* 1999). Brown pelicans prefer salt water habitats year-round, where an adequate and consistent food supply is available. Brown pelicans are colonial nesters and require nesting grounds that are free from both mammalian predators and human disturbance. They also depend on estuarine habitat, including roost sites. This habitat has been extremely reduced along the California coast (U.S. Fish and Wildlife Service 1983).

<u>California least tern</u>. The California least tern (*Sterna antillarum browni*) is the smallest tern in the United States. The birds are about 23 centimeters (9 inches) in length and have a wingspan of about 51 centimeters (20 inches). The least tern historically nested along sandy beaches close to estuaries and embayments along the coast of California from San Francisco Bay to Baja California, Mexico. Human encroachment along California beaches for recreation, residential, and industrial development has severely diminished the availability of suitable nesting habitat. The majority of the least tern population currently is concentrated in

southern California within Los Angeles, Orange, and San Diego Counties. The loss of nesting habitat range-wide in conjunction with increased loss of foraging areas, human disturbance, and predation at remaining breeding colonies resulted in a Federal designation of endangered status in 1970 (U.S. Fish and Wildlife Service 1970).

Pacific pocket mouse. The Pacific pocket mouse *(Perognathus longimembris pacificus*) is a small rodent species that is endemic to the immediate coast of southern California from Marina del Rey and El Segundo in Los Angeles County, south to the vicinity of the border of Mexico in San Diego County (Hall 1981, Williams 1986, Erickson 1993). The species inhabits, or was known to inhabit, coastal strand habitats, coastal dunes, river alluvium, and coastal sage scrub growing on marine terraces (Grinnell 1933, Meserve 1972, Erickson 1993). Available data indicate that the historical distribution of the Pacific pocket mouse was much more extensive prior to the large-scale development of the coastal lowlands of southern California. Between 1894 and 1972, the Pacific pocket mouse was recorded from 8 general locales and 29 specific localities from Los Angeles County south to the border of Mexico in San Diego County. Approximately 80 percent of all Pacific pocket mouse records were from 1931 or 1932 (Erickson 1993). Prior to the rediscovery of the Pacific pocket mouse on the Dana Point headlands in Orange County, California (Brylski 1993), the species had not been observed in over 20 years. In 1995, Pacific pocket mice subsequently were discovered near two historically occupied locales on Camp Pendleton Marine Corps Base in San Diego County, California. Current occupied habitat for the Pacific pocket mouse is estimated to be less than 400 hectares (988 acres). None of the eight historic locales are protected and all have been damaged by or are threatened by habitat destruction or fragmentation, fire, or other disturbances.

Tidewater goby. The tidewater goby (*Eucyclogobius newberryi*) is a small fish characterized by large pectoral fins and a ventral sucker-like disk formed by the complete fusion of the pelvic fins. Gobies are mainly tropical and tend to be bottom dwelling, shallow bay and marine intertidal animals. The tidewater goby ranges from Agua Hedionda Creek, Carlsbad, San Diego County, north to Lake Earl, Del Norte County (Irwin and Soltz 1984). They are common in San Luis Obispo County streams and uncommon from San Francisco Bay to Humboldt Bay

(Moyle 1976). Threats include coastal development, dredging of coastal waterways, coastal road construction, and upstream diversions (U.S. Fish and Wildlife Service 1994*b*).

<u>Coho salmon</u>. The general biology of coho salmon (*Oncorhynchus kisutch*) is described in detail in McMahon (1983), Hassler (1987), and Sandercock (1991). The coho salmon is an anadromous species; coho salmon generally return to their natal streams to spawn after spending 2 years in the ocean. The spawning migrations begin after heavy late-fall or winter rains breach the sandbars at the mouth of coastal streams, allowing the fish to move into them (Moyle *et al.* 1989). Spawning occurs in small to medium-sized gravel at well-aerated sites, typically near the head of a riffle (Moyle 1976). These streams have summer temperatures seldom exceeding 21 degrees Centigrade (70 degrees Fahrenheit). Emergent fry utilize shallow near-shore areas, whereas optimal habitat conditions for juveniles and sub-adults seem to be deep pools created by rootwads and boulders in heavily shaded stream sections. Because of dramatic declines in population numbers, the National Marine Fisheries Service was petitioned to list this species coast wide. As a result, the species is listed as threatened in southern Oregon, northern California, and along the central California coast. It is listed as endangered in the upper Columbia River, Washington, and as threatened in Puget Sound, Washington, and the lower Columbia River (in Washington and Oregon). Causes of coho salmon declines in California and other states include incompatible land-use practices such as logging and urbanization, loss of wild stocks, introduced diseases, over harvesting, and climatic changes.

<u>Steelhead trout</u>. Steelhead trout (*Oncorhynchus mykiss*) are also anadromous fish. Adult steelhead typically spawn in the spring, from February to June (Moyle 1976), in gravel riffles. Optimum temperatures for growth range from 13 to 21 degrees Centigrade (55 to 70 degrees Fahrenheit) (Moyle 1976). Steelhead typically spend 2 to 3 years in freshwater (Moyle 1976). Like coho fry, steelhead fry reside in near-shore areas. In the presence of coho juveniles, steelhead juveniles tend to utilize riffles. The National Marine Fisheries Service was petitioned to list this species coastwide. Steelhead trout are listed as threatened along the northern, central, and south-central California Coast, and endangered in southern California and the Central Valley.

