POPULATION VIABILITY ANALYSIS OF WESTERN GULLS ON THE FARALLON ISLANDS IN RELATION TO POTENTIAL MORTALITY DUE TO PROPOSED HOUSE MOUSE ERADICATION

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REPORT TO THE U.S. FISH AND WILDLIFE SERVICE
FARALLON NATIONAL WILDLIFE REFUGE
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June 2013

Any reference to or use of this report or any portion thereof shall include the following citation:

Nur, N.,R.W. Bradley, D.E. Lee, P.M. Warzybok, and J. Jahncke. 2013. Population Viability Analysis of Western Gulls on the Farallon Islands in Relation to Potential Mortality Due to Proposed House Mouse Eradication. Unpublished report. PRBO Conservation Science, Petaluma, California. PRBO Contribution Number 1868.

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SUMMARY

Proposed invasive house mouse eradication efforts on the Farallon National Wildlife Refuge have identified Western Gulls (*Larus occidentalis*) as a species at risk of non-target mortality. Analyses of potential population level-impacts to the world's largest colony of this species are important for evaluating the feasibility of this proposed project. Using PRBO's long term datasets, we conducted a population viability analysis to model future trends for this population, assessing scenarios with and without eradication mortality, under varying environmental conditions. Scenarios were classified as: "Optimistic" assuming moderately high gull productivity (based on historic data, but with no recurrence of near-failure in reproduction); "Realistic", assuming long-term average productivity with historic frequency of near breeding failure; and "Pessimistic", assuming higher incidence of near-failure in reproduction at the recent frequency.

- Our analysis to assess the population viability of Farallon Western Gulls has been conducted using the best available demographic data for this species, in the population of interest, accounting for strong stochastic variability in parameters over a multi-decadal time scale.

- Future population trends for Farallon Western Gulls, in the absence of any mouse eradication-related mortality, will depend on likelihood of reoccurrence of years with especially low reproductive success, as was observed from 2009 to 2011, which were likely driven by environmental conditions.

- Under "Optimistic" environmental conditions, the model predicts that this Western Gull population would grow by 10.6% after 20 years (median result; quartile range +41% [first quartile] to -14% [third quartile]).

- Under "Realistic" environmental conditions, the model predicts that the population would decline by 8.7% after 20 years (median result; quartile range +18% to -29%).

- Under "Pessimistic" conditions, the model predicts that the population would decline by 27% after 20 years (median result; quartile range -4% to – 45%).

- We determined what level of project-related gull mortality would be ecologically indistinguishable from population trends in the absence of the eradication project (≥ 95% overlap in expected outcomes after 20 years). The threshold was 1700 gulls for the "Realistic" scenario. Under assumptions of our modeling, mortality less than this value would be ecologically indistinguishable after 20 years.

- Under "Realistic" conditions, additional mortality of 1700 gulls would cause the population to demonstrate a cumulative decline of 12.7% after 20 years relative to initial conditions (median result, quartile range +4% to -47%).

- Given assumptions of the model and the demonstrated high variability of parameters, additional mortality less than 1700 gulls would not result in outcomes that, after 20 years, are effectively distinguishable when comparing project mortality and no-project mortality scenarios.

- We conclude that a mortality event of less than 1700 Western Gulls, given an overall population of 32,200 birds, would be unlikely to cause long term irreversible population impacts for this population. However, we acknowledge uncertainty associated with this modeling exercise and that this analysis is independent of assessments of actual gull mortality associated with this proposed project.

INTRODUCTION

The South Farallon Islands, California harbor the world's largest known colony of Western Gulls (*Larus occidentalis*) (Ainley and Boekelheide 1990). Proposed invasive house mouse eradication planning on the Farallon National Wildlife Refuge has identified Western Gulls as a species potentially at risk of non-target mortality, due to direct or indirect consumption of toxic rodenticide. While several mitigation measures are being considered to minimize any mortality, analysis of potential population level impacts on Farallon Western Gulls is needed for evaluating the potential impacts to this species from this proposed project. Our goals were to assess the future trajectory of this population, under varying environmental conditions, and to evaluate the long-term impacts of any potential increased mortality on a twenty-year time scale.

Scope of Study

To meet our goals, we conducted a population viability analysis (PVA) of the Western Gull population on the Farallon Islands to contrast scenarios with additional mortality and scenarios without additional mortality (Nur & Sydeman 1999). This study builds on data collection, compilation, previous demographic modeling, and analysis of demographic parameters of recent data for Farallon Western Gulls presented by Spear & Nur (1994), Nur et al. (1994), Pyle et al. (1997), and Lee (2011). The demographic modeling presented here relies on detailed observations and statistical analysis of the Farallon breeding population, covering the period 1986 to 2011, though the set of parameter values used focused on the latter half of the time series, because that time period is most relevant for this assessment.

An important strength of Population Viability Analysis is that it incorporates stochasticity, the unpredictable variation in demographic parameters that reflects underlying environmental variability (Burgman et al. 1993, Beissinger 2002). This allows for a probabilistic assessment of future populations and evaluation of actions that may reduce or increase risk (Nur & Sydeman 1999, Akçakaya et al. 2004).

Using information on the Western Gull population and how it may be impacted by additional mortality resulting from proposed eradication efforts, we develop projections for the future using a time-frame of 20 years. We evaluate three scenarios that make different assumptions about future Western Gull productivity, likely a proxy for environmental conditions, and their impacts on the population dynamics of the Farallon population. For each scenario we contrast the "no additional mortality" scenario with a scenario of a specified level of mortality, the number of Western Gulls that may die, which we call C). One goal of the study is to determine the value of C such that mortality below this level cannot effectively be distinguished from no mortality 20 years into the future, given assumptions regarding unpredictable variability in environmental and demographic parameters.

The population model presented here assumes that immigration equals emigration. We do not assume a closed population, but rather that there is no "net immigration" (Nur & Sydeman 1999). The three different scenarios that we model all incorporate information on variation in demographic parameters observed during the recent time period (from 1999 to 2008 or 2009, depending on the parameter), and differ with respect to levels of reproductive success. Reproductive success in 2009, 2010, and 2011 was extremely low, less than 0.15 chicks fledged per pair in each of the three years. In the 23 years preceding, reproductive success had never been less than 0.30 chicks fledged per pair and was usually much higher. The cause of this near-failure in 2009-2011 has not been identified, but is likely linked to reduced food availability for this species, as a result of both marine and human influences, during the breeding season, as well as increased intra-specific predation on chicks, itself likely due to reduced food availability.   Thus, the three scenarios evaluated are:

(1)   Optimistic -"Near-failure" does not reoccur in the future. Reproductive success is variable but reflects observations made prior to 2009.

(2)   Realistic - "Near-failure" occurs at the historic frequency of 3 times per 26 years in the period analyzed 1986-2011.

(3)   Pessimistic - "Near-failure" occurs at the "recently observed frequency" of 3 times per 12 years.

It is possible that near-failure may occur at a frequency even higher than that recently observed, but we have not evaluated that possibility in this report. Our "Pessimistic" scenario accounts for unprecedented rates of near breeding failure in our long term Western Gull time series.

