RECOVERY PLAN FOR THE STELLER SEA LION

Eastern and Western Distinct Population Segments (*Eumetopias jubatus*)

REVISION

National Marine Fisheries Service
National Oceanic and Atmospheric Administration

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(*Eumetopias jubatus*)

REVISION

Original Version: December 1992

Prepared by

National Oceanic and Atmospheric Administration
National Marine Fisheries Services
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PREFACE

Congress passed the Endangered Species Act of 1973 (16 USC 1531 *et seq.*) (ESA) to protect species of plants and animals endangered or threatened with extinction. The National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (FWS) share responsibility for the administration of the Act. NMFS is responsible for most marine mammals including the Steller sea lion**.**

Section 4(f) of the ESA directs the responsible agency to develop and implement a Recovery Plan, unless such a plan will not promote the conservation of a species. NMFS has determined that a Recovery Plan would promote the conservation of the eastern and western distinct population segments of Steller sea lion.

NMFS completed the first recovery plan for Steller sea lions in December 1992. At that time, the entire species was listed as threatened under the ESA. Because that recovery plan became obsolete after the reclassification of Steller sea lions into two distinct population segments (DPS) in 1997, and because nearly all of the recovery actions contained in the first plan had been completed, NMFS assembled a new Steller Sea Lion Recovery Team (Team) in 2001 to assist NMFS in revising the Plan to promote the conservation of the Steller sea lion. The first draft of the revised plan was written by the Team at the request of the Assistant Administrator for Fisheries. The recovery team included: experts on marine mammals from the private sector, academia, and government; experts on endangered species conservation; and representatives of the commercial fishing industry, the Alaska Native Steller sea lion subsistence hunting community, and the environmental community.

In March 2006, the Team submitted a draft of the Recovery Plan to NMFS, at which time it became an agency document. NMFS made minor editorial changes prior to releasing the first draft for public review and comment in May 2006. Upon review of the comments and recommendations submitted by peer reviewers and the public, and in light of new information available, NMFS further revised and updated the Plan. The changes made by NMFS were reflected in the Agency's updated (May 2007 version) Draft Revised Steller Sea Lion Recovery Plan, released by NMFS for further public review and comment on May 21, 2007 (72 FR 28473), with the comment period closing on August 20, 2007.

NMFS received 8,058 letters of comment on the May 2007 draft of the revised Plan. Comments were provided by a wide range of interested parties: members of the fishing industry, non-governmental organizations (NGOs), members of academia, the public, and other interested parties. In response to two solicitations, from NMFS and the North Pacific Fishery Management Council (NPFMC), peer review comments were received from the Center for Independent Experts and from scientific experts commissioned by the North Pacific Research Board, at the request of the NPFMC. NMFS reviewed the comments and recommendations submitted by peer reviewers and the public on the 2007 version of the draft revised plan and modified the plan as appropriate to produce this Final Revised Steller Sea Lion Recovery Plan (Plan). NMFS's response to comments on the May 2007 draft of the Plan is available at http://www.fakr.noaa.gov/

NMFS believes that the goals and objectives of the Plan can be achieved only with a long-term commitment to support the actions recommended here. Achievement of these goals and objectives will require the continued cooperation of the governments of the United States, Canada, and Russia. Within the United States, the shared resources and cooperative involvement of federal, state (especially the State of Alaska) and local governments, industry, academia, non-government organizations, and individual citizens will be required throughout the recovery period.

DISCLAIMER

Recovery plans delineate such reasonable actions as may be necessary, based upon the best scientific and commercial data available, for the conservation and survival of listed species. Plans are published by the National Marine Fisheries Service (NMFS), sometimes prepared with the assistance of recovery teams, contractors, State agencies and others. Recovery plans do not necessarily represent the views, official positions or approval of any individuals or agencies involved in the plan formulation, other than NMFS. They represent the official position of NMFS only after they have been signed by the Assistant Administrator. Recovery plans are guidance and planning documents only; identification of an action to be implemented by any public or private party does not create a legal obligation beyond existing legal requirements. Nothing in this plan should be construed as a commitment or requirement that any Federal agency obligate or pay funds in any one fiscal year in excess of appropriations made by Congress for that fiscal year in contravention of the Anti-Deficiency Act, 31 U.S.C. 1341, or any other law or regulation. Approved recovery plans are subject to modification as dictated by new findings, changes in species status, and the completion of recovery actions.

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Recovery plans can be downloaded at no cost from:
http://www.nmfs.noaa.gov/pr/recovery/plans.htm

Cover photo by Lowell Fritz, National Marine Mammal Laboratory, NMFS.

ACKNOWLEDGEMENTS

The National Marine Fisheries Service gratefully acknowledges the Steller Sea Lion Recovery Team for their work in developing their draft (the first draft) of this Recovery Plan for the Steller Sea Lion. That draft was subsequently modified by NMFS, and revised again after peer review and public review to produce this final plan. The members of the Steller Sea Lion Recovery Team are given below

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ACKNOWLEDGEMENTS FROM THE STELLER SEA LION RECOVERY TEAM

The Steller Sea Lion Recovery Team acknowledged and thanked the following individuals for their expert advice, research results, and general guidance and assistance that allowed them to complete the monumental task of writing the first draft of the Steller Sea Lion Recovery Plan: Shane Capron, the Steller sea lion coordinator for NMFS, invested a tremendous amount of time and energy in guiding the Team through the recovery planning process. He also engaged the Team in critically important discussions that resulted in substantial improvements to the Plan. Don Calkins, Doug Eggers, Tom Loughlin, and Robin Samuelsen all contributed as members of the Team when our endeavor began. Al Didier demonstrated an amazing ability to accurately transcribe the important and pertinent points of the discussions at our meetings into useful meeting summaries. The following NMFS staff, especially of the Alaska Region and the Alaska Fisheries Science Center, provided unpublished data and reports, and substantial time and advice on most aspects of the Plan: Tammy Adams, Robyn Angliss, Vladimir Burkanov, Marilyn Dalheim, Robert DeLong, John Durbin, Tom Eagle, Brandee Gerke, Jim Hale, Lee Hulbert, Peggy Krahn, Marina Lindsey, Libby Loggerwell, Mark Lowry, Greg O'Corry-Crowe, Erika Green Phillips, Sharon Melin, Mike Payne, Susan Pultz, Mike Sigler, Beth Sinclair, Barbara Taylor, Johanna Vollenweider, Paul Wade, Bill Wilson, and Anne York. Several other individuals provided expert advice, including Lance Barrett-Lennard (Vancouver Aquarium), Kimberlee Beckmen (ADF&G), John Bickham (Texas A&M University), Kathy Burek (Alaska Veterinary Pathology Services), Michael Castellini (University of Alaska), Dan Hennen (Montana State University), Ken Goldman (California State University - Long Beach), Tracey Goldstein (Alaska SeaLife Center), Judy Jacobs (FWS), Lloyd Lowry (Marine Mammal Commission), Nate Mantua (University of Washington), Craig Matkin (North Gulf Oceanic Society), Lorrie Rea (ADF&G), David Rosen (University of British Columbia), and Arliss Winship (University of British Columbia). Dan Goodman (Montana State University) developed a Population Viability Analysis that required the Team to transparently integrate their knowledge. Ed Bangs (FWS), Don Siniff (University of Minnesota), Don Bowen (Bedford Institute of Oceanography), Bob Hofman (Marine Mammal Commission – retired), and Terry Quinn (University of Alaska) reviewed the Plan and provided comments that improved it. Teresa Fairchild and Sharon Perkins of the Pacific States Marine Fisheries Commission undertook the numerous tasks associated with putting on our meetings, including travel arrangements and lodging. Brock Bernstein (Consultant) facilitated the final two meetings.

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ACRONYM LIST

The following is a list of acronyms and terms used throughout the plan

ADF&G – Alaska Department of Fish and Game
AKR – Alaska Regional Office
AFSC – Alaska Fisheries Science Center
ANO – Alaskan Native Organization
BEST – Bering Ecosystem Study
BSAI – Bering Sea/Aleutian Islands
DEIS – Draft Environmental Impact Statement
DPS – Distinct population segment
Delisting – removal from the List of Endangered and Threatened Wildlife and Plants
EBS – Eastern Bering Sea
EEZ – Exclusive Economic Zone
EIS – Environmental Impact Statement
EPA – U.S. Environmental Protection Agency
ESA – Endangered Species Act
FOCI – Fisheries-Oceanography Coordinated Investigations (NOAA)
FMP – Fisheries Management Plan
FR – Federal Register
FWS – Fish and Wildlife Service
GOA – Gulf of Alaska
Team – Steller sea lion recovery team
List – Federal List of Endangered and Threatened Wildlife and Plants
MARPOL – International Convention for the Prevention of Marine Pollution
MMC – Marine Mammal Commission
MMPA – Marine Mammal Protection Act
mtDNA – Mitochondrial DNA
nm – Nautical Miles
NMML – National Marine Mammal Laboratory
NMFS – National Marine Fisheries Service
NOAA – National Oceanic and Atmospheric Administration
NRC – National Research Council
PBR – Potential Biological Removal
Plan – Steller Sea Lion Recovery Plan
PVA – Population viability analysis
TDR – Time-depth recorder
UME – Unusual mortality event
USCG – United States Coast Guard
VMS – Vessel monitoring system

EXECUTIVE SUMMARY

CURRENT SPECIES STATUS: The Steller sea lion (*Eumetopias jubatus)* was listed as a threatened species under the ESA on April 5, 1990 (55 FR 12645[1]) due to substantial declines in the western portion of the range. At the time of listing, the overall abundance of sea lions in the eastern portion of the range (in southeastern Alaska and Canada) was increasing at approximately 3% per year. Critical habitat was designated on August 27, 1993 (58 FR 45269) based on the location of terrestrial rookery and haulout sites, spatial extent of foraging trips, and availability of prey. In 1997, based on demographic and genetic dissimilarities, NMFS designated two distinct population segments (DPSs) of Steller sea lions under the ESA: a western distinct population segment (DPS) and an eastern DPS (62 FR 24345, 62 FR 30772). Due to persistent decline, the western DPS was reclassified as endangered, while the increasing eastern DPS remained classified as threatened. Through the 1990s, the western DPS continued to decline. The western population showed an increase of approximately 3% per year between 2000 and 2004. This was the first recorded increase in the population since the 1970s. However, the most recent available data from incomplete non-pup surveys in 2006 and 2007 suggest that the overall trend for the western DPS, through 2007, is either stable or slightly declining. Data indicate there are significant trend differences amongst sub-regions within the western DPS. Based on 2004-2005 data, the total population size of western Steller sea lions in Alaska is estimated to be approximately 45,000 animals. The current (as of 2005) population of Steller sea lions in Russia (part of the western DPS) is estimated to be about 16,000. The eastern DPS was estimated to number between 46,000 and 58,000 animals in 2002, and has been increasing at approximately 3% per year since the late 1970s (Pitcher *et al.* 2007).

RECOVERY PLAN: The first recovery plan for Steller sea lions was completed in December 1992 and covered the entire range of the species, which was, at that time, listed as threatened under the ESA. However, that recovery plan became obsolete after NMFS designated two distinct population segments (DPS) of Steller sea lions under the ESA in 1997. The eastern DPS was listed as threatened and the western DPS) was listed as endangered. Nearly all of the recovery actions contained in the first plan had also been completed. Therefore, in 2001, NMFS assembled a new recovery team (Team) to assist NMFS in revising the Plan. Team members represented marine mammal and fishery scientists, the fishing industry, Alaska Natives, and the environmental community. The Team completed a draft revision of the Steller Sea Lion Recovery Plan in March 2006 and submitted their draft to NMFS, at which time it became an agency document. In May 2006, NMFS released the first draft of a revised Plan for public review and comment (71 FR 29919) and extended the period of comment in July 2006 (71 FR 41206). Upon review of the comments and recommendations submitted by peer reviewers and the public, and in light of new information available, NMFS further revised and updated the Plan. The changes made by NMFS were reflected in the Agency's updated 2007 Draft Revised Steller Sea Lion Recovery Plan, released by NMFS for further public review and comment in May 2007 (72 FR 28473). NMFS reviewed the comments and recommendations submitted by peer reviewers and the public on the 2007 version of the draft revised plan and modified the plan as appropriate to produce this Final Revised Steller Sea Lion Recovery Plan. Responses to the comments are posted at: http://www.fakr.noaa.gov/.

[1] Refers to Federal Register, Volume 55, page 12645.

The Plan contains: (1) a comprehensive review of Steller sea lion status and ecology, (2) a review of previous conservation actions, (3) a threats assessment, (4) biological and recovery criteria for downlisting and delisting, (5) actions necessary for the recovery of the species, and (6) estimates of time and cost to recovery.

OVERVIEW: There appear to be two very distinct phases in the decline of the western DPS. The population declined about 70% between the late 1970s and 1990, but the initial decline likely began as early as the late 1950s in some areas. The rate of decline in the 1980s was very rapid, reaching about 15% per year during 1985-89. During this period, mortality incidental to commercial fishing was thought to contribute to perhaps as much as 25% of the observed decline. In addition, during that period it was legal for fishermen to protect their gear and catch by shooting Steller sea lions. Unfortunately, adequate records on the magnitude of such takes are not available. Some evidence indicates that animals in this population were nutritionally stressed during this time period, while other sources of mortality (e.g., predation by killer whales, mortality associated with disease) cannot be quantified due to a lack of information. There were distinct differences in the rates and pattern of decline in the six subareas used to monitor this population: the eastern Gulf of Alaska, central Gulf, western Gulf, eastern Aleutians, central Aleutians, and western Aleutians. Therefore, it is possible that several factors were important in driving the population decline during this time period.

In the 1990s, the rate of decline in the western DPS decreased from 15% to 5% per year. This decrease in the rate of decline followed further environmental changes in the 1990s and the implementation of extensive fishery regulations intended to reduce direct impacts, such as shooting, and indirect impacts, such as competition for prey. During this decade, Steller sea lions did not appear to be nutritionally stressed. The primary factors associated with the decline during this period have not been identified. As was the case in the 1980s, the pattern and rate of declines in abundance varied significantly by subregion.

Critical habitat for Steller sea lions was designated by NMFS on August 27, 1993 to respond to requirements of the Endangered Species Act (50 CFR 226.202). Steller sea lion critical habitat includes a 20 nautical mile buffer around all major haulouts and rookeries, as well as associated terrestrial, air and aquatic zones, and three large offshore foraging areas (see http://alaskafisheries.noaa.gov/protectedresources/stellers/habitat.htm).

In the late 1990s and early 2000s, NMFS reviewed and evaluated the potential impacts of federally managed groundfish fisheries in Alaska on Steller sea lions through a series of consultations under section 7 of the ESA. Two of those consultations resulted in a determination that the commercial fisheries were likely to jeopardize the continued existence of the western DPS of Steller sea lion and adversely modify its critical habitat. Therefore, as required under the ESA, additional conservation measures were implemented to avoid jeopardy and adverse modification. In 2002, NMFS implemented a set of regulations to change spatial and temporal patterns of the pollock, Pacific cod and Atka mackerel fisheries throughout the range of the western stock in U.S waters (Angliss and Outlaw, 2006) which have been amended over time (see Sea Lion Protection Measures at website: http://alaskafisheries.noaa.gov/sustainablefisheries/2003hrvstspecssl.htm). The management measures were intended to disperse fishing over time and area to protect against potential competition for important Steller sea lion prey species near rookeries and important haulouts. These measures were expected to promote the recovery of Steller sea lions in areas where potential competition from commercial fisheries may have contributed to the population decline.

It is plausible that the conservation measures implemented since 1990 are positively affecting the recovery of the western DPS. Between 2000 and 2004, survey data suggested that the estimated overall abundance of the western DPS of Steller sea lions increased for the first time in decades. However, an increasing trend was not detected in all subregions, and incomplete data from 2006 and 2007 indicate the population overall is either stable or declining slightly. It is not known whether the slow down in decline, the period of increase, and the current stability or near stability is a result of management actions, natural changes in the ecosystem, or other factors.

COMPLETED RECOVERY ACTIONS: The 1992 recovery plan included 61 discrete recovery actions (or tasks) with estimated costs and responsible parties associated with those tasks. In our review, we determined that each of the 61 tasks has been accomplished to a substantial degree with one exception -- the development of international conservation agreements. Much of the effort was focused on eliminating the most direct and certain causes of decline (e.g., shooting, incidental take). These efforts are detailed in the Plan, and include the following:

- substantial reduction in disturbance of important rookeries and haulouts;
- substantial reduction in the incidental catch of Steller sea lions in commercial fishing operations, particularly the groundfish trawl fishery;
- significant efforts to reduce intentional take by prohibiting shooting at or near Steller sea lions
- intensive research to better describe the threats to Steller sea lions and provide management with options for recovery actions;
- potential reduction in the competitive interactions between Steller sea lions and commercial fisheries for pollock, Atka mackerel, and Pacific cod in Alaska;
- acquired additional information on the status, foraging ecology, and survivorship of Steller sea lions.

THREATS TO THE RECOVERY OF STELLER SEA LIONS: The extensive research program has increased the understanding of the relative impacts of threats that potentially impede the recovery of Steller sea lions. For the western DPS, the threats assessment concludes that the following threats are relatively minor: (1) Alaska Native subsistence harvest, (2) illegal shooting, (3) entanglement in marine debris, (4) disease, and (5) disturbance from vessel traffic and scientific research. Although much has been learned about Steller sea lions and the North Pacific ecosystem, considerable uncertainty remains about the magnitude and likelihood of the following potential threats to the recovery of the western DPS (relative impacts in parenthesis): competition with fisheries (potentially high), environmental variability (potentially high), incidental take by fisheries (low), toxic substances (medium) and predation by killer whales (potentially high). Uncertainty, controversy, and disagreement within the scientific and stakeholder communities with regards to the potential threat posed by killer whale predation is especially great, with conclusions about the magnitude of that threat being fairly polarized (low vs. high). However, due to the uncertainty and the need to be precautionary in our assessment of possible threats to the recovery of this endangered DPS, NMFS has categorized the relative potential impact of this threat as "potentially high", and we have expanded our presentation and critical evaluation of the major studies and viewpoints of this threat in the Plan.

In contrast, no threats to continued recovery were identified for the eastern DPS. Although several factors affecting the western DPS also affect the eastern DPS (e.g., environmental variability, killer whale predation, toxic substances, disturbance, shooting), these threats do not appear to be at a level sufficient to keep this population from continuing to recover, given the long term sustained growth of the population as a whole. However, concerns exist regarding global climate change and the potential for the southern part of the range (i.e., California) to be adversely affected. Future monitoring should target this southern portion of the range.

RECOVERY GOAL: The goal of this recovery plan is to restore endangered and threatened Steller sea lion populations to the point at which they are again secure, self-sustaining members of their ecosystems, allowing initially for reclassification of the western DPS to threatened status and, ultimately, removal from the List of Endangered and Threatened Wildlife (List). The eastern DPS has been recovering for since the late 1970s and should be considered for removal from the List.

RECOVERY CRITERIA:

The western DPS of Steller sea lions will be considered for reclassification to "threatened" when all of the following conditions are met:

1. The population for the U.S. region has increased (statistically significant) for 15 years on average, based on counts of non-pups (i.e., juveniles and adults). Based on an estimated population size of roughly 42,500 animals in 2000 and assuming a consistent but slow (e.g. 1.5%) increasing trend, this would represent approximately 53,100 animals in 2015.
2. The trends in non-pups in at least 5 of the 7 sub-regions are consistent with the trend observed under criterion #1. The population trend in any two adjacent sub-regions cannot be declining significantly. The 7 sub-regions are:
    a. Eastern Gulf of Alaska (US)
    b. Central Gulf of Alaska (US)
    c. Western Gulf of Alaska (US)
    d. Eastern Aleutian Islands (including the eastern Bering Sea) (US)
    e. Central Aleutian Islands (US)
    f. Western Aleutian Islands (US)
    g. Russia/Asia

3. The ESA listing factor criteria are met.

The western DPS of Steller sea lions will be considered for delisting if all the following conditions are met:

1. The population for the U.S. region of this DPS has increased (statistically significant) for 30 years (at an average annual growth rate of 3%), based on counts of non-pups (i.e., juveniles and adults). Based on an estimated population size of about 42,500 animals in 2000, this would represent approximately 103,000 animals in 2030.

2. The trends in non-pups in at least 5 of the 7 sub-regions are stable or increasing, consistent with the trend observed under criterion #1. The population trend in any two adjacent sub-regions can not be declining significantly. The population trend in any sub-region cannot have declined by more than 50%. The 7 sub-regions are:
    a. Eastern Gulf of Alaska (US)
    b. Central Gulf of Alaska (US)
    c. Western Gulf of Alaska (US)
    d. Eastern Aleutian Islands (including the eastern Bering Sea) (US)
    e. Central Aleutian Islands (US)
    f. Western Aleutian Islands (US)
    g. Russia/Asia

3. The ESA listing factor criteria are met.

The eastern DPS of Steller sea lion will be considered for delisting if all the following conditions are met:

1. The population has increased at an average annual growth rate of 3% per year for 30 years.

2. The ESA listing factor criteria are met.

ACTIONS NEEDED: The Plan identifies 78 substantive actions needed to achieve recovery of the western DPS by addressing the broad range of threats. These actions are aimed at addressing three main objectives: (1) the collection of information on status and vital rates, (2) research programs to collect information on the remaining threats to recovery, including natural and anthropogenic factors, and (3) the implementation of conservation measures to remove impacts of anthropogenic threats to recovery. The Plan highlights four actions (below) that are especially important to the recovery program for the western DPS:

Continue population monitoring and research on the key threats potentially impeding sea lion recovery (Action 1.1.1 and others)
Estimates of population abundance, trend, distribution, health, and essential habitat characteristics are fundamental to Steller sea lion management and recovery. Further, current information on the primary threats is insufficient to assess their impact on recovery. Focused research is needed on how these threats impact sea lion population growth and how they may be mitigated in order to facilitate recovery. In addition to studies on individual threats, the

dynamics between threats needs to be better understood to assess the cumulative effects on sea lions.

Maintain current or equivalent level of fishery conservation measures (Action 2.6.6)
After a long term decline, the western DPS may be stabilizing. The first slowing of the decline began in the 1990s, suggesting that the management measures implemented in the early 1990s may have been effective in reducing some anthropogenic effects (e.g., shooting, harassment, and incidental take). The apparent relative population stability observed in the last 6 years is correlated with comprehensive fishery management measures implemented since the late 1990s. The current suite of management actions (or their equivalent protection) should be maintained until substantive evidence demonstrates that these measures can be reduced without limiting recovery.

Design and implement an adaptive management program to evaluate fishery conservation measures (Action 2.6.8)
Due to the uncertainty as to how fisheries affect Steller sea lions and their habitat, and the difficulty in extrapolating from individual scientific experiments, a properly designed adaptive management program should be implemented. This type of program has the potential to assess the relative impact of commercial fisheries and to better distinguish the impacts of other threats (including killer whale predation). This program will require a robust experimental design with replication at the proper temporal and spatial scales with the appropriate levels of commercial fishing as experimental treatments. It will be a challenge to construct an adaptive management plan that meets the requirements of the ESA, is statistically sufficient, and can be implemented by the commercial fisheries. Acknowledging these hurdles, a significant effort must be made to determine the feasibility of such a program.

Develop an implementation plan (Action 1.5)
An implementation plan will be developed that includes a comprehensive ecological and conceptual framework that integrates and further prioritizes the numerous recovery actions provided in this plan. The implementation plan will contain a synthesis of, and establish priorities among, the individual actions, as well as coordinate their implementation in a cohesive strategy. Several components will be integrated in the conceptual framework of the implementation plan: (1) the complex dynamics of the North Pacific marine ecosystem, (2) multiple causation in those systems, (3) the need for long-term research, (4) the monitoring required to assess the effectiveness of management regulations, and (5) the development of a modeling approach that examines possible effects of multiple threats on sea lion population dynamics to evaluate the strength of the evidence for different hypotheses.

PERIODIC REVIEW OF THE PLAN: NMFS (2007) recognizes that recovery planning is an iterative process. Data generated through careful monitoring and other research should feed back into refinements of recovery plans and actions. It is a goal of the NMFS to review recovery plans and the status of listed species every five years.