Federally-proposed plants

La Graciosa thistle (*Cirsium loncholepis)* is a short-lived, spreading, mound-like or erect and often fleshy, spiny member of the sunflower family (Asteraceae). This plant is endemic to the coastal wetlands of southern San Luis Obispo County and northern Santa Barbara County from the Pismo Dunes lake area and south historically to the mouth of the Santa Ynez River. The historic distribution of the species included areas that have been converted from wetland habitat to agriculture and development. Currently, the species is restricted to marshes and the edges of willow thickets in damp swales in the Guadalupe dune system (Hendrickson 1990). Groundwater pumping, off-road vehicle use, and coastal development are continuing threats to this species (California Department of Fish and Game 1992).

Nipomo mesa lupine *(Lupinus nipomensis)* is an annual member of the pea family (Fabaceae). This plant grows in stabilized, back dune habitat in the southwestern corner of San Luis Obispo County. The plant occurs as 1 extended population in 5 occurrences with fewer than 700 plants. The high quality occurrences are situated in dune swales and contain a higher diversity of native annuals. This plant requires pockets of bare sand, probably indicating a low tolerance for competition (Walters and Walters 1988). Impacts from off-road vehicles continue to degrade habitat, and the species is threatened by further habitat degradation resulting from expansion of introduced weedy plants. This plant is also threatened by coastal development (U.S. Fish and Wildlife Service 1998*c*).

Federal candidate animal

Streaked horned lark. The streaked horned lark (*Eremophila alpestris strigata*) is found in lowland areas of western Washington and Oregon. The streaked horned lark, as is typical of all horned larks, nests on the ground in sparsely vegetated sites in short-grass dominated habitats, such as prairies, fallow agricultural fields, lightly to moderately grazed pastures, seasonal mudflats, airports, and dredged materials islands in the Columbia River (Gabrielson and Jewett 1940, Altman 1999, Rogers 1999*a*). However, they also are found in dune habitats along the coast (Rogers 1999*a*), where their distribution in Washington coincides with western snowy plover nesting habitat. The streaked horned lark is currently a candidate for listing and has been extirpated from much of its range, particularly in Washington. In 2000, 58 streaked horned larks (51 males and 7 females) were detected at the 11 known breeding sites in the south Puget Sound lowlands and the outer coast (MacLaren 2000). The breeding population in Oregon is estimated to include less than 200 pairs (Altman 1999). The species is most common in the central Willamette Valley, particularly in and around Baskett Sough National Wildlife Refuge. Little information is available for the Oregon Coast. The greatest threat to the streaked horned lark is the loss of habitat. Native prairies and grasslands have been virtually eliminated throughout the range of the species as a result of human activity. In coastal areas, the introduction of Eurasian beach grass (*Ammophila arenaria*), currently found in high densities on most of coastal Oregon and Washington, has drastically altered the structure of dunes on the outer coast. The tall, dense, leaf canopy of this plant creates unsuitable habitat for streaked horned larks (Rogers 1999*b*, MacLaren 2000). The vegetation density of this beach grass has increased in the fore and secondary dunes where this species is likely to nest (Wiedemann 1987).

Animals delisted or proposed for delisting

<u>American bald eagle</u>. The bald eagle (*Haliaeetus leucocephalus*) is a large eagle, weighing up to 7 kilograms (15.5 pounds) and measuring 84 to 95 centimeters (33 to 37 inches) in length in the northern race (Stalmaster 1987). Bald eagles are found in coastal areas throughout the year, but are present in greatest numbers around seabird and marine mammal colonies, waterbird concentrations, and estuaries where food abundance is highest and easily available. Marine mammals and seabirds are available primarily as carrion in the beach/dune ecosystem on a temporary or localized basis. Use of this ecosystem by bald eagles is therefore likely to be opportunistic, occur most frequently during the migration and wintering periods, and be greatest where reliable food sources occur nearby. The bald eagle historically ranged throughout North America except extreme northern Alaska and Canada, and central and southern Mexico. The population was estimated at 250,000 to 500,000 eagles. However, populations began to decline significantly in the mid- to late-1800's as eagles were killed, prey numbers were reduced, and nesting habitat was destroyed. In the 1940's, the use of DDT and other organochlorine pesticides became widespread, causing further declines in

numbers. In 1963, only 417 active nests were reported in the lower 48 states (U.S. Fish and Wildlife Service 1995). The number of occupied territories has greatly increased since the banning of DDT and other organochlorines and habitat protection and other recovery measures have been instituted. The bald eagle was delisted (removed from the list of endangered and threatened species) in the lower 48 states on August 8, 2007 (U.S. Fish and Wildlife Service 2007).

