

NON TECHNICAL SUMMARY

Estimation of Steller sea lion population dynamics parameters

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OBJECTIVES:

1. Examine the scientific value of data to be collected to reduce the variances of the estimates of key population model outputs by means of Monte Carlo simulation.
2. Examine whether experimental management based on spatial replication of treatments can be used to discriminate among the various possible hypotheses for the decline in Steller sea lion population size.

NON TECHNICAL SUMMARY:

A spatially-structured individual-based population dynamics model for Steller sea lions is developed to form the basis for the evaluation of questions related to the ability to estimate model quantities. The model is also used for the evaluation of alternative experimental management regimes, in terms of whether they are able to determine whether the cause of the decline in Steller sea lion abundance was due to a fishing-induced impact or some other factors. The model includes spatial-structure by associating individual Steller sea lions with one of 32 rookeries and by modelling movement to incorporate depth preference and natal homing. The model can generate the types of data available for Steller sea lions (pup counts, counts at haul-outs, estimates of survival rate from mark-recapture studies, and age-composition data).

Monte Carlo simulation is used to evaluate different data collection schemes in terms of the ability to distinguish between whether the decline was caused by a change in survival rate, pregnancy rate or both. The ability to correctly identify the cause of the decline is high if the time-dependence in the impact is assumed correctly but can be poor otherwise. The ability to determine pup production and its trend is adequate but estimates of total population size and the number of mature females can be severely biased.

The ability of experimental treatments based on spatial zoning to determine whether the decline was caused by fishing-induced impacts is also evaluated using Monte Carlo simulation. Experiments based on splitting four of the regions identified by York et al. (1996) into sectors open and closed to fishing are considered. The performance of the experiments is poor, owing to the impact of movement, different historical trends in different areas, demographic stochasticity, and the likely effect size.

A number of areas for future research are identified including: (a) assessing the ability to distinguish between whether the decline was caused by a change in survival rate,

pregnancy rate or both when the confounding impact of movement is included, (b) evaluating experiments when the impact of opening an area to fishing is larger than assumed in this study, and (c) extending the individual-based model to link with ecosystem and fishery models to possibly represent some of the possible causes for the declines more explicitly. A major source of uncertainty when parameterising the Steller sea lion model is lack of data on movement. Studies which increase understanding of this aspect of the ecology of Steller sea lions could substantially reduce the uncertainty associated with any model predictions.

KEYWORDS: Steller Sea Lions, Monte Carlo simulation, Spatial Management

1. BACKGROUND

Steller sea lions (Otariidae; *Eumetopias jubatus*) are distributed across the North Pacific rim from the Kuril Islands, through the Aleutian Islands, to the Channel Islands in southern California (Loughlin *et al.*, 1984). Assessments of the status and trends of Steller sea lion populations in Alaska are based on the results of visual counts conducted by the National Marine Fisheries Service (NMFS) and the Alaska Department of Fish and Game (ADF&G) of pups at rookeries and of non-pups at rookeries and non-breeding haul-out sites (e.g. Sease and Loughlin, 1999). Data from these counts suggest that the Alaskan population of Steller sea lions declined by approximately 85% between 1956 and 1998 (York *et al.*, 1996; Sease and Loughlin, 1999). Concerns over the decline in abundance inferred primarily from these counts led the Steller sea lion to be declared as *threatened* under the US Endangered Species Act (ESA) in 1990 (NMFS, 1992). Evidence that temporal trends in Steller sea lion abundance were not geographically uniform (e.g. York *et al.*, 1996), listing by the IUCN (International Conservation Union) of the Steller sea lion as endangered, and genetic evidence of distinct breeding populations led to the classification of Steller sea lions into two stocks. In 1997, the population of Steller sea lions to the west of Prince William Sound, Alaska (the western stock) was declared *endangered* under the ESA, with the population to the east of this point (the eastern stock) remaining *threatened* (Loughlin, 1997).

A variety of reasons for the decline of the western stock of Steller sea lions have been postulated. York (1994) examined changes in the age-composition of samples collected in 1975-78 and 1985-86, and concluded that a 20% decline in the annual survival of juvenile females was the simplest explanation for the reduction in abundance. Pascual and Adkison (1994) analyzed several possible reasons for the decline in Steller sea lion abundance, and concluded that transient age-structure dynamics, historical pup harvesting, and short-term environmental stochasticity are unlikely causes, and that long-term environmental changes or a catastrophe of some sort were probably responsible. Other hypotheses regarding the Steller sea lion decline involve reduction in birth rate, nutritional stress, predation, competition with fisheries, migration, disease, pollution, and the impact of a regime shift or trophic cascade (e.g. Calkins and Goodwin, 1988; Calkins and Pitcher, 1982; Hoover, 1988; York, 1994; Merrick *et al.*, 1987; Loughlin and Merrick, 1989). To date, it has not been possible to exclude any of these hypotheses definitively although past research efforts have eliminated redistribution, pollution, predation, subsistence harvest, disease, and natural fluctuations as the principal causes for the decline (NMFS, 1992). Several authors suggest that the cause of the decline may be a combination of various factors (Loughlin and York, 2000; Hunter and Trites, 2001), and that indeed the primary cause has likely changed over the period of decline (Loughlin and York, 2000). Current information is, however, insufficient to determine adequately (i.e. with sufficient accuracy and precision) values for key population dynamics parameters such as juvenile survival rate. Both of these problems could be addressed through the collection and analysis of additional data.

The timing and extent of the negative trend in the size of the western stock has not been uniform over the geographical range of the population. Trites and Larkin (1996) and

York *et al.* (1996) both identified several spatially distinct trends in abundance of Steller sea lions within the western stock. This suggests that the factors responsible for the decline have not been spatially homogenous. The behavior of Steller sea lions does create distinct localized populations that may well enable spatial differences in the factors affecting population processes, such as survivorship and pregnancy rate, to be expressed differentially within the overall population. In addition to evidence that Steller sea lions do not breed other than with their natal stock [based on analysis of mitochondrial DNA showing there is little incidence of female interchange between the eastern and western stocks (Bickham *et al.*, 1996) - also corroborated by tagging studies (Raum-Suryan *et al.*, 2002)], Steller sea lions show a degree of tendency toward natal site fidelity in that females return to breed at either the site of birth, or at a site close to the natal site (Pitcher and Calkins, 1981).

A NMFS-funded project (Punt and Fay, 2002) describes a spatial age-lumped model of Steller sea lion population dynamics. This model includes demographic and environmental stochasticity in birth and death processes and is fitted (to the extent possible) to population counts, survival rate estimates and age-composition data, and incorporates existing information on survival, pregnancy rate and movement. The model includes features to allow the reason(s) for the decline in population size to be input and therefore provides a necessary basis for the evaluation of the ability of future data to resolve current uncertainties.

2. OBJECTIVES

The objectives for the study were:

- 1) Examine the scientific value of data to be collected to reduce the variances of the estimates of key population model outputs by means of Monte Carlo simulation.
- 2) Examine whether experimental management based on spatial replication of treatments can be used to discriminate among the various possible hypotheses for the decline in Steller sea lion population size.

3. METHODS - OVERVIEW

Monte Carlo simulation is used in this study to determine the value of alternative data collection schemes and their associated data analysis methods and of alternative experimental management approaches. Monte Carlo simulation methods have been used extensively in the past to evaluate data collection schemes and analysis methods in terms of their ability to estimate various quantities of interest to management accurately and precisely. This is particularly the case for the methods of analysis applied to marine fish species (e.g. Patterson and Kirkwood, 1995; Sampson and Yin, 1998; Punt *et al.*, 2002) although Monte Carlo simulation methods have also been used for this purpose for problems involving marine mammal populations (e.g. Kirkwood, 1981; de la Mare, 1986).

Monte Carlo simulation has been used less extensively in the marine environment to evaluate the relative benefits of alternative experimental management approaches involving spatial zoning with different experimental “treatments” (where “treatment” in this case involves whether fishing is or isn’t allowed in a particular spatial zone). Monte

Carlo simulation methods form the basis for the calculations that underlie many of the techniques of Adaptive Management (Walters, 1986; McDonald and Smith, 1997). The experiments to resolve key uncertainties regarding the dynamics of benthic communities off Australia's North West Shelf following the introduction of fishing (e.g. Sainsbury *et al.*, 1997) and to estimate the quantities needed to evaluate management strategies for the exploitation of coral trout on the Great Barrier Reef, Australia (e.g. Mapstone *et al.*, 1996; Punt *et al.*, 2001) were both originally based on a Monte Carlo evaluation of the potential of different experimental management regimes to discriminate among alternative hypotheses.

The key steps in evaluating the value of data collection schemes and their associated data analysis methods, either in terms of estimating quantities of interest to management or in terms of discriminating among alternative hypotheses, using Monte Carlo simulation methods are:

1. Selection of the key quantities of interest / hypotheses to consider.
2. Identification of potential data sources / alternative spatial experimental regimes and the approaches that would be used to analyse the data from the experiments.
3. Development of a model of the system of interest (the operating model), which can be used to generate data typical of those that would result from field experiments.
4. Parameterization of the operating model for various alternative hypotheses for the decline in the abundance of the Steller sea lion, for example, by fitting it to the existing data.
5. Simulation of the process of conducting experiments and analysing the results thereof using the methods selected at step 2.
6. Evaluation of the success of each data source and its associated analysis method though the magnitude of the bias and variance for the estimates of the quantities of interest.
7. Evaluation of the success of different experimental regimes in terms of the whether the probabilities associated with the various hypotheses for the reasons for the decline in Steller sea lion abundance are updated in the correct direction.

Both of the objectives require an appropriate operating model. The following section therefore outlines the age-, sex- and spatially-structured operating model that underlies the analyses of this study.

4. METHODS - THE OPERATING MODEL

4.1 Basic overview

The operating model considered in this report is age- and sex-structured and individual-based. The time-step of the current implementation of the model is monthly (i.e. $N_m=12$) but the model is coded so that any time-step step of a year or shorter can be implemented straightforwardly. The information available for each animal includes its age, sex, mother, whether it is mature or not, its rookery of birth (all animals are assumed to home to their natal sites) and where it is in the North Pacific at any point in time. The location of rookery k is denoted (X^k, Y^k) .

The steps that are carried out each month to project the model forward are:

- If the month is July (the start of the modelled year and the assumed pupping season), generate the number of pups. For each mature female, there is a probability of giving birth that depends on rookery. By definition, all pups are age 0 and the pups are located on their natal rookeries at birth. The probability of a newborn being male or female is assumed to be 0.5¹.
- For females that are immature, whether they mature during the current year (and hence have to move during the year to ensure that they are on their natal rookery at the start of the next year) occurs with probability:

$$\tilde{\beta}_a = \frac{\beta_a - \beta_{a-1}}{1 - \beta_{a-1}} \quad (1)$$

where $\tilde{\beta}_a$ is the probability that an immature female that will be age a at the start of the following year will mature during the current year, and β_a is the fraction of females of age a that are mature.

For ease of calculation, the application of Equation 1 occurs at the start of the modelled year.

- Whether each animal dies during the current time-step is determined. The probability of death depends on age and location in the present operating model. In the absence of additional mortality, the mortality rate for pups is assumed to be density-dependent, i.e.:

$$S_{y,0}^k = \bar{S}_0 + (\bar{S}_0^{\max} - \bar{S}_0)(1 - N_y^{k,\text{tot}} / K^k) \quad (2)$$

where $S_{y,0}^k$ is the survival rate for pups from rookery k during year y ,
 \bar{S}_0 is the survival rate for pups in an equilibrium state,
 \bar{S}_0^{\max} is the survival rate for pups in the limit of zero population size,
 $N_y^{k,\text{tot}}$ is the total number of animals aged 1 and older that were pupped at rookery k and are still alive at the start of year y , and
 K^k is the carrying capacity of rookery k , expressed in terms of animals aged 1 and older.

The actual probability of death depends on the survival rate in the absence of additional mortality multiplied by a factor that reduces the survival rate to the some extent. This factor depends on age, year and location, as detailed below.

¹ The sex ratio for pups aged 1 day to 3 weeks has been estimated to be 51:49 males to females (Calkins and Pitcher, 1982).

- Each animal is allowed to move from its current location. The algorithm applied to determine the location of an animal at the start of month $m+1$, given its location at the start of month m is as follows:
 - 1) Generate a bearing at random from $U[0, 2\pi]$.
 - 2) Generate the net distance moved from a normal distribution, $N(\bar{D}, \sigma_D^2)$ where \bar{D} is the average net distance moved during a month and σ_D is a measure of the variation in the net distance moved during a month.
 - 3) Compute a “target location” (X, Y) based on the current location of the animal and the distance and bearing generated at steps 1) and 2).
 - 4) If it is after mid-winter, allowance for made for animals to move back to towards their natal rookeries. This is implemented by modifying the “target location” according to the equation:

$$\begin{aligned} X' &= X + R(X^k - X)/(N_m - m + 1) \\ Y' &= Y + R(Y^k - Y)/(N_m - m + 1) \end{aligned} \quad (3)$$

where R is a factor which determines the rate at which animals return to their natal rookeries (assumed to be 1 for mature animals and 0.5 for immature animals).

- 5) The probability of moving to a location depends on its depth. The probability of moving to a location is likely to be higher the shallower its depth because: (a) there is more likely to be an island or haulout area in the vicinity, and (b) (presumably) less energy would be needed to forage on the bottom. Therefore, the depth at the (possibly revised) target point is calculated and a $U[0,1]$ random variate generated. Whether the animal moves to the target point depends on a depth-specific probability (see Table 1). If the animal does not move the target point, steps 1) - 5) are repeated.

The movement model is very simple and makes several assumptions, the implications of which, given additional data, could be examined. For example, the average net distance moved (and its variation) are assumed to be independent of age, sex and time.

- If it is the end of the year, the age of each animal is incremented by one.

The initial conditions (nominally year $y=1950$) are defined to be a population in an equilibrium state. The input parameters that determine the number of animals of each rookery at the start of 1950 are the numbers of pups by rookery at the start of 1950. Given specifications for the survival rates for animals aged 1 and older, and the pregnancy rates by age (assumed, in absence of data to the contrary, to be independent of rookery) in an equilibrium state, the value of \bar{S}_0 is computed so that the population remains in equilibrium.