TOTAL ESTIMATED COST OF RECOVERY:

Western DPS: $93,840,000 for the first five fiscal years; $430,425,000 to full recovery assuming 30 years for recovery starting in 2000 and using Year 5 costs in this Plan as the cost for all future years

Eastern DPS: $ 150,000 for the first year; $1,050,000 total, including 10 years post-delisting monitoring

ANTICIPATED DATE OF RECOVERY: The time to recovery for the western DPS will be dependent upon population trajectories over time. While increasing trend counts were observed between 2000 and 2004, data from incomplete counts in 2006 and 2007 suggest that the population is stable or declining slightly. However, if one assumes that the population can and will achieve a modest, but steady, rate of increase, such as the 3% annual increase observed for the eastern DPS, it would be eligible for consideration for downlisting to threatened status within roughly seven years (i.e., by about 2015). If that trend continues further, as has been the case for the eastern DPS, then consideration for delisting is possible shortly after 2030. As more information is obtained on the threats, their impact on sea lions, and how they can be effectively mitigated, more robust projections will be developed about the time to recovery, and its expense.

The eastern DPS appears to have recovered from predator control programs in the 20th century which extirpated animals at rookeries and haulouts. Currently, no substantial threats are evident, and the population continues to increase at approximately 3% per year. The primary action in the plan is to initiate a status review for the eastern DPS and consider removing it from the federal List of Endangered Wildlife and Plants.

I. BACKGROUND

In the 1950s, the worldwide abundance of Steller sea lions (*Eumetopias jubatus*) was estimated at 240,000 to 300,000 animals, with a range which stretched across the Pacific rim from southern California, Canada, Alaska, and into Russia and northern Japan (Figure I-1). By 1990, the U.S. portion of the population had declined by about 80%, which prompted NMFS to list the Steller sea lion as threatened under the ESA. The listing was based primarily on substantial declines that occurred in the 1980s (as high as 15% per year) in the population currently designated as the western distinct population segment (DPS) as well as on a reduced population size in the population now designated as the eastern DPS. After listing in 1990, the rate of decline decreased to about 5% per year.

In 1997, after continued declines in Alaska and the availability of new genetics information that revealed further population structure, NMFS designated two distinct population segments with different statuses under the ESA (Figure I-1). The western DPS, extending from Japan around the Pacific rim to Cape Suckling in Alaska (144°W), was up-listed to endangered due to its continuous decline and lack of recovery. The eastern DPS, extending from Cape Suckling east to British Columbia and south to California, remained on the list as threatened because of concern over western DPS animals ranging into the east, human interactions, and the lack of recovery in California.

The decline continued in the western DPS until about 2000. Data suggest that between 2000 and 2004 the population increased at about 3% per year and was relatively consistent across the U.S. portion of the range with the exception of the central Gulf of Alaska and the western Aleutian Islands areas. Data from incomplete non-pup surveys in 2006 and 2007 indicate that the size of the non-pup portion of the population remained largely unchanged through much of the range (Cape St. Elias to Tanaga Island) from 2004 to 2007. However, recent trends in the western central Aleutians and western Aleutians (through 2004 and 2006) have been negative, suggesting an overall stable or slight negative trend for the population as a whole (Fritz *et al.* 2007). The Russian component of the western DPS has been relatively stable overall, but with regional differences. The eastern DPS has been increasing since the late 1970s (Pitcher *et al.* 2007) with the greatest increases in southeast Alaska and British Columbia, but generally poor performance in California at the southernmost extent of its range.

A. Species Description

Sea lions belong to the Order Carnivora, Suborder Pinnipedia, Family Otariidae, and Subfamily Otariinae. The family contains the extant genera *Arctocephalus*, *Callorhinus*, *Eumetopias*, *Neophoca*, *Otaria*, *Phocarctos*, and *Zalophus*. The genus *Eumetopias* contains one species, the Steller (also called northern) sea lion, *E. jubatus*. Unless noted otherwise, all references to sea lions in this document are to Steller sea lions.

Steller sea lions are the largest otariid and show marked sexual dimorphism with males larger than females. The average standard length is 282 cm for adult males and 228 cm for adult females (maximum of about 325 cm and 290 cm, respectively); weight of males averages 566 kg and females 263 kg (maximum of about 1,120 kg and 350 kg) (Fiscus 1961, Calkins and Pitcher 1982, Loughlin and Nelson 1986, Winship *et al*. 2001). The pelage is light buff to reddish brown and slightly darker on the chest and abdomen. Naked parts of the skin are black (King 1954). Adult males have long, coarse hair on the chest, shoulders, and back; the chest and neck are massive and muscular. Newborn pups are about 1 m long, weigh 16-23 kg, and have a thick, dark-brown coat that molts to lighter brown after 6 months (Daniel 2003). A more detailed physical description is given in Loughlin *et al*. (1987) and Hoover (1988).

Female Steller sea lions attain sexual maturity and first breed between 3 and 8 years of age (Pitcher and Calkins 1981). The average age of reproducing females (i.e., generation time) is about 10 years based on the life tables from Calkins and Pitcher (1982) and York (1994). They normally ovulate and breed annually after maturity although because of a high rate of reproductive failures, estimated birth rates have ranged from 55% to 63% (Calkins and Goodwin 1988, Pitcher and Calkins 1981). They give birth to a single pup from late May through early July and then breed about 11 days after giving birth. They undergo delayed implantation and the blastocyst implants about 3.5 months after breeding. Some offspring are weaned near their first birthday while others continue suckling for an additional year or more. While males may attain physiological maturity before 7 years of age, they are seldom able to establish and defend a territory until 8 years or older (Thorsteinson and Lensink 1962, Pitcher and Calkins 1981).

B. Distribution and Population Structure

The present range of Steller sea lions (Figure I-1) extends around the North Pacific Ocean rim from northern Japan, the Kuril Islands and Okhotsk Sea, through the Aleutian Islands and Bering Sea, along Alaska's southern coast, and south to California (Kenyon and Rice 1961, Loughlin *et al*. 1984, 1992, Burkanov and Loughlin 2005). Seal Rocks, at the entrance to Prince William Sound, Alaska, is the northernmost rookery (60°09'N). Año Nuevo Island off central California is the southernmost rookery (37°06'N), although some pups were born at San Miguel Island (34°05'N) up until 1981. At present, the only active rookeries along the Asian coast are in Russia (Figure 2 of Burkanov and Loughlin 2005). Prior to the decline in the west, most large rookeries were in the Gulf of Alaska and Aleutian Islands (Kenyon and Rice 1961, Calkins and Pitcher 1982, Loughlin *et al*. 1984, 1992, Merrick *et al*. 1987). As the decline continued, rookeries in the west became progressively smaller; consequently, the largest rookeries are now in Southeast Alaska and British Columbia. In 2005, the Forrester Island complex produced 3,429 pups and Hazy Islands 1,286 pups (both in Southeast Alaska). About 2,500 pups were counted at the Scott Islands rookery in British Columbia in 2002. Pitcher et al. (2007) reported that about 55% of the North American pup production of Steller sea lions currently occurs in the eastern population. In 2005, Ugamak Island (687 pups) and Pinnacle Rock (643 pups) were the largest rookeries in the Gulf of Alaska and Aleutian Islands.

Most adult Steller sea lions occupy rookeries[1] during the pupping and breeding season, which extends from late May to early July (Pitcher and Calkins 1981, Gisiner 1985). During the breeding season some juveniles and non-breeding adults occur at or near the rookeries, but most are on

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[1] Throughout this document a rookery refers to a site where breeding occurs and sea lions may haulout during the non-breeding period; a site designated as a rookery will be called a rookery the entire year, even though breeding occurs there only from late May to early July.

haulouts. Adult males, in particular, may disperse widely after the breeding season. Males that breed in California move north after the breeding season and are rarely seen in California or Oregon except from May through August (Mate 1973). During fall and winter many sea lions disperse from rookeries and increase use of haulouts, particularly terrestrial sites but also sea ice in the Bering Sea.

Steller sea lions are not known to make regular migrations, but they do move considerable distances (Baba *et al.* 2000). Animals marked as pups on rookeries in the Gulf of Alaska have been sighted in Southeast Alaska and British Columbia; some marked in British Columbia have been seen at Cape Saint Elias, Alaska; some marked in the eastern Aleutians have been seen in eastern Bristol Bay, Alaska; and some marked in Oregon have been seen in northern California, Washington, British Columbia, Southeast Alaska, and the northern Gulf of Alaska (Calkins and Pitcher 1982, Calkins 1986, Loughlin 1997). Raum-Suryan *et al*. (2002) analyzed resightings of 8,596 pups that were branded from 1975-1995 on rookeries in Alaska and reported that almost all resightings of young-of-the-year were within 500 km of the rookery where the pup was born, although subsequent observations documented movements of 11 month-old pups with their mothers of over 800 km. Juvenile animals were seen at much greater distances from their rookery of birth (up to 1785 km). Sightings of adults were generally less than 500 km away from the natal rookery although adult males have since been seen over 1000 km from the rookery where they held a territory (also their natal rookery).

Steller sea lion pups tagged in the Kuril Islands commonly moved northward to the east and west coasts of Kamchatka (Burkanov *et al*. 1997) and have also been seen as far south as Yokahama, Japan (Baba *et al.* 2000, NMFS unpublished data). Pups tagged on the Commander Islands have moved to the east coast of Kamchatka (Burkanov *et al.* 1997). Juveniles marked in the central Aleutian Islands have been observed in the Commander Islands.

NMFS reclassified Steller sea lions as two distinct population segments under the ESA based on genetic studies and phylogeographical analyses from across the sea lion's range (62 FR 24345). The eastern DPS includes sea lions born on rookeries from California north through Southeast Alaska; the western DPS includes those animals born on rookeries from Prince William Sound westward (Bickham *et al*. 1996, Loughlin 1997). The regulatory division between DPSs is Cape Suckling (144° west longitude) in the northeast Gulf of Alaska. However, frequent movement is seen across this boundary by animals from both populations, particularly juvenile animals (Raum-Suryan et al. 2002).

Mitochondrial DNA (mtDNA) has been the primary type of genetic material examined to evaluate patterns of genetic variability within and among sea lions from various populations and rookeries throughout their range. This type of DNA typically exhibits maternal inheritance, meaning that offspring inherit their mother's sequence (barring mutation). Bickham *et al*. (1996) reported on analyses of characteristics of mtDNA from 224 Steller sea lions sampled between the Commander Islands and Oregon. The researchers found a high level of genetic diversity with a large number of haplotypes occurring at a relatively low frequency (46 of 52 haplotypes with a frequency less than 0.03). Additional analyses from over 1200 sea lions identified over 130 haplotypes range-wide (Bickham *et al*. 1998a, Ream 2002). A distinct break in the distribution of haplotypes was found between locations sampled in the western part of the range (Russia to the eastern Gulf of Alaska) and eastern locations (Southeast Alaska and Oregon), indicating restricted gene flow between two populations (Figure I-1). These researchers speculated that the two populations did not evolve from a single maternal ancestor but rather descended from the genetic makeup of two populations that inhabited separate glacial refugia during the last ice age.

Loughlin (1997) reviewed information on genetics, together with what is known about distribution, population response, and phenotypic characteristics, to identify Steller sea lion populations. He found that the strongest support for multiple populations came from the genetics results described above, but information on distribution and movement patterns and population responses provided additional support. Loughlin concluded that Steller sea lions should be managed as two populations, an eastern population that includes all animals born on rookeries east of Cape Suckling, Alaska, and a western DPS that includes all animals born at rookeries west of Cape Suckling. NMFS accepted this recommendation and in 1997 reclassified Steller sea lions as two distinct population segments under the ESA (62 FR 24345, 62 FR 30772).

Bickham *et al.* (1998a) analyzed mtDNA from an additional 191 Steller sea lions, mostly from regions not sampled in their previous study, e.g., Kuril Islands, British Columbia, and California. The results from those samples combined with previous results confirmed the high degree of genetic differentiation between eastern and western DPSs. Bickham *et al.* (1998b) also analyzed mtDNA from 36 Steller sea lions sampled in the Gulf of Alaska in 1976-1978 and compared the results with samples collected in the 1990s following the steepest population decline (Bickham *et al.* 1996). They found that the high level of haplotypic diversity previously noted for the present population had been maintained between the two sampling periods. Thus, genetic diversity of Gulf of Alaska sea lions had been retained in spite of the recent major decline in abundance. Phylogenetic analysis by Harlin-Cognato *et al*. (2006) suggests that the current genetic structure of sea lions is the result of Pleistocene glacial geology which influenced the availability of suitable rookery habitat.

Substantial additional genetic research was conducted with larger samples from throughout the Steller sea lion range, including most rookeries in Russia. The results of these studies generally confirm the strong east/west population delineation, but differ in their description of further structure within the western DPS when looking either at mtDNA or nuclear DNA (Trujillo *et al.* 2004, Baker *et al.* 2005, Hoffman *et al.* 2006, NMFS unpublished data). A further complexity is the possibility that the geographic boundary between the western and eastern populations may be changing or possibly disappearing (Pitcher *et al.* 2007, NMFS unpublished).

Trujillo *et al.* (2004) examined mtDNA and nuclear DNA from the same samples to show that the population separation apparent from the mtDNA work was not clearly defined when males were taken into account. There was no clear separation of populations based on genetics when markers from both parents were included. They suggested that the difference was either due to a faster population divergence at the mtDNA locus or that, like many other mammals, Steller sea lions show a greater level of male-mediated gene flow via immigration than in females, e.g. males tend to disperse more than females and do not show the same philopatry for their natal areas as females.

Support for this result comes from observational work in the eastern DPS with the monitoring of branded animals. Resights of animals branded as pups in one DPS have been reported occasionally at haulouts and rookeries within the other DPS. In addition, recent mtDNA work with large samples of pups from newly established rookeries in the eastern DPS has shown that some females born in the western DPS are pupping in the eastern DPS (NMFS unpublished data; see below). Because these samples were collected from rookeries that were not yet established at the time of the ESA designation, they were not included in the original genetic studies.

Based on analyses of sequence variability at a segment of the mtDNA control region in samples from 1,568 individuals representing nearly every significant rookery rangewide, Baker *et al.* (2005) hypothesized that a third population (termed by the authors the Asian stock, including rookeries from the Kamchatka Peninsula, Kuril Islands, and Sea of Okhotsk) may exist just west of the Commander Islands in Russia. They recommended that the western stock be partitioned west of the Commander Islands, yielding a western stock that ranges from Prince William Sound west to the Commander Islands, and an Asian stock that includes rookeries from the Kamchatka Peninsula, Kuril Islands, and Sea of Okhotsk. However, the level of differentiation between the putative Asian and western stock was not as great as that between the eastern and western stocks. Other significant findings from this study included: no evidence for significant genetic bottleneck effects; support for significant divergence of eastern stock (southeastern Alaska to California) animals from western stock animals; significant differences between rookeries and regions along Asia from all other western stock rookeries; clear association of the Commander Islands rookery with Alaskan western stock rookeries, not with the Russian (termed Asian) rookeries; and significant isolation by distance among rookeries within, but not among, the stocks, indicating that there may be important gene flow barriers among stocks. Based on the findings from this study, conclusions about the eastern stock would remain unchanged, i.e., that stock would include rookeries from southeastern Alaska through California.

Hoffman *et al.* (2006) followed up on the research by Baker *et al.* (2005) by genotyping over 700 individuals from across the species range at 13 highly polymorphic nuclear (inherited from both parents) microsatellite loci. They found that although there was strong female philopatry (as described by mtDNA methods), there was little evidence to support the separation of an Asian DPS due to potentially extensive male gene flow. These investigators also detected a clear phylogenetic break between populations of the western and Asian stocks and those of the eastern stock. Hoffman *et al.* (2006:2821) concluded that "mtDNA structuring is not due simply to female philopatry, but instead reflects a genuine discontinuity within the range, with implications for both the phylogeography and conservation of this important marine mammal."

Conclusions of Raum-Suryan *et al.* (2002) support the hypothesis that Steller sea lions conform to the metapopulation model. Similarly, other unpublished research funded by NOAA Fisheries focusing on population structure within the western and eastern DPSs in the U.S. has also shown that there may be additional population structure within the western DPS, specifically with a split at Samalga Pass (O'Corry-Crowe *et al* 2006). Conversely, of the two most recently established rookeries in the eastern DPS, about 70% of the pups born on Graves Rock were from western DPS females, and about 45% of the pups born at White Sisters were from western DPS females (NMFS unpublished, Gelatt *et al*. 2006). Movement inferred from the genetics data has been confirmed by the sighting of western branded females with pups at Graves Rock and White Sisters (NMFS unpublished). This movement of females from the western stock to locations within range of the eastern stock for the purpose of pupping and presumably, for breeding, has potential long term implications to the viability of these populations and their management. It is possible that we are witnessing in real-time a very infrequent event in which female sea lions from one population cross over to breed in another. At this point, it appears that sea lions are only crossing (in detectable numbers) from west to east.

Dispersal of animals from their natal rookeries may have important consequences for expansion of a population and possible recovery as it provides a mechanism for occupying new territory or re-occupying vacant areas (Raum-Suryan *et al*. 2002). In Southeast Alaska, new rookeries were established as population size increased, at least partially the result of dispersal from the large Forrester Island rookery (Calkins *et al*. 1999, Raum-Suryan *et al*. 2002, Pitcher et al. 2007, ADF&G unpublished data) and from the western DPS (NMFS unpublished, Pitcher *et al.* 2007).

While new data about the dispersal, patterns of genetic variation, and breeding site selection continue to refine NMFS's understanding of the underlying population structure, this plan must, and does, address the current, legally delineated structure of a western DPS and an eastern DPS. However, NMFS will continue to obtain and to evaluate additional data about the underlying population structure of sea lions. These data are important to achieving a full understanding of population processes and to inform management-related decisions.

C. Overview of Population Status

Count data used to estimate population trend and evaluate status are of two types: counts of pups about 1 month of age and counts of animals over 1 year of age (i.e., non-pups). Counts of pups were usually made by observers on rookeries, herding the non-pups into the water, and walking through the rookery and counting the pups (Calkins and Pitcher 1982, Sease *et al*. 2001). Beginning in 2002, 126mm format aerial photography has also been used to count pups (Westlake *et al*. 1997, Snyder *et al*. 2001). In British Columbia, pup counts were made from 35mm slides taken during aerial surveys flown specifically to facilitate pup counts (vertical orientation).

Counts of pups on rookeries conducted near the end of the birthing season are nearly complete counts of pup production. These counts can be expanded to estimate approximate total population size based on an estimated ratio of pups to non-pups in the population (Calkins and Pitcher 1982, Trites and Larkin 1996). Based on estimates of birth rate and sex and age structure of a stable sea lion population from the Gulf of Alaska, Calkins and Pitcher (1982) estimated total population size was 4.5 times the number of pups born. Some pups die and disappear before the counts are made and a few are born after the counts are conducted (Trites and Larkin 1996); because of this the researchers selected 5.1 as a correction factor. It should be emphasized that this is a very general estimate of population size as several factors can affect the accuracy of this correction factor. Sex and age structure and mortality and birth rates may vary over time and among populations and require different correction factors.

Non-pups were counted in most instances from 35 mm color slides taken from aircraft during the breeding season (Calkins and Pitcher 1982, Merrick *et al.* 1987, Sease *et al.* 2001), although in recent years some counts were made from 126mm format aerial photographs. Counts from 35 mm slides and medium format photographs were highly correlated but, on average, slightly higher counts were obtained from medium-format photographs (Fritz and Stinchcomb 2005).

Counts of both pups and non-pups were used to estimate trend for the various geographic areas depending on availability of data. While time series counts of pups and non-pups are available to assess population status and trend, non-pup counts have been used most often. This is because non-pup counts have been obtained across broad areas of the range (e.g., all of the western stock in Alaska) in a single year using aerial surveys, and because more years of such data exist. By contrast, an assessment of pup production across such a broad area used to take 2-3 years to complete. Consequently, even though a time series of annual pup production estimates could provide more information on sea lion vital rates, it was not possible to obtain these data on a consistent, range-wide basis until 2005, when high resolution medium format photographic aerial surveys were initiated.

Trend analysis was conducted by linear regression of the natural logarithms of the counts by year. For the western DPS, estimates of population trend (an index to changes in absolute population abundance) were based on comparisons of counts among years at a group of sites consistently monitored since the 1970s (trend sites). Trend sites include the majority of animals observed in each survey (e.g., 72% in 1998, 75% in 2000; Sease *et al.* 2001). “Trend rookeries” are a subset of all trend sites and include all major rookeries except those on Outer and Attu Islands. Counts of pups on rookeries are also used to estimate population trend.

Non-pup numbers used for population trend assessment are sums of counts at sites within sub-areas or across the range of the western stock in Alaska. Replicate surveys conducted in 1992 and 1994 confirmed NMFS understanding of sea lion haul-out behavior patterns. The number of sea lions on individual haul-outs can vary considerably from day to day, while numbers on rookeries tend to be more stable. However, if surveys are conducted in mid-June during the height of the breeding season, the sum of counts at all consistently surveyed sites within a sub-area has a much lower variance than the counts at any individual site. This is due to movement between sites within the same sub-area. Coefficients of variation associated with sub-area non-pup totals range between 5-15% (NMFS, unpublished data). NMFS designed a monitoring plan using the established survey techniques to estimate the impact of the fishery management measures (Reasonable and Prudent Alternatives) contained in the 2000 Biological Opinion, and determined that there was a greater than 90% chance of detecting a 1% per year change in population size over 8 years (4 surveys) (NMFS 2000).

From the late 1960s through 2000, the western DPS declined over 80% in abundance, with steepest declines of approximately 15% per year occurring in the late 1980s and slower declines of about 5% per year in the 1990s (based on non-pup counts; Loughlin *et al.* 1992, Trites and Larkin 1996, Loughlin 1997, Sease and Loughlin 1999). Between 2000 and 2004, counts of non-pups on western DPS trend sites increased or were stable through much of the Alaskan range. A partial survey in 2006 suggested that this increasing trend did not continue in some sub-areas, and that the population may have stabilized (Sease and Gudmundson 2002, Fritz and Stinchcomb 2005, Fritz *et al.* 2006[2]). However, joint consideration of data from both the 2006 and the most recent 2007 non-pup surveys (also incomplete) suggests that the overall trend for the western DPS, through 2007, is either stable or slightly declining. Data also indicate there are significant trend differences amongst sub-regions within the western DPS. (Fritz *et al.* 2007[3]). Based on 2004-2005 data, the western DPS was composed of approximately 45,000 sea lions in Alaska and approximately 16,000 in Russia (Angliss and Outlaw, 2006). The estimated population size of the eastern DPS in 2002 was between 46,000 and 58,000 animals. The eastern DPS has been increasing at approximately 3% per year since the late 1970s (Pitcher *et al*. 2007).

The specific causes of the decline are not known, and the relative importance of various factors may have changed over time. While there is no consensus on the causes of the sharp decline in the 1980s or consensus on why the population declined at a slower rate through the 1990s, several factors have been proposed and have some degree of support. Direct mortality through incidental take in fisheries, commercial harvests, and illegal shooting (Perez and Loughlin 1991, Alverson 1992, Trites and Larkin 1992) has been proposed as one mechanism in the decline. A reduction in survival and possibly natality due to a reduced or modified prey base has frequently been proposed as a factor in the decline. This could have resulted from commercial fisheries (Fritz *et al*. 1995, Loughlin 1998) or by a major regime shift in the mid-1970s (Trenberth 1990, Springer 1998, Benson and Trites 2002, Le Boeuf and Crocker 2005, Trites *et al.* 2006a). Predation by killer whales, alone or in conjunction with other factors, may also have contributed to the declines of sea lions and other species of marine mammals in Alaska (Barrett-Lennard *et al.* 1995, Springer *et al*. 2003). It should be noted that Steller sea lions are not the only population of marine mammals to undergo a substantial decline in portions of western Alaska. Harbor seals (Pitcher 1990, Frost *et al*. 1999, Small *et al.* 2003, Ver Hoef 2003), northern fur seals (Trites 1992, Towell *et al.* 2006), and sea otters (Estes *et al*. 1998, Doroff *et al*. 2003) have all declined substantially over at least portions of the range of the western DPS of Steller sea lion.

During approximately the same period, the eastern DPS has more than doubled in size and is at its highest level in recent history, numbering 45,000 to 51,000 animals in 2002 (Pitcher *et al.* 2007). This population increased at about 3% per year from the late 1970s through 2002. Recent data from Southeast Alaska (2005) and California (2004) suggest continued population growth. Legal protection, both in the United States and Canada, probably played an important role in population growth.

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[2] Memo from Fritz *et al.* to the Record describing the 2006 aerial non-pup sea lion survey, results, and implications on the population trajectory. Dated September 25, 2006, Alaska Fisheries Science Center.

[3] Memo from Fritz *et al.* to the Record describing the 2007 survey of adult and juvenile Steller sea lions, June-July 2007. Dated October 23, 2007, Alaska Fisheries Science Center.