For this exercise, we focus on modeling the Farallon population as observed during the recent time period, 1999 to 2011. We use population trend data for this period to derive a Leslie matrix population dynamic model that incorporates stochasticity (Nur & Sydeman 1999). We consider the recent time period to be most relevant for this exercise, as demographic data from the 1980's and early 1990's reflects a different population than exists at present – with the earlier part of the time series showing higher population numbers, lower recapture probability and survival, and higher reproductive success (Figures 1, 2, and 3). Therefore, we maintain that only the more recent demographic data are appropriate as a baseline for predicting future change, as the goal of this study is to assess impacts of a one-time mortality event on the current population in the near future.

Specific objectives addressed by this study are to:

(1) Evaluate future population dynamics based on demographic parameter values and observed population trend, assuming no additional mortality, but considering different scenarios for future environmental conditions. This component of the study quantified the median (expected) behavior of the population as well as the risk of more extreme results (upper quartile and lower quartile of population results) under three different productivity scenarios.

(2) Evaluate future population dynamics as in (1) but include impact of mortality of C gulls at the outset of the simulation. Part of this objective entailed determining the level of mortality (C) such that any mortality below this level, given the variability in parameters, cannot be effectively distinguished from the "no additional mortality" scenarios in this modeling exercise. For the purpose of this exercise, we considered the mortality scenarios to be effectively indistinguishable from each other if the overlap in terms of expected simulation results between

one probability distribution and the other (i.e., with and without mortality) was at least 95%.

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METHODS

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Rationale of Our Approach

The basis of the PVA is a Leslie matrix whose values (i.e. elements) are allowed to fluctuate in relation to variation in the future environment (Nur & Sydeman 1999, Caswell 2001). Here we first briefly describe the demographic parameters being modeled: survival, reproductive success, and probability of breeding. Variation in demographic parameters with respect to age and environmental variability were simultaneously estimated.

i)   **Survival of adults.** Annual survival was determined through capture/recapture analysis of banded gulls from 1986-2011, with respect to age and year-specific variation.

ii)   **Survival of juveniles and subadults.** This refers to annual survival of first-year, second-year, and third-year individuals. By the fourth year of life, evidence indicates that Western Gulls have reached adult levels of survival (Spear & Nur 1994, Pyle et al. 1997). Farallon Western Gulls generally disperse widely during the first one to three years of life (Spear & Nur 1994). Therefore it was not possible to derive accurate estimates of survival from capture/recapture using island-based observational data. Instead, we relied on empirical and statistical studies of age-specific survival of this population (Spear & Nur 1994, Pyle et al. 1997).

iii)   **Reproductive Success** is the number of young reared to fledging per breeding pair per breeding season. We used data from 1986 to 2011from three plots on Southeast Farallon Island, called C,H, and K plots, used to monitor gull

reproductive success. This estimate is conditional on an individual attempting to breed.

iv)   **Probability of Breeding** is a demographic component that reflects the likelihood that an individual that has survived to the beginning of the breeding season, attempts to breed in that season. This parameter potentially varies with the age of the individual. Almost all adults were resighted only when attempting to breed; for that reason, recapture probability is used as an estimate of breeding probability. Note that, in terms of the demographic model, we partitioned probability of breeding into two components: 1) the probability an individual is breeding for the first time and 2) probability that an individual that has previously bred, is currently attempting to breed (see Nur & Sydeman 1999). We used demographic parameter estimates for both probabilities based on the capture/recapture analyses of individuals previously banded as well as observations of age of first-time breeders (see also Pyle et al. 1997).

We incorporate information on annual variation in these four demographic parameters based on observations made during the period 1986 to 2011, as described below, focusing on the most recent period, 1999 to 2011. An important feature of our study is that we calibrated the demographic parameter values used so that the model reproduced the observed population trend data during the recent time period, 1999 to 2011. We assume that all age classes are considered equally at risk to any mortality associated with the proposed project, due to extensive observations of Western Gulls utilizing supplementary food resources during recent field studies (PRBO unpublished data).

Population Trend Data

We used whole colony counts of Western Gulls on the South Farallon Islands at the time of peak incubation for the period 1999 to 2011 and estimated the annual constant rate of change by conducting linear regression on ln-transformed counts (Nur et al. 1999). Results were very similar whether we considered the periods 1999 to

2011, 2000 to 2011, or 2001 to 2011. The observed trend over 1999 to 2011 was a modest growth of 0.74% per year (Figure 1). Therefore, our population model was calibrated to reproduce this growth rate.

Estimation of Demographic Parameters in Relation to Annual Variation in

Survival, Recapture Probability, and Reproductive Success:

Annual survival (symbolized phi) and recapture probability (symbolized p) were estimated over the period 1986 to 2009, for both males and females (Figure 2). It was not possible to estimate year-specific survival beyond 2009 while simultaneously estimating year-specific recapture probability due to limitations of capture-recapture analysis (Cooch et al. 1996). For the initial parameter values in the population model we used mean survival estimates, averaged across the two sexes, based on the most recent 10 years, 1998/1999 to 2008/2009. We also assessed variation in survival and reproductive success across the entire time series (1986 to 2009), but found that the magnitude of annual variation differed between the two time series. The between year standard deviation (SD) was much greater for the 1986-2009 time series, specifically 15% greater for survival and 31 % greater for reproductive success. The between-year SD includes not only variation in underlying demographic parameters among years, but also variation due to sampling error (Gould & Nichols 1998). Recognizing that, we chose to use the smaller of the two between-year estimates of variance (1998/1999 – 2008/2009 time period) for modeling survival and reproductive success. By using the smaller estimate from the recent 10-year period rather than the 24-year period, we were reducing the effect of over-estimation of process variance due to inclusion of sampling error.

The between year SD in adult survival was determined from the year-specific analyses (above). For juvenile and subadult survival, we scaled the between year SD relative to that of adults, given that survival is a binomially distributed random variable and its variance = phi (1-phi) (Mood et al. 1974). That is, the closer survival is to 0.50, the greater is its variance. See Table 1 for SD values used.

Reproductive success (RS; the number of fledged young per breeding pair) was determined each year for our 3 study plots and then averaged across plots and years to determine a mean RS for the period from 1999 to 2008 (Figure 3). The poor reproductive success observed in 2009 to 2011 was modeled separately (see below).

We also quantified the mean annual capture probability (p), which we use as a measure of breeding probability for individuals that have bred before, and the between year variation observed for this parameter. Here, capture probability, refers to the probability that an individual that has bred before breeds in a given year. This assumes that resight probability, probability an individual is resighted and identified given that it is breeding in a given year, is effectively equal to 1. This assumption is justified because breeding birds are highly site tenacious, and once having bred, nearly all surviving individuals return each year to attempt reproduction (Pyle et al. 1991, 1997, Spear et al. 1987), Quantitative estimates of resight probability for breeding birds using program MARK =0.953 (see below). However, we must also consider the probability that an individual that has never bred before, breeds in a given year (Nur & Sydeman 1999). While we were not able to explicitly estimate this latter parameter on a year by year basis over the 24 year time series, we were able to estimate how this probability varies with age, and used that in the modeling.