4.2 Imposing additional mortality / lower pregnancy rate

The model outlined above includes demographic stochasticity in birth and death rates and individual variation in movement rates. However, to be able to mimic the observed population trajectory for Steller sea lions in Alaska even qualitatively, it is necessary to have model formulations that capture hypotheses for the decline in the population. These hypotheses can be divided into those in which some unknown (but perhaps spatially-determined) process (or processes) impact the pregnancy rate of females, those in which some unknown process(es) impact the survival rate of pups, juveniles and adults (again possibly with spatial differences in the size of the effect) and those in which some unknown process(es) impact both the pregnancy rate and the survival rate. There is no formal way to identify how the (possible) impacts on pregnancy rate and survival rate may have changed over time so the analyses of this report consider alternative functional forms for the time-dependence in the impacts.

4.2.1 Modelling impacts on pregnancy rate

The pregnancy rate of mature animals is defined to be the product of an average pregnancy rate (0.65) and a factor which allows for reductions in pregnancy rate due to unknown causes, h_y^A . The factor modifying pregnancy rate is assumed to depend on rookery. Two alternative functional forms for the impact on pregnancy rate are considered (smooth and knife-edged). The smooth function has the form:

$$h_y^A = \begin{cases} 1 - h_F \exp(-[y - y_F]^2 / \sigma_F^1) & \text{if } y \leq y_F \\ 1 - h_F \exp(-[y - y_F]^2 / \sigma_F^2) & \text{otherwise} \end{cases} \quad (4)$$

where h_F is the parameter that determines the magnitude of the impact on pregnancy rate,
 y_F is the year in which the impact on pregnancy rate is greatest,
 σ_F^1 is the parameter that determines the rate at which the impact on the pregnancy rate changes with time prior to year y_F , and
 σ_F^2 is the parameter that determines the rate at which the impact on the pregnancy rate changes with time after year y_F .

The knife-edged function has the form:

$$h_y^A = \begin{cases} 1 - h_F & \text{if } y_F^A \leq y \leq y_F^B \\ 1 & \text{otherwise} \end{cases} \quad (5)$$

where y_F^A is the first year in which pregnancy rate is reduced, and
 y_F^B is the last year in which pregnancy rate is reduced.

4.2.2 Modelling impacts on survival rate

The survival rate for an animal of age a during year y is assumed to be the product of an expected survival rate ($S_{y,a}^k$ - see Equation 2 for the equation defining the survival rate for pups) and an age- and location-specific time-dependent reduction in survival rate, $k_{y,a}^A$, i.e.:

$$k_{y,a}^A = \begin{cases} 1 - \phi_1 \tilde{k}_y^A & \text{if } a = 0 \\ 1 - \phi_2 \tilde{k}_y^A & \text{if } 1 \leq a \leq 4 \\ 1 - \tilde{k}_y^A & \text{otherwise} \end{cases} \quad (6)$$

where ϕ_1 is the relative impact of changes in adult survival rate compared to that for pups,
 ϕ_2 is the relative impact of changes in adult survival rate compared to that for juveniles, and
 \tilde{k}_y^A is the impact during year y on the expected survival rate of adults.

The model for the impact on survival rate therefore allows for different impacts on pups (age 0), juveniles (ages 1-4) and adults (ages 5+):

Two alternative functional forms for the impact on survival rate (smooth and knife-edged) are considered. The smooth function has the form:

$$\tilde{k}_y^A = \begin{cases} h_s \exp(-[y - y_s]^2 / \sigma_s^1) & \text{if } y \leq y_s \\ h_s \exp(-[y - y_s]^2 / \sigma_s^2) & \text{otherwise} \end{cases} \quad (7)$$

where h_s is the parameter that determines the magnitude of the impact on survival rate,
 y_s is the year in which the impact on survival rate is greatest,
 σ_s^1 is a parameter that determines the rate at which the impact on the survival rate changes with time prior to year y_s , and
 σ_s^2 is a parameter that determines the rate at which the impact on the survival rate changes with time after year y_s .

The knife-edged function has the form:

$$\tilde{k}_y^A = \begin{cases} 1 - h_s & \text{if } y_s^A \leq y \leq y_s^B \\ 1 & \text{otherwise} \end{cases} \quad (8)$$

where y_s^A is the first year in which survival rate is reduced, and

y_S^B is the last year in which survival rate is reduced.

4.3 Parameterization of the model

The values for the biological parameters of the model are listed in Table 2. The value of the parameter \bar{S}_0^{\max} is assumed to 0.99 for the purposes of the analyses of this report. The values for the remaining parameters (those that define the rookery-specific carrying capacities and the reductions in survival and pregnancy rate) differ between the analyses related to the two objectives and are detailed below.

5. EVALUATING ALTERNATIVE DATA COLLECTION SCHEMES

5.1 Overview

The model outlined above was used as the basis to evaluate alternative (historical) data collection schemes in terms of their ability to distinguish among scenarios in which the decline in Steller sea lion numbers was caused by (a) additional mortality (smooth or knife-edged), (b) a reduction in pregnancy rate (smooth or knife-edged) or (c) both additional mortality and reduced pregnancy rate. The data collection schemes are also evaluated in terms of how well it is possible to estimate the number of pups and the number of mature females for the years 1950-2000.

For ease of interpretation, the population dynamics model was set up for a single rookery and a single haulout site, and additional mortality was assumed to be the same across the entire area included in the model. In essence, this reduces the model to an individual-based model with demographic stochasticity but no spatial structure. The total number of pups in 1950 was assumed to be 18,000 – a value close to the number of pups estimated for the Central Gulf of Alaska region by Punt and Fay (2002).

One aim of the analysis was to determine the probability that an estimation framework applied to historical data can correctly detect whether the cause of an observed decline is additional mortality, reduced pregnancy rate or both, but not necessarily the form of the change in these variables (smooth or knife-edged; see Equations 4-8). The steps involved in evaluating a combination of a data set and an estimation framework are:

- 1) Selection of a model to represent the “truth” for the simulations (the “operating model”). In this case the model is a variant of the individual-based model outlined in Section 4 of this report.
- 2) Identification of various “true” scenarios among which it is desirable that the estimation frameworks to be evaluated are able to distinguish.
- 3) Identification of alternative models that characterize each of the three scenarios regarding the cause for the decline.
- 4) Generation of the data sets using the operating model. In this example, the data sets include pup counts, non-pup counts, estimates of survival rate and age-composition information. This represents the information available for the most data-rich of the regions defined by York *et al.* (1996) (the Central Gulf of Alaska).

- 5) Application of the each of the alternative models to each of the data sets and selection of one of these models for each data set based on whichever leads to the lowest value for Akaike's Information Criterion (AIC; Burnham and Anderson, 1998).
- 6) Comparison of the models selected at step 5 with the "true" scenario for the simulation.

5.1.1 Specifications of the operating model

Six underlying biological scenarios (see Table 3 for parameter values) are considered:

- 1) The decline was due to a "smooth" reduction in survival rate (Equation 6; abbreviation "Smooth mortality").
- 2) The decline was due to a "knife-edged" reduction in survival rate (Equation 7; abbreviation "Knife-edged mortality").
- 3) The decline was due to a "smooth" reduction in pregnancy rate (Equation 4; abbreviation "Smooth pregnancy").
- 4) The decline was due to a "knife-edged" reduction in pregnancy rate (Equation 5; abbreviation "Knife-edged pregnancy").
- 5) The decline was due to a "smooth" reduction in survival and pregnancy rate (Equations 4 and 6; abbreviation "Smooth both").
- 6) The decline was due to a "knife-edged" reduction in survival and pregnancy rate (Equations 5 and 7; abbreviation "Knife-edged both").

Examples of two sets of population trajectories for these six cases are shown in Figure 1. As expected given the parameter values, all six cases exhibit a large decline in pup production (Figure 1a), total population size (Figure 1b), and number of mature females (Figure 1c). The total pup counts for the Central Gulf of Alaska estimated by Punt and Fay (2002) (Table 4 – see Table 5 for the rookeries included in each of the regions considered in this report) are included in Figure 1(a). Note, however, that the scenarios in Table 3 were not chosen to mimic the actual pup count data for any particular region closely but rather to capture a variety of possible time-trajectories of pup production. The actual pup count data for the Central Gulf of Alaska are, in fact, not used explicitly in any of the analyses of this section. For all scenarios, however, the magnitude of the reduction in pups between 1950 and 2001 is roughly the same.

The two sets of time-trajectories in Figure 1 (solid and dotted lines) do not differ much. This is primarily because both time-trajectories start with the same number of pups but, more importantly, because the impact of demographic uncertainty is inconsequential for population sizes of the magnitude considered in Figure 1 (a total of more than 80,000 animals – Figure 1b). The time-trajectories of pup numbers and those of mature females and total numbers exhibit quite similar patterns with time when the decline is caused by additional mortality but much less so when the decline is caused by a decrease in pregnancy rates. The question addressed in this section is therefore whether the data available are adequate to distinguish among the various causes for the decline given these differences in the shape of the time-trajectory of pup counts.

The data generated by the operating model are:

A) Age-composition data

Two age-composition samples (in simulation years 1973 and 1985) are generated by sampling animals randomly from the 1+ population. The base-line sample size is 100 animals. This sample size is slightly less than the actual sample sizes for Steller sea lions in these years to account (to some extent) for non-random sampling.

B) Pup count data

Estimates of pup numbers are generated for simulation years 1976, 1978-79, 1984, 1986-94, 1996-98, and 2000-01. The pup count estimates are assumed to be log-normally distributed about the true numbers of pups with a baseline coefficient of variation of 0.05.

C) Non-pup count data

Estimates of the number of animals aged 1 and older are available for simulation years 1976, 1986, 1989-92, 1994, 1996-98, and 2000. The non-pup counts are assumed to be log-normally distributed with baseline coefficients of variation of 0.1.

D) Tag-recapture-based estimates of survival

Survival rate estimates over the period 1988-93 for animals that were age 0 in 1988 and those were aged 5+ in 1988 are generated by “tagging” the animals of age 0 and ages 5+ at the start of 1988 and “recapturing” the “tagged” animals that are still alive at the start of 1993. The ratio of the number of animals “recaptured” to the number “tagged” defines the expected survival rate. The actual estimates available to the estimation frameworks are log-normally distributed about these expected values with coefficients of variation of 0.1 (age 0 animals) and 0.05 (animals aged 5+). Separate estimates are generated for males and females.

5.1.2 Estimation models

The estimation models applied to the simulated data sets are deterministic non-spatial versions of the operating model. The estimable parameters of these models include the carrying capacity and parameters that depend on which of (a) increased mortality, (b) reductions in pregnancy rate, and (c) both increased mortality and reductions in pregnancy rate, is assumed to be the cause of the decline. The change in survival rate / pregnancy rate is always assumed to be smooth. The values for the parameters of the model are estimated by maximizing a likelihood function that includes each of the four data types listed above.

5.2 Results and discussion

5.2.1 The base-case trials

Table 6 lists the probability of selecting each of the three causes for the decline in Steller sea lion abundance (i.e. reduced survival, reduced pregnancy rate, or both – “Mortality only”, “Pregnancy only” and “Mortality and Pregnancy” in Table 6) for six variants of a base-case trial. This base-case trial includes all of the four data sources listed above while the six variants are based on each of the six underlying scenarios for actual cause of the decline in the simulated Steller sea lion population.

The probability of correctly identifying the cause for the decline is very high (>90%) when the change in pregnancy rate / survival rate is really smooth with time. However, there are quite substantial probabilities of selecting the wrong model when the change in pregnancy rate / survival rate is knife-edged (but the estimators assume that the change is smooth). For example, the probability of correctly identifying that a change is caused by a decline in pregnancy rate (only) drops from 93% (row “Smooth pregnancy” in Table 6) to only 19% (row “Knife-edged pregnancy” in Table 6). The probability of incorrectly selecting the model in which both pregnancy rate and survival changed over time is high when the change is actually due to a single factor but the change is knife-edged in time. This suggests that the AIC statistic leads to the selection of an overly complicated model because the simple “smooth” model is unable to mimic the observed time-trajectory of pup counts adequately (see, for example, Figure 1a) – adding some extra complexity due to a change in another factor leads to an improved fit.

Figure 2 shows the errors when estimating the annual pup numbers (1950-2001) in terms of the time-trajectories of the medians and 95% intervals for the relative errors (expressed as percentages). Results are shown in Figure 2 for each of the six underlying scenarios and for the three alternative hypotheses for the decline (“Mortality only”, “Pregnancy only”, and “Mortality and Pregnancy”). The estimates of pup numbers are unbiased and precise when the assumed cause for the decline in the estimation model is the same as the true cause in the operating model while the “Mortality and Pregnancy” estimator is close to unbiased after about 1980 for all six scenarios. The “Mortality only” and “Pregnancy only” estimators exhibit bias when the hypothesis underlying the estimation model differs from that underlying the operating model and when the cause for the decline is correctly assumed but that true functional form for the change is knife-edged rather than being smooth, but the magnitude of the bias is only substantial for the years prior to about 1980 (Figure 2).

As expected, the estimates of the number of mature females are essentially unbiased and fairly precise when the hypothesis underlying the estimation model is correct (Figure 3), although some bias is evident for the “Mortality and Pregnancy” estimator for recent years even when the operating model is based on the assumption that both survival and pregnancy rate were impacted. The sizes of the biases are, however, very substantial when the hypothesis underlying the estimation model differs from that underlying the operating model and, furthermore, the biases is often largest (less than -50% and greater than 100%) for recent years unlike the case for pup numbers (Figure 2).

5.2.2 Sensitivity tests

Table 7 examines the sensitivity of the results in Table 6 to changing the data available to the estimation methods.

Eliminating the age data (row “No Age data” in Table 7) has a relatively small impact on the performance of the estimation methods – there are slight increases to the probability of correctly identifying the cause of the decline when the factor causing the decline changes smoothly over time and a fairly large increase in correctly identifying a change in survival as the cause of the decline for the knife-edged mortality scenario. Not having information on survival reduces the probability of correctly selecting the “Mortality only”

model if the “Smooth mortality” scenario is true (from 90 to 87%) but has almost no impact on the results for the other scenarios.