D. Western DPS Status and Trend

The western DPS of Steller sea lion breeds on rookeries in Alaska (the U.S. portion of the western DPS) from Prince William Sound (144°W) west through the Aleutian Islands and in Russia on the Kamchatka peninsula, Kuril Islands and the Sea of Okhotsk (Bickham *et al.* 1996, Loughlin 1997). Loughlin *et al.* (1984) estimated the worldwide population of Steller sea lions was between 245,000 and 290,000 animals (including pups) in the late 1970s (1974-80). Though the genetic differences between the eastern and western DPSs were not known at the time, Loughlin *et al.* (1984) noted that 90% of the worldwide population of Steller sea lions was in the western DPS in the early 1980s (75% in the U.S. and 15% in Russia) and 10% in the eastern DPS. Loughlin *et al.* (1984) concluded that the total worldwide population size (both DPSs) was not significantly different from that estimated by Kenyon and Rice (1961) for the years 1959 and 1960, though the distribution of animals had changed. After conducting a range-wide survey in 1989, Loughlin *et al.* (1992) noted that the worldwide Steller sea lion population had declined by over 50% in the 1980s, to approximately 116,000 animals, with the entire decline occurring in the range of the western DPS.

1. Alaska (U.S. portion of the range)

Steller sea lions use 38 rookeries and hundreds of haul-out sites within the range of the western DPS in Alaska (Figures I-2 and I-3). The first reported counts of Steller sea lions in Alaska were made in 1956-1960 (Kenyon and Rice 1961, Mathisen and Lopp 1963), and these totaled approximately 140,000 for the Gulf of Alaska (GOA) and Aleutian Islands (AI) regions (Merrick *et al.* 1987)[3]. Subsequent surveys showed a major decline in numbers first detected in the eastern AI in the mid-1970s (Braham *et al.* 1980). The decline spread eastward to the central GOA during the late 1970s and early 1980s and westward to the central and western AI during the early and mid 1980s (Merrick *et al.* 1987, Byrd 1989). Approximately 110,000 adult and juvenile sea lions were counted in the Kenai-Kiska region in 1976-1979, and by 1985 and 1989, counts had dropped to about 68,000 (Merrick *et al.* 1987) and 25,000 (Loughlin *et al.* 1990), respectively. Since 1990 when Steller sea lions were listed under the ESA, complete surveys have been conducted throughout their range in Alaska every one or two years (Merrick *et al.* 1991, 1992, Sease *et al.* 1993, 1999, 2001, Strick *et al.* 1997, Sease and Loughlin 1999, Sease and Gudmundson 2002, Sease and York 2003, Fritz and Stinchcomb 2005).

Steller sea lion populations in parts of the Alaskan range of the western DPS may have begun to drop between the late 1950s and the mid 1970s (Table I-1[4]). From the mid-1970s to 1990 the overall western DPS in Alaska declined by over 70%, with the largest declines in the AI (76% to 84%) and smaller declines in the GOA (23% to 71%; Table I-1). Between 1990 and 2000, trend site

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[3] For the western DPS of Steller sea lion in Alaska, count data have generally been combined and analyzed in six subareas (Figures I-2 and I-3), which are geographically convenient but do not necessarily reflect biologically important units. Because earlier efforts to count sea lions were concentrated in the center of their Alaskan range, evaluations of long-term trends have often been calculated for the "Kenai to Kiska" index area, which includes the central and western Gulf of Alaska and the eastern and central Aleutian Islands.

[4] In some cases the counts shown in this table are lower than total survey counts given above (and used in some other reports) because not all sites counted in a survey are trend sites.

counts continued to decline, though more slowly than in the 1980s, resulting in a total reduction of almost 90% since the 1950s and 83% since the 1970. Sub-area declines from 1990 to 2000 had a different pattern than in the 1970s-1990 period, with smaller changes in the center of the Alaskan range (western GOA and eastern and central Aleutians: -32% to +1%) and larger declines at the edges (eastern and central GOA and western Aleutians: -54% to –64%). The average rate of decline between 1990 and 2000 for all trend sites in the western DPS was 5.1% per year (Sease *et al.* 2001).

Between 2000 and 2004, Kenai-Kiska and western Alaska population trend site counts of non-pup Steller sea lions increased by 12% (Table I-1; Fritz and Stinchcomb 2005). Increases were not spread evenly across the range in Alaska, however. Non-pup counts increased by over 20% in the eastern Aleutian Islands and in the eastern and western GOA, and by 10% in the central Aleutian Islands, but were lower by as much as 16% in the central GOA and western Aleutians (Table I-1; Figures I-2 and I-3). While overall non-pup counts from 2000 to 2004 increased, counts in the western GOA and eastern AI had essentially no trend between 1990 and 2004, suggesting that western Steller sea lions in the core of their Alaskan range may currently be oscillating around a new lower mean level.

Counts of non-pup sea lions on 1990s trend sites in the eastern GOA, eastern AI, and western GOA were essentially unchanged between 2004 and 2006. It was the increase in counts in these three sub-areas that largely drove the 3% per year increase observed between 2000 and 2004 in the Alaskan western DPS as a whole. Counts in these three sub-areas increased by 3,151 sea lions between 2000 and 2004, while counts in the remaining sub-areas (central GOA, central AI, and western AI) declined by 548. If the population in the three previously increasing sub-areas stabilized between 2004 and 2006, then it appears unlikely that the Alaskan western DPS as a whole continued to increase through 2006. However, lack of 2006 counts in the largest of the previously declining sub-areas (central GOA and central AI) precludes a definitive conclusion about trends in the western DPS across Alaska, a gap which the aerial non-pup survey planned for June 2007 was designed to fill. However, that survey was also incomplete.

Using the methods described in Loughlin *et al.* (1992), Loughlin (1997) estimated that the non-pup U.S. portion of the western DPS totaled approximately 177,000 animals in the 1960s; 149,000 in the 1970s; 102,000 in 1985; 51,500 in 1989; and only 33,600 in 1994. Using similar methods, Loughlin and York (2000) estimated the number of non-pups in the U.S. portion of the western DPS in 2000 at about 33,000 animals. Using a different method[5], Ferrero *et al.* (2000) and Angliss and Outlaw (2005) estimated the minimum abundance of the U.S. portion of the western DPS in 1998 at 39,031 and in 2001-2004 at 38,206, respectively, a decline of over 80% since the late 1970s.

As noted previously, pups have been counted less frequently than non-pups. However, the overall trends since the late 1970s have been similar to counts of non-pups (Table I-2). The number of pups counted in the Kenai-Kiska region declined by 70% from the mid-1980s to 1994, with large declines (63% to 81%) in each of the four sub-areas. From 1994 to 2001-02, Kenai-Kiska

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[5] Estimated population numbers were based on a pup multiplier (e.g., 5.1 and 4.5 were used), while the minimum population estimates were based on adding the total number of non-pups counted in an aerial survey with the "best" estimate of pups counted.

pup counts decreased another 19%, with the largest change (-39%) observed in the central GOA. The overall decline in the number of pups in the Kenai-Kiska region from the mid-1980s through 2002 was 76%. Pup counts in the eastern GOA (not included in the Kenai-Kiska region) declined by 35% from 1994 to 2002, while in the western Aleutian Islands, pup counts declined by 50% between 1997 and 2002 (Table I-2). Between 2001-02 and 2005, increases in pup counts were noted in the eastern and western GOA and eastern AI, while pup counts declined in the central GOA and central and western AI. In June-July 2005, a medium format aerial survey for pups was conducted from Prince William Sound to Attu Island, which provided the first complete pup count for all western DPS rookeries in Alaska (n = 9,951 pups; NMFS unpublished data). Using the "pup" estimator (4.5) yields an estimate of approximately 45,000 Steller sea lions in the range of the western DPS in Alaska.

The population of Steller sea lions on the Pribilof Islands has seen similar declines, although the trends were initiated much earlier. Elliott (1880) reported that approximately 10,000 to 12,000 animals were distributed at rookeries on both St. Paul and St. George Islands in the 1870s. Osgood *et al.* (1916) described the importance of Steller sea lions to the local community for both food and material for clothing and boats. The pups especially were favored for their meat. Between 1870 and 1890, at least 4,000 sea lions were killed on St. Paul Island and by the early 1900s the local agent noted that the hunt should cease due to a reduced population (Osgood *et al.* 1916). In 1940, Scheffer counted 800-900 adults and 300-400 pups on St. Paul and noted that the population was growing and that the sea lions interfered with the management of the fur seal herd by competing for both food and space and "creating a nuisance to the men who drive and kill the seals" (Scheffer 1946). This competition initiated a request to cull part of the population. The recommendation was to kill 50 pups a month during June, July, and August to assess the seasonal quality of the pelts.

The combination of hunting and culling appears to have kept the Pribilof sea lion population at reduced numbers, and Loughlin *et al.* (1984) reported that the breeding rookeries on St. George Island were extirpated by 1916. No pups have been reported on St. George since. In the summer of 1960, 4,000 to 5,000 non-pups and 2,866 pups were counted on Walrus Island, just offshore of St. Paul (Kenyon 1962). Between the 1960s and 2005, however, numbers of non-pups and pups on Walrus Island declined over 90%, to 322 non-pups in 2001 and only 29 pups in 2005 (Figure I-3 and Table I-2; Loughlin *et al.* 1984, NMFS unpublished data). The cause of the declines during the last 50 years remains unexplained. Subsistence takes of non-pups have continued on the main islands of St. Paul and St. George averaging 141 during 1992-1998, but declined to less than 100 sea lions in the latter half of the 1990s (Wolfe and L.B. Hutchinson-Scarbrough 1999). Walrus Island is the only Steller sea lion rookery still active in the Pribilofs, but pup production has declined steadily from 2,866 in 1960 to approximately 334 in 1982, 50 in 1991, 39 in 2001, and only 29 in 2005 (NMFS 1992, NMFS unpublished data, Fritz and Stinchcomb 2005).

Modeling studies based primarily on data collected in the central GOA indicate that the decline experienced by the western sea lion population in Alaska in the 1980s may have been caused by a steep drop in the survival rate of juveniles, perhaps by as much as 20-30% (York 1994, Pascual and Adkison 1994, Holmes and York 2003). However, the models suggest that the decline at this time was also associated with smaller decreases in adult survival and female natality (Holmes and York 2003). The drop in natality would not have been predicted based on density-dependence alone. Subsequent to the 1980s, demographic models indicate that juvenile and adult survival rates rebounded to levels similar to those of the stable equilibrium population of the 1970s, but that natality continued to decline into the 1990s (Holmes and York 2003).

2. Russia and Other Parts of Asia

Steller sea lions use 10 rookeries and approximately 77 haul-out sites within the range of the western DPS in Russia (Figure I-5). Of these 77 haul-outs, three had previously been rookeries, but presently no breeding occurs on these three sites, 49 are active haul-out sites, 20 have been abandoned (no sea lions seen there for the past 5-10 years), and five have inadequate information to assess their status. Analysis of available data collected in the former Soviet Union indicates that in the 1960s, the Steller sea lion population totaled about 27,000 (including pups), most of which were in the Kuril Islands. Between 1969 and 1989, numbers of adult and juvenile sea lions at major rookeries and haul-outs in the Kuril Islands alone declined 74% (Merrick *et al.* 1990). By the late 1980s and early 1990s, the total Russian population had declined by approximately 50% to about 13,000 (including pups) (Burkanov and Loughlin 2005). Since the early 1990s, the population in Russia has increased in most areas and, in 2005, is estimated to number approximately 16,000 (including pups), of which about half inhabit waters around the Kuril Islands (Burkanov and Loughlin 2005).

Trends in counts of non-pup and pup Steller sea lions on selected rookeries and haulout sites have varied by subarea within Russian waters (Tables I-3 and I-4; Figure I-5). In the Kuril and Commander Islands and in eastern Kamchatka, Steller sea lion numbers declined through the 1970s and 1980s, but increased slightly or were stable from the early 1990s through 2005. In the western Bering Sea, there are no rookeries; numbers of non-pups have plunged over 90% and since 2000, have totaled less than 100 (Table I-3). By contrast, Steller sea lion numbers on Tuleny Island and at two rookeries in the Sea of Okhotsk (on Iony and Yamsky Islands) have increased considerably in the last 15 years. Overall, counts of non-pups on all Russian trend sites were essentially stable between 1989 and 2004 (an annual rate of change of -0.02%, which is not significantly different from 0; p=0.96).

The Steller sea lion is listed as an endangered species under Russian legislation. While the Russian government currently has no organized program of monitoring and research, both NMFS and the Alaska SeaLife Center have programs to monitor population trends (non-pup and pup counts), estimate vital rates (branding and re-sighting), collect food habits data, and conduct other research on Steller sea lions in Russia. It is anticipated that research on Russian-Asian sea lions will continue to be supported by both institutions in the near future.

Burkanov and Loughlin (2005) summarized that Steller sea lions may have bred in Japan in the past, but at present the only rookeries in the northwestern Pacific are in Russia. Historically Steller sea lions were common along the entire Asian coast from the Amur River to the Korean Peninsula and occasionally in the Yellow Sea, but are not known to have bred there. Currently, Steller sea lions do not inhabit the Yellow Sea. In the Sea of Japan, evidence indicates their current range is limited to coastal waters along the southwest coast of Sakhalin Island and the west coast of Hokkaido. In the northwestern Pacific, the southern border of the Steller sea lion range appears to have moved northward 500-900 km over the past 50 years (Burkanov and Loughlin

2005). Few data exist regarding historical or current distribution along the west coast of the Sea of Japan (Burkanov and Loughlin 2005).

3. Western DPS overall

The western DPS of Steller sea lions decreased from an estimated 220,000-265,000 animals in the late 1970s to fewer than 50,000 in 2000. The decline began in the 1970s in the eastern Aleutian Islands (Braham *et al.* 1980), western Bering Sea/Kamchatka and the Kuril Islands. In Alaska, the decline spread and intensified east and west of the eastern Aleutians in the 1980s and persisted at a slower rate through 2000 (Sease *et al.* 2001). The 12% increase in numbers of non-pups counted in the Alaskan range of the western DPS between 2000 and 2004 was the first region-wide increase observed during more than two decades of systematic surveys. The observed increase, however, has not been spread evenly among all regions of Alaska. Increases were noted in the eastern and western Gulf of Alaska and in the eastern and central Aleutian Islands, while the decline persisted through 2004 in the central Gulf of Alaska and the western Aleutian Islands. Non-pup counts at all western DPS trend sites in Alaska in 2004 were similar to the 1998 total, but were still 33% lower than the number counted in 1990. In Russia, both pup and non-pup data indicate that sea lion numbers are increasing at Sakhalin Island and in the Sea of Okhotsk and likely at the Commander Islands. However, non-pup numbers in Kamchatka and the Kuril Islands, the former core of the Russian range, declined substantially through the late 1980s, but have increased slightly through 2005. The number of western Steller sea lions throughout its range in Alaska and Russia in 2005 is estimated at approximately 61,000 (45,000 in Alaska and 16,000 in Russia).

E. Eastern DPS Status and Trend

1. Overview

The available historical records of Steller sea lion abundance were reviewed for the eastern DPS in an attempt to relate current population size with levels prior to the initiation of standardized surveys (Figure I-6). These records provide interesting insights into relative population levels but must be interpreted with caution because the older counts were obtained by a variety of methods and during varying times of the year. Count data obtained prior to 1970 were not subjected to quantitative analyses because of intermittent availability and concerns about comparability with more recent count data. Counts of both pups and non-pups were used to estimate trends for the various geographic areas depending on availability of data (Figures I-7 and I-8). Trend analysis was conducted by linear regression of the natural logarithms of the counts by year.

Population trend was analyzed by geographic regions (Southeast Alaska, British Columbia, Washington, Oregon, and California) as the data were collected by various state and federal agencies in each area. Steller sea lions, particularly juveniles, range widely (Raum-Suryan *et al.* 2002), and therefore population estimates for a particular geographic area represent the number of animals supported by the rookeries in that area and not the exact number of animals present in the area at any time. This is particularly true when large rookeries are located near jurisdictional borders such as the boundaries between Southeast Alaska and British Columbia and between Oregon and California.

2. Southeast Alaska

Numbers of pups counted on rookeries increased from 2,219 in 1979 to 5,510 in 2005, an annual rate of increase of 3.1% (Table I-5). In 1979, the Forrester Island rookery complex was the only rookery in Southeast Alaska. During the early 1980s, a rookery developed at Hazy Islands, and in the early 1990s at White Sisters. Recently, two additional sites, Graves Rocks and Biali Rocks, appear to have developed into rookeries with 175 and 100 pups counted respectively at the two sites in 2005. Since 1990, nearly all the increase in pup numbers has been at the newer rookeries, as pup numbers at the Forrester Island rookery were stable. In addition to the five rookeries, sea lions used 30 major haulouts, plus several other sites for brief periods each year, probably in conjunction with seasonal prey concentrations.

At four of five rookeries in Southeast Alaska, counts of non-pups increased substantially from 1979 to 2005 (Table I-6). Based on 2002 pup counts, estimated Steller sea lion abundance (all age classes) in Southeast Alaska was 21,947 animals (with the 4.5 pup multiplier) or 24,873 (with the 5.1 pup multiplier); by comparison, a total of 20,160 sea lions (pups plus non-pups) were counted during the 2002 survey.

Historical data for this region are scant, yet numbers of Steller sea lions were likely relatively low during the early 1900s when there may not have been any rookeries in Southeast Alaska (Rowley 1929, Imler and Sarber 1947). Numbers have progressively increased since that time (Calkins *et al.* 1999) and are now believed to be at a historical high.

3. British Columbia

Counts of Steller sea lion pups increased from 941 in 1971 to 3,281 in 2002 (Table I-7; Olesiuk and Trites 2003), an annual rate of increase of 3.2% closely paralleling the trend in Southeast Alaska. Rookeries occur at North Danger Rocks, Cape St. James, and the Scott Islands (Maggot, Triangle, Sartine, and Beresford Islands). Sea lions also use 24 major haulout sites in British Columbia (Olesiuk 2001) plus a number of other seasonal haulouts (Bigg 1988) (Figure I-7).

Extensive sea lion reduction programs were conducted at many locations in British Columbia from 1912 through 1966, and sea lions were commercially exploited during the 1960s, resulting in the population being reduced to about 30% of peak levels of the early 1900s (Bigg 1988). A major rookery, the Sea Otter Group, was eradicated by about 1940 as a result of intensive control efforts and while sea lions still used it as a haulout it no longer serves as a rookery.

The most recent survey occurred in summer, 2002 and counted 15,402 sea lions including 3,281 pups and 12,121 non-pups (Table I-7; Olesiuk and Trites 2003). Steller sea lion abundance (all age classes) in British Columbia, based on 2002 pup counts at rookeries, was 14,765 animals (with the 4.5 pup multiplier) or 16,733 (with the 5.1 pup multiplier). Olesiuk and Trites (2003) used the raw counts and a multiplier to estimate the total number of animals present in British Columbia waters during the breeding season of 2002 at 18,400 – 19,700 individuals of all ages, including non-breeding animals associated with rookeries in Southeast Alaska and Oregon. It appears that the British Columbia Steller sea lion population has largely recovered from the low levels of the 1970s, particularly when considered in conjunction with the adjoining Southeast Alaska population (Olesiuk 2001).

4. Washington

No rookeries exist in the state of Washington, but Steller sea lions are present along the coast throughout the year. Four major haulouts are used (Figure I-8), and counts of non-pups have been made during the breeding season during most years since 1991, when numbers of sea lions increased at an average of 9.2% annually (Table I-8). These animals are assumed to be immature animals and non-breeding adults associated with rookeries from other areas. Branded juvenile sea lions from the Forrester Island rookery in Southeast Alaska (Raum-Suryan *et al*. 2002) and from the Rogue Reef rookery in Oregon (Brown unpublished data) have been observed in Washington. Older records suggest that current numbers are reduced from historical levels. Between 2,000 and 3,000 Steller sea lions were reported during August and September of 1914, 1915, and 1916 in the Carroll Island area (Kenyon and Scheffer 1959, Scheffer 1950) while the maximum observed during 60 complete surveys of Washington haulouts between 1980 and 2001 was 1,458 in October, 2000 (non-breeding season count).

5. Oregon

Steller sea lions occupy two rookeries, located at Rogue Reef and Orford Reef, and eight haulout sites in Oregon (Figure I-8). The total number of non-pup sea lions counted during the breeding season surveys at all of these sites has increased from 1,461 in 1977 to 4,169 in 2002 (Table I-8; Brown *et al*. 2002), an annual rate of increase of about 3.7%. Although not nearly as well documented, pup numbers also appear to have increased. In 1996, 685 and 335 pups were counted at Rouge Reef and Orford Reef respectively, whereas in 2002, 746 and 382 pups were counted at the two sites. These counts were made from 126mm format, aerial photographs. Steller sea lion abundance (all age classes) in Oregon, based on 2002 pup counts at rookeries, was 5,076 animals (with the 4.5 pup multiplier) or 5,753 (with the 5.1 pup multiplier). A total of 5,297 animals were actually counted during the 2002 surveys.

Historical data on Steller sea lion abundance in Oregon are sketchy. Pearson and Verts (1970) estimated the population at 1,078 animals in 1968, somewhat lower than the 1977 count of 1,461. Population size was believed to be substantially smaller than in 1925 due to extensive human-caused mortality, in part stimulated by a bounty (Pearson and Verts 1970). After three decades of growth, this population has recovered substantially, but the relationship of present numbers to levels during the 1800s and early 1900s is not known.

6. California

Steller sea lions historically occupied five major rookeries and haulouts in California (San Miguel Island, Año Nuevo Island, the Farallon Islands, Sugarloaf Island/Cape Mendocino, and Saint George Reef) that have been surveyed periodically over the last 75 years (Figure I-8). While there is a long, intermittent time series of counts for California (Bonnot 1928, Bonnot and Ripley 1948, Bartholomew and Boolootian 1960, Orr and Poulter 1967, LeBoeuf *et al.* 1991, Westlake *et al.* 1997), standardized counting techniques for state-wide surveys were not implemented until 1996. For this reason some caution is warranted when attempting to evaluate population trend from the older data. Population trends have differed markedly at the major sites; therefore, each site is discussed separately.

Previously, Steller sea lions ranged to the Channel Islands in Southern California, primarily using San Miguel Island but also Santa Rosa Island, which were considered the southernmost rookeries and haulouts (Bonnot 1928, Rowley 1929). It appears that sea lions used these sites seasonally and bred in small numbers (Stewart *et al.* 1993). In the early and middle 20th century, perhaps 2,000 Steller sea lions occupied the Channel Islands (Bonnot and Ripley 1948). Numbers appear to have begun declining about 1938 (Bartholomew 1967), and no adults have been seen there since 1983 and no births recorded since 1982 (Stewart *et al.* 1993). Additionally, several rookery and haulout sites along the California coast, primarily south of Año Nuevo, have been abandoned, as well as a documented rookery at Seal Rocks near San Francisco (Bartholomew and Boolootian 1967, Bonnot 1928, Bonnot and Ripley 1948, Rowley 1929).

Numbers of non-pup Steller sea lions at the two central California sites, Año Nuevo and the Farallon Islands, are currently only about 20% of the levels reported between 1927 and 1964 (Table I-9). There appears to have been a particularly steep decline in the 1960s and 1970s. Counts appear to have recently stabilized or at least the rate of decline has lessened (Hastings and Sydeman 2002). Numbers of pups born on Año Nuevo declined from about 600 to 800 during the 1960s Le Boeuf *et al*. 1991, Orr and Poulter 1967) to 152 in 1999. However, between 1996 and 2004 the number of pups counted stabilized (P = 0.656). In 2004, 221 pups were counted at Año Nuevo. Recent pup production on the Farallons has been low (Hastings and Sydeman 2002) with a maximum of 22 pups counted in 2004. During the 1920s, the Farallon Islands and Año Nuevo were identified as the most important rookeries in California (Rowley 1929), with estimates of pup production at 400 and 625, respectively (Bonnot 1928).

Steller sea lions have been counted sporadically at the Sugarloaf/Cape Mendocino rookery and haulout during breeding seasons since 1927. Non-pup numbers appear to have been relatively stable, although highly variable, since 1996. The two highest counts were 900 in 1930 and 740 in 2001 suggesting that the current population is comparable to historical levels. Pups have been counted in recent years and numbers have increased (62 in 1996 to 131 in 2004; +12.9% per year, $R^2 = 0.725$, $P = 0.007$).