American peregrine falcon. The American peregrine falcon is a medium-sized raptor. Three subspecies of the peregrine falcon (*Falco peregrinus*) are recognized in North America (Brown and Amadon 1968). The Peale's falcon (*Falco peregrinus pealei*) is a year-round resident of the northwest Pacific Coast, from northern Washington through British Columbia to the Aleutian Islands. The arctic peregrine falcon (*Falco peregrinus tundrius*) nests in the tundra of Alaska, Canada, and Greenland and is typically a long-distance migrant, wintering as far south as South America. The American peregrine falcon (*Falco peregrinus anatum*) occurs throughout much of the remainder of North America, from the subarctic boreal forest south to Mexico. American peregrine falcons that nest in subarctic areas generally winter in South America, and those that nest in lower latitudes exhibit variation in migration behavior or are nonmigratory (Yates *et al.* 1988). The most common habitat characteristic of this species is the presence of tall cliffs which serve both as nesting and perching sites for roosting and hunting. Also required is a source of nearby water (river, coast, lake, wetland, *etc.*) which supports populations of small- to medium-sized resident or migratory birds upon which the American peregrine falcon preys. Organochlorine pesticides were the primary cause of a rapid and significant decline in the number of American peregrine falcons in many areas of North America between the 1940's and early 1970's. The American peregrine falcon was removed from the list of endangered and threatened wildlife on August 25, 1999 (U.S. Fish and Wildlife Service 1999*b*).

Plant species of concern

Northcoast phacelia *(Phacelia insularis* var. *continentis)* is a delicate, annual plant in the borage family (Boraginaceae). The California Natural Diversity Data Base lists occurrences for variety *continentis* in the following habitats: coastal terrace, coastal bluff, coastal scrub, and some stabilized dunes. Clark and Fellers (1986) found that var. *continentis* is restricted to sandy or rocky soils; at Point Reyes, it is found with annual grasses, annual lupines (*Lupinus* spp.), goldfields (*Lasthenia macrantha*), bedstraw (*Galium* sp.), and thistle (*Cirsium* sp.). They also found it only occurs in Marin and Mendocino Counties, California. There are four localities where the plant has been found at Point Reyes, Marin County, in either 1983 or 1984. Two of the populations were found near the tip of the Point Reyes Peninsula (lighthouse and Chimney Rock areas); the other two populations were found along the north and south side of Abbott's Lagoon. *Phacelia insularis* var. *continentis* has also been found at dunes along the coast at Fort Bragg, Mendocino County, including Gold Beach and along Ten Mile Beach, MacKerricher State Park (S. Smith *in litt.* 1994). Dr. Gregory Lee (*in litt.* 1984) reported his suspicion that construction near the Point Reyes lighthouse in the early 1980's may have adversely impacted this population. Both Mendocino County populations are threatened by invasive weeds, trampling by people and horses, and cattle grazing; the Gold Beach population is also threatened by development (S. Smith *in litt.* 1994).

<u>Pink sand-verbena</u> *(Abronia umbellata* ssp. *breviflora)* is a succulent, prostrate herb in the four o'clock family (Nyctaginaceae). It blooms in delicate pink flowers arranged in umbellate heads. *Abronia umbellata* ssp. *breviflora* is confined to sand dunes and disturbed sandy areas along the Pacific Coast (Meyers 1990). Historically, populations of this species were known from beaches along the Pacific Coast from Vancouver Island, British Columbia, south to northern California (Kaye 1997). The species is now believed to be extinct in British Columbia and Washington, and is known from only a few populations in Oregon and California (Kaye 1997). The pink sand-verbena is frequently found in association with yellow sand verbena (*Abronia latifolia*). In northern California, this plant has been found at Gold Bluffs Beach in Prairie Creek State Park, Redwood National Park, and the southern end of the Samoa Peninsula in Humboldt County (Meyers 1990, Arguello 1994). It also has been found at MacKerricher State Park in Mendocino County and Point Reyes National Seashore in Marin County (Duebendorfer 1987). In Oregon, pink sand verbena has been reestablished as part of western snowy plover habitat restoration projects at the North Spit of Coos Bay, Tenmile and Tahkenitch Creeks, and Siltcoos River

mouths. The U.S. Bureau of Land Management, U.S. Forest Service, and Oregon Department of Agriculture have been experimenting with broadcast seeding and out-planting of greenhouse stock as part of Challenge Cost Share Programs. Reestablishment appears successful. However, it is too early to state whether the populations are self-sustaining (E.Y. Zielinski and R.W. Williams *in litt.* 1999). Threats to *Abronia umbellata* ssp. *breviflora* include habitat encroachment by European beachgrass (*Ammophila arenaria*), destruction by vehicular traffic, human recreational use, and driftwood collection where the *Abronia* is locally abundant (Meyers 1990, Arguello 1994).

San Francisco spineflower *(Chorizanthe cuspidata* var. *cuspidata)* is an annual herb in the buckwheat family (Polygonaceae). Most populations occur on coastal sand dunes; a few occur on weakly consolidated sandstone. Usually found in the rear sand dunes on more stabilized, consolidated soils, this plant occurs along the California coast from San Mateo County to southern Sonoma County. It has been found at Dillon Beach and Point Reyes National Seashore in Marin County (Howell 1970), and southwestern portions of the Presidio, San Francisco (Howell *et al.* 1958).