The demographic model also required estimation of variance in "net fecundity" where net fecundity is defined as the product of RS * p *0.5. We calculated variance in net fecundity based on the product of these individual parameters (Mood et al. 1974), assuming no covariance between RS and p. Thus, our estimate of variance in net fecundity is conservative because inclusion of positive covariance (likely the case: in "good" years both RS and p tend to be high and in "bad" years both tend to be low) would have increased the variance of net fecundity beyond what we were able to calculate. In general, we have attempted to be conservative with respect to variance estimation in order to avoid over-estimating annual variance. Over-estimating annual variance would have resulted in over-estimating the mortality level C that the Western Gull population could tolerate with no detectable long-term effects.

Poor Reproductive Success in Recent Years

      An important feature of the Farallon Western Gull population for the purposes of this modeling is that there was unusually low reproductive success observed in the last three years of the data set (2009 to 2011).   From 1986 to 2008, annual reproductive success ranged from 0.30 to 1.55 fledged young per pair (Figure 3). However in the most recent three years, an average of only 0.06 to 0.13 fledged young were produced per pair. Comparing 2009-2011 to the 10 years previous to that (1999 to 2008), indicated a reduction of 86.2% in mean reproductive success (Figure 3). We believe that this recent "near-failure" could significantly impact population modeling results if it were to continue over the coming years or repeat at some time in the future. Therefore, to model reproductive success we used the mean value over the recent period (1999 to 2008), with between-year variability for the same period (1999 to 2008, excluding 2009-2011). To this we then added the probability of near-failure in reproduction occurring at three different probability levels, one for each scenario.

Age-specific Estimation of Parameters for the Population Matrix

Survival and Fecundity

      Survival by age was estimated using the program MARK (Cooch and White 2012) for individuals banded as chicks and subsequently captured or identified at the South Farallon Islands. Age-specific estimates were then incorporated into the model as appropriate. For adults, age 4 and older, annual survival showed no clear pattern with respect to age, for either males or females (Lee 2011). Therefore the model assumed that all adults had the same survival value (see Table 1). Survival prior to age 4 could not be estimated from these capture-recapture analyses since a very small number of marked subadult gulls have been identified at the colony before breeding. Therefore, to estimate juvenile and subadult survival, we relied on prior analyses based on intensive field observations and statistical analysis by Spear & Nur (1994) and Pyle et al. (1997).

We used mean values for males and females, for all ages, prior to calibration for the initial survival values in the model (Table 1).

The first component of fecundity, age-specific reproductive success (RS), was directly estimated from females of known-age (Lee 2011). We assumed that patterns for males were similar to that of females (Pyle et al. 1997). RS appeared to differ with respect to age. RS increases with age up to age 7, then is fairly level through age 16, and then declines subsequently. On the basis of age by age estimates, we developed a simplified table, categorizing adults into four groups: Young adults (ages 4-5 yrs), transitional adults (age 6), prime-age adults (ages 7 to 16 yrs), and old adults (ages 17 and older) (Table 1).

Capture or resighting probability (p) was used to estimate breeding probability. Age-specific estimates were obtained as part of the survival modeling described above (see Lee 2011). Results indicated that p differed little with age for either sex and remained high throughout life (mean = 0.953 averaging across the two sexes; Lee 2011). Therefore we assumed that once an individual bred it did so with probability of 0.953 (see Table 1).

Age-specific breeding probability includes a second component, the probability an individual breeds for the first time. Capture-recapture analyses provided estimates of the transition from pre-breeder (never having bred before) to breeder (Lee 2011). The model assumed the earliest age of breeding is 4 years, with probability of breeding at age 4 being 19% (mean value for males and females). For 5 year olds, 52% attempt to breed, composed of individuals that bred the year before (as 4 year olds; 19%, see above) and an additional 33% that are breeding for the first time as 5-year olds. Similar calculations apply to age 6, at which age 81% are attempting to breed. By age 7, we assume that individuals reach the full-adult value of 95.3% breeding probability. Age-specific breeding probability is summarized in Table 1.

Post-breeding Census and Density Dependence

The Leslie matrix population model can be implemented with respect to either a pre-breeding or post-breeding census (Caswell 2001, Akçakaya 2005). We chose the latter, primarily because it splits first-year survival into its own row, which can easily be manipulated. As a result, the youngest age class in the simulations refers to individuals who have just fledged (juveniles). There is no evidence that survival or reproductive rates vary in relation to population size or density for this population (Nur and Sydeman 2003, unpublished). Therefore we assumed population parameters to be independent of density (Nur & Sydeman 1999) .

Calibration

Estimates of survival, whether of sub-adults or adults, will underestimate true survival due to permanent emigration of individuals from the study area (Clobert and Lebreton 1991). Such emigration could be from one part of the island to another, or off of the island altogether. The dispersal can be of pre-breeders or of individuals that have already bred. We acknowledge the occurrence of permanent emigration from the study area, but assume (in the absence of other information) that emigration equals immigration. In other words, individuals that leave the study area never to return are replaced by individuals moving into the study area. Given immigration/emigration, it is important to attempt to obtain an unbiased estimate of survival. Failure to do so would result in under-estimating true survival rates.

To allow us to correct for this under-estimation, we calibrated the performance of the population model such that the set of demographic parameter values used produced a population whose median trajectory corresponded to the observed population behavior. From 1999 to 2011, the breeding population demonstrated an average (time-constant) increase of 0.74% per year (Figure 1). To replicate these conditions, we were required to increase survival by a small amount. For first-year survival, we increased the value from 0.582 to 0.610, but note that female survival was estimated by Spear & Nur (1994) at 0.61, so this simply means using the higher of the two sex-specific values, an adjustment needed to allow for some emigration at a relatively low rate. For second-year survival, we increased the value from 0.794 to 0.810, but note that female survival

was estimated by Spear & Nur (1994) at 0.81, so this, too, means simply using the higher of the two sex-specific values to allow for some emigration. For third-year survival, we increased the value from 0.854 to 0.875, but note that female survival was estimated by Spear & Nur (1994) at 0.89, so this reflects a value that is in between the male and female estimates but slightly closer to the female value. For survival in the fourth-year of life, we assumed the same value as adults (Pyle et al. 1997). For all individuals four years old and older, we adjusted calculated survival from 0.885, the mean value for males and females, to 0.890, a very slight adjustment to allow for some emigration. Note that extensive evidence for gulls in general and for this population specifically indicates that adult dispersal is less than that of juveniles and subadults, consistent with a smaller adjustment (Nur & Sydeman 1999). To reiterate, the population model allows for some emigration but assumes that emigration equals immigration. We could not verify this assumption directly, but given the general absence of quantitative estimates of emigration rates for seabirds, this was the approach we took. We did not adjust fecundity values. All the simulations used the survival values adjusted through this calibration process. Survival and fecundity values used in the simulations, once the model was calibrated, are listed in Table 1.