The Saint George Reef rookery, located near the California/Oregon border, appears to be at a fairly high level relative to historical measures and counts of non-pups have been stable, although variable, since 1990 (Table I-10). During 2004, 444 pups and 738 non-pups were counted at this site. Bonnot (1928) reported 1,500 Steller sea lions at Saint George Reef in 1927 and Bonnot and Ripley (1948) counted 700 animals in 1930. Pups have been counted since 1996 (except for 1997) and have increased (243 in 1996 to 444 in 2004; +9.8% per year, $R^2 = 0.703$, $P = 0.009$).

Statewide in California, total non-pup counts at these five major rookery and haulouts during the first half of the last century ranged from 4,500 to 5,600. The 2004 count at these same five sites was 1,578 non-pups and 818 pups suggesting that only about a third as many animals are currently present in the state. Nearly all of the reduction has occurred at the three southern sites. From 1996 through 2004, statewide non-pups numbers were stable, while pup numbers increased at 7.5% per year, $R^2 = 0.679$, $P = 0.112$).

An additional 1,418 Steller sea lions were counted during the 2002 survey at 41 haulout sites (with counts raging from 1 to 692 animals on these haulouts and with 15 sites with more than 25 animals) along the California coast between Saint George Reef and Año Nuevo Island. Steller sea lion abundance (all age classes) in California, based on 2002 pup counts at rookeries, was 3,209 animals (with the 4.5 pup multiplier) or 3,636 (with the 5.1 pup multiplier). However, 3,815 animals were actually counted during the 2002 survey.

7. Eastern DPS Overall

Overall, the eastern DPS has increased at over 3% per year since the 1970s, more than doubling in Southeast Alaska, British Columbia, and Oregon. The robustness of the observed positive trend for the eastern population over the past 25-30 years was confirmed by Bayesian trend analyses conducted by Goodman (Appendix 1). He estimated annual growth at 3.64% for nonpups in Oregon with a 95% confidence interval of 2.42 to 4.44% and concluded that there was an extremely low probability (0.01) that the actual growth rate was lower than 2% per year. For pups in Southeast Alaska he estimated annual growth at 3.13% (95% confidence interval of 2.29 to 3.95%). The probability of a growth rate below 1.5% per year was estimated at 0.1% for the Southeast counts.

Saint George Reef rookery and Sugarloaf rookery in northern California are near levels recorded early in the 20th century, and pup production has increased since 1996. This increase is probably at least partially the result of protective legislation, enacted in both the United States and Canada during the early 1970s, that reduced mortality at a time when the population was below carrying capacity. However, numbers of animals at the Año Nuevo rookery and the Farallon Islands in central California are substantially reduced (-90%) from those reported early in the 20th century (Bonnot 1928), despite legal protection from directed human take. The former haulout/rookery at San Miguel Island is now extinct, as are several other sites previously used in California (Rowley 1929). The reason for the large declines, since the mid-1900s, in southern and central California are not known. However, sympatric populations of other pinnipeds have grown greatly over the past 75 years (Stewart *et al.* 1993). In particular, a closely related species, the California sea lion (*Zalophus californianus*), has increased greatly from at most a few thousand in the 1920s (Bonnot 1928) to between 237,000 and 244,000 in 2004 (Carretta *et al.* 2005); some aspect of a competitive relationship may have been involved in the Steller sea lion decline. Changes in the ocean environment, particularly warmer water temperatures, have also been proposed as possible factors that favored California sea lions and other pinnipeds over Steller sea lions through changes in the distribution of favored prey (Bartholomew and Boolootian 1960).

The eastern population was subjected to substantial mortality by humans, primarily due to commercial exploitation and both sanctioned and unsanctioned predator control, (Bonnot 1928, Scheffer 1946, Rowley 1929, Bonnot and Ripley 1948, Pearson and Verts 1970, Bigg 1988, Scheffer 1950). Commercial exploitation occurred primarily in the 1800s and early 1900s while unsanctioned predator control probably persisted into the 1970s in some locations. Although not well documented, there is little doubt that numbers of Steller sea lions were greatly reduced in many locations.

Within the eastern DPS, 13 rookeries and about 85 major haulout sites currently exist from Cape Fairweather (58.8°N, 137.9°W) to Año Nuevo Island (37.1°N, 122.3°W). Populations associated with 12 of these rookeries have either increased or stabilized at relatively high levels in recent years. Total population size of the eastern DPS in 2002 was estimated to range between 45,000 and 51,000 animals of all ages (Table I-11). Additional surveys in California during 2003 and 2004 and in Southeast Alaska during 2005 suggest the population has continued to increase since the 2002 survey and likely exceeds 50,000 animals.

Conditions for Steller sea lions in the eastern DPS appear to be most favorable in the northern portion of their range. Southeast Alaska and British Columbia together account for nearly 82% of total pup production. All four rookeries founded in the past 25 years are located in northern Southeast Alaska at the northern extent of the population range. The southernmost portion of the range has contracted and the southernmost active rookery, at Año Nuevo Island, appears to have stabilized at a low population size. A somewhat similar change in Steller sea lion distribution and the establishment of new breeding sites have been noted in the northwestern Pacific, where the southern range limit moved northward by 500-900 km over the past 50 years and several new rookeries were established (Burkanov and Loughlin 2005).

Currently, no Steller sea lion rookeries exist within a geographical gap (993 km) between the Scott Islands Rookery off northwest Vancouver Island and Orford and Rogue Reef Rookeries in southern Oregon. It is possible that additional rookeries were once located along this coastline, and it would not be surprising to see new rookeries founded or re-established, as has occurred in Southeast Alaska, if the population continues to increase. Steller sea lion rookeries are normally located on remote, offshore islands or reefs and require adequate areas above high water levels where young pups can survive most weather conditions and adequate prey is available on a consistent basis within the foraging range of lactating females. Perhaps the limited availability of such sites has prevented the establishment of additional new rookeries.

During the 1970s the eastern DPS contained only about 10% of the total number of Steller sea lions in the U.S. With the large decline in the western DPS in conjunction with the increase in the east, this has changed dramatically with over half of U.S. Steller sea lions now belonging to the eastern DPS.

F. Habitat Characteristics and Use

Steller sea lions use a variety of marine and terrestrial habitats. Haulouts and rookeries tend to be preferentially located on exposed rocky shoreline and wave-cut platforms (Ban 2005, Call and Loughlin 2005). Some rookeries and haulouts are also located on gravel beaches. Rookeries are nearly exclusively located on offshore islands and reefs. Terrestrial sites used by Steller sea lions tend to be associated with waters that are relatively shallow and well-mixed, with average tidal speeds and gradual bottom slopes (Ban 2005). When not on land, Steller sea lions are seen near shore and out to the edge of the continental shelf and beyond.

1. Terrestrial habitat use

Female sea lions appear to select places for giving birth that are gently sloping and protected from waves (Sandegren 1970, Edie 1977). Pups normally stay on land for about two weeks, and then spend an increasing amount of time in intertidal areas and swimming near shore. Mothers spend more time foraging as pups grow older and less time on shore nursing (Milette and Trites 2003). Females with pups begin dispersing from rookeries to haulouts when the pups are about 2.5 months-of-age (Raum-Suryan *et al*. 2004, Maniscalco *et al.* 2002, 2006).

Haulout is the term used to describe terrestrial areas used by adult sea lions during times other than the breeding season and by non-breeding adults and subadults throughout the year. Sites used as rookeries in the breeding season may also be used as haulouts during other times of year. Some haulouts are used year-around while others only on a seasonal basis. Sea lions are sometimes seen hauled out on jetties and breakwaters, navigational aids, floating docks, and sea ice. Many animals also use traditional rafting sites, which are places where they rest on the ocean surface in a tightly packed group (Bigg 1985, NMFS unpublished data).

Although rookeries and haulouts occur in many types of areas, sea lions display strong site fidelity to specific locations from year to year. Factors that influence the suitability of a particular area may include substrate, exposure, proximity to food resources, oceanographic conditions, tradition of use, and season (Calkins and Pitcher 1982, Ban 2005), as well as the extent and type of human activities in the region (Johnson *et al*. 1989). Thermoregulatory factors may play an important role in site selection (Gentry 1970, Sandegren 1970).

2. Marine habitat use

Telemetry studies show that in winter adult females may travel far out to sea into water greater than 1,000 m deep (Merrick and Loughlin 1997), and juveniles less than 3 years of age travel nearly as far (Loughlin *et al*. 2003). The Platforms of Opportunity (POP) database maintained by NMFS shows that sea lions commonly occur near and beyond the 200 m depth contour (Kajimura and Loughlin 1988, NMFS POP data). Some individuals may enter rivers in pursuit of prey (Jameson and Kenyon 1977). In summer while on breeding rookeries, adult females attending pups tend to stay within 20 nm of the rookery (Calkins 1996, Merrick and Loughlin 1997).

Studies using satellite-linked telemetry have provided detailed information on movements of adult females and juveniles (Table I-12). Merrick and Loughlin (1997) found that adult females tagged at rookeries in the central Gulf of Alaska and Aleutian Islands in summer made short trips to sea (mean distance 17 km, maximum 49 km) and generally stayed on the continental shelf. In winter, adult females ranged more widely (mean distance 133 km, maximum 543 km) with some moving to seamounts far offshore. Most of the pups tracked during the winter made relatively short trips to sea (mean distance 30 km), but one moved 320 km from the eastern Aleutians to the Pribilof Islands. Adult females with satellite transmitters in the Kuril Islands in summer made short at-sea movements similar to those seen in Alaska (Loughlin *et al.* 1998).

Behavioral observations indicate that lactating females spend more time at sea during winter than in the summer. Attendance cycles (consisting of one trip to sea and one visit on land) averaged about three days in winter and two days in summer (Trites and Porter 2002, Milette and Trites 2003, Trites *et al.* 2006b, Maniscalco *et al.* 2006). Time spent on shore between trips to sea averaged about 24 hours in both seasons. The winter attendance cycle of dependent pups and yearlings averaged just over two days, suggesting that they do not accompany their mothers on foraging trips (Trites and Porter 2002, Trites *et al.* 2006b). Foraging trips by mothers of yearlings were longer on average than those by mothers of pups (Trites and Porter 2002).

Additional studies on immature Steller sea lions indicate three types of movements: long-range trips (greater than 15 km and greater than 20 h), short-range trips (less than 15 km and less than 20 h), and transits to other sites. Long-range trips started around 9 months of age and may occur most frequently around the time of weaning, while short-range trips happened almost daily (0.9 trips/day, n = 426 trips). Transits began as early as 2.5-3 months of age, occurred more often after 9 months of age, and ranged between 6.5 - 454 km (ADF&G unpublished data, Loughlin *et al.* 2003). Some of the transit and short-range trips occurred along shore, while long-range trips were often offshore, particularly as ontogenetic changes occurred.

Overall, available data suggest two types of distribution at sea by Steller sea lions: 1) less than 20 km from rookeries and haulout sites for adult females with pups, pups, and juveniles, and 2) much larger areas (greater than 20 km) where these and other animals may range to find optimal foraging conditions once they are no longer tied to rookeries and haulout sites for nursing and reproduction. Loughlin (1993) observed large seasonal differences in foraging ranges that may have been associated with seasonal movements of prey, and Merrick (1995) concluded on the basis of available telemetry data that seasonal changes in home range were related to prey availability.

3. Designated critical habitat

On August 27, 1993 NMFS published a final rule to designate critical habitat for the threatened and endangered populations of Steller sea lions (58 FR 45269). The areas designated as critical habitat for the Steller sea lion were determined using the best information available at the time (see regulations at 50 CFR part 226.202), including information on land use patterns, the extent of foraging trips, and the availability of prey items. Particular attention was paid to life history patterns and the areas where animals haul out to rest, pup, nurse their pups, mate, and molt. Critical habitat areas were finally determined based upon input from NMFS scientists and managers, the Steller Sea Lion Recovery Team, independent marine mammal scientists invited to participate in the discussion, and the public (Figures I-9 and I-10).

Physical and biological features of Steller sea lion critical habitat:

Two kinds of marine habitat were designated as critical: “aquatic zones” around rookeries and haulouts and three special aquatic feeding areas in Alaska (Figures I-9 and I-10).

Critical habitat includes so called “aquatic zones” that extend 3,000 feet (0.9 km) seaward in State and Federally managed waters from the baseline or basepoint of each major rookery and major haulout in Alaska that is east of 144 deg. W. longitude and that extends 3,000 feet (0.9 km) seaward in State and Federally managed waters from the baseline or basepoint of each major rookery in California and Oregon. Aquatic zones in the U.S. breeding range of the western stock extend 20 nm (37 km) seaward in State and Federally managed waters from the baseline or basepoint of each major rookery and major haulout in Alaska that is west of 144 deg. W. longitude. These "aquatic zones" around rookeries and haulout sites were chosen based on evidence that many foraging trips by lactating adult females in summer may be relatively short (20 km or less; Merrick and Loughlin 1997). Also, mean distances for young-of-the-year in winter may be relatively short (about 30 km; Merrick and Loughlin 1997, Loughlin *et al*. 2003). These young animals are just learning to feed on their own, and the availability of prey in the vicinity of rookeries and haulout sites may be crucial to their transition to independent feeding after weaning. Similarly, haulouts around rookeries are important for juveniles, because most juveniles are found at haulouts not rookeries. Evidence indicates that decreased juvenile survival may be an important proximate cause of the sea lion decline (York 1994, Chumbley *et al*. 1997) and that the growth rate of individual young sea lions was depressed in the 1980s (Calkins and Goodwin 1988). These findings are consistent with the hypothesis that young animals were nutritionally stressed. Furthermore, young animals are almost certainly less efficient foragers and may have relatively greater food requirements, which, again, suggests that they may be more easily limited or affected by reduced prey resources or greater energetic requirements associated with foraging at distant locations. Therefore, the areas around rookeries and haulout sites must contain essential prey resources for at least lactating adult females, young-of-the-year, and juveniles, and those areas were deemed essential to protect.

Second, three "special aquatic foraging areas in Alaska" were chosen based on 1) at-sea observations indicating that sea lions commonly used these areas for foraging, 2) records of animals killed incidentally in fisheries in the 1980s, 3) knowledge of sea lion prey and their life histories and distributions, and 4) foraging studies. In 1980, Shelikof Strait was identified as a site of extensive spawning aggregations of pollock in winter months. Records of incidental take of sea lions in the pollock fishery in this region provide evidence that Shelikof Strait is an important foraging site (Loughlin and Nelson 1986, Perez and Loughlin 1991). The southeastern Bering Sea north of the Aleutian Islands from Unimak Island past Bogoslof Island to the Islands of Four Mountains is also considered a site that has historically supported a large aggregation of spawning pollock, and is also an area where sighting information and incidental take records support the notion that this is an important foraging area for sea lions (Fiscus and Baines 1966, Kajimura and Loughlin 1988). Finally, large aggregations of Atka mackerel are found in the area around Seguam Pass. These aggregations have supported a fishery since the 1970s and are in close proximity to a major sea lion rookery on Seguam Island and a smaller rookery on Agligadak Island. Atka mackerel are an important prey of sea lions in the central and western Aleutian Islands. Records of incidental take in fisheries also indicate that the Seguam area is important for sea lion foraging (Perez and Loughlin 1991).

G. Vital Rates

Changes in the size of a population are ultimately due to changes in one or more of its vital demographic rates. Inputs to the population are provided by reproduction of adults (e.g., birth rates; probability that a female of a given age will give birth to a pup each year) and immigration. Outputs from the population include those that leave the population through emigration or death, which can also be inversely described by rates of adult and juvenile survivorship. Estimates of vital rates are best determined in longitudinal studies of marked animals, but can also be estimated through population models fit to time series of counts of sea lions at different ages or stages (e.g., pups, non-pups).

1. Survival

Causes of pup mortality include drowning, starvation caused by separation from the mother, disease, parasitism, predation, crushing by larger animals, biting by other sea lions, and complications during parturition (Orr and Poulter 1967, Edie 1977, Maniscalco *et al.* 2002, 2006 ADF&G and NMFS unpublished data). Older animals may die from starvation, injuries, disease, predation, subsistence harvests, intentional shooting by humans, entanglement in marine debris, and fishery interactions (Merrick *et al*. 1987).

Calkins and Pitcher (1982) estimated mortality rates using life tables constructed from samples collected in the Gulf of Alaska in 1975-1978. The estimated overall mortality from birth to age 3 was 0.53 for females and 0.74 for males; i.e., 47% of females and 26% of males survived the first 3 years of life. Annual mortality rate decreased from 0.132 for females 3-4 years of age, to 0.121 for females 4-5 years old, to 0.112 for females 5-6 years old, and to 0.11 by the seventh year; it remained at about that level in older age classes. Male mortality rates decreased from 0.14 in the third year to 0.12 in the fifth year. Females may live to 30 years and males to about 20 (Calkins and Pitcher 1982).

York (1994) produced a revised life table for female Steller sea lions using the same data as Calkins and Pitcher (1982) but a different model. The estimated annual mortality from York's life table was 0.22 for ages 0-2, dropping to 0.07 at age 3, then increasing gradually to 0.15 by age 10 and 0.20 by age 20. Population modeling suggested that decreased juvenile survival likely played a major role in the decline of sea lions in the central Gulf of Alaska during 1975-1985 (Pascual and Adkison 1994, York 1994, Holmes and York 2003). This is supported by field observations on two major rookeries in the western DPS. The proportion of juvenile sea lions counted at Ugamak Island was much lower in 1985 and 1986 than during the 1970s, suggesting that the mortality of pups/juveniles increased between the two periods (Merrick *et al*. 1988). A decline in the proportion of juvenile animals also occurred at Marmot Island during the period 1979-1994. A very low resighting rate for pups marked at Marmot Island in 1987 and 1988 suggested that the change in proportions of age classes was due to a high rate of juvenile mortality (Chumbley *et al.* 1997).

2. Reproduction

Detailed information on Steller sea lion reproduction has been obtained from examinations of reproductive tracts of dead animals. These studies have shown that female Steller sea lions reach sexual maturity at three to six years of age and may produce young into their early 20s (Mathisen *et al.* 1962, Pitcher and Calkins 1981). Adult females normally ovulate once each year, and most breed annually (Pitcher and Calkins 1981). Males reach sexual maturity between three and seven years of age and physical maturity by age 10 (Perlov 1971, Pitcher and Calkins 1981). Males are territorial during the breeding season, and one male may breed with several females. Thorsteinson and Lensink (1962) found that 90% of males holding territories on rookeries in the western Gulf of Alaska were between nine and 13 years of age, while Raum-Suryan *et al.* (2002) found that males marked on Marmot Island as pups first became territorial at 10 and 11 years of age.

One of the key parameters governing population growth is reproductive output (birth rate). Reproductive output may be affected by nutrition, diseases, contaminants, and other factors (Merrick *et al.* 1987, Pitcher *et al.* 1998) that are discussed below and in detail in Sections III and IV.

Samples collected in the Gulf of Alaska in the mid-1980s, showed evidence of reproductive failure and reduced rates of body growth that were consistent with nutritional stress (Calkins and Goodwin 1988). During the 1970s and 1980s, 97% of sexually mature females in the western DPS were pregnant in early gestation. However, the percentage of females that carried their pregnancy to late gestation fell to 67% during the 1970s and to 55% in the 1980s indicating that a considerable amount of intrauterine mortality and/or premature births occurred after implantation; the difference was statistically insignificant, yet the statistical power to detect the difference was less than 0.50 (Pitcher *et al.* 1998). Lactating females were less likely to become pregnant than non-lactating females during the early decline, indicating that the energetic stress of nursing while being pregnant with another pup may have prevented some females from giving birth each year (Pitcher *et al.* 1998. The difference in pregnancy rates of the lactating females between the 1970s (63%) and 1980s (30%) was significant ($P = 0.059$). Examination of reproductive tracts from female Steller sea lions killed near Hokkaido, Japan in 1995-96 showed that the pregnancy rate for females that had ovulated was 88% (23/26) (Ishinazaka and Endo 1999). These samples were collected in January and February, so this estimated pregnancy rate was much higher compared to the late-term rates of 55-67% estimated for sea lions from Alaska.

It is important to obtain current estimates of birth rate since the most recent estimates are from 1985-86. Historically, birth rates were estimated from the examination of reproductive tracts from collected animals, which is not currently feasible. Estimates will need to be derived from alternative techniques such as mark-resight estimation, analysis of reproductive hormone levels in feces or tissue samples, or population modeling.

Better body condition was found to increase the probability that a female would maintain pregnancy. Comparatively low birth rates for females from the western DPS during the 1970s and 1980s (Pitcher and Calkins 1981) coupled with elevated embryonic and fetal mortality appear to have contributed to decreased reproductive performance during the period of early decline (Pitcher and Calkins 1981, Calkins and Goodwin 1988, Pitcher *et al.* 1998, NMFS 1998a, 1998b, 2000). Age-structured models fit to observed time series of pup and non-pup counts suggest that declines in reproductive performance of females in the western DPS continued into the 1990s in some or major parts of the Alaskan range (Holmes and York 2003, Fay 2004, Holmes *et al.* 2007), but may have increased in the late 1990s and 2000s in most areas (Winship and Trites 2006). However, Holmes *et al.* (2007) make a strong case that at least in the central GOA, natality rates have continued to decline in the 1990s and 2000s from pre-decline levels (see below).

3. Demographic modeling

Estimates of birth rates and survival rates

Demographic analysis of age distribution information has been used to estimate demographic rates in an attempt to identify the combination of changes in birth and survival rates that might account for the observed past changes in pup and non-pup numbers across the range of the western DPS in Alaska. These analyses are hampered by sparseness and spottiness of data. The only large sample of Steller sea lion that has been submitted for tooth-ring age determinations was a collection from one location, Marmot Island, in a very restricted time frame (2 years in the 1970s) in circumstances that primarily sampled breeding age animals at a rookery. The females in the same sample were examined for reproductive status. York (1994) created a life table estimate from these data by assuming (1) that this collection was representative of age distributions and reproductive frequencies in the entire population, (2) that the population was in stable age distribution, and (3) that there was no population growth.

At a much less detailed level, some censusing techniques distinguish between pups and non-pups in the counts at many rookeries. There are over 30 rookeries that have been censused over the years in a regular, but much less frequent than annual, rotation. If assumptions are made about the tendency of non-breeding animals of breeding age, as well as animals of below breeding age, to be present on rookeries and be included in the counts (this is not actually known yet, and is a matter of ongoing investigation in the analysis of sighting records of branded animals), the time series of counts of pups and non-pups allow some inference about crude per capita birth rates to adults, crude per capita survival rates of the adults, and rates of survival from birth until recruitment to the breeding segment of the population.

York (1994) concluded from her life table analysis that the population decline observed in the 1980s at Marmot likely was primarily due to a large drop in juvenile survivorship compared to the 1970s, a conclusion also reached by Pascual and Adkison (1994). Holmes and York (2003) extended these analyses of central Gulf of Alaska sea lions through the late 1990s and added an index of juvenile recruitment to the model. Their results, along with those of Fay (2004), indicated a drop in juvenile survivorship from the 1970s to the 1980s, and that the slower decline rate in the 1990s was associated with increases in juvenile and adult survivorship compared to the 1980s. However, their analyses also showed a decline in natality (birth rates plus pup mortality through one month of age) that began in the late 1970s and early 1980s (Holmes and York 2003, Holmes *et al.2007*).

Holmes and York (2003) and Holmes *et al.* (2007) fit an age-structured model with temporally varying vital rates to time series of pup and non-pup counts, and an index of juvenile recruitment to the sea lion population in the central Gulf of Alaska (CGOA). Holmes *et al.*

(2007:2214) concluded "The model fits indicate that birth rate in the CGOA steadily declined from 1976 to 2004. Over the same period, survivorship first dropped severely in the early 1980s…then survivorship steadily recovered." Findings from their study indicate that in 2004, the birth rate in the central Gulf of Alaska was 36% lower than it had been in the 1970s. However, their model indicated that the survivorships of juveniles and adults were very near or slightly above the levels in the 1970s. Key insights from this study are that:

1) the current relative stability is very dependent on adult survival;
2) declining birth rates may be a problem for western DPS sea lions across the Gulf of Alaska and into the Aleutians.

A multiple hypothesis modeling exercise by Wolf *et al.* (2006) highlighted the likelihood that multiple mechanisms were at work in influencing the decline of the western population of Steller sea lions. They found that total prey availability could have affected fecundity and that the Pollock fraction in the environment (by CPUE weight) affects pup recruitment. They did not attempt to answer the mechanistic question but acknowledged possibilities. Ultimately Wolf *et al*. (2006) suggested that both the quantity and the quality of available food are important but that the relationship may be complicated.