Surf thistle *(Cirsium rhothophilum)* is a fleshy, gray tomentose, bush-like or low-mounded biennial to short-lived perennial member of the sunflower family (Asteraceae). This species is known from Pismo Beach, Oso Flaco Lake, Nipomo Mesa, and the Guadalupe dunes in San Luis Obispo County, and from the coastal dunes from Point Sal to Point Conception, Santa Barbara County. This plant typically occurs only in the strip of habitat between the wind-blown beach and the stabilized dunes, a zone that for the majority of its distribution is only a few meters (several feet) wide. Vegetative reproduction is uncommon for this plant in habitats dominated by species that have vigorous vegetative reproduction (Zedler 1979, Zedler and Frazier 1991). Vandenberg Air Force Base contains 57 percent of the recorded locations, with 80 percent of the total number of plants of *Cirsium rhothophilum*. Foot access to the Vandenberg dune system via Surf, California, allows some recreational trampling to occur and aggressive competition and displacement by non-native species continue to threaten the species. Nine locations occurring just to the south and north of the base are subject to threats from facility development at Point Conception by the U.S. Coast Guard, cattle grazing and trampling impacts, habitat disturbance from oil production on private lands, and trampling by beach users at a small county park. The populations in the Pismo Dunes State Vehicular Recreation Area continue to be threatened by destruction from recreational vehicle activity.

Animal species of concern

<u>Barrier beach tiger beetle</u>. See Tiger beetles section.

<u>Belkin's dune fly</u>. The Belkin's dune fly (*Brennania belkini*) is a member of the Order Diptera and Family Tabanidae. The adult resembles a bee. The range of this fly includes coastal sand dunes from Playa del Rey, Los Angeles, County, south to Ensenada, Baja California Norte, Mexico (Middlekauff and Lane 1980). The Belkin's dune fly breeds only on coastal sand dunes. Threats to this fly include destruction of coastal dunes by off-road vehicles, urban development, and dune stabilization with non-native plants.

<u>Globose dune beetle</u>. The globose dune beetle (*Coelus globosus*) belongs in the Order Coleoptera and Family Tenebrionidae. It is a dark, flightless beetle, about 6 to 8 millimeters (0.3 inch) long. The globose dune beetle inhabits foredunes and sand hummocks immediately bordering the coast. This flightless beetle spends most of its life buried under the sand, beneath native dune vegetation. The beetle often lives around the bases of beach bursage (*Ambrosia chamissonis*), saltbush (*Atriplex leucophylla*), sea-rocket (*Cakile edentula*), and yellow sand-verbena (*Abronia latifolia*) (Doyen 1985). The globose dune beetle's range was formally from coastal Mendocino County south to Baja California Norte, Mexico. Its current patchy distribution occurs in Mendocino County (Ten Mile River), Sonoma County (Bodega Head), Marin County (Point Reyes), San Mateo County (Butano Creek), Santa Cruz County (north of the mouth of the Pajaro River), Monterey County (Salinas River and Point Sur), Santa Barbara County (Dos Pueblos Canyon), Ventura County (Punta Gorda), Los Angeles County (Venice and Topanga), San Diego County (Tijuana River), and the California Channel Islands (except for San Clemente). The globose dune beetle's habitat is threatened by development, heavy foot or vehicle traffic, and the invasion of non-native beach grass (*Ammophila*) or iceplants (*Carpobrotus* and *Mesembryanthemum*).

Little bear scarab beetle. The little bear scarab beetle (*Lichnanthe ursina*) is a member of the Order Coleoptera and Family Scarabaeidae. This beetle varies in color from light brown to nearly black. Its flight behavior is characterized by males flying close to the sand surface in search of females (Carlson 1980). The little bear scarab beetle occurs on coastal dunes at Point Reyes and likely in Sonoma, Marin, San Francisco, and San Mateo Counties (U.S. Fish and Wildlife Service 1998*a*). This species has been found at Dillon Beach and Point Reyes Beach, Marin County and Ocean Beach, San Francisco County (Carlson 1980).

Mimic tryonia snail. The mimic tryonia snail (*Tyronia imitator*) is also commonly known as the California brackish water snail. It belongs in the Class Gastropoda and Family Hydrobiidae. The shell of the mimic tryonia snail is 3 to 5 millimeters (0.1 to 0.2 inch) long; the fine spiral shell has four to five whorls (Taylor 1978). The mimic tryonia snail inhabits coastal brackish water sloughs, lagoons, and estuaries. Historically, this snail was distributed from Salmon Creek Lagoon, Sonoma County (California) to Ensenada, Baja California (northern Mexico). Its current patchy distribution is now found in the counties of Alameda, Santa Clara, San Mateo, San Luis Obispo, Monterey, Santa Barbara, San Diego, Ventura, Los Angeles, and Orange. The dredging and filling of lagoons and estuaries for flood control and other purposes (*e.g.*, creation of small boat harbors and construction of roads) have destroyed mimic tryonia snail habitats, and closed the lagoons' and estuaries' mouths. This action has created an unsuitable freshwater environment for this snail.