Details of the Stochastic Modeling

The stochastic population modeling was carried out with RAMAS GIS version 5 (Akçakaya 2005). The primary outcome variable of the modeling was the number of individuals in each age class of the population in each year of the simulation, as a function of environmental variability and starting population size. The simulations depict results in which the demographic parameter values for survival and fecundity in a given year in a given simulation are randomly chosen from a distribution whose mean and variance were determined as described above.

In these analyses, we present results for a hypothetical 20-year simulation using the best data appropriate to the present state of the Farallon Western Gull population. Projections beyond 20 years would be excessively uncertain. In the output, years since the beginning of the simulation are shown as year 0, 1, 2, etc., up to 20.

Starting Population Size, Mortality Scenarios, and Simulations

The starting total population size for the simulations is 32,200 individuals of all age classes, in the absence of any additional mortality. This corresponds to a breeding population size of 17,400 individuals, the best recent estimate, from 2011 (Warzybok and Bradley 2011), assuming a stable age structure as determined by the Leslie matrix (Caswell 2001), and assuming average breeding probability. In other words, our results indicate that given the calibrated demographic parameter values used and a breeding population size of 17,400 individuals, there are on average an additional 14,800 sub-adults and non-breeding adults. Note that the 3-year average for 2009-2011 is 17,100 breeding individuals, within 1.6% of the 2011-only value. Therefore, our results are robust to whether we use the most recent year (2011) or the 3-year average.

In scenarios with mortality, the starting population size in year 0 was $32,200 - C$ gulls, where C was determined to be 1700 gulls (see Results and Figure 7). For these scenarios, we assumed that C gulls were removed in proportion to the age distribution of the total population, as there are no data to suggest otherwise. In other words, 5.3% (=1700/32200) of all age classes were removed at the start of the simulation.

This value of C was determined from an assessment of whether the set of outcomes under a "no-additional-mortality" scenario, henceforth "no mortality", is different from the set of outcomes under an "additional mortality" scenario – under the "realistic" scenario productivity values, as described above. We did this by assessing overlap of the modeled distributions for 20 years in the future. We defined two probability distributions to be different if the overlap of one with the other was less than 95%. In other words, if the no-mortality distribution overlapped the additional mortality distribution by 95% or more, we considered the two distributions to be effectively indistinguishable even though statistically they may be distinguishable (e.g., their medians may be statistically different).

To operationalize this definition we first identified the median of the no mortality distribution, call this m_{no}. For example, this value might be 29,400. By definition, 50% of all outcomes were below this value, m_{no} = 29,400. We then analyzed the distribution of outcomes under the same conditions except that C gulls were removed at the outset.

We then identified the value of *C* such that, with *C* gulls removed, the distribution of outcomes had been shifted by 5%, i.e., 55% of outcomes were now below the original median. A displacement in the distribution of 5%, from 50% below m_{no} to 55% of outcomes below m_{no}, is equivalent to an overlap of 95% between two distributions, *assuming the two distributions differ only in their location and they have the same shape and spread*. Note that a displacement of 0% means an overlap of 100%, whereas a displacement of 50% entails an overlap of 50%. In the latter case, 100% of the new distribution lies below m_{no} which in turn corresponds to the value below which 50% of the original distribution lies, i.e., the overlap is 50%: 50% of the original distribution lies above the maximum value observed for the new distribution.

To be clear, the value of *C* used in these modeling exercises was determined as the maximum level of mortality that produced ecologically indistinguishable differences in scenarios, defined here as 95% overlap, in the probability distributions of Western Gull population size 20 years in the future. This included scenarios with and without mortality, under "Realistic" productivity conditions, given our estimates of the total Farallon population. This level of mortality is completely independent of any assessment of acceptable level of mortality by any partners of the proposed mouse eradication project, or predicted mortality based on gull attendance during any proposed eradication action, exposure to toxic rodenticide, or toxicity of rodenticide.

All scenarios depict results based on 10,000 simulations, the maximum for the RAMAS program. For the calculations of overlap of distributions we used 30,000 simulations, combining results of three different runs of 10,000 simulations each. The simulations consider the 3 scenarios of Western Gull productivity: "Optimistic", "Realistic", and "Pessimistic" and 2 levels of mortality (i.e., no mortality or removal of *C* gulls).

RESULTS

Results of the population viability analyses are summarized in Figures 4, 5, and 6, corresponding to "Optimistic", "Realistic", and "Pessimistic" scenarios. For each

scenario we depict results with either no additional mortality (starting population size is 32,200 individuals) or with removal of C gulls at the outset. By simulating results with different mortality levels, we determined that removal of 1700 gulls results in a shifting of the distribution by 5% and thus represents 95% overlap between the no mortality and removal of C gulls options on a 20 year time horizon. This is the case assuming "Realistic" environmental conditions where "near-failure" occurs at historic frequency (p = 0.1153 per year). The overlap in the two distributions under the "Realistic" scenario, with and without additional mortality, is depicted graphically in Figure 7.

Figure 4 depicts results under the "Optimistic," no near-failure scenario. In the absence of additional mortality, the population is expected to grow by 10.6% after 20 years, to 35,600 individuals, using the median result of the modeling. However, there is a 25% probability of a decline of 14% or more after 20 years, and a 25% probability that the total increase will be 40% or more after 20 years. If the population incurs mortality in year 0, after 20 years it is expected to be at median value of 33,500, an increase of 4.0% compared to the pre-mortality population size of 32,200. Under the same set of assumptions, there is a 25% probability that there will be 26,100 individuals or fewer, which represents a population decline of 18.9% or greater compared to the pre-mortality population size. Thus, under this scenario, but not the other two, the population will have likely increased after 20 years, even with additional mortality. However, as in the other scenarios, there is also a substantial probability that the population will be at lower levels than it was prior to the mortality event in year 0.

Figure 5 depicts results under the scenario under "Realistic" conditions, of near-failure occurring at the historic frequency of 3 times per 26 years. In the absence of additional mortality, the population is expected to decline by 8.7% after 20 years, to a median outcome of 29,400 individuals. However there is a 25% probability of a decline of 29% or more after 20 years, and a 25% probability that the total increase will be 32% or more after 20 years. If the population incurs mortality in year 0, after 20 years it is expected (median value) to be at 28,100, a decline of 12.7% compared to the pre-mortality population size of 32,200. Under the same set of assumptions, there is a 25% probability that there will be 21,500 individuals or fewer, which represents a population decline of 33.2% or greater compared to the pre-mortality population size. That said,

there is also a 25% probability that after 20 years, under this scenario, the population will have grown to 36,500 or more individuals, a 13.4% or greater increase compared to the pre-mortality size of 32,200, even though the population sustains a loss of 1700 gulls.