Fay (2004) and Winship and Trites (2006) broadened the geographic scope by estimating time series of vital rates for metapopulations and for subpopulations at each rookery in the Gulf of Alaska and Aleutian Islands from 1978-2002. Results of these studies suggest that the changes in vital rates responsible for the declines likely varied among subpopulations and with time. Juvenile and adult survival rates appear to have been lowest during the 1980s for many, but not all subpopulations, while juvenile survival in the western Aleutians appears to have been lower during the 1990s than during the 1980s. With regard to changes in natality, Fay (2004) found evidence of DPS-wide declines in birth rates beginning in the early 1980s with little or no rebound through 2000. Winship and Trites (2006) found declines in natality in the central Gulf of Alaska (similar to Holmes and York 2003, Fay 2004), but not elsewhere in the range of the western Steller sea lion.

The studies attempting to estimate past demographic rates were motivated in part by a hope that these could shed light on the various possible causes for the changes in vital rates responsible for the population decline. In this, the retrospective studies have been largely inconclusive. One exception is the study by Hennen (2006) which found an association between rate of by-rookery decline and the fishing activity around the respective rookies, for the period of the 1980s but not continuing into the 1990s. Hennen (2006) did not investigate how this effect might have been partitioned among birth rates and survival rates of various age classes.

Models extrapolating the population into the future

Population viability analysis (PVA) attempts to predict the probability of a population going extinct, or crossing a specified threshold, over a specified period. Four simulation models of varying complexity have been constructed to assess the likelihood that Steller sea lions will go extinct in western Alaska (see Appendix ; York *et al.* 1996, Gerber and VanBlaricom 2001, Winship and Trites 2006). Some of the models treated each rookery as independent populations, while others considered metapopulations (i.e., groups of rookeries), or combined counts from all rookeries between the eastern Gulf of Alaska and the western tip of the Aleutian Islands into a single population estimate.

The rookery-based and metapopulation modeling requires assumptions about rates of migration and recolonization. Those rates are not presently known, though they are the subject of ongoing monitoring of branded animals. Each of the models used information about rates of population change that occurred in the past to infer what might happen to sea lion populations in the future.

York *et al.* (1996) developed three models corresponding to three spatial scales (a rookery model, a cluster of rookeries model, and an aggregate model for the Kenai – Kiska area). They used a model of exponential growth randomly changing annually from a distribution that remains constant over time to model counts of adult female sea lions made at the peak of the breeding season. Using counts from 1976-1994 in their retrospective analysis, the rookery model predicted that the median number of adult females on each rookery between Kenai-Kiska would decline to fewer than 50 animals with 80% of the rookeries disappearing within 100 years, and fewer than 5,000 females remaining by 2015. However, some sites (Akutan, Clubbing Rocks, Ugamak Island, Sea Lion Rocks, and Akun Island) were predicted to persist beyond 100 years despite extinctions at other rookeries. The cluster model grouped Steller sea lion rookeries into 5 clusters and found a relatively high probability of persistence of the western DPS due to positive growth rates in the western Gulf of Alaska cluster. However, pooling all rookery counts within the Kenai – Kiska area to form a single breeding population, and using the rates of decline that occurred from 1976-1994 to project the future, resulted in a predictable continued decline of the western DPS. York *et al.* (1996) concluded that there was no indication that the entire population would likely go extinct within 30 years, but that populations on some rookeries would probably be reduced to low levels (fewer than 200 adult females). The rookery-based model predicted the longest mean persistence time for the Kenai-Kiska population, while the geographic model (pooling all rookery counts) predicted the shortest.

Gerber and VanBlaricom (2001) used count data from 1965-1997 to develop two viability models that evaluated the sensitivity of extinction risk to various levels of stochasticity, spatial scale, and density dependence, again assuming that annual variation was the predominant process driver. The first was a metapopulation simulation model that suggested a median time to extinction of about 85 years, based on the dynamics of groups of rookeries in the Central Gulf of Alaska, Western Gulf of Alaska, Eastern Aleutian Islands, and the Central Aleutian Islands. The second model was exploratory rather than tied strictly to the retrospective analysis and considered population size and population growth rates corresponding to the lowest 5% of the frequency distribution of likely growth rates. This model suggested the time to extinction was about 62 years. Gerber and VanBlaricom (2001) concluded that results from their analysis were consistent with a population threatened with extinction.

Winship and Trites (2006) used counts of both pups and non-pups from 33 rookeries between 1978 and 2002 to estimate the combination of birth and survival rates operating during the population decline. They then projected each of the 33 rookery populations into the future using these estimated site-specific life tables (with associated uncertainties). Using Bayesian statistical methods to quantify uncertainty, Winship and Trites (2006) explored 3 scenarios that incorporated different assumptions about carrying capacities and the presence or absence of density-dependent regulation. Results of all 3 scenarios indicated an overall low risk of extirpation of Steller sea lions as a species in western Alaska in the next 100 years. However, most rookeries had high probabilities of going extinct if trends observed in the 1990s continued — while fewer were predicted to go extinct if trends observed since the late 1990s persisted. All simulations identified two clusters of contiguous rookeries that had relatively low risks of extinction if their dynamics continued to be independent of the rest— the Unimak Pass area in the western Gulf of Alaska / eastern Aleutian Islands, and the Seguam – Adak region in the central Aleutian Islands. Risks of rookeries going extinct were particularly small when density-dependent compensation in birth and survival rates was assumed. Winship and Trites (2006) did not include the more drastic decline rates from the 1980s in their analysis, thereby treating this time period as a catastrophic event which was unlikely to occur again.

Goodman (Appendix) also used a Bayesian framework to quantify uncertainty in model parameters and propagate this through the risk calculation. However, he treated the western Steller sea lions as a single population by combining counts made at all rookeries and regions of western Alaska, and treated the dominant environmental variation as occurring on a larger than annual time scale. Population-wide estimates were available for six time periods over the 46 years that sea lions have been counted (i.e., 1958, 1977, 1985, 1989, 2000 and 2004). In this analysis the probability of sea lions persisting for 100 to 500 years depended upon assumptions about the past operation of anthropogenic factors that will not play such a large role in the future. These specific assumptions were a joint product of a subgroup of the recovery team. Overall, this model suggested significant probabilities of sea lions declining below a threshold of 4,743 individuals (i.e., quasi-extinction) for the population as a whole within 100 years. This model allows a parsing of how unfavorable parameter values and uncertainty about parameter values both play a role in the calculated risk.

There is some degree of consistency between the predictions of all four sets of PVA models (Appendix ; York *et al.* 1996, Gerber and VanBlaricom 2001, Winship and Trites 2006) due in large part to their use of some of the same base population data and to the fundamental assumption of all PVA models that populations will continue to behave as they have in the past after correction for factors that will be different in the future. As such, sea lion populations (i.e., individual rookeries, clusters of rookeries, or the entire western DPS) that declined at fast rates were predicted to go extinct sooner than populations that had declined slowly. Results from the four Paves conducted to date indicate that the western Steller sea lions have a high probability of declining to a low level if they are considered as a single homogeneous population (by combining all rookery counts and assuming an overarching population trend). However, the prognosis for the species is considerably more optimistic if each of the 33 rookeries is considered as distinct, independent populations with its own probability of persistence, and assuming that differing environmental factors around the respective rookeries remain stationary for the long term (as opposed to the possibility of rolling declines). Under this scenario, PVA models at a spatial scale smaller than the DPS predict that many rookeries will go extinct, but that the species will persist on the time frame considered, most especially if assumed density dependence plays a positive role.

The large potential influence of assumed density dependence is a common feature in the literature of PVA applications, but the statistical estimation of the strength of operation of density dependence in any particular population is notoriously problematic. Density dependence has not been established empirically in the dynamics observed in the Steller sea lion western DPS over the past 40 years.

H. Feeding Ecology

The feeding ecology of Steller sea lions has been described in detail in the initial Steller Sea Lion Recovery Plan (NMFS 1992), and the ESA Biological Opinion on Groundfish Fisheries in the Gulf of Alaska, and Bering Sea and Aleutian Islands area (NMFS 2000). Readers are referred to those documents for additional information.

1. Foods consumed

Steller sea lions are generalist predators that eat a variety of fishes and cephalopods (Pitcher 1981, Calkins and Goodwin 1988, NMFS 2000, Sinclair and Zeppelin 2002), and occasionally other marine mammals and birds (Gentry and Johnson 1981, Pitcher and Fay 1982, Daniel and Schneeweis 1992, Calkins 1988).

The diet of Steller sea lions in the eastern part of their range was not well studied prior to the early 1990s. In California and Oregon they are known to have eaten rockfish, hake, flatfish, salmon, herring, skates, cusk eel, lamprey, squid, and octopus (Fiscus and Baines 1966, Jameson and Kenyon 1977, Jones 1981, Treacy 1985, Brown *et al.* 2002). Principal prey in British Columbia has included hake, herring, octopus, Pacific cod, rockfish, and salmon (Spalding 1964, Olesiuk *et al.* 1990). The most commonly identified prey items in Southeast Alaska were walleye pollock, Pacific cod, flatfishes, rockfishes, herring, salmon, sand lance, skates, squid, and octopus (Calkins and Goodwin 1988, Trites *et al.* 2007a).

Considerable effort has been devoted to describing the diet of Steller sea lions in the Gulf of Alaska, Aleutian Islands, and Bering Sea (Table I-13). In the mid 1970s and mid 1980s, Pitcher (1981; n = 250) and Calkins and Goodwin (1988; n = 178) described Steller sea lion diet in the Gulf of Alaska by examining stomach contents of animals collected for scientific studies. Walleye pollock was the principal prey in both studies; octopus, squid, herring, Pacific cod, flatfishes, capelin, and sand lance were also consumed frequently. Stomachs of Steller sea lions collected in the central and western Bering Sea in March-April 1981 contained mostly pollock and also Pacific cod, herring, sculpins, octopus, and squid (Calkins 1998).

Merrick and Calkins (1996) analyzed Kodiak Island region sea lion stomach contents (n = 263) data from the 1970s and 1980s for seasonal patterns of prey use. They found a significant seasonal difference in diet for the 1970s. Walleye pollock was the most important prey in all seasons except summer in the 1970s, when the most frequently eaten prey type was small forage fishes (capelin, herring, and sand lance). No significant seasonal differences were found in the 1980s. Researchers noted that, overall, small forage fishes and salmon were eaten almost exclusively during summer, while other fishes and cephalopods were eaten more frequently in spring and fall.

Since 1990, additional information on Steller sea lion diet in Alaska has been obtained by analyzing scats collected on rookeries and haulouts (Merrick *et al.* 1997, NMFS 2000, Sinclair and Zeppelin 2002). Scat data, like stomach contents, may be biased (e.g., prey species may have hard parts that are more or less likely to make it though the digestive tract; see Cottrell and Trites 2002, Tollit *et al.* 2003, 2004a), but they allow a description of prey used over a wide geographic range from Kodiak Island through the western Aleutian Islands and for both summer and winter (Table I-13). Analysis of scats collected in the 1990s showed that pollock continue to be a dominant prey in the Gulf of Alaska and that Atka mackerel was the most frequently occurring prey in central and western Aleutian Islands scats. Pacific cod has also been an important food, especially in winter in the Gulf of Alaska, while salmon was eaten most frequently during summer months. Results also indicated a wide variation; certain species that appear to be minor dietary items when data are tabulated for large regions may actually be highly ranked prey for specific rookeries and seasons.

At the far western end of the Steller sea lion range, Atka mackerel, sand lance, rockfish, and octopus were identified as important foods at the Kuril Islands in collections made in 1962 (Panina 1966), and pollock, Pacific cod, saffron cod, cephalopods, and flatfish were the main prey of 62 animals collected near Hokkaido, Japan in 1994 - 1996 (Goto and Shimazaki 1998).

NMFS (2000) compiled all the available data on prey occurrence in stomach contents samples for the eastern and western Steller sea lion populations for the 1950s - 1970s and the 1980s. For both populations the occurrences of pollock, Pacific cod, and herring were higher in the 1980s than in the 1950s -1970s. These results suggest that the dominance of pollock in the Steller sea lion diet over much of its range may have changed over time. However, studies completed prior to the mid-1970s had small sample sizes and more limited geographic scope. As such, caution should be exercised when extrapolating from these limited samples to a description of the diet composition of Steller sea lions in the 1950s - 1970s.

Stomach contents analysis indicates that Steller sea lions have a mixed diet. Although it is not uncommon to find stomachs that contain only one prey species, most collected stomachs contained more than one type of prey (Merrick and Calkins 1996, Calkins 1998). Merrick and Calkins (1996) found that the probability of stomachs containing only pollock was higher for juveniles than for adults, and small forage fish were eaten more frequently by juveniles while flatfish and cephalopods were eaten by adults more frequently.

Steller sea lion scat and stomach contents data have not been extensively examined for possible sex-related differences in diet. However, Trites and Calkins (unpublished data) collected scat on three rookeries and a nearby male haulout and found that salmon and herring dominated the summer scats of lactating females, while pollock and rockfish dominated the scat of breeding-age males.

2. Prey characteristics

The primary prey of Steller sea lions are fish and cephalopods, which tend to have a broad, but predictable range in temporal, spatial, and seasonal nearshore availability. Typically, many prey species make predictable seasonal migrations from pelagic to nearshore waters where they form large spawning concentrations. Prey is then further concentrated by local transition boundaries such as frontal zones and bathymetric features such as submarine channels (Sinclair *et al.* 1994). Steller sea lions appear to have the foraging flexibility to take advantage of both the predictable behavioral traits of these prey species (Sigler *et al.* 2004), as well as the localized oceanographic conditions that enhance prey concentrations (Sinclair and Zeppelin 2002, Trites *et al.* 2006a). Steller sea lions are able to respond to changes in prey abundance. An example is the increase in consumption of arrowtooth flounder in the Gulf of Alaska between the 1970s (Pitcher 1981) and the 1990s (Sinclair and Zeppelin 2002). Another example is the geographic variation in diet observed during the 1980s and 1990s; east to west the primary prey varies from Pacific hake (Brown *et al.* 2002) to walleye pollock and then to Atka mackerel (Sinclair and Zeppelin 2002).

Prey species can be grouped into those that tend to be consumed seasonally, when they become locally abundant or aggregated when spawning (e.g., herring, Pacific cod, eulachon, capelin, salmon and Irish lords), and those that are consumed and available to sea lions more or less year-round (e.g., pollock, cephalopods, Atka mackerel, arrowtooth flounder, rock sole and sand lance, based on Pitcher 1981, Calkins and Goodwin 1988, Sinclair and Zeppelin 2002, Trites *et al.* 2007a). Some of the seasonal prey species occur most frequently in summer and fall (e.g., salmon and Irish lords) or winter and spring (e.g., herring, Pacific cod, eulachon, capelin). There are also significant regional differences in the occurrences of some species (e.g., Atka mackerel are only in the Aleutian Islands, and arrowtooth flounder occur in the Gulf of Alaska).

Prey size varies greatly ranging from several centimeters in length for species such as sandlance and capelin to over 60 cm in length such as salmon, skates, Pollock and cod. Remains of pollock exceeding 70 cm in length have been recovered in Steller sea lion scats (Tollit *et al.* 2004b, Zeppelin *et al.* 2004). Walleye pollock otoliths recovered from stomachs collected in the Bering Sea and Gulf of Alaska have shown that all age classes of sea lions eat a wide range of sizes (Calkins and Goodwin 1988, Frost and Lowry 1986, Lowry *et al.* 1989, Merrick and Calkins 1996, Calkins 1998).

3. Foraging behavior based on telemetry studies

Adults

Limited data are available concerning the foraging behavior of adult Steller sea lions. Adult females alternate trips to sea to feed with periods on shore when they haul out to rest, care for pups, breed, and avoid marine predators. Conversely, territorial males may fast for extended periods during the breeding season when they mostly remain on land (Spalding 1964, Gentry 1970, Withrow 1982, Gisiner 1985). Females with dependent young are constrained to feeding relatively close to rookeries and haulouts because they must return at regular intervals to feed their offspring.

Merrick *et al.* (1994) and Merrick and Loughlin (1997) present information on the dive characteristics and foraging behavior of a small sample of adult Steller sea lions in Alaska; Loughlin *et al.* (1998) provided similar information for the Kuril Islands, Russia. Merrick *et al.* (1990) and Brandon (2000) presented information on attendance behavior of adult females with VHF radio-transmitters in the Kuril Islands and Alaska, respectively. Trites and Porter (2002) and Milette and Trites (2003) documented attendance patterns from behavioral observations. These studies showed that during the breeding season, adult female Steller sea lions generally spent about half their time at sea on relatively brief (about 0.8 days) foraging trips. Dives tended to be shallow (mean = 21 meters), brief (mean = 1.4 minutes), and frequent (about 13 per hour) (Table I-14). Observations during winter showed that females with suckling yearlings (19-21 months of age) had feeding trips of about 2.5 days while those with young-of-the-years (7-9 months of age) had trips lasting 2.0 days ; time on shore for lactating females averaged 15.4 hours (Trites and Porter 2002). Merrick and Loughlin (1997) found that during summer adult females made trips to sea that averaged 17 km from the rookery (range 3-49 km; SE = 4.6; Table I-12). Outside of the breeding and pupping season, movements may be less constrained although animals still return to coastal haulouts to rest. For adult females tracked during winter by Merrick and Loughlin (1997), the mean trip duration was 204 hours and average distance moved offshore was 133 km (range 5-543 km; SE = 59.9).

In Southeast Alaska, adult females with pups made relatively brief foraging trips (mean 19.1 hr) while those with yearlings or without pups were much longer in duration; during winter female trips to sea had a mean of 56.1 hours with a maximum of 169 hours. (Swain 1996). Those females with pups remained within 20 nm of the rookery and mean foraging distance from the Hazy Island and Forrester Island rookery complex was 14.5 km offshore (Calkins 1996).

Additional research integrating three separate electronic devices has provided some fine-scale information on Steller sea lion foraging. The combined data (collected from a stomach temperature transmitter that indicates when Steller sea lions ingest prey, a data logger that records depth and velocity, and a satellite-linked time-depth recorder (SDR) to determine locations) provide insights to when and where Steller sea lions may be foraging. Andrews *et al.* (2002) used this approach on adult females in summer at Forrester Island (SE) and Seguam Island (BSAI) in 1994 and 1997; the data indicated nearly all prey ingestion occurred when animals repeatedly exhibited deep dives (greater than 10m), and that prey was ingested during all at-sea trips that included such foraging dives. However, long periods of time often elapsed and large distances were covered between successful foraging events. Adult females began foraging dives greater than 10 m within 8-26 minutes after departing a rookery, yet the first prey was not ingested until 0.9 to 5.1 hours after departure.

Juveniles

The need to understand the behavior of juvenile Steller sea lions has focused research effort in recent years and resulted in a relatively large sample data set for animals less than three years of age (Loughlin *et al.* 2003, Raum-Suryan *et al.* 2004, Rehberg 2005, Pitcher *et al.* 2005, Fadely *et al.* 2005) (Tables I-12 and I-14). In general, juveniles in their second year are capable of diving to adult depths but tend not to as often as older animals (Loughlin *et al.* 2003, Rehberg 2005). Rehberg (2005) found that young-of-year sea lions also tend to increase the greater relative proportion of their swimming and diving behavior from diurnal to nocturnal periods. Mean dive depth and duration increases with age and is predicted to increase in a positive relationship with body mass up to about 10 years of age (Pitcher *et al.* 2005). Tagged young-of-the-year animals during winter made trips offshore and along shore that averaged 15 hours long and extended to an average of 30 km (range 1-320 km; SE = 14.5). Loughlin *et al.* (2003) defined three types of movements that vary with age and body mass for juvenile Steller sea lions at sea: (1) transits between land sites with a mean distance of 66.6 km; (2), long-range trips (less than 15 km and greater than 20 hours); and (3) short-range trips (less than 15 km and less than 20 hours). Likewise, Raum-Suryan *et al.* (2002) reported that greater than 90% of round trips were less than 15 km from haul-outs and 84% were less than 20 hours in duration.

4. Nutritional requirements

The amount of food required to provide for energetic needs can vary greatly depending on the energy content of the food and physiological status of the animal (Innes *et al.* 1987). Steller sea lions pups grow rapidly during their first weeks of life and require a substantial intake of energy that is supplied by the mother. Nursing Steller sea lions pups at Año Nuevo Island consumed 1.5 to 2.4 liters of milk per day with a fat content of 23 to 25% (Higgins *et al.* 1988).

Nutritional requirements for free-ranging Steller sea lions have not been measured. Kastelein *et al.* (1990) provided data on food consumption of 10 animals kept in captivity and fed a diet that included several fish species and squid. Average daily consumption increased from 4 to 6 kg per day for one year olds to 10-13 kg per day at age five, with males generally eating more than females. An adult male ate 18kg per day on average, and females increased their daily requirement by approximately 30% when they became sexually mature and produced pups

Keyes (1968) concluded that adult, nonpregnant, nonlactating pinnipeds would require 6 to 10% of their body weight in food per day. Similarly, captive feeding experiments with one to two year olds indicate that the daily maximum digestive limit of Steller sea lions (in terms of weight of prey consumed) is equivalent to about 14 to 16% of their body weight (Rosen and Trites 2004).

Kastelein *et al.* (1990) estimated that the amount of food found in Steller sea lion stomachs has usually been on the order of one-fourth of their average daily requirements but did not account for digestion suggesting that meal sizes may at times be much larger. The stomach of a 311 kg sea lion collected in the Bering Sea contained 24 kg of partially digested pollock, which amounted to 7.7% of the animal's body weight (L. Lowry unpublished data). Kastelein *et al.* (1990) also reported that after a day of fasting, captive Steller sea lions ate meals that were about 25% larger than their daily average leading the authors to surmise that large sea lions have a relatively large stomach capacity, which is probably an adaptation that allows them to feed at infrequent intervals.

Winship *et al.* (2002) used bioenergetic modeling to estimate the food requirements of free-ranging Steller sea lions. The model incorporated information on age- and sex-specific bioenergetics of individual animals, population size and composition, and the composition and energy content of the diet. Their model predicted that juvenile animals have higher mass-specific food requirements than adults (greater than 10% versus 5 to 6% of body mass per day) and that a lactating female needs to consume about 70% more food on average if her pup is entirely dependent on her for energy during its first year of life. The mean predicted food requirement of an average Steller sea lion consuming an average Alaskan diet was 17 kg per day.

When assessing the suitability of prey for Steller sea lions in the wild, the important issue is the net amount of nutrition that can be gained from time spent feeding. Nutrition to be gained must take into account energy value of the prey as well as protein, vitamins, minerals, and micronutrients. Quantifying the biological value of prey species and the physiological consequences of inadequate prey is an area where laboratory studies can provide important data. For example, the energetic differences between prey species cannot be solely calculated from measures of gross energy content. The differences in energy due to lipid and protein composition are exaggerated by even higher losses from the heat increment of feeding and digestive efficiency of pollock (Rosen and Trites 1997, 2000b).

5. Nutritional Stress

In the sections above we discussed various topics such as sea lion vital rates, nutritional requirements, foraging ecology, prey, etc. In this section we synthesize this information to evaluate the evidence for nutritional stress. Nutritional stress is defined as the result of a species being unable to acquire adequate energy and nutrients from their prey resources. This can be manifested through acute nutritional stress (e.g., emaciation, rapid mortality through starvation, large scale breeding failures) and chronic nutritional stress (e.g., reduction in natality, reduced body size, higher juvenile and adult mortality, increased predation risk)(Trites and Donnelly 2003, NMFS 2000).

NMFS recognizes that nutritional stress could result from changes in prey quality, distribution or abundance. There are two main types of factors that could be affecting sea lion prey: those that are natural and those that are primarily anthropogenic. Anthropogenic factors and natural factors could both be impacting prey availability and could potentially interact.

If nutritional stress is being caused by an anthropogenic factor, it is probably linked to removal of important sea lion prey species by commercial fisheries (Atkinson *et al.* in press), although anthropogenic effects on climate could potentially have increasingly important impacts on sea lion prey in the coming decades. However, environmental features, such as oceanographic regime shifts, could also affect the relative abundance and distributions of key prey.

Hypotheses (such as the “Junk-food” hypothesis”) proposing natural causes of changes in prey quality, distribution, and abundance have centered primarily on the issue of prey quality: the idea is that regime shifts caused changes in prey community composition and sea lion diet which led to greater consumption of low energy fish (e.g., gadids) and less high energy fish (e.g., ‘oily’ fishes such as Pacific herring). Hypotheses positing an anthropogenic effect on Steller sea lion prey have emphasized the potential for fisheries to reduce the available biomass of important Steller sea lion prey especially in key areas and during critical life stages.