Increasing the coefficients of variation of the data from 0.05 and 0.1 (pups and non-pup counts respectively) to 0.1 and 0.2 respectively (row “Higher CVs” in Table 7) increases the probability of correctly identifying the correct cause for the decline. This is particularly the case when survival and pregnancy rate change in a knife-edged way when the probabilities increase from 69 and 19% to 83 and 54%. It may seem surprising that the probability of correctly identifying the true cause for the decline increases when the precision of the data gets poorer. This occurs because, given the very high precision associated with the data for the base-line trial, even a slight improvement in fit by the model in which both survival and pregnancy rate change over time leads to this more complicated model being selected over the true model. Increasing the coefficients of variation associated with the count data reduces the probability of this error occurring.

Performance in terms of correctly identifying a change in survival rate as the cause for the decline deteriorates substantially when the true population size is low (1,800 pups in 1950 rather than 18,000 - row “Low population size” in Table 7). One reason for this is that the dynamics are more impacted by demographic uncertainty than is evident in Figure 1. The more complicated model is selected over the true model in this case because it can better mimic the available information. Fitting the models using only count data (row “Counts only” in Table 7) has little impact on the results.

The final sensitivity test corresponds to a form of “worst case” scenario (but one that is not completely dissimilar to the actual situation for the western Aleutian Islands and eastern Gulf of Alaska regions). This sensitivity test (row “Poor data” in Table 7) involves lower initial (1950) pup numbers, higher coefficients of variation for the pup and non-pup counts and no age or survival rate data. Somewhat surprisingly, the probability of identifying the correct model remains high (in fact the probability of correctly identifying a change in pregnancy rate as the cause of the decline is higher for this sensitivity test than for the base-case analysis). However, in contrast to the other sensitivity tests, there is now a non-zero probability of identifying the cause as “Pregnancy only” when the true cause is “mortality only”.

5.2.3 Discussion

In general, if the change in survival rate, pregnancy rate or both was smooth with time, the estimators considered in this report are able to correctly identify this with high probability. Unfortunately, this is not the case if the change was knife-edged as a function of time when the more complicated model which assumes changes in both pregnancy rate and survival rate led to the decline is selected even if the cause is only one of these factors.

The probability of incorrectly selecting the more complicated model increases if the count data are more precise. This occurs because the estimates of precision used when fitting the model relate only to the extent of observation (measurement) error and do not capture model error (e.g. that the change in the population process that caused the decline was knife-edged rather than smooth) and process error (the impact of demographic

uncertainty if the actual population size is low). One consequence of this is that slight improvements in fit are accorded considerable importance when using AIC to select among models. Future work along these lines should consider imposing a minimum coefficient of variation (e.g. 0.1) when fitting population models to the count data to avoid this problem. Such data-downweighting to account for the impact of model and process error is common when applying fisheries stock assessment models to length-frequency data (e.g. Cope *et al.*, in press).

The results in Figure 1 and Tables 6 and 7 indicate that the primary source of information to discriminate among hypotheses is the count data. Not having data on survival rate appears to impact the ability to correctly identify survival as the cause of the decline.

Perhaps the most noteworthy outcome from this study is the poor ability to estimate the size of population components other than pups (Figures 2 and 3). This result was perhaps not unexpected given that the monitoring data do not relate directly to the absolute size of the mature female component of the population but it does provide a warning about using standard population dynamics models fitted primarily to pup count data to predict the numbers in other population components. Fay (in prep) has found that estimates of the total numbers of animals prior to 1980 are very sensitive to the choice of model structure (what population component has been impacted and what the time-dependence of this impact was).

6. EVALUATING ALTERNATIVE EXPERIMENTAL MANAGEMENT SCHEMES

The model outlined in Section 4 of this report is set up in as realistic a way as possible to evaluate the benefits of alternative experimental management schemes based on spatial zoning. This spatial configuration includes 32 rookeries and 104 haulout sites (see Figures 4 and 5). The objective of the experimental treatments is to determine whether by closing some (but not all) areas to fishing it becomes possible to determine whether fishing (rather than some other factor) is the cause of the decline (directly or indirectly).

6.1 Using experiments to select among causes for the decline

Two types of experimental management regime are considered in this study. Both of these involve closing areas to fishing and then monitoring the changes in pup production at the rookeries that are in the areas open and closed to fishing. There are few rookeries in the easternmost and westernmost regions (“Eastern Gulf of Alaska” and “Western Aleutian Islands” respectively; Table 5) so the experimental “treatments” are assumed to be applied to the central four regions only. There are two types of experimental management regime (A and B). Both involve defining longitudinal sectors and restricting or prohibiting fishing in some of these. Regime A closes all fishing within the “closed” longitudinal sectors (i.e. the area closed is defined by two lines of longitude and all fishing is assumed to cease between those longitudes) while Regime B closes all fishing within 37km of the rookeries in the “closed” longitudinal sectors. There are three implementations of each management regime depending on the choice of the longitudinal sectors and which of these sectors are designated to be open and which are designated to be closed (Table 8). The two sets of longitudinal sectors arise from: (a) grouping

rookeries so that the distance between rookeries within a sector is small relative to the distance between rookeries in adjacent sectors (experimental regimes 1A and 1B), and (b) splitting the existing regions approximately into two and placing the sector boundaries so that only rookeries from one region are in each sector (experimental regimes 2A, 2B, 3A and 3B).

The results from the experiment are counts of pups by longitudinal sector. The experiment (i.e. any closures to fishing) is assumed to occur in 2003, pup counts are assumed to be available for the years for which they are actually available (Table 4) and for all sectors every year after 2001. The coefficient of variation of the pup counts is assumed to be independent of rookery but perhaps time-dependent, and the pup counts are assumed to be log-normally distributed about their expected values.

The pup counts are grouped into longitudinal sectors and analysed by fitting the following model:

$$P_y^{r,s} = \begin{cases} \hat{P}_{1979}^{r,s} e^{-\lambda^r (y-1979)} & \text{if } y \leq 2003 \\ \hat{P}_{1979}^{r,s} e^{-\lambda^r (2005-1979)} e^{-\gamma^{r,s} (y-2005)} & \text{otherwise} \end{cases} \quad (9)$$

where $P_y^{r,s}$ is the pup counts summed over the rookeries with status s (status = in the sector open or in the sector closed to fishing) in region r for year y ,
 λ^r is a measure of the rate of change in pup numbers from 1979 to 2003 for region r , and
 $\gamma^{r,s}$ is a measure of the rate of change in pup numbers for the rookeries with status s in region r after 2003.

Two variants of equation 9 are fitted to each pseudo data set. The first variant (model 1) estimates five parameters for each region (two $P_y^{r,s}$ s, two $\gamma^{r,s}$ s and one λ^r) and the second variant (model 2) estimates four parameters for each region (two $P_y^{r,s}$ s, one γ^r and one λ^r), i.e. γ is assumed to be independent of the status of a rookery for model 2. Model 1 is based on the assumption that the trend after 2003 depends on the status of a rookery (i.e. closing an area changes the factors that determine additional mortality and reduced pregnancy rate) while model 2 is based on the assumption that closing an area has no impact on the magnitude of these factors (because fishing is not the cause of the decline). The fits of models 1 and 2 to the pup count data are compared using AICc (Burnham and Anderson, 1998) and one of these is selected. The ability of experimental treatments to distinguish whether fishing is the cause for the decline can be evaluated by the frequency of time that model 1 is selected when fishing is really the cause for the decline (and hence closing areas does change the trend in population size in the area closed to fishing) and by the frequency of time that model 2 is selected when fishing is not the cause for the decline (and hence closing areas does not change the trend in population size in the area closed to fishing).

6.2 Specifications for the operating model

The hypotheses considered in the operating model examine: (a) the biological processes impacted by whatever has caused the decline (additional mortality, and additional mortality and reduced pregnancy rate), and (b) whether closing an area eliminates the impact in that area. The impact on pregnancy rate is assumed to be related to the rookery (i.e. whether pregnancy rate for a particular rookery is affected by an experiment depends on whether the area around the rookery is open or closed to fishing). Note that closing the area around a rookery will not necessarily prevent an animal from that rookery dying due to additional mortality because the animal may move during the year to a sector that is not closed and hence suffer the additional mortality associated with that sector.

The movement patterns of Steller sea lions remain highly uncertain with data available for only a relatively small number of animals and over a relatively short duration (e.g. Merrick and Loughlin, 1997; Loughlin *et al.*, 2003). For ease of simplification, and in absence of hard data to model movement, two scenarios regarding movement are considered: $\bar{D} = 100$ and $\bar{D} = 50$ (see Section 4.1). For both of these scenarios σ_D is set equal to 40. Figure 6 shows the results of a snapshot from the model in the winter of a year prior to any impact on pregnancy rate and survival rate. Figure 6 shows histograms of the minimum depth of water in the 20" x 20" square in which each animal was located and the distance each animal was from its natal rookery. The depth (mean=131m, SD=415; mean=94m, SD=238 for $\bar{D}=100$ and $\bar{D}=50$ respectively) and the distance from natal rookery (mean=180NM, SD=115; mean=108NM, SD=63 for $\bar{D}=100$ and $\bar{D}=50$ respectively) depend on \bar{D} .

Figures 7 and 8 show the time-trajectories of pup numbers, numbers of mature females, and relative numbers on haulouts by region for the two scenarios related to the biological processes that have caused the decline and Table 9 lists the pup counts in 1950 for each scenario. Results are shown in Figures 7 and 8 when the impact on mortality and pregnancy rate is not modified (note that it changes over time given the functional forms assumed – see Equations 4-8).

One of the key assumptions underlying the analyses of this section is that the extent to which mortality is increased or pregnancy rate is decreased when an area is open to fishing equals that implied by the fits to the historical data even though fishing has been restricted around Steller sea lion rookeries already, i.e. the “open to fishing” treatment is actually “open to restricted fishing”.

6.3 Results and discussion

6.3.1 Results for a single scenario about future data

Table 10 lists the probability of selecting the model in which the trend in pup numbers differs depending on the status (open or closed) of a rookery (model 1). Results are shown for two hypotheses regarding the processes impacted and for whether the cause of the decline was due to fishing or not. The results in this table correspond to the case in which future (post-2002) pup counts have a coefficient of variation of 0.05 and a decision between models 1 and 2 is made in 2010. The results in Table 10 are based on 200 simulations (20 replicates of 10 projections of the model).

For the experiment in which rookeries are grouped by distance (experiments 1A and 1B in Table 10), the probability of selecting model 1 (different trends in the open and closed areas) is 100% irrespective of whether fishing is or is not the true cause of the decline. This result arises because the open and closed sectors within each region exhibit different trends even when fishing is not the cause of the decline. Figure 9 illustrates this by histograms over the 200 simulations for one of the scenarios in Table 10 of the difference between γ for the open and closed sectors by region. Ideally, the difference should be zero when fishing is not the cause for the decline (i.e. closing areas has no impact on the magnitude of the additional mortality) and should be negative when fishing is the cause for the decline. However, this is not the case. For example, the histograms for the Central Aleutian Islands and the Western Gulf of Alaska regions are not clustered about zero even in the case when fishing is not the cause of the decline.

This is illustrated further in Figure 10 by the time-trajectories of pup production by sector. Figure 10a corresponds to the case in which fishing does not impact the extent of additional mortality. It would therefore have been anticipated that the trends for the open (“O”) and closed (“C”) sectors would have exhibited similar trends in these panels. However, this is not the case, particularly for the Central Aleutian Islands and Western Gulf of Alaska regions (Figure 10a). Furthermore, the impact of closing a sector to fishing does not seem to be particularly large (Figure 10b), i.e. the assumed effect size for the experiment seems to be quite low. One reason for the different trends in the open and closed sectors is that the sector boundaries do not always correspond with the boundaries of the regions for experiments 1A and 1B (Table 8) so some sectors include rookeries that have historically exhibited different trends.

Figures 11 and 12 provide the same model outputs as Figure 10, except that they pertain to experiments 2A, 2B, 3A and 3B. The differences in trends in pup production between the sectors open and closed to fishing when fishing is not the cause of the decline is much smaller in Figures 11 and 12 than in Figure 10 confirming that the major differences in trend evident in Figure 10 were due to the sector boundaries not always corresponding to the boundaries of the regions. However, performance is poor even for these experiments. For example, in only a very few cases is model 1 rejected more frequently when the decline is caused by fishing than when it is not (Table 10). In fact, the wrong model was chosen more often than not. One reason for this is that the trends in pup production still differ between the sectors open and closed to fishing even when fishing is not the cause of the decline (Figures 11a and 12a) due to mixing of animals from rookeries associated with one sector into an adjacent sector (and hence experiencing the additional mortality in that sector).

One reason for the difference in the trend in pup production in the open and closed sectors of the Western Gulf of Alaska region is that the population size in the sector open to fishing is small so there is an impact of demographic uncertainty (note that the time-trajectories of pup production for sectors with ‘small’ population sizes (e.g. “Western Gulf of Alaska (O)” and “Eastern Gulf of Alaska”) exhibit much more inter-annual variability than those with “large” population sizes).

Reducing the extent of movement to $\bar{D}=15$ (Figure 13) substantially reduces the impact of animals “wandering” into adjacent regions but cannot (by definition) impact the extent of demographic stochasticity. For example, reducing the extent of movement leads to somewhat more optimistic results for the “Central Aleutian Islands (O)” sector when fishing is not the cause for the decline (compare Figures 11a and 13a). Presumably this is because the reduction in survival in the Western Aleutian region is still high during the projection years, and, when movement is higher, animals from the Central Aleutian Islands (O) sector die due to their mixing into the Western Aleutian region.

The sensitivity of the results to changing the precision of the future pup counts and the year in which a decision between models 1 and 2 is made was examined but these factors were found to be relatively inconsequential in terms of the qualitative features evident in Table 10 and Figures 9-13.

6.3.2 Discussion

The experiments considered in this study generally performed very poorly. The main reasons for the poor performance related to:

1. trajectories of future pup production in longitudinal sectors in the North Pacific will exhibit different trends irrespective of whether they are open or closed to fishing because of (a) the effects of animals moving among regions (and hence being subject to different sources of mortality), (b) experimental units including rookeries that have exhibited different trends in the past (presumably because they were subject to different risks or magnitudes of risks), and (c) demographic stochasticity.
2. The lack of an large effect size.