If near-failure occurs at the recent frequency of 3 times per 12 years, under the "Pessimistic" scenario, then we can expect population declines, at least by year 20 (Figure 6). In the absence of additional mortality, the population is expected to decline by 27% after 20 years, to a median outcome of 23,500 individuals. In addition, there is a 25% probability of a decline of 45% or more after 20 years, and a 25% probability that the decrease after 20 years will be 3.7% or less. In fact, under this scenario, and with no additional mortality, the probability of a net population increase of any magnitude after 20 years is 22%.   If the population incurs additional mortality in year 0, after 20 years it is expected to be at a median value of 22,200, a decline of 31.1% compared to the pre-mortality population size of 32,200. Under the same set of assumptions, there is a 25% probability that there will be 17,900 individuals or fewer, which represents a population decline of 44.4% or greater compared to the pre-mortality population size. That said, there is also a 25% probability that after 20 years, under this scenario, the population will have not declined or declined to 29,300 or more individuals; that is, the net decrease compared to the pre-mortality size of 32,200 is a decline of 9.0% or even less of a decline. Under this scenario, a loss of 1700 gulls would likely leave the population at a lower level than at the outset, prior to incurring additional mortality, with only the magnitude of the decline to be established.

CONCLUSIONS

Our modeling effort indicates that, under "no-additional-mortality" scenarios, the Farallon Western Gull population will increase over the next twenty years with "Optimistic" productivity estimates, but will decline with assumption of "Realistic" productivity, and likely decline 3 times faster if incidence of recent near breeding failures were to occur with probability of 25% per year.

In assessing mortality scenarios, we determined the level of mortality that produced 95% overlap in the probability distributions of Western Gull population size 20 years in the future, for scenarios with and without mortality, under "Realistic" productivity conditions, given our estimates of the total Farallon population. This value was 1,700 gulls, assuming a total starting Farallon population of 32,200 birds. These results are independent of any assessment of actual risk to this Western Gull population from rodenticide exposure. We fully support all efforts to mitigate and minimize any mortality associated with any proposed actions.

If the Western Gull population incurs a one-time loss of 1,700 individuals, this could have a detectable effect on the population dynamics compared to no such additional mortality. For example, an expected 8.7% decline after 20 years could become, instead, after the one-time mortality event, a 12.7% net decline under the "Realistic" productivity scenario (Figure 5). Nevertheless, our results indicate that environmental variability due to "normal" variation in demographic parameters as well as the incidence of "near-failures" of reproductive success will have much greater impact than the effects of a mortality event such as loss of 1,700 gulls. Furthermore, the ability of the population to recover from the loss of 1,700 individuals will very much depend on the incidence of reproductive failures in the future, unrelated to the mouse eradication project; such reproductive failures are difficult to forecast.

Our analysis to assess the population viability of Farallon Western Gulls has been conducted using the best available demographic data for this species, in the population of interest, accounting for strong stochastic variability in parameters over a multi decadal time scale. This information should be strongly considered in assessments of population level impacts to this species for any future management actions.

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Table 1. Summary of compiled demographic parameters for Western Gull in relation to Age.   Calibrated Survival and Net Fecundity values (and Standard Deviation) were used in the Population Dynamic Model Matrix. Excluding "Near-Failure" Years of 2009-2011. Data compiled from: Lee (2011), Spear & Nur (1995), Nur et al. (1994) and Pyle et al. (1997)

Age	Repro Success	Breeding Probability	Adult Survival	Calibrated Survival	SD	Net Fecundity	SD
1	0	0	0.582	0.610	0.060	0	0
2	0	0	0.794	0.810	0.049	0	0
3	0	0	0.854	0.875	0.041	0	0
4	0.436	0.191	0.885	0.890	0.039	0.0367	0.014
5	0.436	0.524	0.885	0.890	0.039	0.101	0.039
6	0.649	0.810	0.885	0.890	0.039	0.233	0.089
7	0.882	0.953	0.885	0.890	0.039	0.372	0.143
8	0.882	0.953	0.885	0.890	0.039	0.372	0.143
9	0.882	0.953	0.885	0.890	0.039	0.372	0.143
10	0.882	0.953	0.885	0.890	0.039	0.372	0.143
11	0.882	0.953	0.885	0.890	0.039	0.372	0.143
12	0.882	0.953	0.885	0.890	0.039	0.372	0.143
13	0.882	0.953	0.885	0.890	0.039	0.372	0.143
14	0.882	0.953	0.885	0.890	0.039	0.372	0.143
15	0.882	0.953	0.885	0.890	0.039	0.372	0.143
16	0.882	0.953	0.885	0.890	0.039	0.372	0.143
17	0.718	0.953	0.885	0.890	0.039	0.303	0.116
18	0.718	0.953	0.885	0.890	0.039	0.303	0.116
19	0.718	0.953	0.885	0.890	0.039	0.303	0.116
20	0.718	0.953	0.885	0.890	0.039	0.303	0.116
21	0.535	0.953	0.885	0.890	0.039	0.226	0.087