Inadequate prey intake by Steller sea lions will eventually be manifested at some level as nutritional stress (chronic or acute) with various changes in vital rates (see Bowen *et al.* 2001 [their Table 1], NRC 2003 [their Table 6.2]). Nutritional stress is a physiological response to suboptimal quantity and/or quality of available food, and may be acute (e.g., starvation occurring over a period of weeks) or chronic (e.g., suboptimal consumption over a period of months or years) (Trites and Donnelly 2003). Nutritional stress has been considered a leading hypothesis to explain the rapid decline of the western DPS of Steller sea lion (NMFS 2000), and has been the subject of considerable debate (NRC 2003, Trites and Donnelly 2003, Fritz and Hinckley 2005). However, it has been a difficult hypothesis to test due to a lack of data for Steller sea lions during the period of decline, the difficulty of working with these animals in remote locations, the long-term nature of the problem, and a poor understanding of the basic nutritional biology of Steller sea lions.

When assessing the potential for nutritional stress in Steller sea lions it is important to distinguish between early and late periods of the decline as well as recent population trends. The decline in the number of Steller sea lions in the western DPS was rapid through the 1980s, but slowed during the 1990s. In terms of testing the nutritional stress hypothesis, this means that the animals currently available in the wild for study may no longer be affected by the factors that caused their initial population decline. Many of the biological indicators of past (or current) nutritional stress may therefore no longer be measurable in direct ways. Nutritional limitation as indicated by reduced body size and reduced late term pregnancy rates during the rapid decline of the 1980s contrasts with most recent studies of Steller sea lions from the western DPS (Table I-15). Yet, if survival has been greatly reduced, then there is potential to have affected animals under-represented in the sample. Modeling results by Malavear (2004) suggests that juveniles less than one year old may die off fairly rapidly, whereas the older juveniles respond by slower growth and maturation times. Frid *et al.* (2006) suggest that because of interactions between energy status, predation risk and prey availability the body condition of animals could remain high while food resources are indeed declining. The marked acute nutritional effects observed for immature and adult otariids when prey biomass is reduced during El Niño events (Trillmich and Ono 1991, Soto *et al.* 2004) have not been observed for Steller sea lions (Table I-15). Therefore, if nutritional stress is acting on the western DPS, then we must look for evidence for or against chronic nutritional stress as opposed to acute nutritional stress (Trites and Donnelly 2003, Rosen *et al.* 2006).

Evidence During The Rapid Decline – The 1980s

Steller sea lions collected in the Gulf of Alaska during the early 1980s showed evidence of reproductive failure and reduced rates of body growth that were consistent with nutritional limitation (Calkins and Goodwin 1988, Pitcher *et al.* 1998, Calkins *et al.* 1998). Lactating females were less likely to become pregnant than non-lactating females during the early decline, indicating that the energetic stress of nursing while being pregnant with another pup may have prevented a significant number of females from giving birth each year (Pitcher *et al.* 1998). During the 1970s and 1980s, 100% and 95%, respectively, of all sexually mature females in the western DPS were pregnant in early gestation. The percentage of those females that carried their pregnancy to late gestation was only 55% to 67% during the 1970s and 1980s and was not statistically different between periods (Pitcher *et al.* 1998). However, among lactating females with higher energy demands, 63% carried their pregnancies to late gestation in the 1970s compared to only 30% in the 1980s, and this difference was significant. Better body condition was found to increase the probability that a female would maintain pregnancy. Comparatively low birth rates for females from the western DPS during the 1970s and 1980s (Pitcher and Calkins 1981) coupled with elevated embryonic and fetal mortality, appear to have contributed to decreased reproductive performance during the period of early decline (Pitcher and Calkins 1981, Calkins and Goodwin 1988, Pitcher *et al.* 1998, NMFS 1998b, 2000). Age-structured models fit to observed time series of pup and non-pup counts suggest that declines in reproductive performance of females in the western DPS continued through the 1990s and into the 2000s within the western DPS (Holmes and York 2003, Fay 2004, Winship and Trites 2006).

Food limitation resulting from the lack of availability of prey, or reduced quality, can result in reduced body size in marine mammals (Scheffer 1955, Laws 1956, Read and Gaskin 1990, Trites and Bigg 1992). Another indication that the western DPS may have been nutritionally compromised during the period of rapid decline in the 1980s was a reduction in average body size (Perez and Loughlin 1991, Castellini and Calkins 1993, Calkins *et al.* 1998). Steller sea lions from the central Gulf of Alaska during the 1980s were smaller in length, girth, and weight compared to the 1950s (Calkins *et al.* 1998, Fiscus 1961, Mathisen *et al.* 1962) and 1970s (Perez and Loughlin 1991, Castellini and Calkins 1993, Calkins and Goodwin 1988). Female sea lions over age 9 in the 1950s were significantly larger (standard length and axillary girth) than in the 1970s and 1980s (Calkins *et al.* 1998).

Since body size is influenced most during the first 8 years of life (Calkins and Pitcher 1982), Calkins *et al.* (1998) backdated 8 years from their mid-1980s sample to determine the break point for the reduction in body size—the late 1970s, or just after the 1977 regime shift. Ages of sea lions from the 1958 collection (Fiscus 1961, Mathisen *et al.* 1962) ranged from 9 to 22. Backdating 9-22 years from 1958, to see when growth was important to setting the size of the older females collected then, yields 1936-1944 as the critical years for the oldest females and 1949-1957 for the youngest. Thus, female sea lions collected in 1958 grew to large sizes from 1936-1957; this was a period when diets, for at least a portion of the interval, apparently were dominated by gadids and flounders (Imler and Sarber 1947). Applying the same procedure to the size data from the mid-1970s yields 8-year growth intervals of approximately 1959-1967 for the oldest (16 years) and 1968-1976 for the youngest; or from 1959-76 for all ages. The oldest animals underwent their 8 critical growth years during a period of what is thought by some to have been rich in high quality prey (Trites and Donnely 2003), yet they were smaller than those animals from the preceding gadid-rich era of the mid-1940s.

Such a change in morphological indices from animals in the wild (Pitcher *et al.* 2000) is consistent with sub-optimal nutritional status in the 1980s compared with the 1970s. Further, adult females in the 1970s were themselves smaller than in the late 1950s (Calkins et al. 1998), indicating that nutritional stress may have occurred prior to the regime shift of the mid-1970s. Sea lions feeding on a gadid-dominated diet in the 1940s appeared larger than in later samples during the 1970s. This is contrary to the prey quality hypothesis for nutritional stress.

Evidence During the Slower Decline – The 1990s

Much of the research from 1990-2004 to determine the extent to which nutritional stress (either acute or chronic) could be a factor in the decline of the western DPS involved comparing individual animals from the western and eastern DPSs. Many of the studies focused on pup condition, as well as maternal attendance patterns, foraging biology and adult dietary analyses. Contrary to what would be expected for animals experiencing acute nutritional stress, Steller sea lion pups in the early 1990s were heavier in the areas of population decline (i.e. the western DPS) than in rookeries where the population was increasing (Merrick *et al.* 1995). Pups at two rookeries within the area of decline were heavier in 1992-93 than prior to the decline in 1965 and 1975. Similar results were reported by Davis *et al.* (1996, 2004) who found no significant differences in pup birth sizes between declining and stable populations in the 1990s; nor were there differences in adult female body mass or composition. Rea *et al.* (2003) found no indication of poor body condition (based on percent total body lipid) in pups from either area. Paradoxically, Adams (2000) found pup growth rates were higher and females were larger by mass and length in declining western DPS areas (see also Brandon 2000).

Using a similar comparative protocol, researchers observed no differences or opposite than expected trends for sea lion milk composition (Davis *et al.* 1996, Adams 2000), pup milk intake rates (Adams 2000), pup growth rates (Davis *et al.* 1996, Adams 2000), maternal attendance patterns and foraging trip duration (Brandon 2000, Milette and Trites 2003, Andrews *et al.* 2002) between western and eastern DPS sea lions. Results from all of these studies suggest that adult females at rookeries in the declining population did not have difficulty finding prey during the summer. Furthermore, no apparent difference was observed between average winter attendance cycles of females from the declining western DPS (Marmot Island and Cape St. Elias) and increasing eastern DPS (Timbered Island) haul out populations (Trites *et al.* 2006b). In the 21st century, no evidence has yet been found of exceptional pup mortality, low birth weights in the western DPS, or poor growth of pups in the area of decline. Body fat contents were highly variable in both areas at 15 months of age (Rea *et al.* 2003). Fadely *et al.* (2004) compared growth rates of 29 sea lions captured in Alaska from 2000-2003 in a longitudinal survey and found that growth rates for juveniles were higher in the western DPS than for the eastern DPS.

Blood chemistry and hematological parameters, including blood urea nitrogen (BUN), ketone bodies (e.g., b-HBA), hematocrit and hemoglobin concentration, show characteristic patterns with changes in nutrition (Keyes 1968, Rea 1995), and have been experimentally induced in fasted Steller sea lion pups and juveniles (Rea *et al.* 1998b, Rea *et al.* 2000). However, Rea *et al.* (1998a) found no evidence of nutritional stress based on these parameters in wild Steller sea lions from areas with the greatest population declines. Red blood cell data from a study by Bishop and Morado (1995) reported elevated target cells and depressed poikilocyte levels in pups from the western DPS compared to those in the eastern DPS, indicative of anemia in the western DPS. Conversely, Castellini *et al.* (1993) reported no obvious differences in hematocrit or hemoglobin levels in pups during the 1990s from the western DPS compared to reference values. In evaluating serum haptoglobin levels (an indicator of acute stress response) in Steller sea lions, Zenteno-Savin *et al.* (1997) reported elevated serum levels in the western DPS compared to the east, but were careful to avoid speculation on the cause of these differences.

The general conclusion from these physiological studies comparing the eastern and western DPS during the 1990s has been that acute nutritional stress was not evident in the adult females or pups from either stock. Whether this was due to inherent biases in the study design is not known. One potential confounding factor in these studies may be habitat differences between the study sites. This would affect prey aggregation (Lowe and Fritz 1997) and thus foraging times for sea lions (Andrews *et al.* 2002). The large reduction in the western DPS Steller sea lion population by 1990 would likely affect relative prey availability for individuals through reduced competition (Winship and Trites 2003). Despite poor knowledge of the underlying mechanisms, morphological (Williams unpublished data) and survey (Fritz and Stinchcomb 2005) data indicate a trend towards improvement for Steller sea lions in the western DPS relative to conditions in the late 1970s and 1980s. Other demographic evidence (Holmes and York 2003, Fay 2004, Fay and Punt 2006) point to low natality that could affect the ability of the western DPS to recover.

The "Junk Food" Hypothesis

Changes in the structure of fish communities in the North Pacific Ocean (Hollowed and Wooster 1992, 1995, Anderson and Piatt 1999) due to oceanic regime shifts could alter the quality or availability of prey for Steller sea lions. Alverson (1992) proposed that changes in the structure of the Bering Sea and Gulf of Alaska ecosystems resulted in the dominance of pollock and other gadids (e.g. Pacific cod), and that the shift to ecosystems dominated by pollock had been the overriding factor in the Steller sea lion decline. He suggested a link between the changes in ecosystem trophic structure and the decline of sea lions based on the notion that pollock are a low quality food and the western population of sea lions has not been able to sustain itself with a larger fraction of its diet comprised of pollock. This became known as the "Junk Food Hypothesis." (Rosen and Trites 2000a, Trites and Donnelly 2003). Initially, proponents of this hypothesis suggested that juveniles and adult females experienced reduced survival and fecundity due to their lower quality, gadid-rich diet.

The Team recognized, and NMFS acknowledges, that the Junk Food Hypothesis has evolved and been refined since it was first proposed. Proponents of the hypothesis now emphasize that adult and juvenile pinnipeds require different amounts of energy (e.g., Winship et al. 2002; Winship and Trites 2003) and have different capacities (e.g., see below) to consume larger amounts of prey to compensate for differences in energy content.

A number of short-term diet manipulation studies on captive pinnipeds have been conducted to determine the effect of nutritional status on sea lion health. One such study reported that young Steller sea lions raised in captivity did not substantially increase food intake when switched from an ad libitum diet of herring to one of pollock (Rosen and Trites 2000a). The implication from this study was that the captive immature sea lions did not consume sufficient quantities of low-energy fish to maintain energy homeostasis, and thus lost weight during the experiments. A similar finding was reported for immature harp seals (Kirsch *et al.* 2000). When mature harbor seals were switched from high-fat herring to low fat herring, there was no difference in digestibility values, suggesting that digestibility may be more dependant on prey species and less dependant on nutrient composition of any particular type of prey (Stanberry 2003). In addition this harbor seal study showed that at least adult harbor seals can maintain body condition and health over a short period on a low-fat diet, mainly by slightly increasing their food intake (Stanberry 2003). Fadely *et al.* (1994) found that a mixed group of a subadult and two adult California sea lions maintained mass equally well on a diet of pollock or herring.

Rosen and Trites (2004) reported that the maximum weight that a juvenile Steller sea lion can digest per day on a sustainable basis appears to be about 14 to 16% of their body mass. This finding is based on offering 1 to 2 year old captive Steller sea lions as much high-energy (herring) or low-energy (capelin) fish as they could eat every day, or every second day. In this study, young sea lions feeding on low energy prey needed to consume more fish than they were physically capable of to meet their energy requirements. In contrast, older sea lions could consume the extra calories required without hitting the upper ceiling on digestive capacity. This was due in part to the lower relative energy needs of the older sea lions compared to young animals (Winship *et al.* 2002). Rosen and Trites (2002, 2004, 2005) found that Steller sea lions could alter their food intake in response to short-term changes in prey quality or availability and that food restrictions are likely to result in a "foraging response" rather than a "fasting response" which could produce a higher net energy deficit than first suspected (Rosen and Trites 2005). A diet composed of predominantly low energy prey combined with an interrupted schedule of feeding (i.e. on alternate days) necessitated food intake levels that apparently exceeded the physiological digestive capacities of young animals (Rosen and Trites 2004). Rosen *et al.* (2006) also found that sea lions can alter food intake levels to account for lower energy density prey but that juveniles may be more susceptible to these changes as well as reduced availability of prey given their consumption requirements. Calkins *et al.* (2005) conducted feeding experiments with 3 sea lions (4.5-9.5 years of age) and concluded that sea lions were able to compensate for lower quality prey (similar to results in Rosen *et al.* 2006) but without reaching satiation as described by Rosen and Trites (2004).

In comparison to adults, juvenile Steller sea lions on a constant "maintenance" level diet of either pollock or herring for 5 weeks over several seasons demonstrated marked seasonal effect on both body mass and composition (Rosen and Trites 2002, Kumagai 2004, Kumagai *et al.* 2006). Sea lions maintained on a low-lipid pollock diet lost significantly more body lipid reserves during periods of high-energy utilization (i.e., growth) than animals on a high-lipid herring diet. Similarly, juvenile Steller sea lions on calorically equivalent, sub-maintenance diets of low lipid Atka mackerel showed a greater reduction in lipid reserves than when fed sub-maintenance quantities of high lipid herring (Rosen and Trites 2002, 2005). While the sea lions fed Atka mackerel lost more of their lipid energy reserves, the sea lions fed herring lost more lean body mass (e.g., muscle). The implication is that if sea lions in the wild are similarly restricted in their energy intake, they could experience detrimental effects on individual fitness regardless of the prey type. However, these theoretical effects remain to be demonstrated in free-ranging populations which do not have mono-specific diets. Low diet diversity may play a role in nutritional stress but reported relationships between the level of population decline and diet diversity has been questioned (Atkinson et al. 2007 and references cited therein).

The duration of nutritional limitation, age of the animals, seasonal changes in energetic demands and effects of captivity appear to be important factors when evaluating the effects of diet on pinniped physiological responses. The aforementioned studies involved relatively short-term (2 to 6 week) changes in the diets of juvenile pinnipeds held in permanent captivity. Calkins *et al.*(2005b), evaluated the effects of diet on free-ranging juvenile sea lions held in temporary captivity. One group of seven 1 to 2 year old sea lions was fed only pollock while another group of eight was fed a mixed diet composed primarily of herring for 2 months. All animals gained weight on both diets, and there were no significant differences in the rate of mass increase between the two groups, nor were there any negative health consequences detected in the treatment (pollock) group. In a four-month study of juvenile and adult harbor seals, Trumble *et al.* (2003) found no overall changes in body mass or composition attributed to ad libitum pollock/herring diet changes. The longest study conducted to date was conducted by Castellini (2002) and Calkins *et al.* (2005) and evaluated three different diets on three sea lions over a three-year period. The diets were designed to reflect the pre- and post-decline diets in the Kodiak area and that of sea lions in Southeast Alaska where the population has increased. Changes in body mass of one adult male and two adult females were not significantly different on the three diet regimes, which led the authors to conclude that sea lions (adults in this instance) could compensate for low energy prey by increasing their ingestion provided sufficient quantity was available. They found that changing seasonal physiology is likely to have more impact on body condition than quality of prey, provided sufficient quantities are available (Calkins *et al.* 2005).

All of these studies have limitations, including small sample sizes. Many of them differ in study design (e.g., age and/or species of the animals studied, diets fed, length of the experiment, etc.). These differences hinder, in some cases, direct comparisons among studies. Concurrence is developing between the various captive animal feeding trials on some points, but not others. For example, data indicate, and there is widespread agreement that adult sea lions can compensate for lower energy prey by increasing the amount of food they eat. It appears that there are no differential effects between high-lipid and low-lipid (or low-protein and high-protein) prey on sea lion body composition when animals are able to consume sufficient prey to meet their energy demands. Therefore, gadids are likely to have been an important component of a healthy sea lion diet for decades (Calkins et al. 2005, Fritz and Hinckley 2005). Nutritional stress may result from the inability of sea lions to acquire sufficient prey to meet the energetic demands, especially during reproduction or seasonal growth. Juveniles are susceptible to nutritional stress due to their high metabolic requirements, potential consumptive limitations as reported by Rosen and Trites (2003), and limited foraging abilities. Females during the summer breeding season (on rookeries) appear to be able to attain adequate energy to nurse their pups. However, pregnant females with and without pups may be experiencing chronic nutritional stress after leaving the rookery, as evidenced by decreased pregnancy rates of lactating females (Pitcher *et al.* 1998), and decreased natality rates overall (Holmes and York 2003, Fay 2004, Holmes *et al.* 2007).

As noted above, low energy prey such as pollock or capelin are likely to be a normal part of Steller sea lion diets. Winship and Trites (2003) concluded that the key difference between the diets of increasing and decreasing sea lion populations in the North Pacific is the overall amount of low energy prey consumed by sea lions in each region (i.e., the average energy density of each meal). Dietary data available for the 1990s (Sinclair and Zepplin 2002) further indicates that higher rates of population decline correlated with meals that had overall lower energy densities. However, pollock makes up a significant portion of the diet of increasing populations of sea lions in Southeast Alaska (Trites *et al.* 2007a), and Pacific hake (*Merluccius productus*) is dominant in the diet of sea lions in Oregon (Riemer and Brown 1997). Furthermore, several stable and increasing populations of otariids including California sea lions (Bailey and Ainley 1982, Riemer and Brown 1997, Gearin *et al.* 1999), Cape fur seals (Punt *et al.*

1995), and South American sea lions (Dans *et al.* 2004) have diets with a high proportion of relatively low energy prey (e.g., gadids).

Summary

Sea lions in the 1970s and 1980s exhibited possible symptoms of nutritional stress (Calkins *et al.* 1998, Pitcher *et al.* 1998, Trites and Donnelly 2003), but there is no comparable evidence that nutritional stress was responsible for the continued decline of the western DPS during the 1990s. This may be due in part to differences in study methodologies between decades, and the focus on comparing increasing and decreasing populations of sea lions during the 1990s rather than comparing pre- and post-decline conditions.

In terms of acute nutritional stress, there is no indication at any time (1970s–2005) of emaciated juveniles or adults[6], of a decrease in pup body size, or of lactating females spending more time searching for prey (Table I-15). However, total birth rates at some rookeries and overall survival rates appeared to be lower during the 1990s. This and a well-documented continued drop in the number of pups and adults counted through the 1990s could be caused by chronic poor nutrition combined with other threats. The 1990s data suggest that (1) although diet composition of western animals had not changed, adult females appeared to secure enough food to adequately nurse their pups within the first 4 to 6 weeks of lactation, and (2) if food limitation was a major cause of continued declines (either through a shortage of prey or a low abundance of high energy prey) it may have affected reproductive performance of adult females. Analysis and synthesis of data collected more recently (2000 to 2005) is underway, but information that could be used to directly assess the nutritional status of Steller sea lions during this period is not yet available.

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[6] The occasional stranded animals have been found emaciated. This is more likely due to disease or other factor than directly from starvation.

I. Ecosystem Interactions

In the North Pacific Ocean, Steller sea lions inhabit a diverse and complex ecosystem, which they share with many other species. Detailed descriptions of physical and biological characteristics of the Gulf of Alaska and Bering Sea have been compiled by Hood and Calder (1981), Hood and Zimmerman (1986), National Research Council (1996), Trites *et al.* (1999, 2006a), Loughlin and Ohtani (1999), and Guénette, and Christensen (2005).

Physical aspects of the environment obviously determine whether or not an area is suitable for sustaining Steller sea lions, or any other life form. Physical parameters that may be important to sea lions include coastal geomorphology, air and water temperatures, wind speeds, wave conditions, tides, currents, etc. A few recent studies have addressed how such factors may influence sea lion distribution and abundance. One showed that the terrestrial sites used by Steller sea lions tend to be associated with waters that are relatively shallow and well-mixed, with average tidal speeds and less-steep bottom slopes (Ban 2005). Another study identified patterns in ocean climate that are consistent with the patterns of sea lion distributions, population trends, numbers and diets (Trites *et al.* 2006a). Thus, there appears to be a linkage between Steller sea lions and the physical environment, which likely plays a major role in determining the northern and southern limits of the Steller sea lion range.

Physical characteristics of the ecosystems inhabited by sea lions are not static, but rather show variations on several time scales (Schumacher and Alexander 1999, Trites *et al.* 2006a). Considerable attention has recently been given to abrupt decadal scale changes in long term data series that describe the climate, oceanic conditions and abundances of a number of species in the North Pacific. The largest such change recorded this past century occurred in the mid 1970s (Ebbesmeyer *et al.* 1991, Graham 1994, Francis *et al.* 1998). In some cases fluctuations in fish, bird, and mammal populations seem to correlate with these decadal scale climate changes (Springer 1998, 2004, Benson and Trites 2002, Trites *et al.* 2006a).

Food web interactions (Trites 2003), predation (Barrett-Lennard *et al.* 1995) and disease (Burek *et al.* 2003, Goldstein 2004) are all biotic components of the ecosystem that are important to Steller sea lions as they function as food, competitors, predators, parasites, and disease agents.

Human exploitation of marine mammals and fishes in the North Pacific Ocean over the past 250 years has undoubtedly modified the environment that Steller sea lions occupy. The precise effects on Steller sea lions have been impossible to determine, but have likely been substantial, variable over time, and both top-down and bottom-up in nature. Large-scale removals of competitors of prey, such as some species of great whales, northern fur seals, and perhaps some fishes may have provided additional food and for some period of time may have increased sea lion carrying capacity. The relationship of Steller sea lions with their primary predator, killer whales, has also likely varied over time with the exploitation of alternative prey such as great whales, northern fur seals, and sea otters and perhaps with the exploitation and recovery of killer whales themselves. Combining this with climatic variability and commercial fisheries that could potentially affect the carrying capacity for Steller sea lions yields an extremely complex history.

Ecosystem models are available for the Aleutian Islands, Eastern Bering Sea and Southeast Alaska; these models can be used to decipher the combined effects that fishing, predation, ocean climate change, and interspecies interactions have had on Steller sea lions and their ecosystems as a whole (Trites *et al.* 1999, Guénette and Christensen 2005). These models indicate that bottom-up and top-down processes occur simultaneously and suggest that Steller sea lions have been both positively and negatively affected by changes in their food base (due to fishing and ocean climate change), as well as by competition with large flatfish, and by the effects of predation by killer whales (particularly when sea lion numbers are low). Further work is continuing with these models to assist in better understanding the complex ecosystem interactions underway in the North Pacific.