This last issue is examined in Figure 14 which shows time-trajectories of population size when the impact of opening an area to fishing increases the extent of additional mortality to the maximum historical level. In this case, there is an enormous effect size, from which it should be easily possible to detect different trends in different sectors (note that the scenario considered in Figure 14 is most optimistic in terms of avoiding the influences of movement and having rookeries with different historical trends within the same sector).

Although the results outlined above are not very optimistic, it should be noted that several of the assumptions underlying the evaluations should favour a successful experiment within the simulation framework of this paper. For example, the simulations are predicated on the assumption that there is only one major cause for the declines (fishing or something else). The power to select between fishing and non-fishing causes will be reduced if there are several causes for the decline and fishing is only one of them. The simulations are also predicated on the assumption that the coefficient of variation of the historical pup count data is 0.05. While this appears to be the case for the central Gulf of Alaska region, the precision of the pup counts for other regions is poorer than this (Punt and Fay, 2002). The population projections ignore environmental stochasticity. Inclusion of this source of uncertainty will add noise to the population trajectories and likely further reduce the ability of differences in trends in pup production to distinguish between causes for the decline.

Improved performance for potential experiments might be achieved by refining both the designs (e.g. by avoiding having sectors in which the population size is “small”) and the analysis methods (e.g. by replacing Equation 9 with a different model).

7. FUTURE WORK

7.1 General

There are many unknown aspects of the population dynamics of Steller sea lions in Alaska. The uncertainty associated with some of these could be reduced through the collection of additional data. In the context of this study, the factor that both substantially impacts uncertainty and should be amenable to future research is movement. The population dynamics model assumes that movement is random, subject to a depth preference and natal homing. Additional movement-related information could be used to improve this aspect of the model substantially.

Although not specifically designed for this purpose, the model could be used as the basis for conducting a population viability analysis for Steller sea lions. Use of the model for this purpose would, however, require that an attempt be made to quantify sources and magnitudes of environmental stochasticity and other factors (besides fishing) affecting Steller sea lions.

The population dynamics model could be extended to link with ecosystem and fishery models to represent some of the possible causes for the declines more explicitly.

7.2 Evaluating data collection schemes

The results in Section 5 of this report do not agree completely with those of Fay (in prep). One reason for this is that the model variant on which the alternative data collection schemes were evaluated ignored spatially structure. Therefore, the analyses did not include any potentially confounding factors such as the indices of abundance from haul-outs reflecting the abundance of animals from more than one region. Fay (in prep) has found that the trends in pup counts and those in haul-out counts are inconsistent for several of the regions defined by York *et al.* (1996). One potential reason for this conflict is the impact of movement.

7.3 Evaluating alternative experimental management schemes

The results in Section 6 do not leave very much hope that experiments are capable of resolving whether fishing or another factor is the cause of the decline in Steller sea lions, particularly given the uncertainties regarding movement. However, there remain some avenues worth investigating:

- (a) Changing the effect size (i.e. what happens if a sector is open to fishing). This will require a detailed examination of when and how the management measures designed to halt the decline in the Steller sea lion population were implemented.
- (b) The sectors could be refined to avoid having sectors with “small” populations.
- (c) Equation 9 could be replaced by a population model-based estimator.

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Table 1 : Probability of moving to a location as a function of its depth

Depth Range	< 50m	50-100m	100-200m	200-500m	>1000m
Probability	0.90	0.50	0.3	0.1	0.01

Table 2 : Expected survival probabilities for females and the probability of being mature as a function of age (source: York (1994), Table 1).

Age	Survival, S_a	Maturity, β_a
0	0.782	0
1	0.782	0
2	0.782	0
3	0.930	0.32
4	0.909	0.57
5	0.895	0.83
6+	0.851 ^a	1

a – average over age-classes 6-30+

Table 3 : The parameters that define the six scenarios.

Parameter	Mortality		Pregnancy		Both	
	Smooth	Knife-edged	Smooth	Knife-edged	Smooth	Knife-edged
Survival-related						
h_S	0.13	0.1			0.065	0.05
y_S	1983.5				1983.5	
σ_S^1	1.6				1.6	
σ_S^2	300				300	
y_S^A		1975				1975
y_S^B		1995				1995
ϕ_1	0.35				0.35	
ϕ_2	2.55				2.55	
Pregnancy-related						
h_F			0.8	0.65	0.6	0.325
y_F			1997.5		1997.5	
σ_F^1			100.6		100.6	
σ_F^2			300		300	
y_F^A				1975		1975
y_F^B				2002		2002

Table 4 : Estimated total pup numbers by region and year with their associated asymptotic standard deviations and coefficients of variation (Source: Appendix A of Punt and Fay (2002)).

Eastern Gulf of Alaska				Central Gulf of Alaska				Western Gulf of Alaska			
Year	Estimate	SD	CV (%)	Year	Estimate	SD	CV (%)	Year	Estimate	SD	CV (%)
1976	383	71	19	1976	19629	2133	11	1978	6559	993	15
1978	476	63	13	1978	18344	1433	8	1979	5814	532	9
1979	533	65	12	1979	17824	1174	7	1984	5948	705	12
1984	1063	256	24	1984	16198	1068	7	1986	5023	571	11
1989	1041	120	12	1986	12373	756	6	1990	2219	256	12
1990	991	129	13	1987	9548	466	5	1991	2109	215	10
1991	936	106	11	1988	7368	321	4	1992	2005	181	9
1993	835	69	8	1989	5686	270	5	1993	1906	153	8
1994	789	58	7	1990	4388	259	6	1994	1812	132	7
1996	703	50	7	1991	3941	205	5	1996	1638	115	7
1997	664	52	8	1992	3539	163	5	1998	1481	122	8
1998	627	57	9	1993	3179	133	4	2000	1338	141	11
2000	559	70	12	1994	2855	113	4	2001	1272	151	12
2001	528	75	14	1996	2303	98	4				
				1997	2068	98	5				
				1998	1858	100	5				
				2000	1499	103	7				
				2001	1346	104	8				

Table 5 : Comparison of the rookeries that formed the clusters used in the analyses of York *et al.* (1996) and those that form the regions for the analyses of this report.

Region	This report		York <i>et al.</i> (1996)	
	Rookeries	Area	Rookeries	Rookeries
6	Seal rocks, Wooded (Fish), Outer (Pye)	Eastern Gulf of Alaska	Not included in York <i>et al.</i> (1996)	
7	Chirikof, Chowiet, Marmot, Outer, Sugarloaf	Central Gulf of Alaska	Outer, Sugarloaf, Marmot, Chirikof, Chowiet, Atkins	
8	Atkins, Chernabura, Clubbing Rocks, Lighthouse Rocks, Pinnacle Rock	Western Gulf of Alaska	Chernabura, Pinnacle-Clubbing (before 1985)	
9	Adugak, Akun / Billing Head, Akutan / Cape Morgan, Bogoslof / Fire Island, Ogchul, Ugamak Complex	Eastern Aleutian Islands	Pinnacle-Clubbing (after 1985), Sea Lion rocks, Ugamak, Akun, Akutan	
10	Adak, Amchitka / Column Rock, Ayugadak, Gramp Rock, Kasatochi / North Point, Kiska / Cape St Stephen, Kiska / Lief Cove, Seguan / Saddleridge, Tag, Ulak / Hasgox Point, Yunaska	Central Aleutian Islands	Yunaska-Seguan, Agligadak-Kasatochi, Adak, Gramp, Tag, Ulak, Ayugak, Kiska, Buldir,	
11	Agattu, Attu / Cape Wrangell, Buldir	Western Aleutian Islands	Agattu, Attu	

Table 6 : Percentage of simulations in which each of the three models are selected for the six base-case scenarios.

True Model	Estimation Model		
	Mortality only	Pregnancy only	Mortality and Pregnancy
Smooth mortality	90	0	10
Smooth pregnancy	0	93	7
Knife-edged mortality	69	0	31
Knife-edged pregnancy	2	19	79
Smooth both	3	2	95
Knife-edged both	29	0	71

Table 7 : Percentage of simulations in which each of the three models are selected for the base-case and the sensitivity analyses.

True Model	Estimation Model		
	Mortality only	Pregnancy only	Mortality and Pregnancy
Base-case analysis			
Smooth mortality	90	0	10
Smooth pregnancy	0	93	7
Knife-edged mortality	69	0	31
Knife-edged pregnancy	2	19	79
No Age data			
Smooth mortality	92	0	8
Smooth pregnancy	0	95	5
Knife-edged mortality	78	0	22
Knife-edged pregnancy	0	17	83
No survival data			
Smooth mortality	87	0	13
Smooth pregnancy	0	92	8
Knife-edged mortality	69	0	31
Knife-edged pregnancy	3	20	77
Higher CVs			
Smooth mortality	93	0	7
Smooth pregnancy	0	94	6
Knife-edged mortality	83	0	17
Knife-edged pregnancy	2	54	44
Low population size			
Smooth mortality	74	0	26
Smooth pregnancy	0	91	9
Knife-edged mortality	35	0	65
Knife-edged pregnancy	1	12	87
Counts only			
Smooth mortality	86	0	14
Smooth pregnancy	0	92	8
Knife-edged mortality	67	0	33
Knife-edged pregnancy	1	25	74
Poor data			
Smooth mortality	88	2	10
Smooth pregnancy	0	90	10
Knife-edged mortality	63	15	22
Knife-edged pregnancy	4	63	33

Table 8 : The longitudinal sectors and whether fishing is (“Open”) or isn’t (“Closed”) allowed within each for the six alternative experiments. Experiment type A closes fishing within the entire longitudinal sector while experiment type B closes fishing within 37 km of the each of the rookeries within the sector.

Longitude range	Experiments 1A/1B	Longitude range	Experiments 2A/2B	Experiments 3A/3B
160 ⁰ E – 175 ⁰	Open	160 ⁰ E – 176 ⁰ 34E	Open	Open
175 ⁰ E – 180 ⁰	Open	176 ⁰ 34E – 178 ⁰ W	Open	Closed
180 ⁰ – 171 ⁰ W	Closed	178 ⁰ W – 169 ⁰ 28W	Closed	Open
171 ⁰ W – 166 ⁰ W	Open	169 ⁰ 28W – 167 ⁰ 30W	Open	Closed
166 ⁰ W – 163 ⁰ W	Closed	167 ⁰ 30W – 164 ⁰ 05W	Closed	Open
163 ⁰ W – 160 ⁰ W	Open	164 ⁰ 05W – 161 ⁰ W	Open	Closed
160 ⁰ W – 158 ⁰ W	Closed	161 ⁰ W – 157 ⁰ 23 ⁰ W	Closed	Open
158 ⁰ W – 153 ⁰ W	Open	157 ⁰ 23 ⁰ W – 155 ⁰ W	Open	Closed
153 ⁰ W – 150 ⁰ W	Closed	155 ⁰ W – 150 ⁰ 19W	Closed	Open
150 ⁰ W – 145 ⁰ W	Open	150 ⁰ 19W – 145 ⁰ W	Open	Open

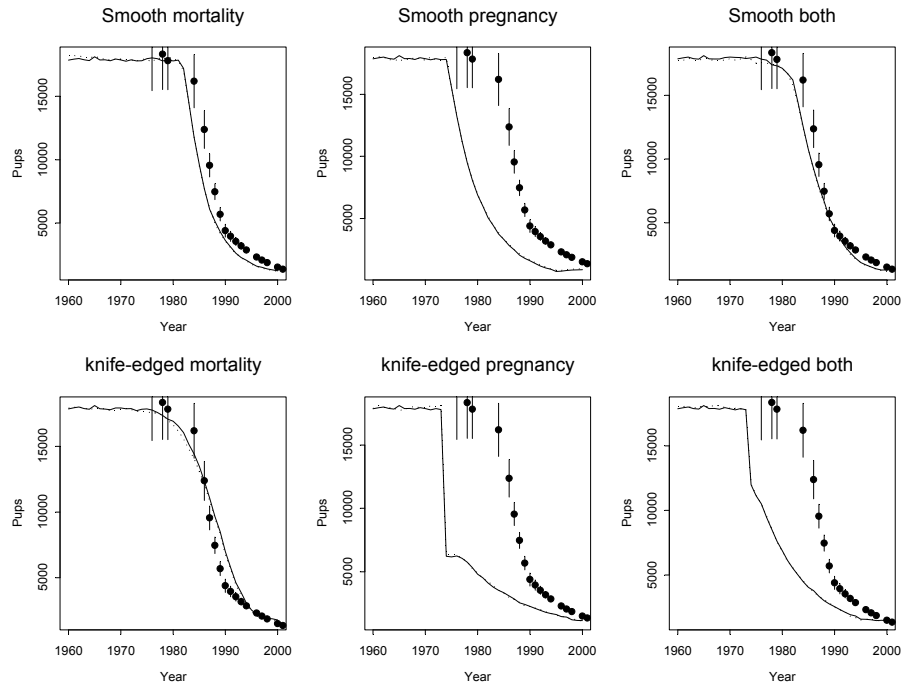
Table 9 : Pup numbers in 1950 by region, rookery and process impacted (source: Fay (in prep)).