Figure 1. Western Gull breeding population trends for the South Farallon Islands, 1986-2011.
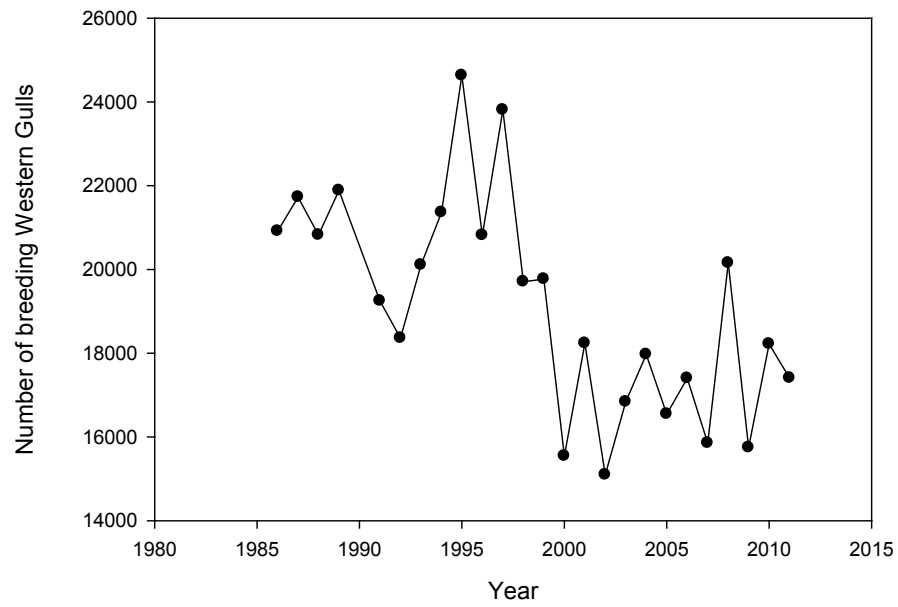
Figure 2. Annual variation in recapture probability and survival (± SE) for Farallon Western Gulls from long term study plots, 1986 to 2009 for both females and males. Missing vales for female recapture probability could not be estimated in program Mark.
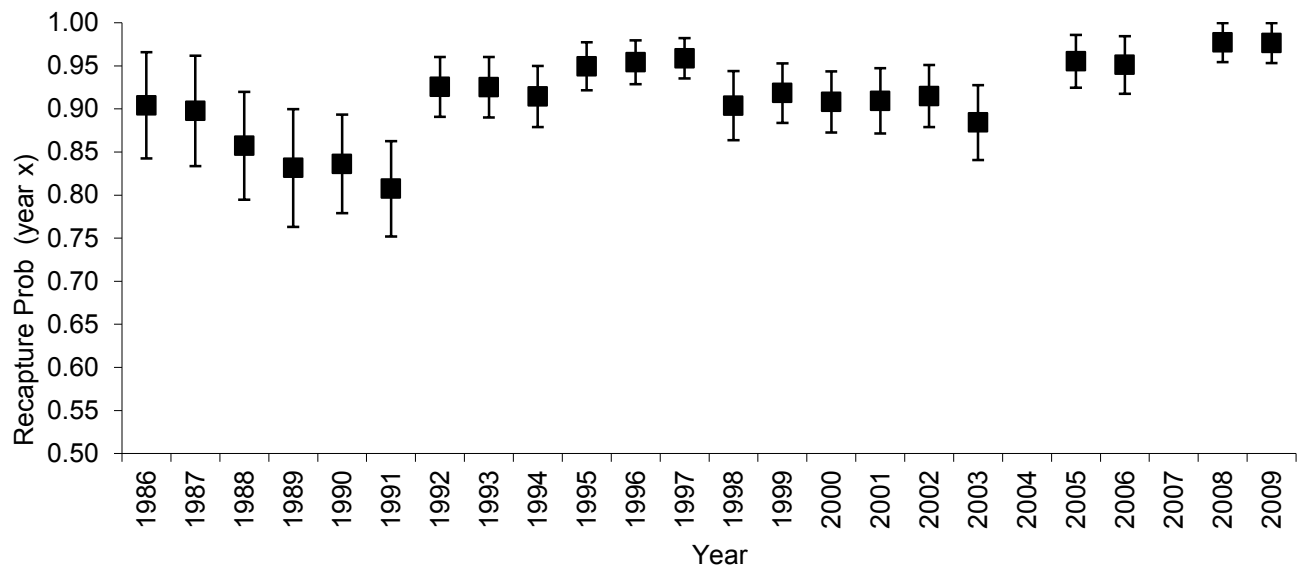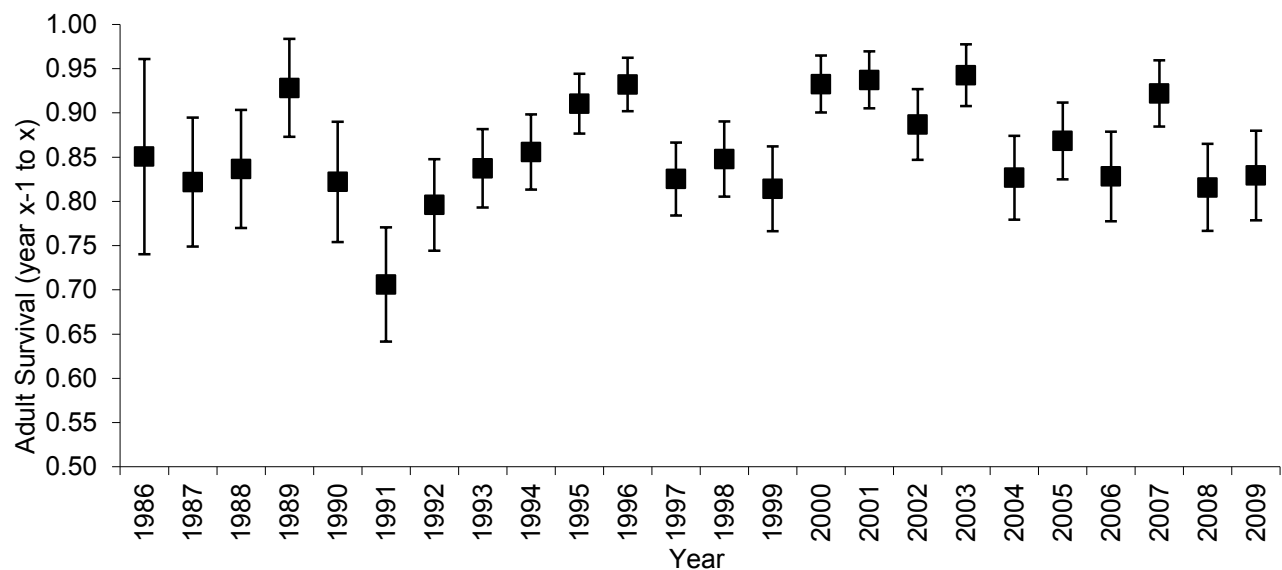