Table I-1. Counts of adult and juvenile (non-pup) Steller sea lions at western DPS rookery and haul-out trend sites in Alaska during June-July surveys from 1976 to 2004 (NMFS 2000, Sease *et al.* 2001, Sease and Gudmundson 2002, and Fritz and Stinchcomb 2005). Numbers in parentheses are the number of trend sites counted in each sub-area. Percentage changes between years are shown in bold.

Year(s)	Gulf of Alaska			Aleutian Islands			Kenai-Kiska (69)	Western DPS in Alaska (82)
	Eastern (9)	Central (15)	Western (9)	Eastern (11)	Central (34)	Western (4)		
1956-60[1]		34,792	15,772	44,020	17,120		111,704	
1962					23,175			
1976-79[2]	7,053	24,678	8,311	19,743	36,632	14,011	89,364	110,428
1985		19,002	6,275	7,505	23,042		55,824	
1989	7,241	8,552	3,908	3,032	7,572		23,064	
1990	5,444	7,050	3,915	3,801	7,988	2,327[3]	22,754	30,525
1991	4,596	6,270	3,732	4,228	7,496	3,083	21,726	29,405
1992	3,738	5,739	3,716	4,839	6,398	2,869	20,692	27,299
1994	3,365	4,516	3,981	4,419	5,820	2,035	18,736	24,136
1996	2,132	3,913	3,739	4,715	5,524	2,187	17,891	22,210
1998	2,110[4]	3,467	3,360	3,841	5,749	1,911	16,417	20,438
2000	1,975	3,180	2,840	3,840	5,419	1,071	15,279	18,325
2002	2,500	3,366	3,221	3,956	5,480	817	16,023	19,340
2004[5]	2,536	2,944	3,512	4,707	5,936	898	17,099	20,533
1950s to 2000		**-91%**	**-82%**	**-91%**	**-68%**		**-86%**	
1970s to 2000	**-72%**	**-87%**	**-66%**	**-81%**	**-85%**	**-92%**	**-83%**	**-83%**
1970s to 1990	**-23%**	**-71%**	**-53%**	**-81%**	**-78%**	**-83%**	**-75%**	**-72%**
1990 to 2000	**-64%**	**-55%**	**-27%**	**+1%**	**-32%**	**-54%**	**-33%**	**-40%**
2000 to 2004	**+28%**	**-7%**	**+24%**	**+23%**	**+10%**	**-16%**	**+12%**	**+12%**

[1] 1956 counts for the western GOA, 1957 counts for the central GOA, 1959 counts for the central Aleutians and 1960 counts for the eastern Aleutians.

[2] 1976 counts for the eastern, central, and western GOA and the eastern Aleutians, and 1979 counts for the central and western Aleutians.

[3] Gillon Point rookery, Agattu Island not surveyed in 1990.

[4] 1999 counts substituted for sites in the eastern Gulf of Alaska not surveyed in 1998.

[5] 2004 counts were from medium format photographs, while all others were from 35 mm photographs, aerial counts or beach counts. 2004 data reflect a –3.64% adjustment to account for film format resolution and count differences (Fritz and Stinchcomb 2005).

Table I-2. Counts of Steller sea lion pups at western DPS rookeries in Alaska during 1979 to 2004 (NMFS 1992, Sease and Loughlin 1999, Fritz and Stinchcomb 2005, NMFS unpublished). Percentage changes between years are shown in bold.

	Gulf of Alaska			Aleutian Islands			Eastern Bering Sea	Kenai-Kiska[7]	Western DPS in Alaska
Year(s)	Eastern[1]	Central[2]	Western[3]	Eastern[4]	Central[5]	Western[6]	Walrus Island		
1979			8,616						
1982							334		
1984			6,435						
1985-89		10,254		4,778	9,428		250	30,895[7]	
1990-92		4,904	1,923	2,115	3,568		63	12,510	
1994	903	2,831	1,662	1,756	3,109		61	9,358	
1996	584								
1997	611					979	35		
1998	689	1,876	1,493	1,474	2,834	803		7,677	9,169
2001-02	586	1,721	1,671	1,561	2,612	488	39	7,565	8,678
2003-04	716	1,609	1,577	1,731					
2005	715	1,651	1,707	1,921	2,551	343	29	7,830	8,917
Earliest count to 1994		**-72%**	**-81%**	**-63%**	**-67%**			**-70%**	
Earliest count to 2001-02	**-35%**	**-83%**	**-81%**	**-67%**	**-72%**	**-50%**	**-88%**	**-76%**	**-5%**
1994 to 2001-02	**-35%**	**-39%**	**+1%**	**-11%**	**-16%**		**-36%**	**-19%**	
2001-02 to 2005	**+22%**	**-4%**	**+2%**	**+23%**	**-2%**	**-30%**	**-25%**	**+4%**	**+3%**

[1] Seal Rocks and Fish (Wooded) Island

[2] Outer, Sugarloaf, Marmot, Chowiet and Chirikof Islands

[3] Atkins and Chernabura Islands, and Pinnacle Rock and Clubbing Rocks

[4] Ugamak, Akun, Akutan, Bogoslof and Adugak Islands

[5] Yunaska, Seguam, Kasatochi, Adak, Tag, Ulak, Ayugadak and Kiska (2) Islands, and Gramp and Column Rocks.

[6] Buldir, Agattu (2), and Attu Islands

[7] Rookeries in the Central and Western Gulf of Alaska, and Eastern and Central Aleutian Islands

Table I-3. Counts of adult and juvenile (non-pup) Steller sea lions on terrestrial trend sites in Russia.

Year	W. Bering Sea	Commander Islands	E. Kamchatka	Kuril Islands	Tuleny Island	Sea of Okhotsk
1963		2,920[1]		14,660	60[2]	
1969				14,184		
1971		2,920				
1973		3,503				
1974					49	1,208
1975				8,397		
1977		4,480				
1978		2,807			26	
1981		2,101		5,921		
1982	4,910	1,577				
1983	3,230	1,761	2,073		65	
1984		1,930				
1985	3,370	1,700			137	
1986		2,633			450	
1987	1,231	2,267	1,690			
1988		1,221			171	1,691[3]
1989	1,199	896	1,519	4,488	190	
1990		865			410	
1991	427	752	794		350	
1992		843			463	
1993		569			549	
1994	200	543	642		557	
1995		653				
1996		804			615	2,429[4]
1997		812			679	
1998		900			836	
1999	180	860	720		770	
2000		741			1,155	
2001		718	669	5,129	857	2,324
2002	16	581	491		1,041	2,072
2003		530		5,178	1,119	
2004	91	674	548		1,084	2,357
2005				5,544	1,218	

[1] 1962 data. [2] 1964 data. [3] 1989 data for Iony Island. [4] 1995 data for Yamsky Islands and 1997 data for Iony Island.

Table I-4. Counts of Steller sea lion pups on rookery trend sites in Russia.

Year	Commander Islands	E. Kamchatka	Kuril Islands	Tuleny Island	Sea of Okhotsk
1962	1				
1963			3,673		
1969	0		3,250		
1970	3				
1971	4				
1972	9				
1973	26				
1974				1	607
1977	19				
1978	26			0	
1980				6	
1981	48				
1982	83			0	
1983	104		1,992	5	
1984	141			0	
1986	151		1,560	25	
1987	197	211			
1988	141			38	712[1]
1989	195		1,442	45	
1990				59	
1991	229			63	
1992	222	108	1,623	90	
1993	224	115		120	
1994	226	93		146	
1995	248	84	1,972		
1996	261	87		219	1,250[2]
1997	244	96		256	
1998	280	91		303	
1999	271	87		291	
2000	180	76	1,824	340	
2001	228	61	1,807	303	1,231
2002	210	84	1,973	410	980
2003	216		2,086	480	
2004	221	107		508	1,868
2005	236		2,306	407	

[1] 1989 data for Iony Island. [2] 1995 data for Yamsky Islands and 1997 data for Iony Island.

Table I-5. Counts of one-month-old Steller sea lion pups on rookeries in southeast Alaska, 1979-2005 (ADF&G and NMFS unpublished data).

Year	Forrester Island	Hazy Island	White Sisters	Graves Rocks	Biali Rocks	Total Pups
1979	2,187	32				2,219
1990	2,932	638	30			3,600
1991	3,261	808	95			4,164
1994	2,757	862	151			3,770
1996	2,764	768	182			3,714
1997	2,798	1,157	205			4,160
1998	2,753	1,199	282	1		4,235
2001	3,152	1,091	371	89	38	4,741
2002	3,060	1,257	403	98	59	4,877
2005	3,429	1,286	520	175	100	5,510

Table I-6. Counts of adult and juvenile (non-pup) Steller sea lions observed at individual rookeries and rookery and haulout trend sites combined in southeast Alaska during June-July aerial surveys from 1979 to 2005 (Sease *et al.* 2001, ADF&G and NMFS unpublished data).

Year	Forrester Island	Hazy Island	White Sisters	Graves Rocks	Biali Rocks
1979	3,121	893	761	-	810
1982	3,777	1,268	934	-	722
1989	4,648	1,462	734	475	794
1990	3,324	1,187	980	937	596
1991	3,970	1,496	975	470	494
1992	3,508	1,576	860	366	398
1994	4,010	1,615	868	733	410
1996	3,551	1,759	894	475	342
1998	3,788	1,962	858	445	476
2000	3,674	1,824	1,398	558	690
2002	3,699	2,050	1,156	1,001	624
2005	5,557	2,293	1,078		598

Table I-7. Counts of Steller sea lions on rookeries and haulouts in British Columbia, 1971-2002 (Olesiuk and Trites 2003).

Year	Non-pups	Pups	Total
1971	4,617	941	5,475
1977	5,219	963	6,274
1982	4,713	1,245	5,956
1987	6,109	1,084	7,193
1992	7,376	1,468	8,844
1994	8,091	1,186	9,277
1998	9,818	2,073	11,891
2002	12,121	3,281	15,402

Table I-8. Counts of non-pup Steller sea lions on rookeries and haulouts in Oregon and of pups counted during ground counts or from medium-format photographs on the Rogue Reef and Orford Reef rookeries 1976-2001 (Brown *et al.* 2002). Mean counts of Steller sea lion non-pups on Washington haulouts during the breeding season, June 16 through July 15, 1991 – 2001 (Washington Department of Fisheries and Wildlife unpublished data).

Year	Oregon Total Non-pups	Rogue Reef Pups	Orford Reef Pups	Washington Total Non-pups
1977	1,461	--	--	--
1979	1,542	--	--	--
1980	1,632	--	--	--
1981	2,105	--	--	--
1982	2,604	--	--	--
1983	2,106	--	--	--
1984	1,867	--	--	--
1985	2,210	--	--	--
1986	2,289	--	--	--
1987	2,709	--	--	--
1988	2,825	--	--	--
1989	2,183	--	--	89
1990	2,414	492	298	--
1991	--	--	--	274
1992	3,581	--	--	278
1993	2,838	--	--	--
1994	3,293	--	--	384
1995	3,837	--	--	409
1996	3,205	685	335	594
1997	3,897	--	--	352
1998	3,971	--	--	470
1999	3,275	--	--	806
2000	2,927	--	--	778
2001	3,648	600	--	516
2002	4,169	746	382	--

Table I-9. Historical compilation of counts of adult and juvenile (non-pup) Steller sea lions on rookeries (current and former) and associated haulouts in California, 1927-2004.

Year	San Miguel Island	Año Nuevo Island	Farallon Islands	Sugarloaf Island/ Cape Mendocino	Saint George Reef	Total
1927	595[a]	1,500[a]	700[a]	700[a]	1,500[a]	4,995
1930	620[b]	2,500[b]	900[b]	900[b]	700[b]	5,620
1936	1,359	1,000	500	700	652	4,211
1938	1,902	2,000	357	500	325	5,084
1947	950[b]	2,050[b]	750[b]	625[b]	200[b]	4,575
1962	--	2,265[c]	--	--	--	--
1964	61[d]	--	--	--	--	--
1974	--	673[e]	133[f]	--	--	--
1983	--	319[e]	141[f]	--	--	--
1990	--	449[g]	206[f]	--	674[g]	--
1991	0	359[g]	178[f]	--	626[h]	--
1992	--	189[g]	87[f]	--	693[h]	--
1993	--	218[g]	107[f]	--	496[g]	--
1994	--	387[g]	121[f]	--	538[g]	--
1995	--	288[g]	138[f]	--	--	--
1996	--	306[g]	76[f]	501[g]	987[g]	1,870
1997	--	--	--	--	937[h]	
1998	--	179[g]	--	256[g]	493[g]	
1999	0	350[g]	214[g]	270[g]	713[g]	1,547
2000	0	270[g]	79[g]	489[g]	866[g]	1,704
2001	0	227[g]	60[g]	740[g]	790[g]	1,817
2002	0	255[g]	125[g]	588[g]	716[g]	1,684
2003	0	254[g]	136[g]	513[g]	803[g]	1,706
2004	0	340[g]	85[g]	415[g]	738[g]	1,578

[a] Bonnot 1928
[b] Bonnot and Ripley 1948
[c] Orr and Poulter 1965
[d] Odell 1971
[e] LeBoeuf *et al.* 1991
[f] Point Reyes Bird Observatory, unpublished data
[g] Southwest Fisheries Science Center, unpublished data
[h] Oregon Department of Fish and Wildlife, unpublished data

Table I-10. Recent counts of Steller sea lion pups on rookeries in California, 1990-2004.

Year	Año Nuevo	Farallons	Sugarloaf/ Cape Mendocino	Saint George Reef	Total Pups
1990	312[a]	4[c]			
1991	287[b]	2[c]			
1992	263[b]	4[c]			
1993	230[b]	5[c]			
1994	244[a]	7[c]		115[a]	
1995	226[a]	6[a]			
1996	236[a]	5[c]	62[a]	243[a]	546
1997	210[a]				
1998	186[a]		61[a]	256[a]	
1999	152[a]	10[a]	86[a]	184[a]	432
2000	184[a]	4[a]	138[a]	293[a]	619
2001	230[a]	2[a]	152[a]	338[a]	722
2002	189[a]	7[a]	150[a]	367[a]	713
2003	226	13[a]	158[a]	458[a]	855
2004	221	22[a]	131[a]	444[a]	818

[a] Southwest Fisheries Science Center, unpublished data

[b] Westlake *et al.* 1997

[c] Point Reyes Bird Observatory, unpublished data

Table I-11. Estimates of the total number of Steller sea lions (pups and non-pups) in the eastern DPS in 2002 based on number of pups counted multiplied by the 4.5 multiplier (Calkins and Pitcher 1982) and the 5.1 multiplier (Trites and Larkin 1996).

Region	Pups Counted	Estimate with 4.5 multiplier	Estimate with 5.1 multiplier
Southeast Alaska	4,877	21,947	24,873
British Columbia	3,281	14,765	16,733
Oregon	1,128	5,076	5,753
California	713	3,209	3,636
Total	9,999	44,997	50,995

Table I-12. Source of literature, age class/group, sample size (n), capture location, season captured, instrument deployed, and mean trip duration, distance, and time at sea for Steller sea lions tagged with radio (VHF) and satellite (e.g. SLTDR) transmitters. Error (SE) is standard deviation unless otherwise indicated.

Source	Age Class/Group	n	Capture Location	Season	Instrument	Mean Trip Duration (h)	Mean Trip Distance (km)	Mean % Time @ Sea
Merrick and Loughlin (1997)	Adult Female	7	Marmot (CGOA)	Summer	VHF	21.0 ± 3.7 (SE)		53
	Adult Female	3	Ugamak (EAI)	Summer	VHF	25.0 ± 3.9		58
	Adult Female	4	EAI to CGOA	Summer	SLTDR	18.0 ± 3.1		50
	Adult Female	5	EAI to CGOA	Winter	SLTDR	204.0 ± 104.6		90
	YOY	5	EAI to CGOA	Winter	SLTDR	15.0 ± 2.2		38
Loughlin *et al.* (1998)	Adult F	8	Kuril Islands, Russia	Summer	SLTDR	short; max = 94 h	94% trips ≤ 10 km (max=263 km)	
Loughlin *et al.* (2003)[1]	YOY	12	CAI, EAI, EGOA, CGOA, and WA	All	SLTDR/SDR	7.5 ± 7.5	7.0 ± 19.0	
	Juv (>10 mo.)	13	CAI, EAI, EGOA, CGOA, and WA	All	SLTDR/SDR	18.1 ± 34.2	24.6 ± 57.2	
	Combined	25	CAI, EAI, EGOA, CGOA, and WA	All	SLTDR/SDR	12.1 ± 23.8		
Raum-Suryan *et al.*(2004)[2]	YOY (75), Juv (28)	103	see below	Spr/Sum/Win	SDR	84% trips ≤ 20 h	90% trips ≤ 15 km	
	Western DPS	29	EAI, CGOA, EGOA	Spr/Sum/Win	SDR		6.5 (5.08-8.26) CI	
	Eastern DPS	74	North, South, and Central SE	Spr/Sum/Win	SDR		4.7 (3.92-5.53)	
Fadely *et al.* (2005)[3]	YOY/Juv	30	CAI, EAI, and CGOA	Feb-April	SDR	8.9 (8.4-9.4) CI	0.56 (0.56-0.74) CI	
				May-July	SDR	12.5 (11.3-13.9)	1.30 (0.93-1.49)	
				Nov-Jan	SDR	10.1 (8.2-12.5)	1.11 (0.74-1.67)	
Rehberg (2005)	YOY	11	CAI and GOA	Spring/Winter	SRDL			42 (38-45) CI
	Juv	12	CAI and GOA	Spring/Winter	SRDL			51 (49-54) CI

Table I-12. Continued.

[1] Trip duration ranged from 1.0 h to 81.3 h (YOY) and 344.0 h (Juv) and trip distance ranged from 1.0 km to 260.7 km (YOY) and 447.3 km (Juv).
[2] Inter-haulout distance averaged 79.3 ± 7.7 km (max = 127 km) and dispersal distances (2 YOY, 2 Juv) included 76, 120, 500, and 1300 km, respectively.
Sea lions in the western and eastern DPSs used an average of 1.6 and 2.1 haulouts, respectively.
[3] Most locations associated with diving were within 9 to 19 km (5-10 nm) of shore and in waters less than 100 m. Trip duration and use of offshore waters increased with age and coincided with spring.
YOY: young-of-the-year; Juv: juvenile (> 1 year unless otherwise specified); VHF: very high frequency radio transmitter; SLTDR: satellite-linked time-depth recorder; SDR: satellite depth recorder; SRDL: satellite relayed dive logger; CAI: central Aleutian Islands; EAI: eastern Aleutian Islands; EGOA: eastern Gulf of Alaska; CGOA: central Gulf of Alaska; SE: Southeast Alaska; WA: Washington State; CI: 95% confidence interval

Table I-13. Food habits information for Steller sea lions collected in the range of the western DPS, 1945-1998. (Reprinted from Fritz and Hinckley 2005).

A. Sample Sizes and Characteristics		Months				Region						
Reference	Years	Jan-Mar	Apr-Jun	Jul-Sep	Oct-Dec	CGOA	WGOA	EBS	EAI	CAI	WAI	Russia
Imler and Sarber (1947)	1945			7		7						
Wilke and Kenyon (1952)	1949, 51			3				3				
Mathisen *et al.* (1962)	1958		94				94					
Thorsteinson and Lensink (1962)	1959		56			9	27		20			
Tikhomirov (1964)	1962	X	X					X				
Fiscus and Baines (1966)	1960, 62		16			4	2	1	9			
Perlov (1975)	1966-69			?								X
Lowry *et al.* (1982)	1976	4						4				
Pitcher (1981)	1975-78	43	54	9	47	136	17					
Calkins (1998) a	1981	60										60
Calkins (1998) b	1981	32						32				
Frost and Lowry (1986)	1985	13						13				
Gearin (unpub)	1985, 86			3	8			11				
Calkins and Goodwin (1988)	1985, 86		X		X	74						
Merrick *et al.* (1997) a	1990-93			76		76						
Merrick *et al.* (1997) b	1990-93			67					67			
Merrick *et al.* (1997) c	1990-93			167						167		
Merrick *et al.* (1997) d	1990-93			28							28	
Goto and Shimazaki (1997)	1994-96	62										62
Sinclair and Zeppelin (2002) a	1990-98	X	X	X	X	574						
Sinclair and Zeppelin (2002) b	1990-98	X	X	X	X		929					
Sinclair and Zeppelin (2002) c	1990-98	X	X	X	X				889			
Sinclair and Zeppelin (2002) d	1990-98	X	X	X	X					1370		

Table I-13. Food habits information of Steller sea lions collected in the range of the western DPS, 1945-1998 (continued).

B. Food habits data	Sample	Sample	Data	Percent of Sample with Prey Item (x=present)												
Reference	Type	Location	Type	Pollock	Cod	Flatfish	Greenling	Rockfish	Smelts	Sandlance	Herring	Salmon	Sculpin	Shrimp/Crab	Squid	Octopus
Imler and Sarber (1947)	Stomach	Land	FO	57		71						28				43
Wilke and Kenyon (1952)	Stomach	Land	PW	7	10	49				32			<1			2
Mathisen *et al.* (1962)	Stomach	Land	FO				13	9	14	1		1	6	10		44
Thorsteinson and Lensink (1962)	Stomach	Land	FO			6	4	11		25			4	2		20
Tikhomirov (1968)	Visual	At-sea									D					
Fiscus and Baines (1966)	Stomach	At-sea	FO	6		12	6	6	56	25			19			
Perlov (1975)	Stomach	At-sea	FO	63			10						1		>30	25
Lowry *et al.* (1982)	Stomach	At-sea	PV	97		1									1	1
Pitcher (1981)	Stomach	Land	FO	67	12	5		3	11		11	4	4	7	23	13
Calkins (1998) a	Stomach	At-sea	FO	83	43	3					17		>12	2	2	18
Calkins (1998) b	Stomach	At-sea	FO	100	28	>19		3			6		6	>10	19	19
Frost and Lowry (1987)	Stomach	At-sea	PV	48							48					
Gearin (unpub)	Stomach	Land	FO	>36	>45	54								18		45
Calkins and Goodwin (1988)	Stomach	Land	FO	58	7	14				7	3	3	1	>1	4	32
Merrick *et al.* (1997) a	Scat	Land	FOSS	66		4	<1		6			20	0		3	
Merrick *et al.* (1997) b	Scat	Land	FOSS	33		2	31		8			17	7		2	
Merrick *et al.* (1997) c	Scat	Land	FOSS	13		0	69		1			6	4		8	
Merrick *et al.* (1997) d	Scat	Land	FOSS	7		0	77					5	5		7	
Goto and Shimazaki (1997)	Stomach	At-sea	FO	89	76	24									69	11
Sinclair and Zeppelin (2002) a	Scat	Land	FO	>50	>5	>20	<5	x	x	>10	>10	>10	<10		<10	<10
Sinclair and Zeppelin (2002) b	Scat	Land	FO	>70	>10	>10	<5	x	x	>10	<10	>10	>10		<5	<5
Sinclair and Zeppelin (2002) c	Scat	Land	FO	>50	>10	<5	>20	x	x	<5	>5	>20	>10		<10	<10
Sinclair and Zeppelin (2002) d	Scat	Land	FO	<10	>10	<5	>60	x		<5	<5	>20	>10		<20	<20

Table I-13 (continued). Abbreviations: CGOA – central Gulf of Alaska; WGOA – western Gulf of Alaska; EBS – eastern Bering Sea; EAI – eastern Aleutian Islands; CAI – central Aleutian Islands; WAI – western Aleutian Islands; X – number for cell is unknown; ? – season of sample collection is unknown but likely to be as indicated; FO=frequency of occurrence; PW=percent by weight; PV=percent by volume; FOSS=Split sample FO.

Table I-14. Source of literature, age class/group, sample size (n), capture location, season captured, instrument deployed, mean depth of dives, mean depth of maximum dives, maximum depth, mean duration of dives, and maximum duration of dives for Steller sea lions tagged with satellite (e.g. SLTDR) transmitters. Units for data coincide with those in the table header unless otherwise indicated and error is standard deviation unless otherwise indicated.