Region	Rookery	Mortality Only	Mortality and Pregnancy	
Eastern Gulf of Alaska	Seal Rocks	216	1757	
	Wooded	480	3912	
Central Gulf of Alaska	Outer	1328	1390	
	Marmot	5977	6257	
	Sugarloaf	6065	6349	
	Chirikof	437	457	
	Chowiet	3670	3842	
Western Gulf of Alaska	Atkins	3275	3631	
	Chernabura	446	495	
	Pinnacle Rock	702	778	
	Clubbing Rocks	207	229	
	Lighthouse Rocks	811	899	
	Ugamak Complex	607	568	
	Akun / Billings Head	147	137	
Eastern Aleutian Islands	Akutan / Cape Morgan	1518	1421	
	Bogoslof / Fire Island	948	887	
	Ogchul	184	176	
	Adugak	1827	1710	
	Adak / Lake Point	542	772	
	Amchitka / Column Rock	246	350	
	Ayugadak	289	412	
	Gramp Rock	947	1349	
Central Aleutian Islands	Kasatochi / North Point	571	813	
	Kiska / Cape St Stephen	318	453	
	Kiska / Lief Cove	528	752	
	Seguam / Saddleridge	1301	1853	
	Tag	614	875	
	Ulak / Hasgox Point	1352	1925	
	Yunaska	506	721	
	Western Aleutian Islands	Buldir	1322	5177
		Agattu	532	2085
		Attu / Cape Wrangell	383	1498

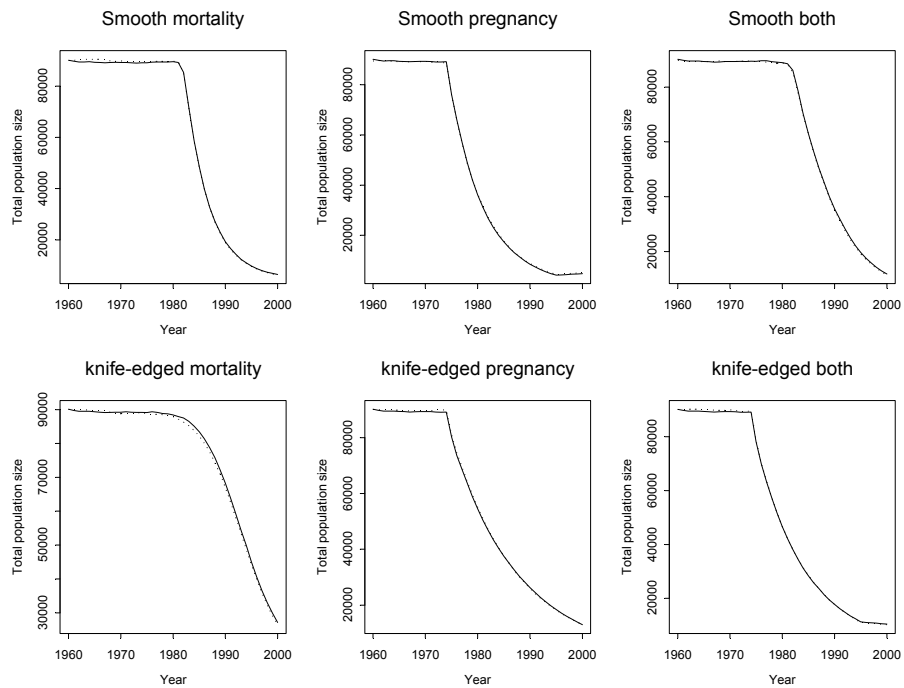
Table 10 : Probability of selecting the model in which the trend in pup numbers differs depending on the status (open or closed) of a rookery (model 1). Results are shown for two hypotheses regarding the processes impacted and for whether the cause of the decline was due to fishing or not. The results in this table correspond to the case in which future (post-2002) pup counts have a coefficient of variation of 0.05 and a decision between models 1 and 2 is made in 2010.

Process impacted	Caused by fishing	\bar{D}	Experiment Type					
			1A	1B	2A	2B	3A	3B
Mort	No	100	100	100	81	81	89	89
Mort	Yes	100	100	100	94.5	75	85.5	82.5
Mort + Preg	No	100	100	100	100	100	66	66
Mort + Preg	Yes	100	100	100	94	94	100	100
Mort	No	50	100	100	60	60	100	100
Mort	Yes	50	100	100	81.5	81.5	68.5	79.5
Mort + Preg	No	50	100	100	86	86	100	100
Mort + Preg	Yes	50	100	100	64.5	64.5	100	66

(a) Pup production



(b) Total population size



(c) Mature females

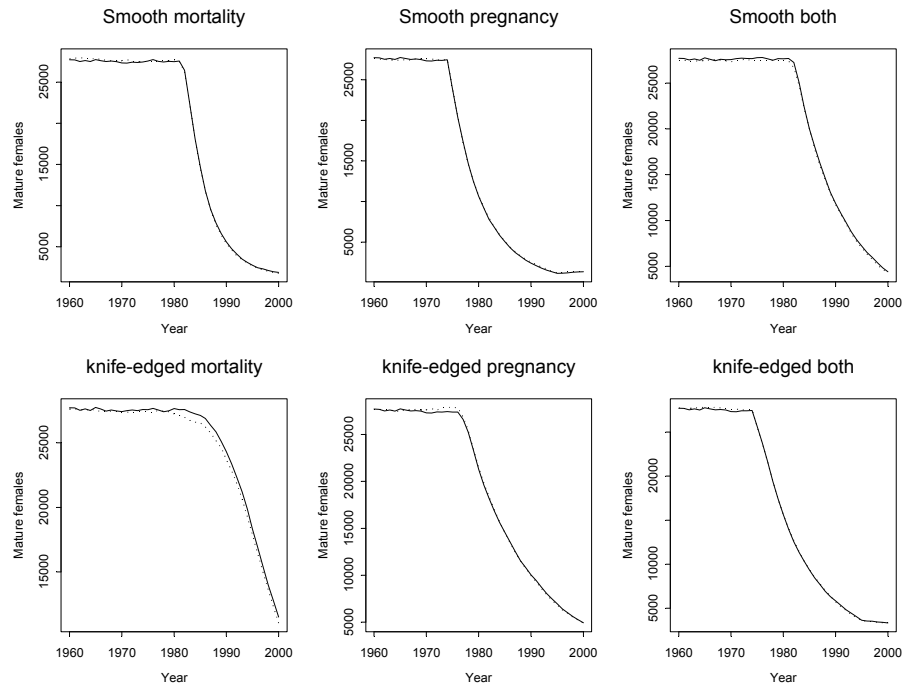
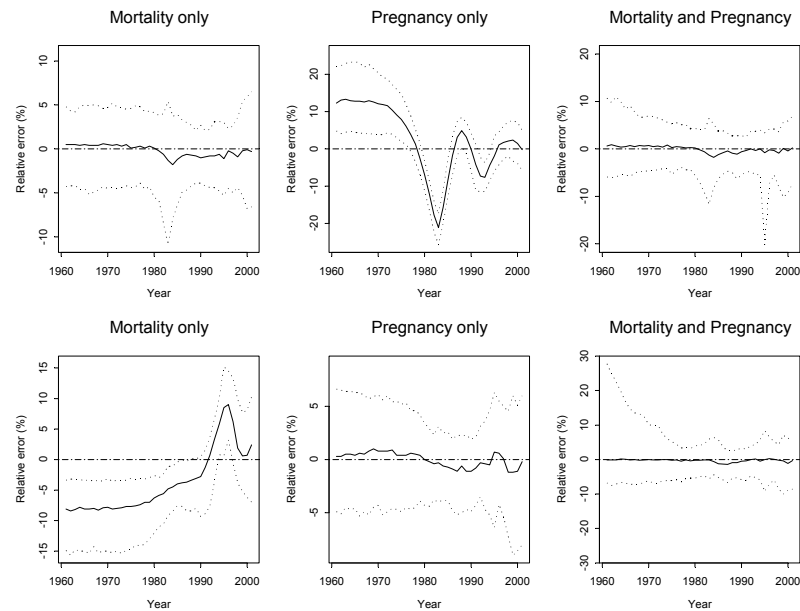


Figure 1 : Examples of two population projections for each of the six scenarios (dotted and solid lines). Results are shown for: (a) pups, (b) the total (1+) population size, and (c) the number of mature females. The actual pup count data for the Central Gulf of Alaska with 95% confidence intervals (see Appendix A of Punt and Fay (2002) and Table 4) are included in Figure 1(a).

(a) Smooth functional form



(b) Knife-edged function form

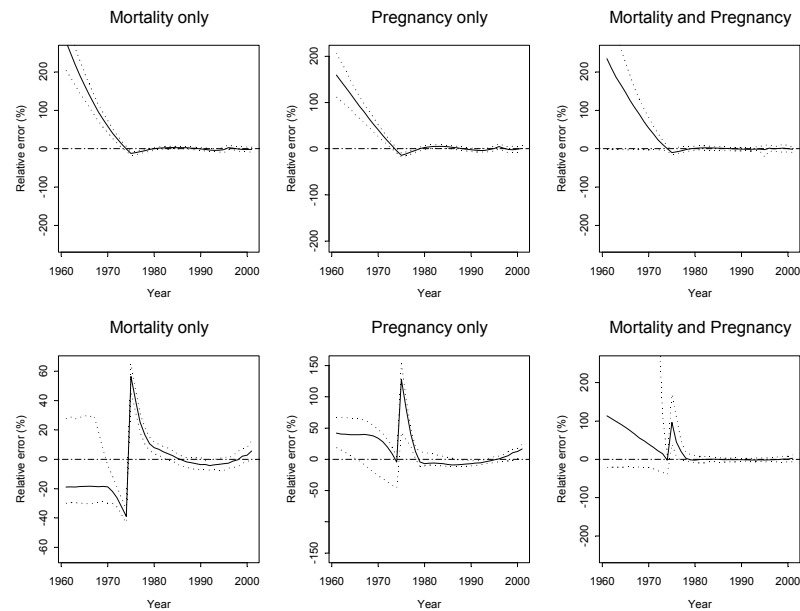


Figure 2 : Relative errors (expressed as percentages) for the number of pups for three alternative estimators. The upper panels in (a) and (b) are when the decline is caused by a change in survival rate while the lower panels in (a) and (b) are when the decline is caused by a change in pregnancy rate. The upper and lower panels in (c) pertain respectively to the smooth and knife-edged versions of the model.

(c) Impact on survival and pregnancy rate

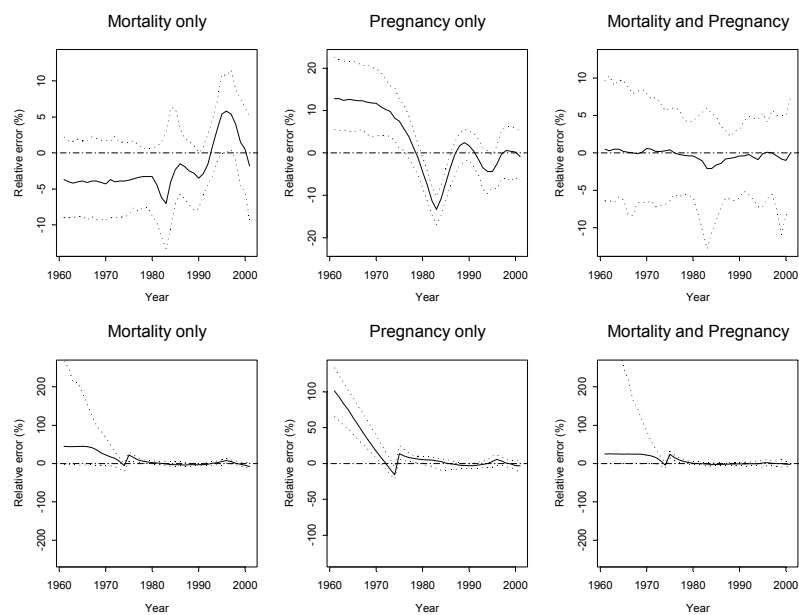
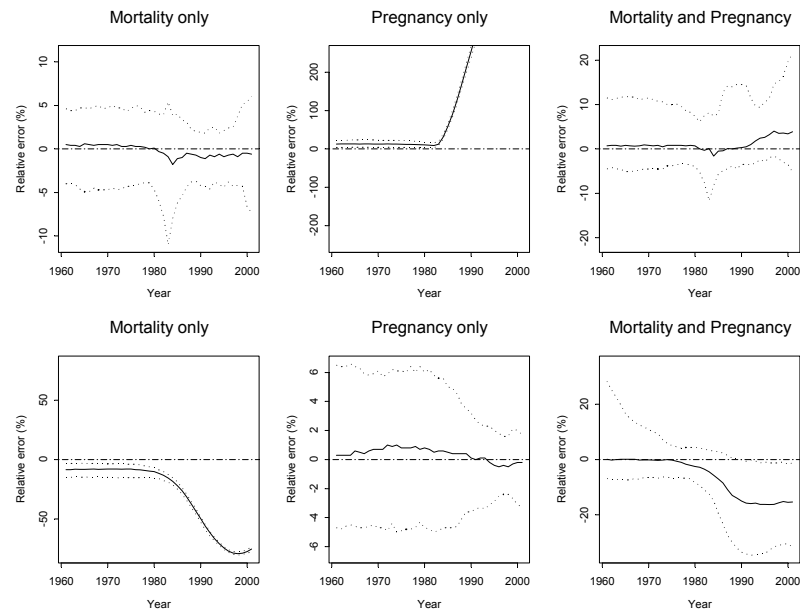


Figure 2 continued

(a) Smooth functional form



(b) Knife-edged function form

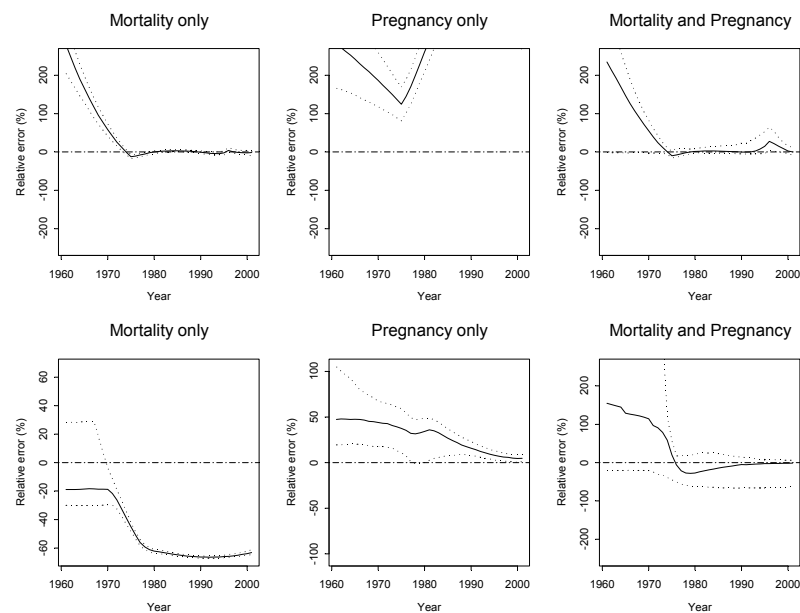


Figure 3 : Relative errors (expressed as percentages) for the number of mature females for three alternative estimators. The upper panels in (a) and (b) are when the decline is caused by a change in survival rate while the lower panels in (a) and (b) are when the decline is caused by a change in pregnancy rate. The upper and lower panels in (c) pertain respectively to the smooth and knife-edged versions of the model.