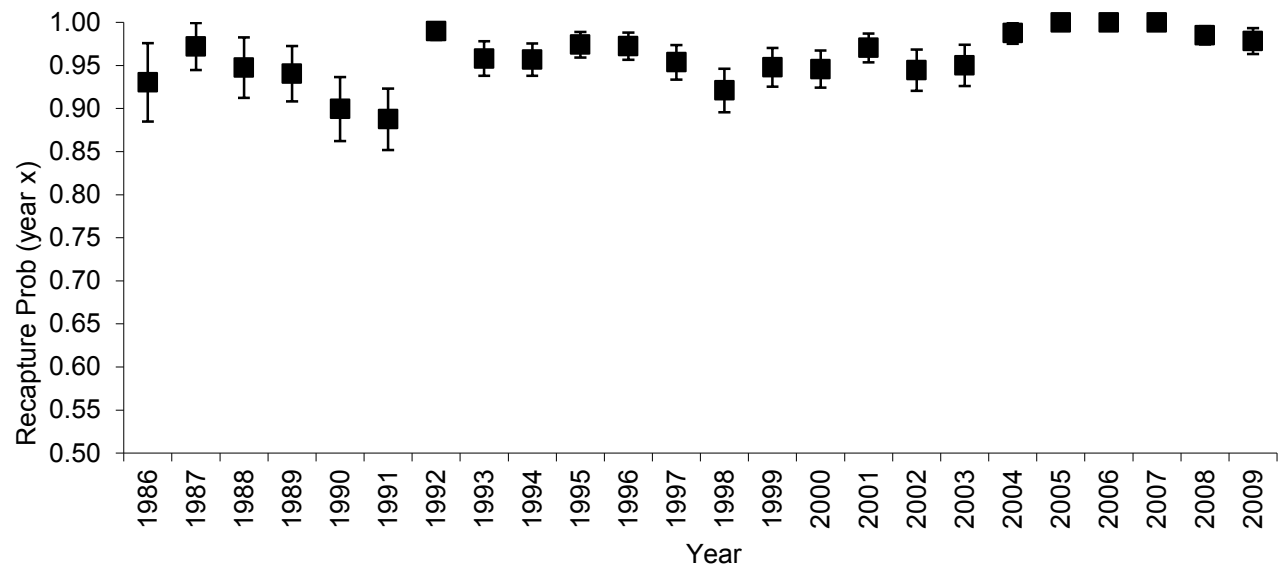Figure 2a. Female recapture probability

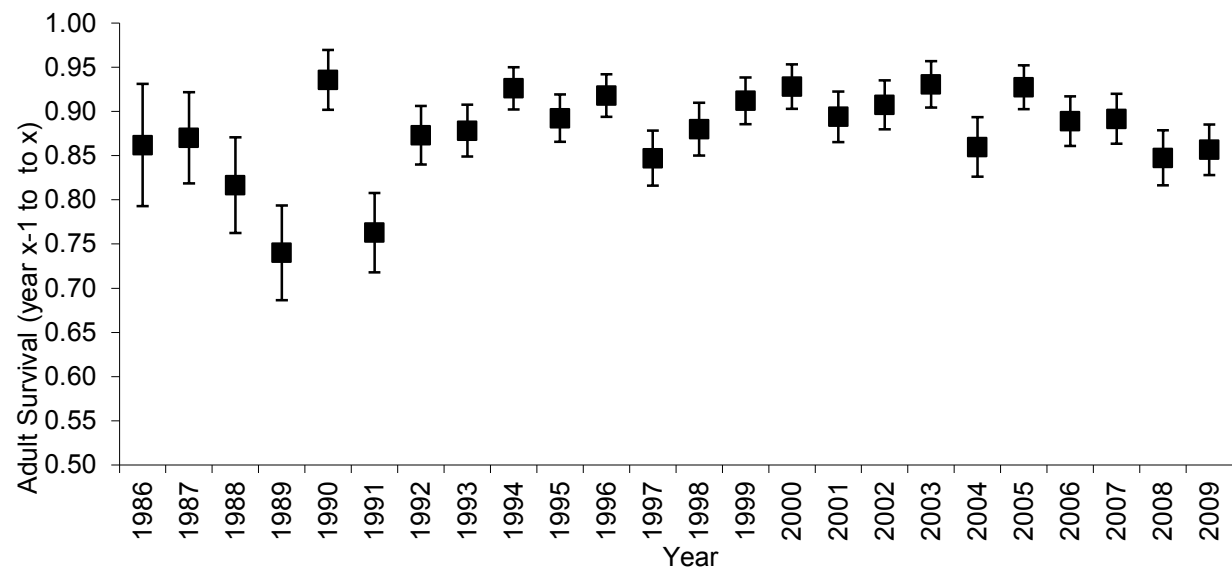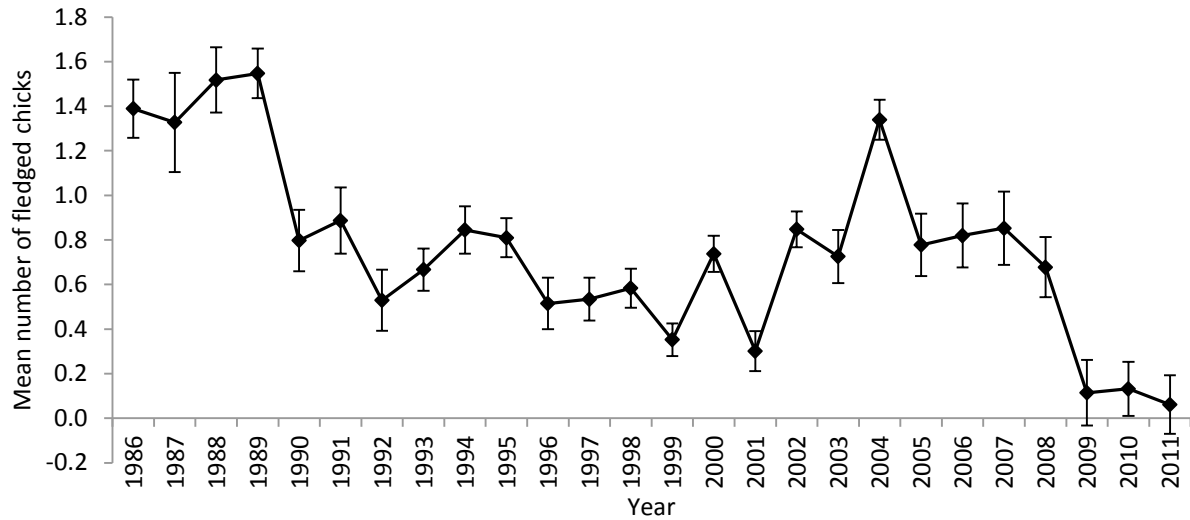Figure 2b. Female Survival

Figure 2c. Male recapture probability

Figure 2d. Male survival

Figure 3. Annual estimates (± SE) for mean number of chicks fledged per female Western Gull breeding in C, H, and K plots combined on Southeast Farallon Island, California 1983-2011.