Morro blue butterfly. The Morro blue butterfly (*Icaricia icarioides morroensis*) belongs to the Order Lepidoptera and Family Lycaenidae. This butterfly has a wingspan of 27 millimeters (1 inch) and can be distinguished from other subspecies of *icarioides* by its true blue coloration (Sternitzky 1930). The Morro blue butterfly inhabits sand dune areas. It feeds on *Lupinus chamissonis*, a large blue-flowered beach lupine (Murphy 1988). The Morro blue butterfly is distributed along the coast in San Luis Obispo County and at two localities outside of its Morro dune area, Nipomo Mesa (9.7 kilometers (6 miles) south of Arroyo Grande) and south of Oso Flaco Lake (Murphy 1988). Historically, its range probably extended south to coastal Los Angeles County (Emmel and Emmel 1973) and on the San Antonio Terrace, Vandenberg Air Force Base (Sheridan 1994).

The Morro blue butterfly's population decline is mainly due to the destruction of its habitat. Heavy use of off-road vehicles and urbanization (*e.g.*, housing development and nuclear power plant construction) have destroyed many of the Morro blue butterfly's habitat localities.

Oso Flaco patch butterfly, Oso Flaco robber fly, and Oso Flaco flightless moth. The Oso Flaco patch butterfly (*Chlosyne leanira*) is a member of the Order Lepidoptera and Family Nymphalidae. This butterfly is highly restricted in distribution and little is known of its biology. The Oso Flaco patch butterfly inhabits the Oso Flaco sand dunes of San Luis Obispo County. Adults have been found in late April and early May. This general dune area is threatened by development and off-road vehicle traffic. The Oso Flaco robber fly (*Ablautus schlingeri*) is a member of the Order Diptera and Family Asilidae. Robber flies have the top of the head hollowed out between the eyes. Adults are predaceous and attack a variety of insects, such as wasps, bees, dragonflies, grasshoppers, tiger beetles, and other flies. The larvae feed chiefly on the larvae of other insects. The Oso Flaco flightless moth (*Areniscythris brachypteris*) is a member of the Order Lepidoptera and Family Scythridae. The historic range of the Oso Flaco robber fly and Oso Flaco flightless moth is in California.

Point Conception Jerusalem cricket. The Point Conception Jerusalem cricket (*Ammopelmatus muwu*) is a member of the Order Orthoptera and Family Stenopelmatidae. Habitat for this species is coastal dunes. The historic range of this cricket is in Santa Barbara County, California.

Point Reyes blue butterfly. The Point Reyes blue butterfly (*Icaricia icarioides* ssp.) is a member of the Order Lepidoptera and Family Lycaenidae. The species pupate in the ground and their larval food is *Lupinus chamissonis.* The Point Reyes blue butterfly occurs in foredunes and rear dunes in the Point Reyes area (U.S. Fish and Wildlife Service 1998*a*). This butterfly is believed to be extinct in San Francisco, California (Powell 1981).

Rude's longhorn beetle. The Rude's longhorn beetle (*Necydalis rudei*) is a member of the Order Coleoptera and Family Cerambycidae. This reddish-brown beetle has a robust form. Its pubescense is moderately dense and golden.

Distinguishing features are the barely, longitudinally impressed, and shining pronotal disk, dilated antennal segments, and shining, coarsely punctate elytra[1] (Linsley and Chemsak 1972). The Rude's longhorn beetle inhabits the coastal sand dunes of San Luis Obispo County. The larvae are found on the root crown and lower stem of mock heather (*Ericameria ericoides*) (Linsley and Chemsak 1972). Oviposition occurs on the stem or root crown at ground level, and the larvae feed upon these areas. The larva forms a pupal chamber in the stem.

<u>Salt marsh skipper (a/k/a wandering skipper)</u>. The salt marsh skipper (*Panoquina erans*) is a member of the Order Lepidoptera and the Family Hespariidae. This butterfly is olive brown, with light spots on the upper portion and undersides of the forewings (Donahue 1975). Although restricted to tidelands and estuarine habitats, the salt marsh skipper is widely distributed along the narrow coastal strand from Santa Barbara and Ventura Counties, California, to the southern tip of Baja California, Mexico (Murphy 1988). Historical records include occurrences of this species at Huntington Beach and Doheny Beach in Orange County, California; and Imperial Beach in San Diego County, California (Murphy 1988). At the Tijuana Slough National Wildlife Refuge, San Diego County, California, adult butterflies have been observed at the barrier beach, tidal channel, and tidal creek near tidal flats (Nagano 1982*a*). They have also been found at the Bolsa Chica wetlands (MITECH 1990). The threats to habitat for the salt marsh skipper include development and habitat conversion.

<u>Tiger beetles</u> (including <u>Barrier beach tiger beetle</u>, <u>Gabb's tiger beetle</u>, <u>Mudflat tiger beetle</u>, <u>Oblivious tiger beetle</u>, and <u>Sandy beach tiger beetle</u>). Tiger beetles are members of the Order Coleoptera and Family Cicindelidae. They are highly active terrestrial predators, eating any arthropod they can overpower. They are fast runners and agile fliers, making them hard to approach. They are most active on warm sunny days from spring to fall, on mud or sand, near permanent bodies of water. Tiger beetle larva build vertical burrows in the sand in the same area as adults. They are commonly found along the southern California coastline (Nagano 1982*b*). Threats to tiger beetles include oil spills, urban expansion, and increased recreational beach use, especially off-road vehicles, which can crush the burrows of the larva.