(c) Impact on survival and pregnancy rate

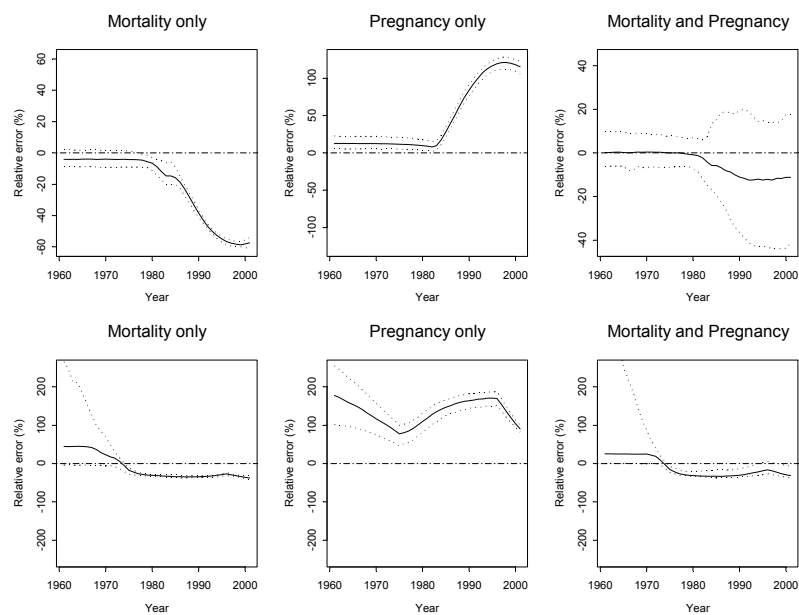


Figure 3 continued

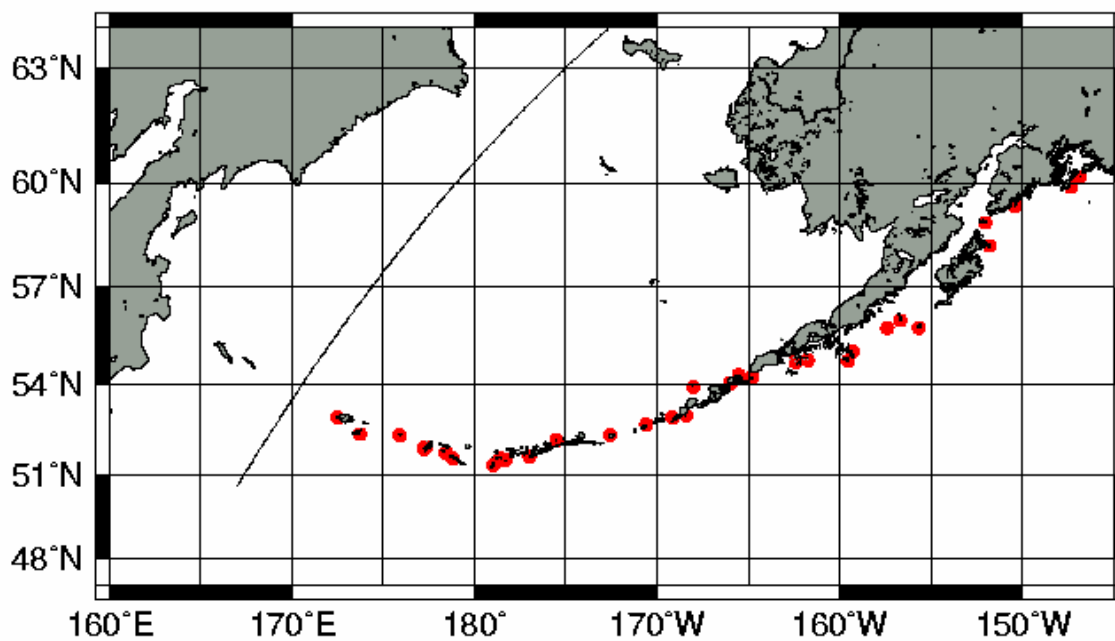


Figure 4. Map of the North Pacific Ocean indicating the 32 rookeries considered in the analyses of this report.

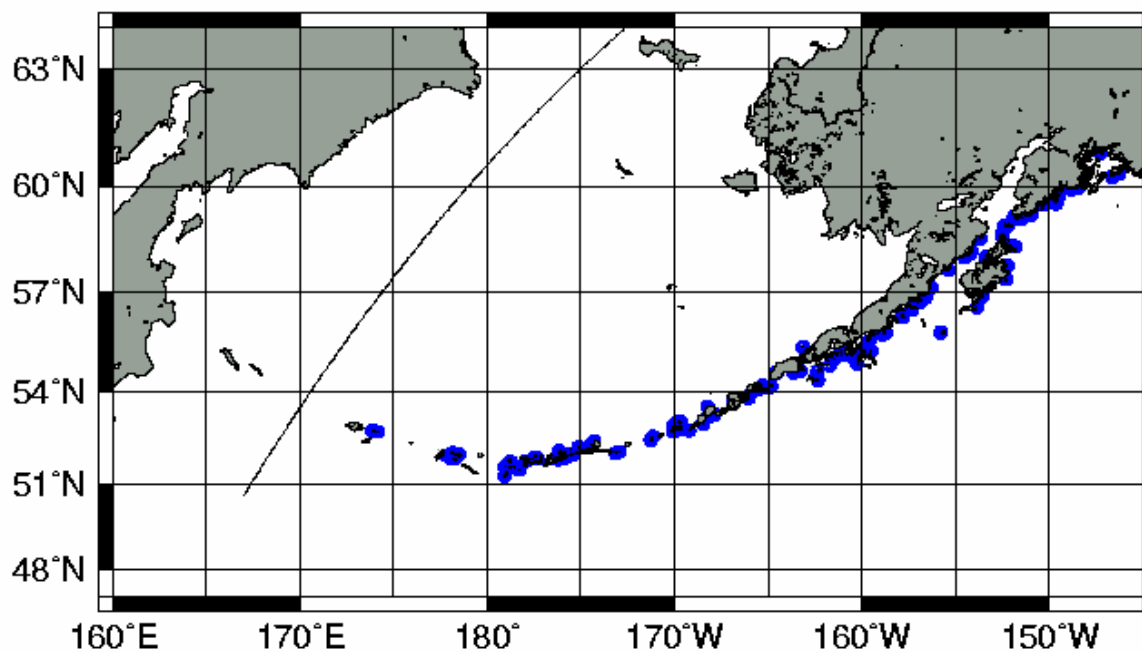


Figure 5. Map of the North Pacific Ocean indicating the 104 haulout sites considered in the analyses of this report.

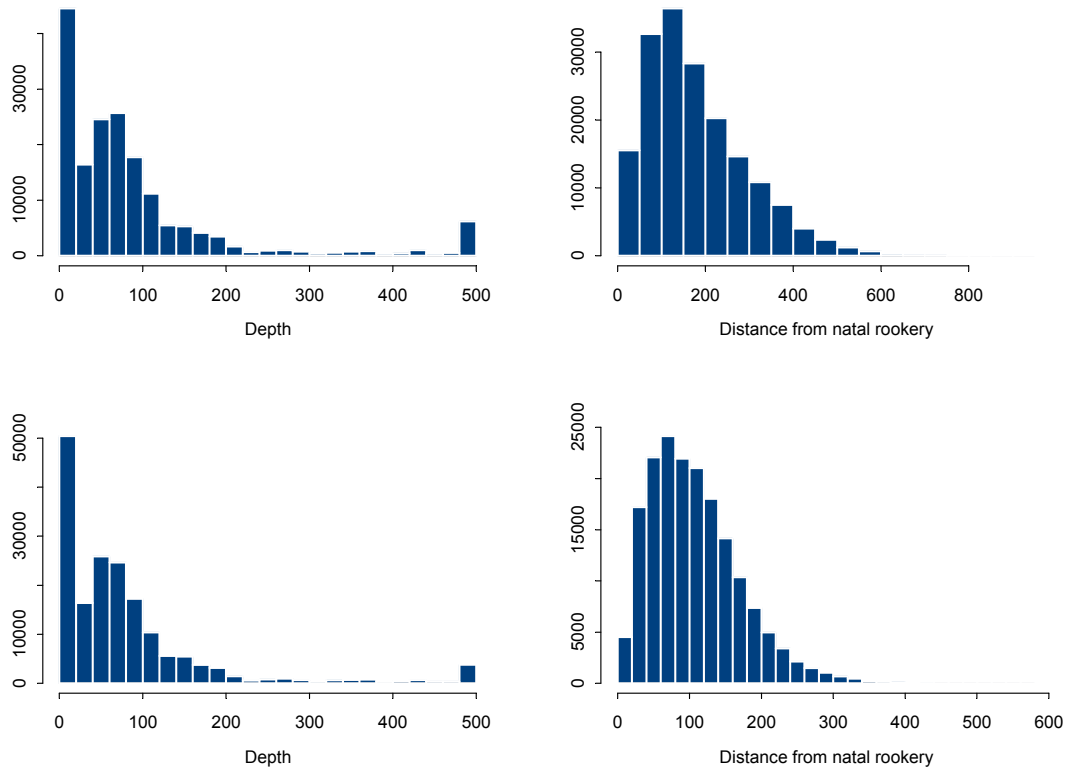


Figure 6 : Histograms of the depth of water in which Steller sea lions are located and the distance they are from their natal rookeries in winter. The upper and lower panels correspond to $\bar{D} = 100$ and $\bar{D} = 50$ respectively.

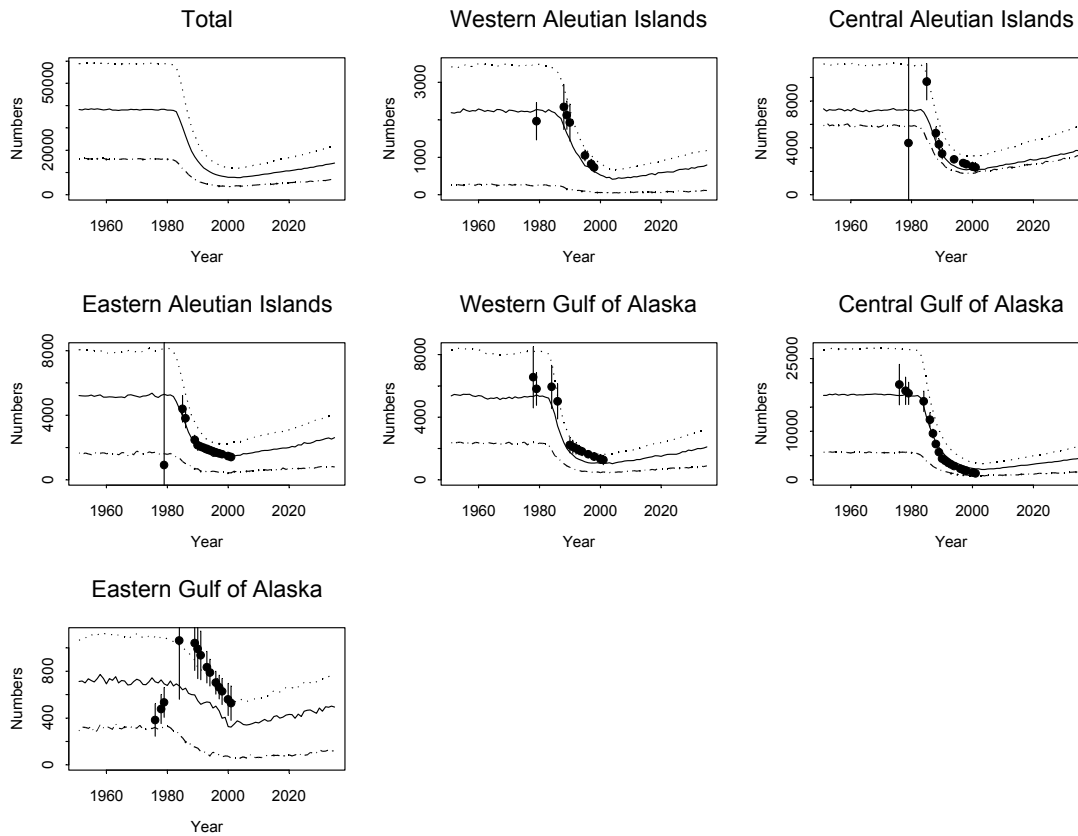


Figure 7 : Time-trajectories of pup production (solid lines), mature females (dotted lines), and relative numbers on haul-outs (dashed lines) for six regions when the decline is due to a reduction in survival rate. The solid dots are the pup counts.