Figure 4. Estimated percent change in the Farallon Western Gull population over 20 years, assuming "Optimistic" conditions (no re-occurrence of near-failure years), with (red) and without (black) eradication-associated mortality. Shown are the 25th percentile, 50th percentile (solid regression line and circles), and 75th percentile outcomes. Mortality scenario removes 1700 birds in year 0. Assumes a starting population of 32,200 birds.
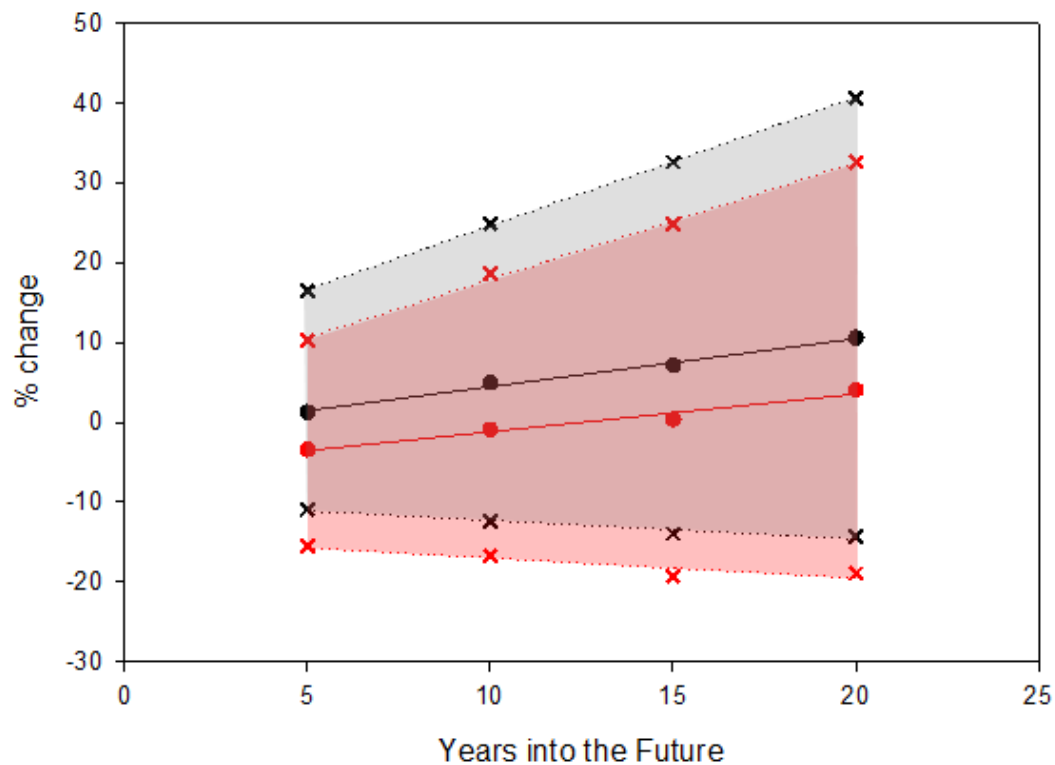
Figure 5. Estimated percent change in the Farallon Western Gull population over 20 years, assuming "Realistic" conditions (re-occurrence of near-failure years at historic frequency of, on average, 3 times per 26 years), with (red) and without eradication-associated mortality (black). Shown are the 25th percentile, 50th percentile (solid regression line and circles), and 75th percentile outcomes. Mortality scenario removes 1700 birds in year 0. Assumes a starting population of 32,200 birds.
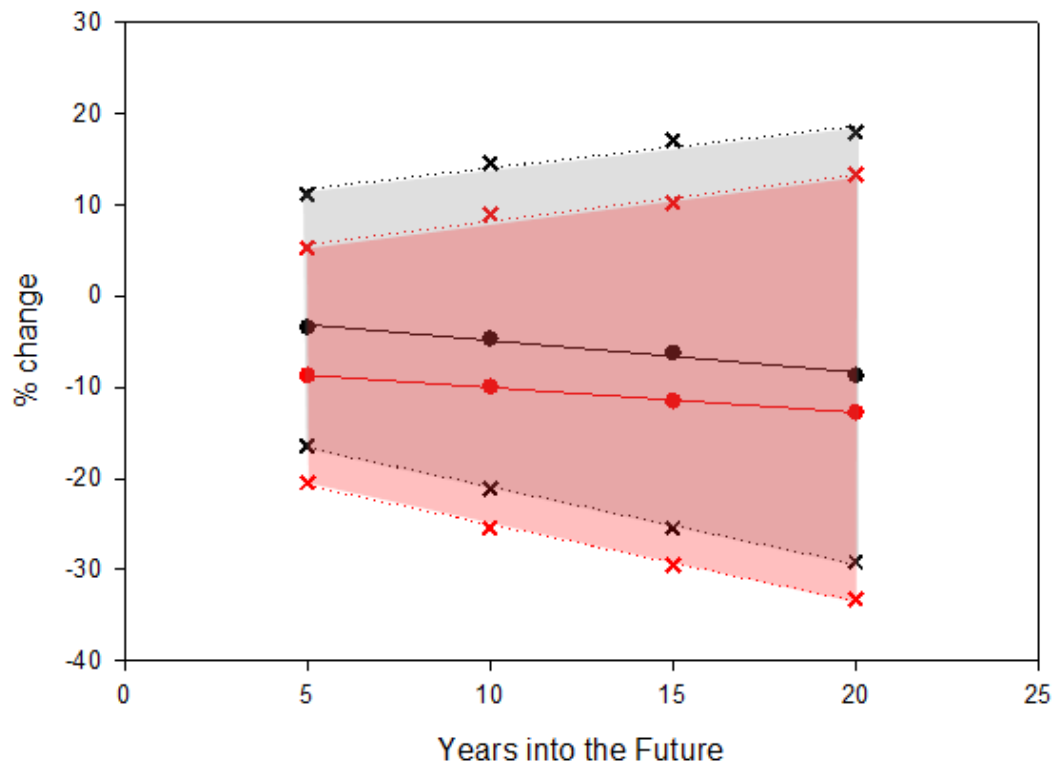
Figure 6. Estimated percent change in the Farallon Western Gull population over 20 years, assuming "Pessimistic" conditions: re-occurrence of near-failure years at recent frequency (on average, 3 times per 12 years), with (red) and without (black) eradication-associated mortality. Shown are the 25th percentile, 50th percentile (solid regression line and circles), and 75th percentile outcomes. Mortality scenario removes 1700 birds in year 0. Assumes a starting population of 32,200 birds.
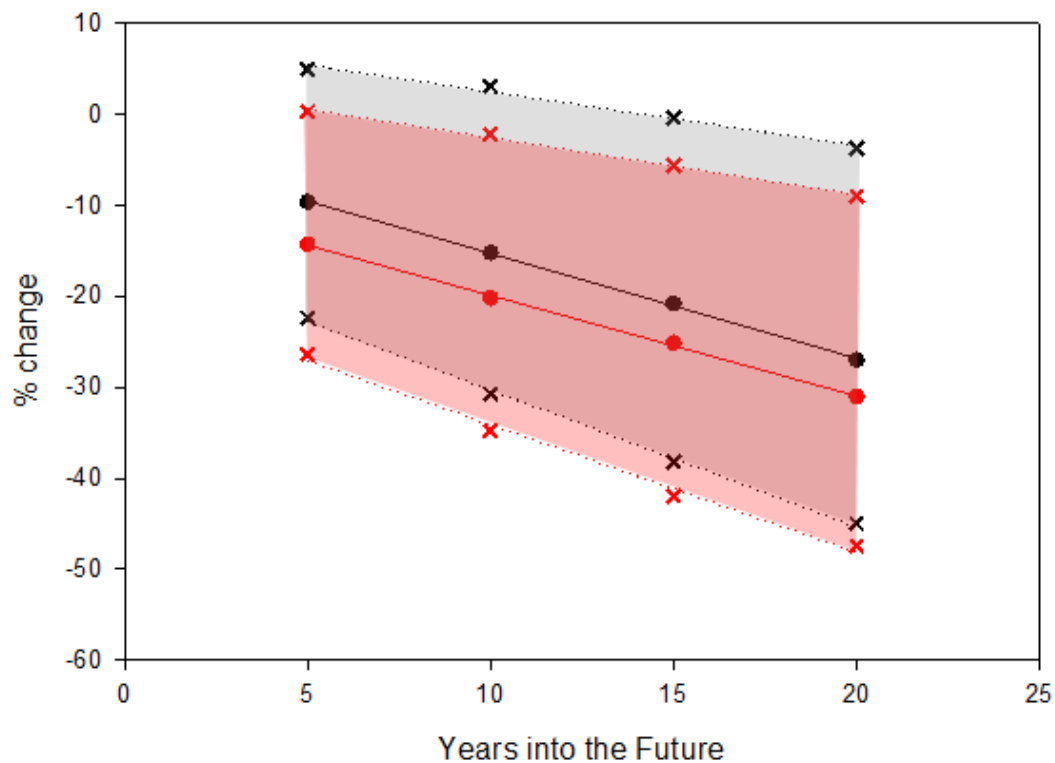
Figure 7. Probability distribution for "no mortality" and "mortality of 1700 gulls" scenarios, after 20 years, under "Realistic" Conditions: "historic" frequency of near-failure (results of 10,000 simulations for no mortality and 30,000 simulations for mortality of 1700 gulls). Note initial population size, with no mortality, is 32,200 individuals. Results binned into bins of 2,000 and then a polynomial (fourth-order) smoothing function was applied, except that the extreme tails are actual values. The two probability density functions overlap by approximately 95%.