The range of the barrier beach tiger beetle (*Cicendela latesignata latesignata*) is from San Pedro, Los Angeles County, south to the Orange/San Diego County line and from Mission Bay, San Diego County, to the Cape region of Baja California, Mexico (Nagano 1982*b*). Habitats of this subspecies include mudflats and sandy areas in coastal estuaries. It has been found at the Tijuana Estuary National Wildlife Refuge (Nagano 1982*a*), the Border Field State Park in San Diego County (Nagano 1982*b*), and Silver Strand in San Diego County (Rumpp 1979).

The range of the Gabb's tiger beetle (*Cicendela gabbi*) is from San Pedro, California, south along the coastline to the Cape region of Baja California, Mexico. Gabb's tiger beetles inhabit mudflats and salt flats in estuarine areas. This subspecies has been found at the Tijuana Estuary National Wildlife Refuge (Nagano 1982*b*).

The range of the mudflat tiger beetle (*Cicendela trifasciata sigmoidea*) is from Morro Bay, San Luis Obispo County, south to the Cape region of Baja California, Mexico. The habitats of this subspecies are mudflats and dark-colored moist to wet sand in coastal estuarine areas. This subspecies has been found at the Tijuana Estuary National Wildlife Refuge (Nagano 1982*b*).

The oblivious tiger beetle (*Cicendela latesignata obliviosa*) inhabits the seashore from La Jolla north to the Orange County line, including Mission Beach and the mouth of the Santa Margarita River at Camp Pendleton, San Diego County (Nagano 1982*b*); it has also been found at the estuary of Los Penasquitos Creek in San Diego County (Rumpp 1979).

The range of the sandy beach tiger beetle (*Cicendela hirticolis gravida*) is from the San Francisco Bay region south along the coast to Baja California Norte, Mexico. This subspecies is generally found on sand in estuarine areas, and has been found at Point Mugu Naval Air Station, Ventura, California, and the Tijuana Estuary National Wildlife Refuge, San Diego County, California (Nagano 1982*b*).

<u>White sand bear scarab beetle</u>. The white sand bear scarab beetle (*Lichnanthe albopilosa*) is a member of the Order Coleoptera and Family Scarabaeidae. A distinguishing characteristic of the white sand bear scarab beetle is the presence of

white setae[1]along the elytra[2]and dorsum[3] (Carlson 1980). The elytra are light brown and the clypeus is rectangular. Males range in length from 13.5 to 15 millimeters (0.5 to 0.6 inch); whereas the females are slightly larger, ranging in length from 15 to 17.5 millimeters (0.6 to 0.7 inch) (Carlson 1980). The white sand bear scarab beetle is found in the coastal sand dunes of San Luis Obispo and Santa Barbara Counties. The activity period of the adults is probably from mid-morning to mid-afternoon on sunny days. Little is known regarding this beetle's life history. The white sand bear scarab beetle's habitat is threatened by development and off-road vehicle use.

Marine mammals

California sea lion**.** *Zalophus californianus* are an eared seal (Family Otariidae) that display strong sexual dimorphism. Females are smaller than males, measuring 1.8 meters (6 feet) long and weighing around 113 kilograms (250 pounds). Males measure 2.3 meters (7.5 feet) and weigh around 338 kilograms (750 pounds). The fur coloration is brown to tan. California sea lions were hunted commercially in the mid to late 1800's for their hides and for glue stock. By the 1930's, only 7,000 California sea lions were seen in California. They were given special protection by the California Department of Fish and Game and the Marine Mammal Protection Act of 1972. The population recovered rapidly, and Bonnell *et al*. (1983) estimated the world population to be 156,000, 50 percent of which resides in California. Currently, the non-breeding range of California sea lions extends from British Columbia, Canada, south to Tres Marias Islands in Mexico, and the breeding range extends from the Farallon Islands south to the tip of Baja California, Mexico. Archaeological data, though, indicate that California sea lion rookeries were in existence prior to 100 years ago in Oregon. All pinnipeds require birthing on land. The breeding season occurs in May through July but most pups are born in June. Pupping and breeding sites are primarily on sandy beach and rocky flat areas on islands. The largest breeding colony occurs on San Miguel Island, California. After the breeding season, seals migrate away from their breeding grounds but still come onshore to rest at traditional haul out sites. In recent years, immature sea lions are increasingly present on northern California

1. setae- slender, typically rigid or bristly, and springy parts/organs of animals or plants.
2 elytra- thickened, sclerotized anterior wing in beetles and other insects, serving to protect the posterior wings.
3. dorsum-entire dorsal surface of an animal or upper surface of an appendage or part.

haul-out sites such as Ano Nuevo, Point Reyes, and the Farallon Islands during the summer. Sea lions will stampede into the water when resting onshore and disturbed by people on foot, low flying aircraft, or vessel traffic. Chronic human disturbance causes California sea lions to abandon rookeries.