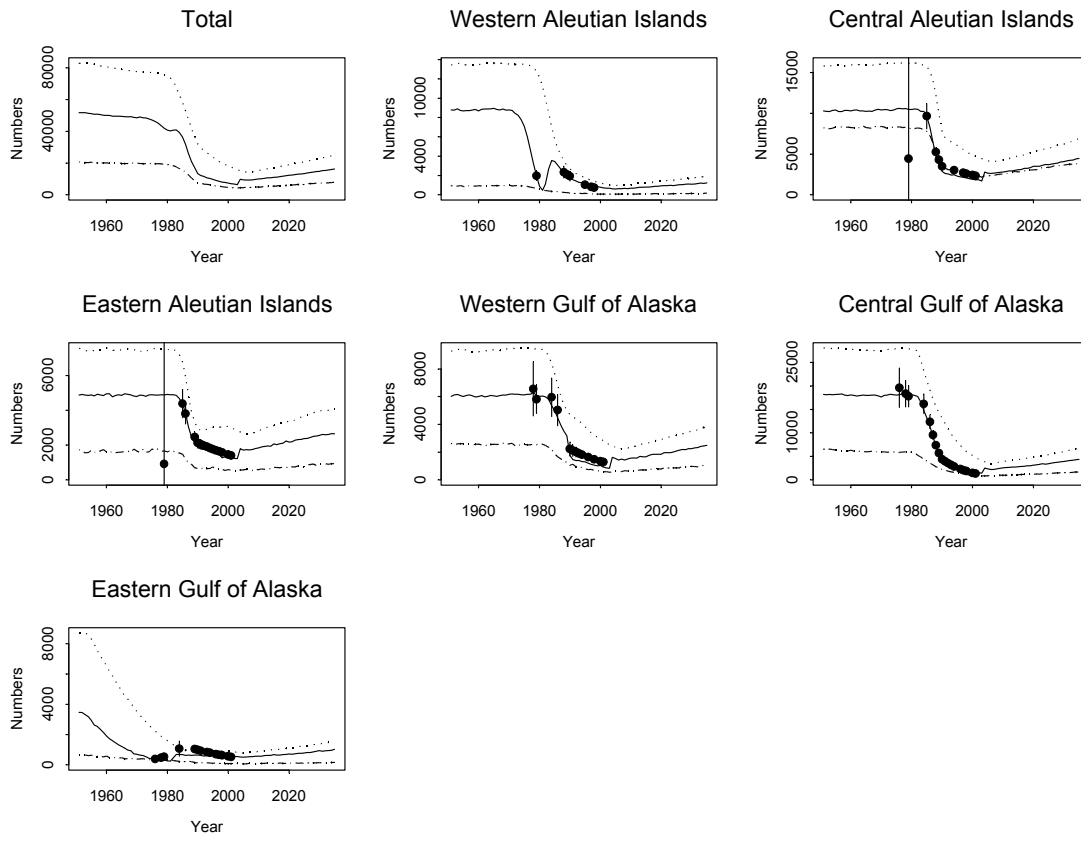
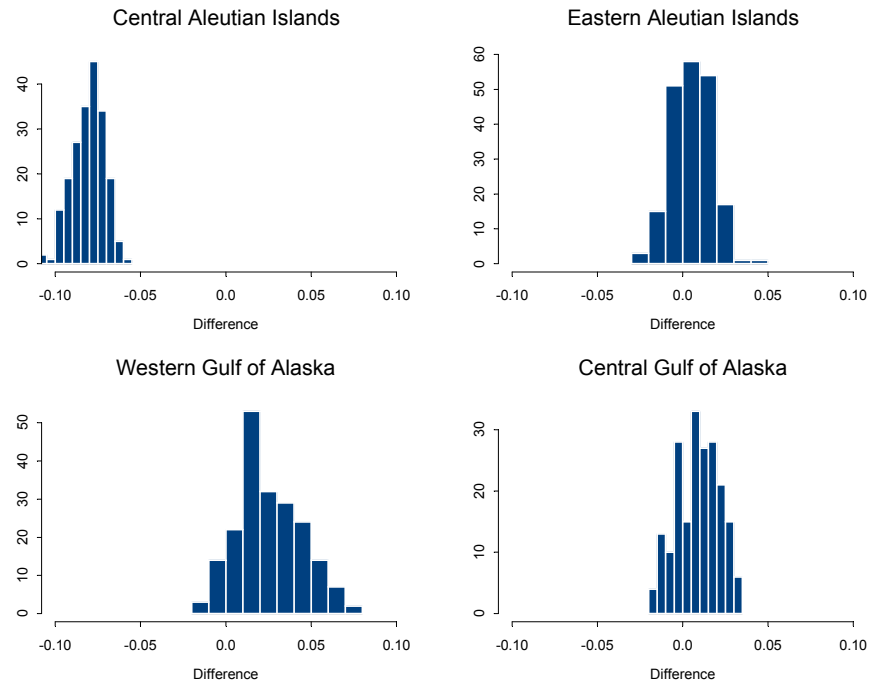


Figure 8 : Time-trajectories of pup production (solid lines), mature females (dotted lines), and relative numbers on haul-outs (dashed lines) for six regions when the decline is due to a reduction in survival rate and pregnancy rate. The solid dots are the pup counts.

(a) Fishing is not the cause of the decline



(b) Fishing is the cause of the decline

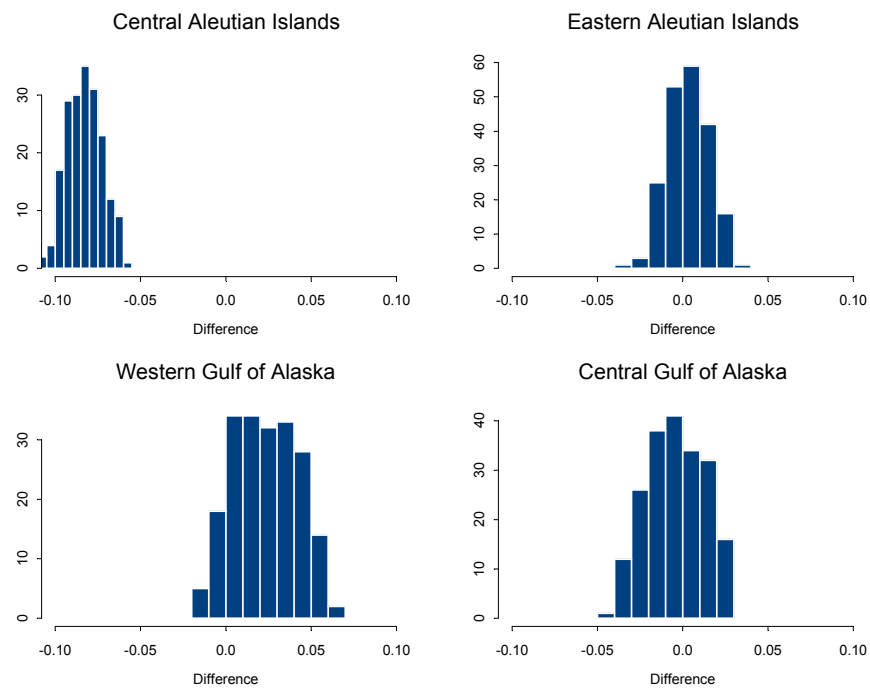
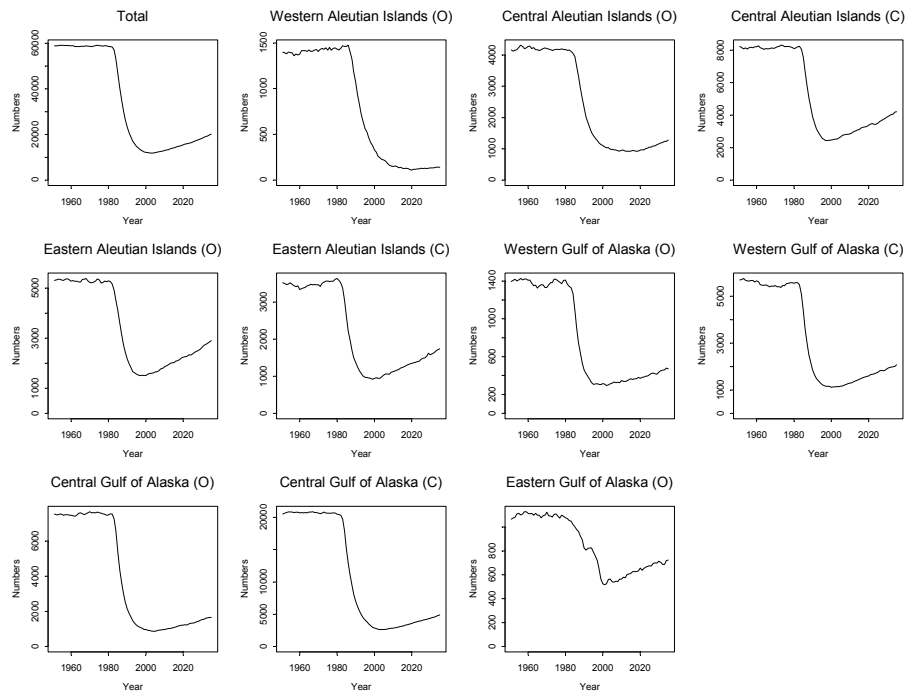


Figure 9 : Estimated differences between the open and closed sectors within each of the four regions (i.e. a negative number indicates that the pup counts in the area closed to fishing are increasing relative to those for the area open to fishing). The results in the figure are based on the scenario in which the cause of the decline is additional mortality and $\bar{D}=100$.

(a) Fishing is not the cause of the decline



(b) Fishing is the cause of the decline

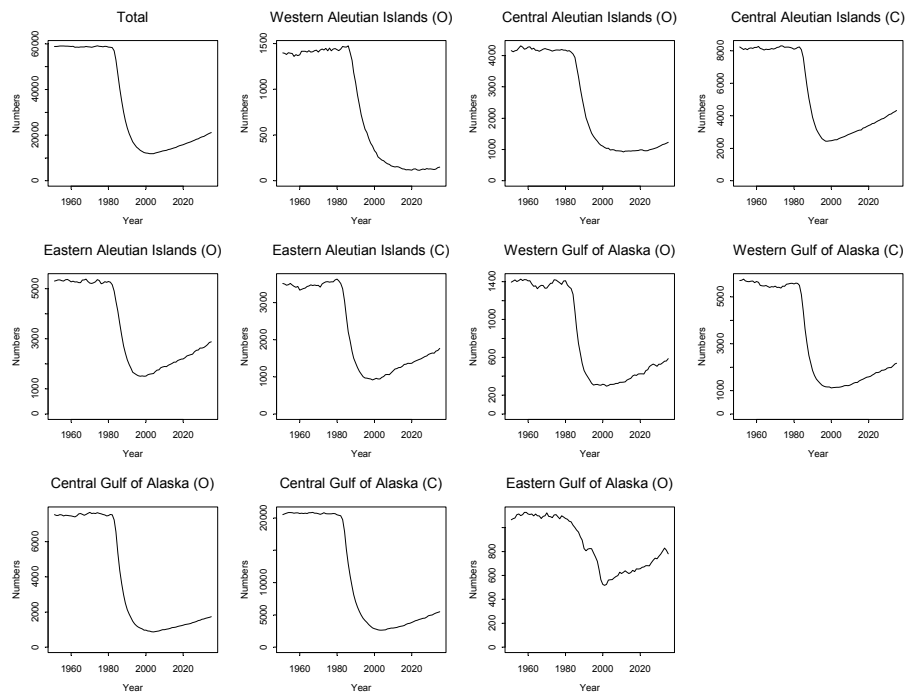
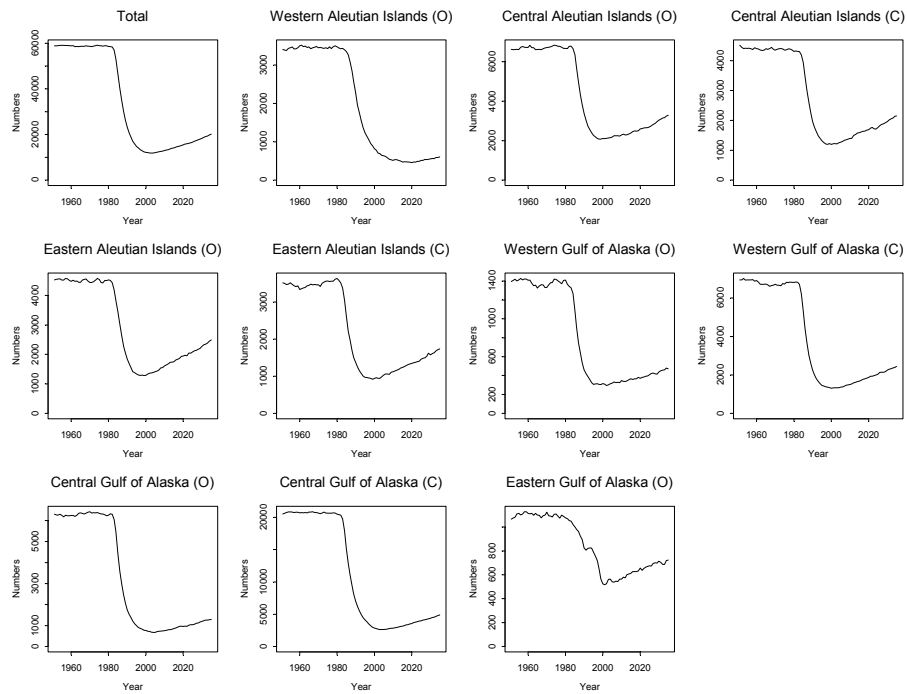


Figure 10 : Time-trajectories of pup production for the total population and for each of the sectors for experiments 1A and 1B. “O” indicates “open to fishing” and “C” indicates “closed to fishing”. The results in the figure are based on the scenario in which the cause of the decline is additional mortality and $\bar{D}=100$.