Guadalupe fur seal. *Arctocephalus townsendi* is distinguished from other fur seals by its large head and long, pointed snout. Currently, the species breeds only on Isla de Guadalupe, off Baja California, Mexico (Fleischer 1978). Like the northern fur seal, they have a thick layer of underfur that prevents heat loss and gives buoyancy by trapping air. Males are much larger than females, measuring 1.8 meters (6 feet) in length and weighing about 158 kilograms (350 pounds), compared to the average weight of 45 kilograms (100 pounds) for females (Orr and Helm 1989). Historically, the Guadalupe fur seal ranged from the Farallon Islands south to Revillagigedo Islands off of Mexico; however, the species was nearly exterminated by commercial seal hunters (Fleischer 1978). Currently, their range is from Guadalupe Island, Mexico, north to the California Channel Islands. The estimated population at Guadalupe Island in 1977 was less than 2,000 seals (Bonnell *et al*. 1983). The Guadalupe fur seal is currently rare. Guadalupe fur seals prefer to haul out on solid rocky shores at the base of cliffs; however, they also occur on sandy beaches on San Miguel Island, California. The breeding season extends from late spring to summer and most pups are born in June.

Harbor seal. Harbor seals, also known as the common or spotted seal, are the smallest and the most widespread of all pinnipeds in the eastern Pacific (Bigg 1981). Males are only slightly larger than females and both measure around 1.5 to 1.8 meters (5 to 6 feet) in length and weigh 58.5 to 90 kilograms (130 to 200 pounds). Harbor seals are the only pinniped species found throughout the northern latitudes of the world and are separated into five subspecies based on morphology and geography. The subspecies found in California ranges from the Bering Sea, Alaska, south to Isla San Martin, Baja California, Mexico (Bigg 1981). Rough estimates of the total population of harbor seals of the subspecies, *Phoca vitulina richardsi*, range from 300,000 to 350,000 (Boveng 1988). However, there is not a free exchange of seals throughout this range, and instead, the population is comprised of regional stocks. For example, seals on the southern Channel Islands, and in central and northern California are thought to form separate stocks (Boveng

1988). Sixty percent of seals counted in 1987 occurred north of San Francisco. Point Reyes and the southern Channel Islands were the areas of highest concentration accounting for 15 and 22 percent, respectively. Bonnell *et al*. (1983) considered Point Reyes to be the most important harbor seal hauling ground in central and northern California. Harbor seals characteristically congregate onshore in groups to rest and rear their young at traditional sites that are generally used year round. The abundance onshore at any particular location varies with season, time of day, state of sea, tide, age and sex class, and human disturbance (Brown and Mate 1983, Allen *et al*. 1985, Yochem *et al*. 1987). The substrates upon which they prefer to haul out range from rocky intertidal areas to tidal mudflats and sandy beaches. They are a nearshore seal and are found primarily in protected bays and estuaries. Harbor seals are the least pelagic (ocean-going) of the pinnipeds and haul-out on an almost daily basis (Yochem *et al*. 1987). Daily activity pattern studies indicate that seals spend between 30 to 44 percent of the time per day resting, and 56 to 70 percent either traveling to feeding areas or engaged in foraging activities. Seals, though, are seasonally abundant onshore with more seals hauled out during the breeding (March through June) and molt (June through August) periods than during the winter (Yochem *et al*. 1987). Harbor seals breed throughout their geographic range; however, there is a latitudinal birthing cline. Seals are born progressively later in the season as one moves north from Baja California, Mexico, where pups are born in February, to Alaska, where they are born in June. Harbor seals generally feed alone or in small groups in nearshore waters and at night on primarily small benthic and schooling fish (Bigg 1981).

<u>Northern elephant seal</u>. Northern elephant seals (*Mirounga angustirostris*) are the largest in size of all pinnipeds, weighing up to 2,300 kilograms (5,083 pounds). Adult males physically mature at 9 years with secondary sexual characteristics such as a large proboscis (long flexible snout). Females lack these features and are much smaller in size. The current world population is estimated at around 150,000. The population is expanding rapidly, doubling every 5 years with growth rates averaging around 14 percent per year (LeBoeuf and Laws 1992). Associated with this rapid increase has been the colonization of many areas along the mainland California coast. At Point Reyes Headland, for example, the colony has grown at an average rate of 16 percent per year and is expanding onto adjacent

beaches (Allen *et al.* 1989). Northern elephant seals prefer to congregate onshore in large groups on sandy or cobblestone beaches with a gradual slope. There is a pronounced annual pattern in seal abundance onshore with seals most abundant during the molt (April through July) and breeding season (December through March). The breeding range extends from southern Oregon to Baja California, Mexico. Currently in California, elephant seals breed on the southern Channel Islands (Santa Barbara County), Ano Nuevo Island and mainland (San Mateo County), the Farallon Islands (San Francisco County), Diablo Cove (San Luis Obispo County), Cape San Martin (Monterey County), Point Reyes (Marin County), and Point Saint George (Del Norte County). There is also a new colony in southern Oregon near Cape Blanco. The protracted molt period is due to seals of different age and sex classes molting in sequence; however, peak numbers occur in April and May when immatures and adult females are onshore. When onshore, seals remain hauled out continuously, fasting.