(a) Fishing is not the cause of the decline



(b) Fishing is the cause of the decline

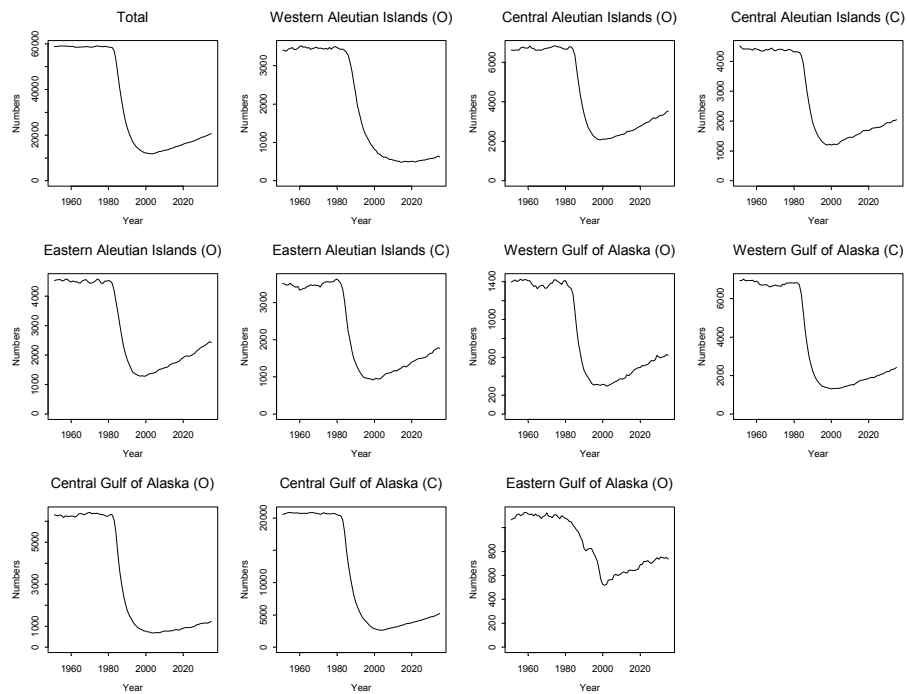
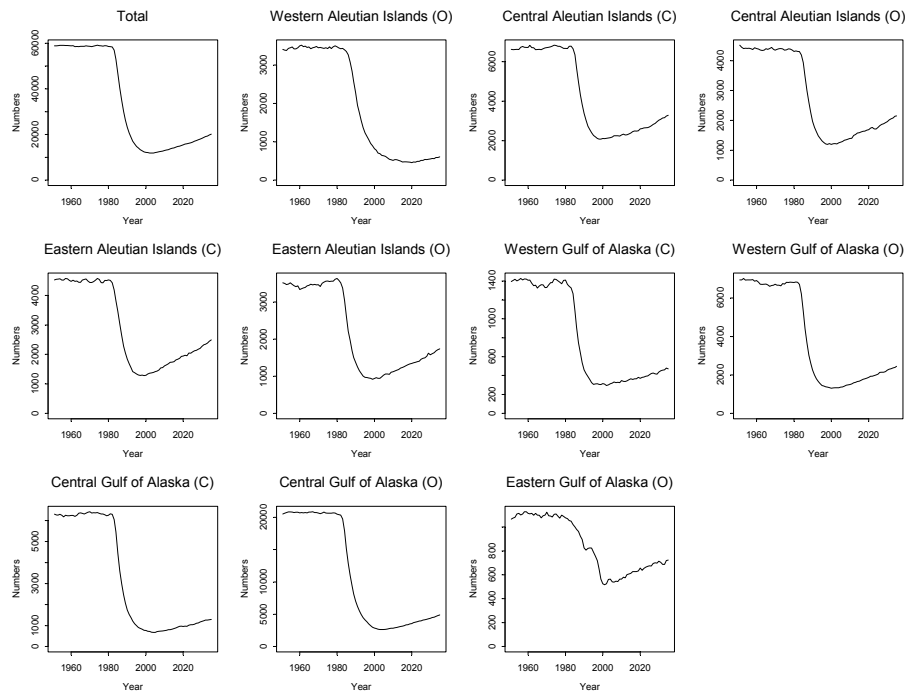


Figure 11 : Time-trajectories of pup production for the total population and for each of the sectors for experiments 2A and 2B. “O” indicates “open to fishing” and “C” indicates “closed to fishing”. The results in the figure are based on the scenario in which the cause of the decline is additional mortality and $\bar{D}=100$.

(a) Fishing is not the cause of the decline



(b) Fishing is the cause of the decline

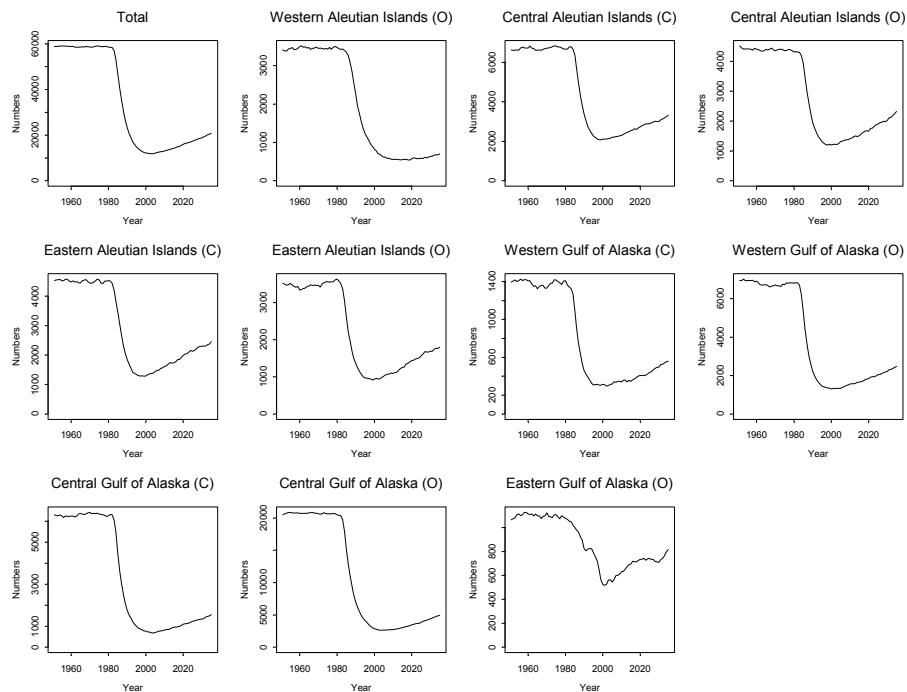
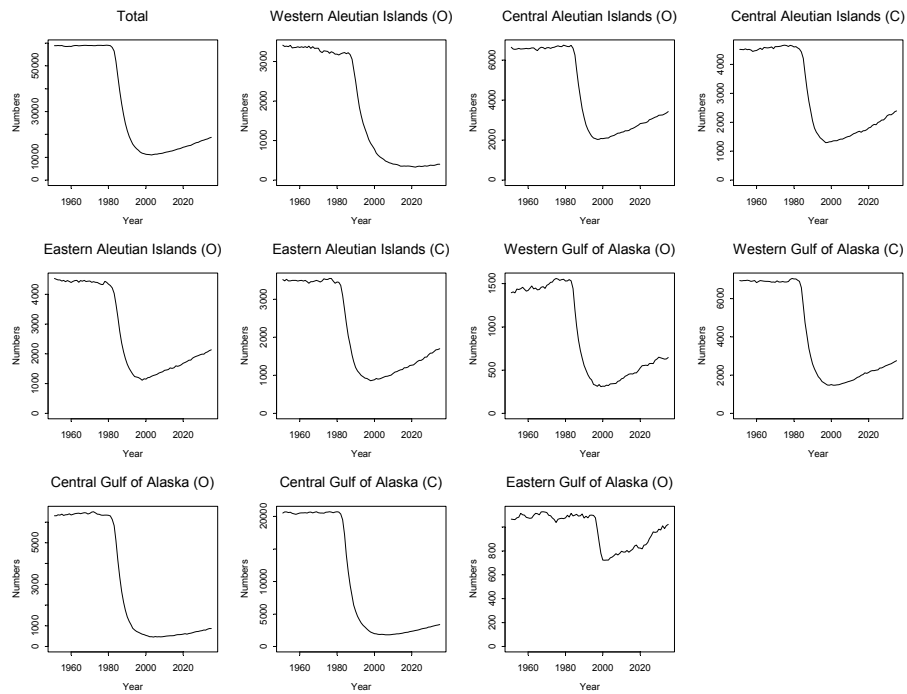


Figure 12 : Time-trajectories of pup production for the total population and for each of the sectors for experiments 3A and 3B. “O” indicates “open to fishing” and “C” indicates “closed to fishing”. The results in the figure are based on the scenario in which the cause of the decline is additional mortality and $\bar{D}=100$.

(a) Fishing is not the cause of the decline



(b) Fishing is the cause of the decline

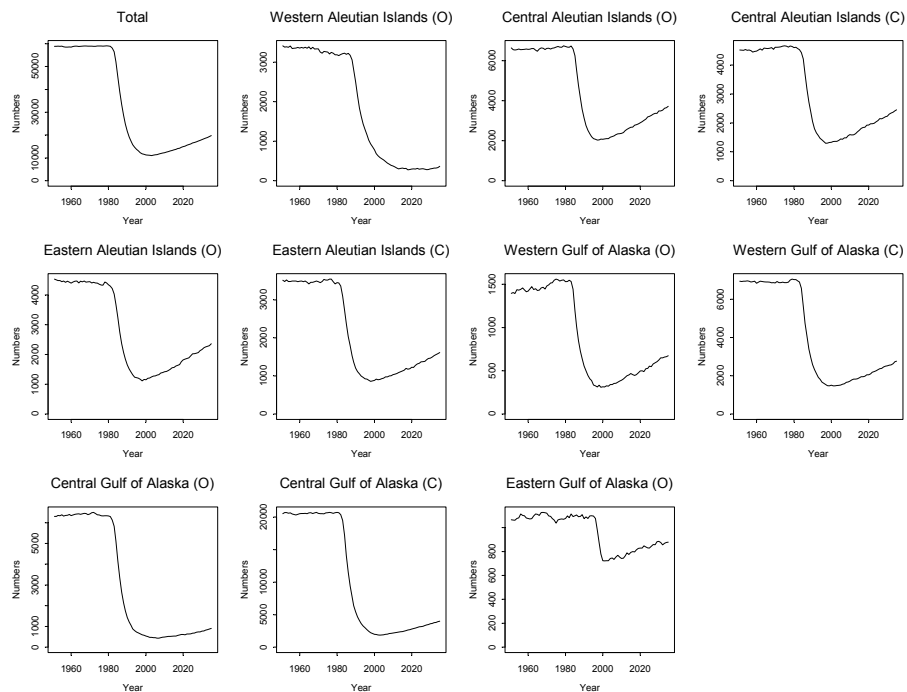
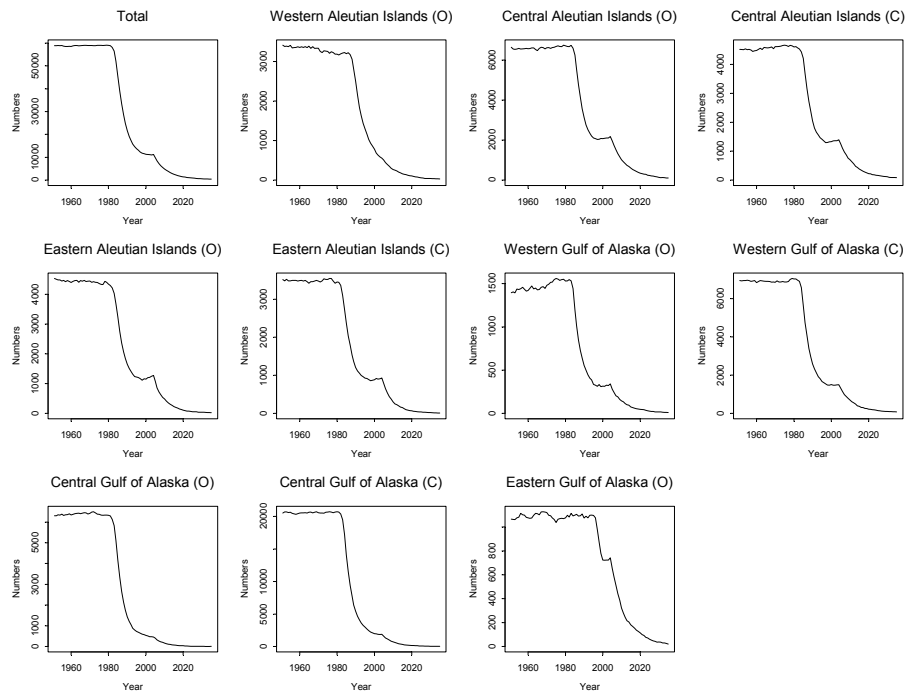


Figure 13 : Time-trajectories of pup production for the total population and for each of the sectors for experiments 2A and 2B. “O” indicates “open to fishing” and “C” indicates “closed to fishing”. The results in the figure are based on the scenario in which the cause of the decline is additional mortality and $\bar{D}=15$.

(a) Fishing is not the cause of the decline



(b) Fishing is the cause of the decline

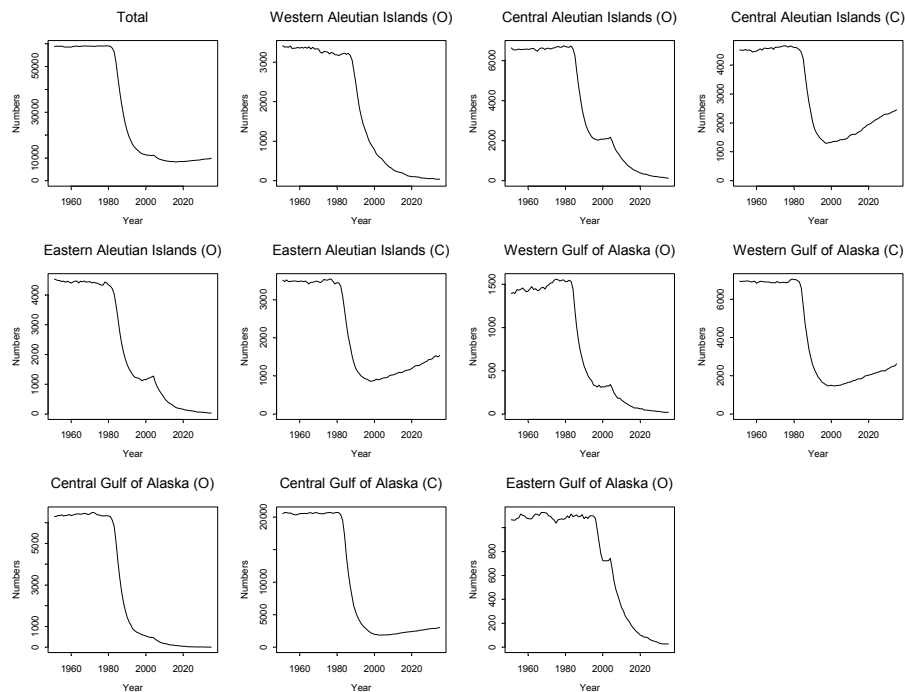


Figure 14 : Time-trajectories of pup production for the total population and for each of the sectors for experiments 2A and 2B. “O” indicates “open to fishing” and “C” indicates “closed to fishing”. The results in the figure are based on the scenario in which the cause of the decline is additional mortality and $\bar{D}=15$. Mortality in the areas open to fishing is assumed to be equal to that in the year of maximum impact.