<u>Northern fur seal</u>. Fur seals are members of the family of eared seals (Family Otariidae) and are unique among seals because of a thick layer of underfur that insulates them from their environment. Northern fur seal (*Callorhinus ursinus*) males weigh about four times more than females, measuring up to 2 meters (6.6 feet) and weighing 270 kilograms (600 pounds). Fur seals were hunted for their fur but were given special protection by the North Pacific Fur Seal Convention in 1911. The population recovered until 1974 when it began to decline at an average annual rate of 5 to 8 percent. In 1985, the United States ceased annually harvesting fur seals, and the Marine Mammal Commission has designated northern fur seals a depleted species (Marine Mammal Commission 1988). The current world population of northern fur seals is around 1 million. The breeding population on San Miguel Island is around 11,000. The first documentation of northern fur seals breeding on San Miguel Island was in 1961, and between 1969 and 1978, the rate of increase in pups grew 46 percent annually from a total of 28 to 635 pups. Northern fur seals lead a mostly pelagic life (9.5 months) and come onshore only during the breeding season, from May to August. San Miguel Island is the southernmost breeding location of the northern fur seal. The breeding colonies occur in the north Pacific extending from Robben Island in the Okhotsk Sea, the Pribilof Islands, and Commander Islands of Alaska, south to San Miguel Island, California, and more recently the Farallon Islands of California. Fur seals

have a polygynous reproductive system whereby males hold territories with females. Females give birth to a single pup, and a few days after giving birth, females go on feeding cycles at sea, returning to nurse pups on land. Unattended pups form pods on the beach until females return. The pups remain at rookeries until November and then go to sea (Orr and Helm 1989).

Steller sea lion. Steller sea lions (*Eumetopias jubatus*) are the largest member of the family of eared seals, Otariidae, and are sexually dimorphic in size and appearance. Males weigh 1 metric ton (2,204 pounds) and are about 2.9 meters (9.5 feet) long, whereas females weigh about 0.2722 metric ton (600 pounds). The mane and roar of the adult males gives the impression of an African lion, and accounts for their name (Orr and Helm 1989). Steller sea lions are widely distributed around the Pacific from Hokkaido, Japan, north to the Bering Sea and south to the Southern California Bight. The breeding range of Steller sea lions, however, has been shrinking steadily in California since the 1930's and more sharply throughout the range since the 1960's (King 1983, National Marine Fisheries Service 1992). The number of animals in the central Gulf of Alaska has declined about 52 percent (down 2.7 percent per year) from 140,000 in 1956 to 1960 to 68,000 in 1985. The species was listed as threatened under the Endangered Species Act in 1991. In Oregon, the estimated population is around 3,000 animals concentrated at only a few coastal rocky locations (Bonnell *et al*. 1983). In California up until the 1970's, Steller sea lions bred regularly in small groups on San Miguel Island, the Farallon Islands, and at Point Reyes Headland, but no pups have been born at San Miguel Island or Point Reyes Headland since then. The population of Steller sea lions in California is currently estimated to be around 2,000 animals (Bonnell *et al*. 1983). Steller sea lions are present on haul-out sites year round, but the highest numbers occur between June and August during the breeding season. Steller sea lions give birth and breed on sloping, flat rocky areas and cobblestone or coarse sand beaches that are protected from high waves. A female may nurse a yearling and newborn at the same time but nursing usually lasts from 32 to 44 weeks. Steller sea lions eat primarily fish and squid but also will prey on crustaceans and mammals. They are believed to feed on what is seasonally abundant. They also feed on harbor seals, northern fur seal pups, and sea otters (Antonelis and Fiscus 1980).

Cetaceans. There are several federally-listed species of large whale that occur in the inshore waters of California, Oregon, Washington, and Baja California, Mexico. Blue, sperm, and humpback whales are still listed as endangered under the Endangered Species Act, and good population estimates are lacking. On occasion, whales are known to strand onshore when alive or dead. Examples of stranded cetaceans in California include gray whale (*Eschrichtius robustus*), sperm whale (*Physeter macrocephalus*), blue whale (*Balaenoptera musculus*), and humpback whale (*Megaptera novaeangliae*). Other species occur regularly nearshore, are not listed, but are protected by the Marine Mammal Protection Act. Examples of these species include minke whale (*Balaenoptera acutorostrata*) and killer whale (*Orcinus orca*). Most species have recovered in number substantially during the past two decades. The current population estimate of eastern Pacific gray whales is 24,000, and in 1993 the species was removed from the endangered species list (Marine Mammal Commission 1996).

Humpback and gray whales regularly occur in coastal areas. Both species engage in long migration from northern latitudes south during the winter months, and both forage in the Bering Sea. Much is known of the migratory habits of the gray whale which travels close to shore and calves in lagoons of Baja California, Mexico, and in southern California; however, less is known of where humpback, blue, or sperm whales calf. Given the species' ability to travel great distances, calving could occur anywhere in the Pacific Ocean. Despite their recovery, whales remain vulnerable to the effects of various human activities including coastal development, commercial whale watching, oil and gas development, and salt recovery operations in breeding lagoons of Baja California, Mexico. Development in breeding lagoons is of particular concern because whales have departed from lagoons temporarily when underwater noise levels were excessive. Every year whales are entangled and drowned in fishing nets or hit by ships (Marine Mammal Commission 1996).

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