Source	Age Class/Group	n	Capture Location	Season	Instrument	Mean Dive Depth (m)	Mean Max Depth (m)	Max Depth (m)	Mean Dive Duration (min)	Max Dur (min)
Merrick and Loughlin (199	Adult F	4	Alaska (EAI to CGOA)	Summer	SLTDR	21.0 (med)		150-250		
	Adult F	5	Alaska (EAI to CGOA)	Winter	SLTDR	24.0		> 250		
	YOY	5	Alaska (EAI to CGOA)	Winter	SLTDR	9.0		72		
Loughlin *et al.* (1998)	Adult F	8	Kuril Islands, Russia	Summer	SLTDR	53.0		250	1.9	8
Loughlin *et al.* (2003)[1]	YOY	13	CAI, EAI, EGOA, and CGOA	All	SLTDR/SDR	7.7 ± 1.7	25.7 ± 16.9	252	0.8 ± 0.1	
	Juv	5	EAI, EGOA, and CGOA	All	SLTDR/SDR	16.6 ± 10.9	63.4 ± 37.7	288	1.1 ± 0.4	
	Juv	7	WA	All	SLTDR/SDR	39.4 ± 14.9	144.5 ± 32.6	328	1.8 ± 0.6	
Pitcher *et al.* (2005)	YOY	75	Alaska (EAI to SE)	All	SDR	87% dives < 10 m		252	82% dives < 2 min	>12
	Juv	36	Alaska (EAI to SE)	All	SDR			452		
Fadely *et al.* (2005)[2]	YOY	26	CAI, EAI, and GOA	Spr/Sum/Win	SDR	10.3				
	Juv	4				13.0				
Rehberg (2005)	YOY	11	CAI and GOA	Spring/Winter	SRDL	12.4 (11.0-14.0) CI			0.87 (0.7-1.0) CI	
	Juv	12	CAI and GOA	Spring/Winter	SRDL	22.9 (20.0-28.0)			1.71 (1.5-2.0)	

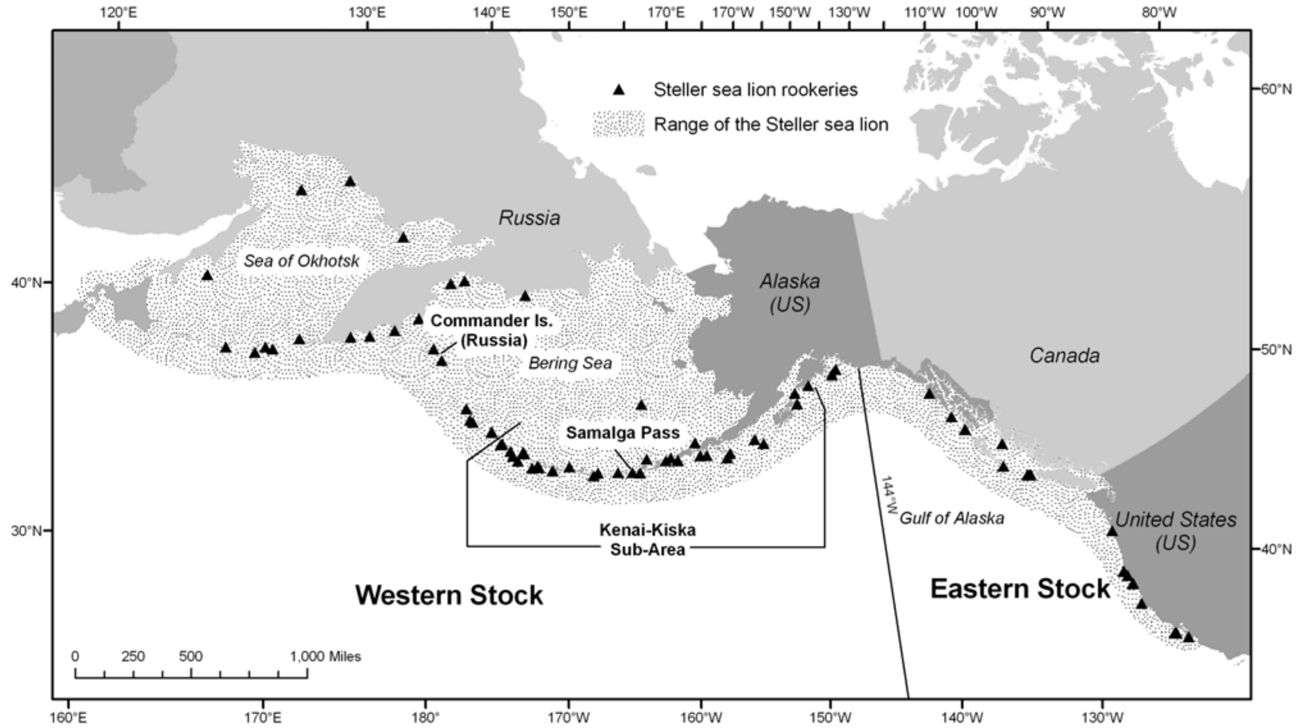
[1] Dive parameters did not differ among regions in the Aleutian Islands and Gulf of Alaska.

[2] Increase in diving activity coincided with increases in SST and chlorophylll-a, but also with age.

YOY: young-of-the-year; Juv: juvenile (> 1 year unless otherwise specified); VHF: very high frequency radio transmitter; SLTDR: satellite-linked time-depth recorder; SDR: satellite depth recorder; SRDL: satellite relayed dive logger; CAI: central Aleutian Islands; EAI: eastern Aleutian Islands; EGOA: eastern Gulf of Alaska; CGOA: central Gulf of Alaska; SE: Southeast Alaska; WA: Washington State; CI: 95% confidence interval

Table I-15. Data gaps for assessing potential biological manifestations of nutritional stress in the western DPS of Steller sea lions. Evidence is based on a comparison with the previous decade (H=historical) or with the eastern DPS (G=Geographic). Y=Yes, data are available to make a comparison and an effect was as indicated; N=No, data are available to make a comparison but the effect was opposite to that indicated; U=Unknown, no data are available; U*=Unknown, data available but not analyzed. Range-wide versus local data sets are identified by superscript "R" and "L", respectively. See text and Appendix for details and references.

Potential Biological Effects	**1980s**	**1990s**	**2000-2004**
More emaciated pups (<4 wks)	U	U*	N(H)
More emaciated pups (>4 wks)	U	U	U
More emaciated juveniles	U	N(H,G)	U
More emaciated adults	U	N(H,G)	U
Reduced pup survival (to 4 wks)	U*	U*	U
Reduced adult body size	Y(H)	U	U
Reduced juvenile body size	Y(H)	U*	U
Reduced pup body size	U	N(G), U*(H)	N(H)
Reduced birth weight	N or U?	U	U
Reduced pup weight	?	N(G),U*(H)	N(H)
Reduced growth rate	Y(H)	N(G)	N(H)
Reduced pup survival	? OR U	U*	N(H)
Reduced juvenile survival	Y(H)	Y	N(H)
Reduced adult survival	Y(H)	N	N(H)
Reduced overall survival	Y(H)	Y(H,G)	N(H)
Reduced birth rate	Y(H)	Y(H)	Y(H)
Reduced pup counts	Y(H)	Y(H)	N(H)
Reduced non-pup counts	Y(H)	Y(H)	N(H)
Increased reproductive failure	Y(H)	U	U
Change in pup blood chemistry (increased fasting)	U	N(G)	N
Change in juvenile blood chemistry (increased fasting)	U	U*	N
Delayed sexual maturity	U	U	U
Change in metabolic rate	U	U	U
Decreased body condition (adult females on rookeries)	U	U* (N(G))	U
Reduced adult perinatal fast	U	N(G)	U
Longer foraging trip duration	U	N(G)	U*
Increased susceptibility to disease (haptoglobin)	U	U*	U
Increased incidence of disease	U	N(G)	N(H,G)
Increased susceptibility to predation	U	U	U
Altered weaning age	U	U*(G)	U*
Decreased weaning size	U	U	U
Traditional ecological knowledge re. body condition	?	U*	U*



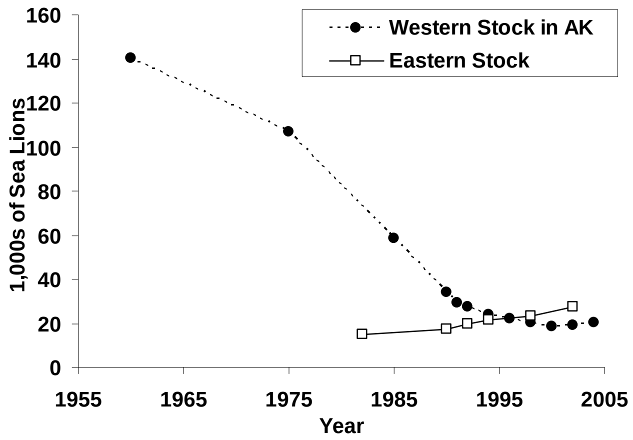


Figure I-1. A. At-sea and breeding ranges (rookeries) of the western and eastern stocks (DPSs) of Steller sea lion in the North Pacific Ocean. B. Counts of adult and juvenile (non-pup) Steller sea lions at index (trend) rookery and haulout sites in the range of the eastern and western (AK only) stocks.

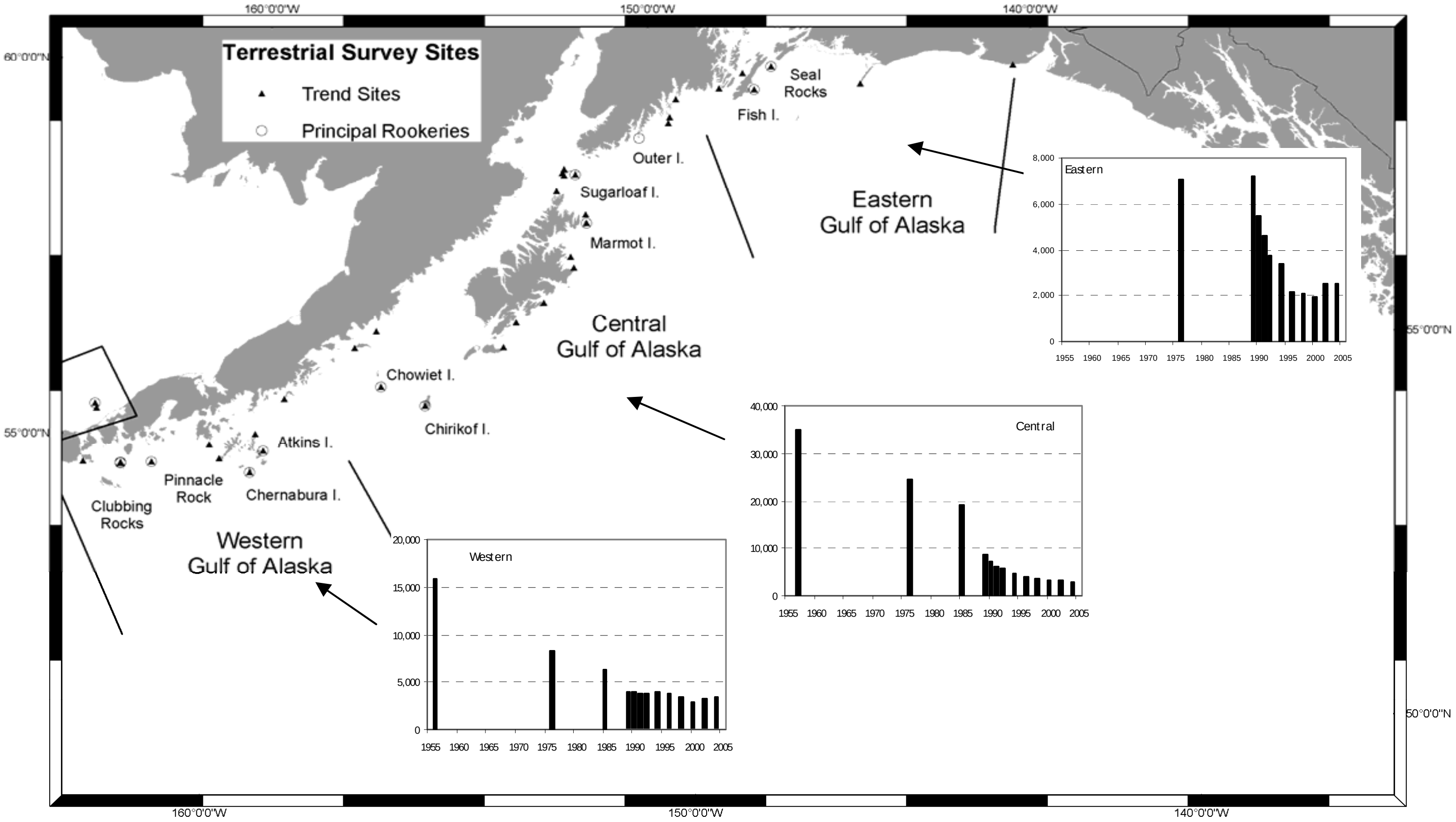


Figure I-2. Counts of adult and juvenile Steller sea lions on western DPS trend sites in three sub-areas of the Gulf of Alaska, 1950s through 2004. Principal rookeries (named) and major terrestrial haul-out trend sites are shown (NMFS 1992; Fritz and Stinchcomb 2005).

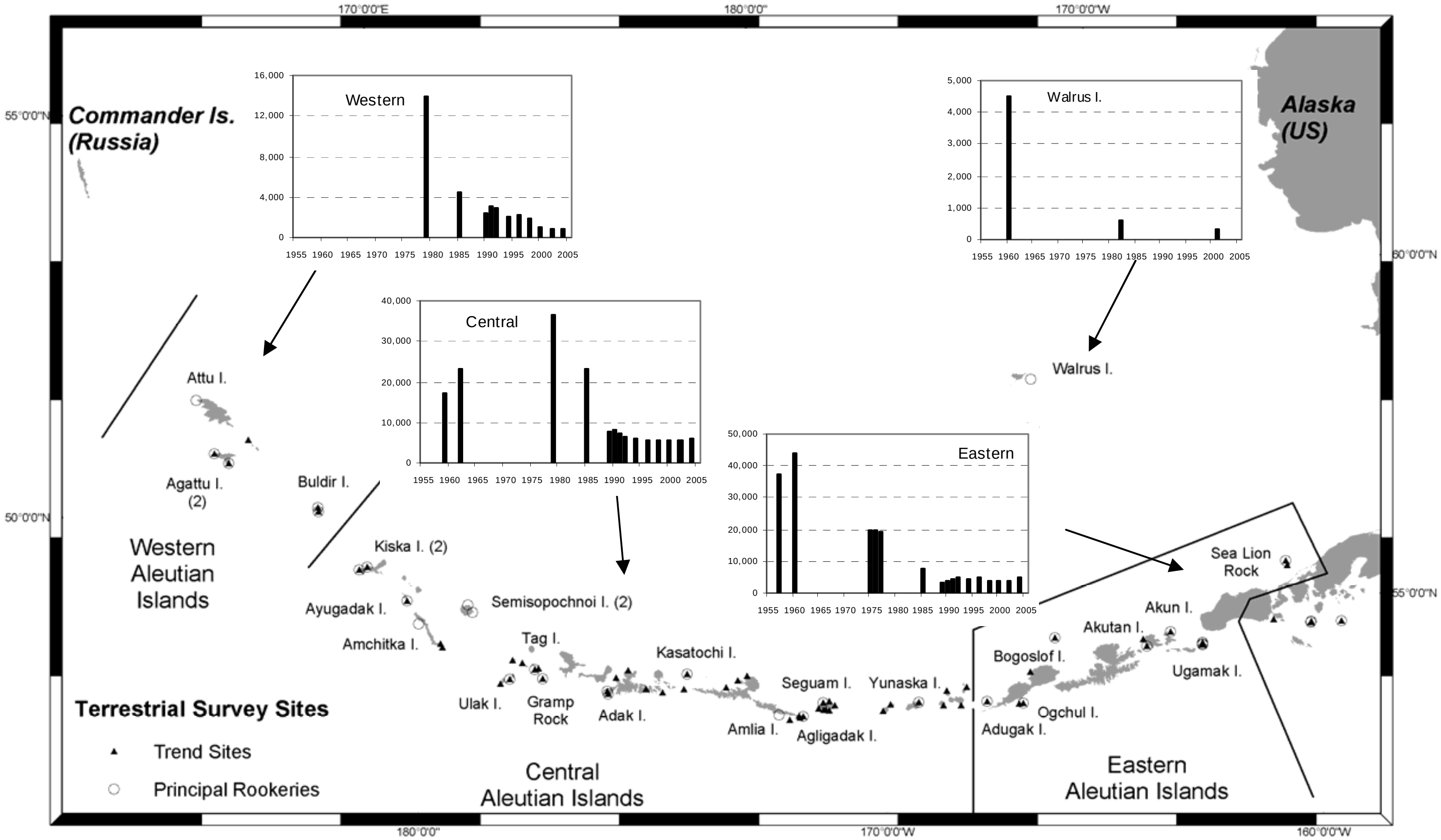


Figure I-3. Counts of adult and juvenile Steller sea lions on western DPS trend sites in three sub-areas of the Aleutian Islands, 1950s through 2004. Counts on Walrus Island in the eastern Bering Sea are also shown, as are the location of principal rookeries (named) and major terrestrial haulout trend sites (NMFS 1992; Fritz and

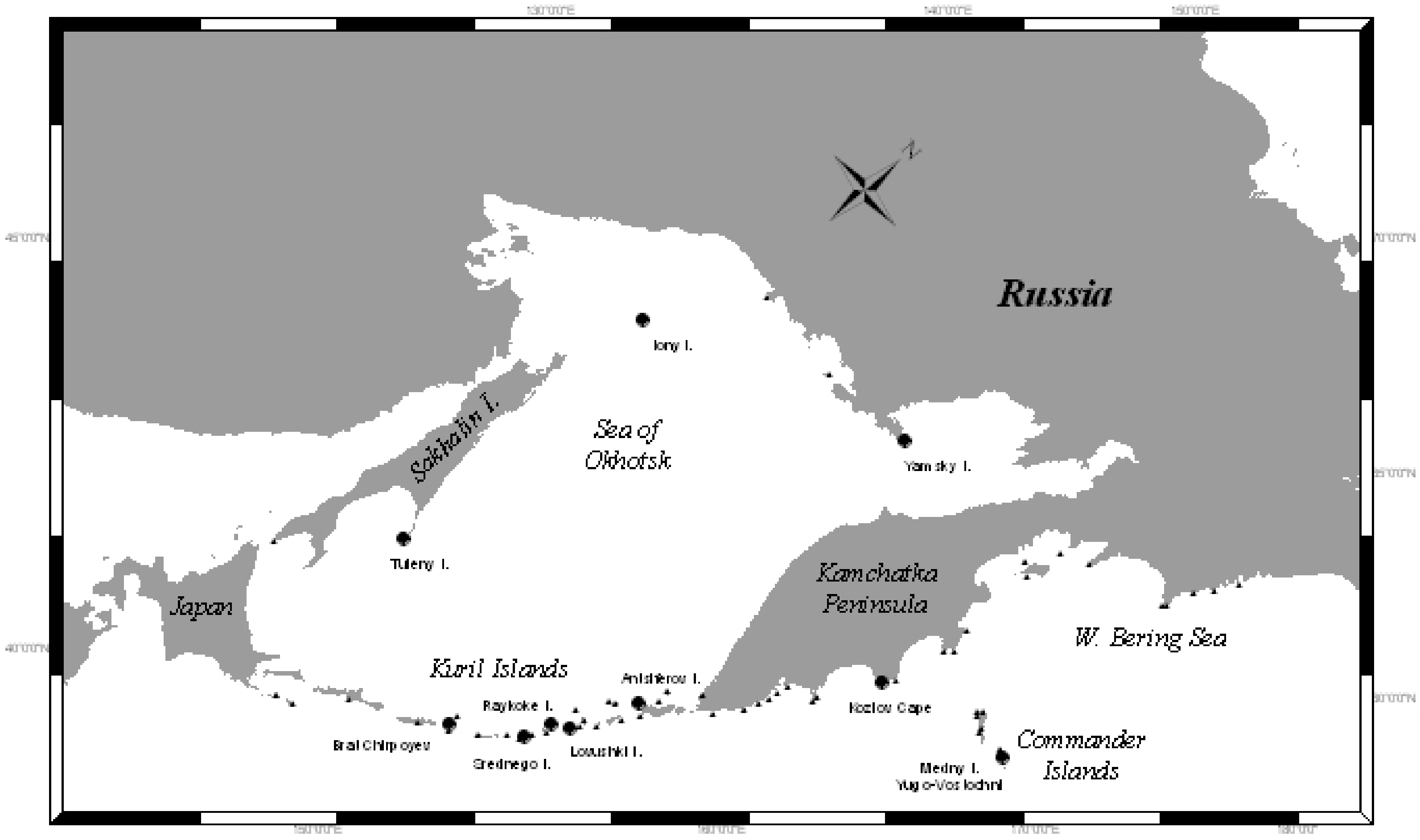


Figure I-4. Locations of Steller sea lion rookeries (named) and haul-out sites in Russia.

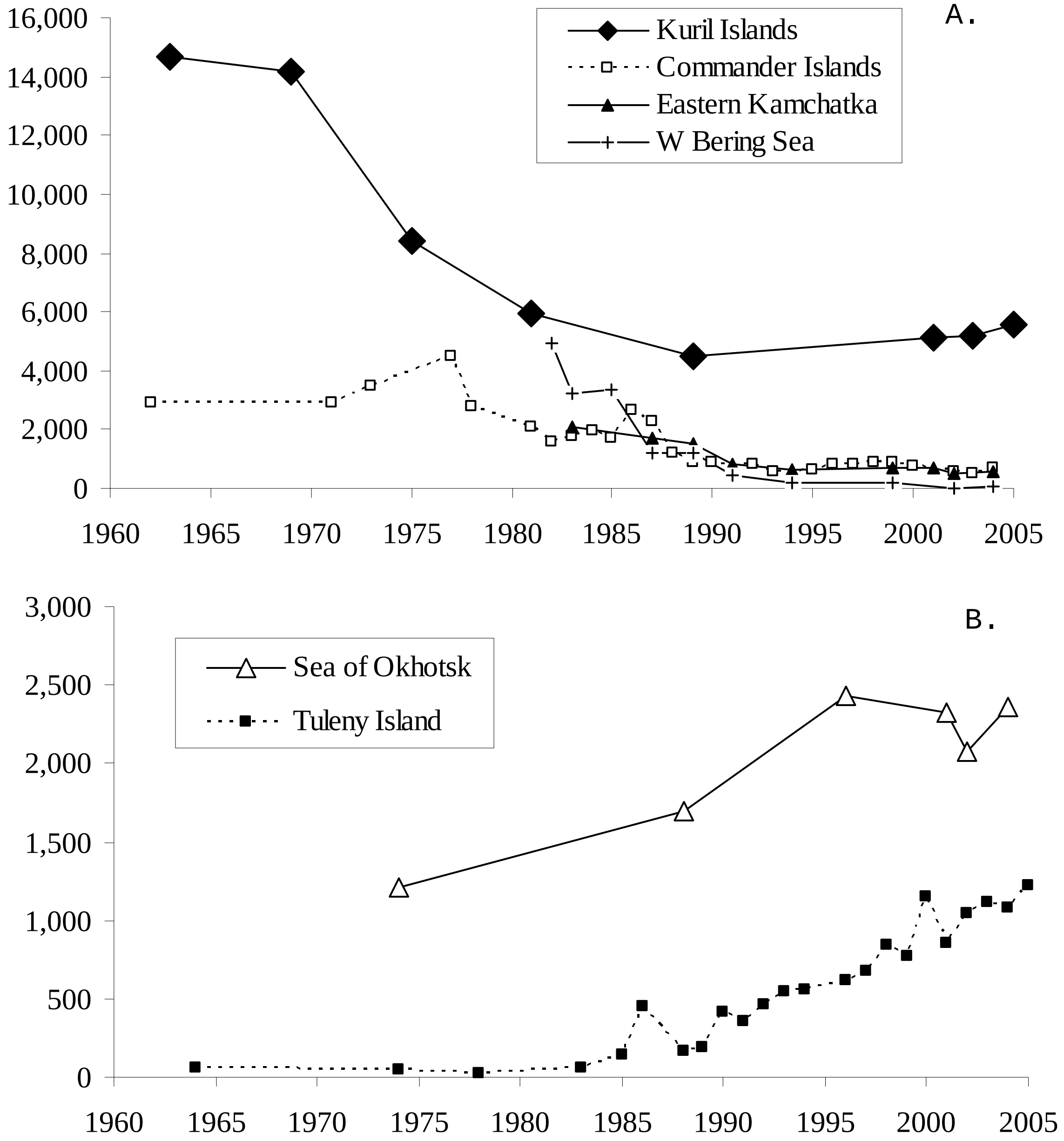


Figure I-5. Counts of non-pup Steller sea lions in Russia at trend haul-out and rookey sites by sub-area. A. Kuril Islands, eastern Kamchatka Peninsula, western Bering Sea (no rookeries) and the Commander Islands. B. Sea of Okhotsk and Tuleny Island near Sakhalin Island (only rookeries).

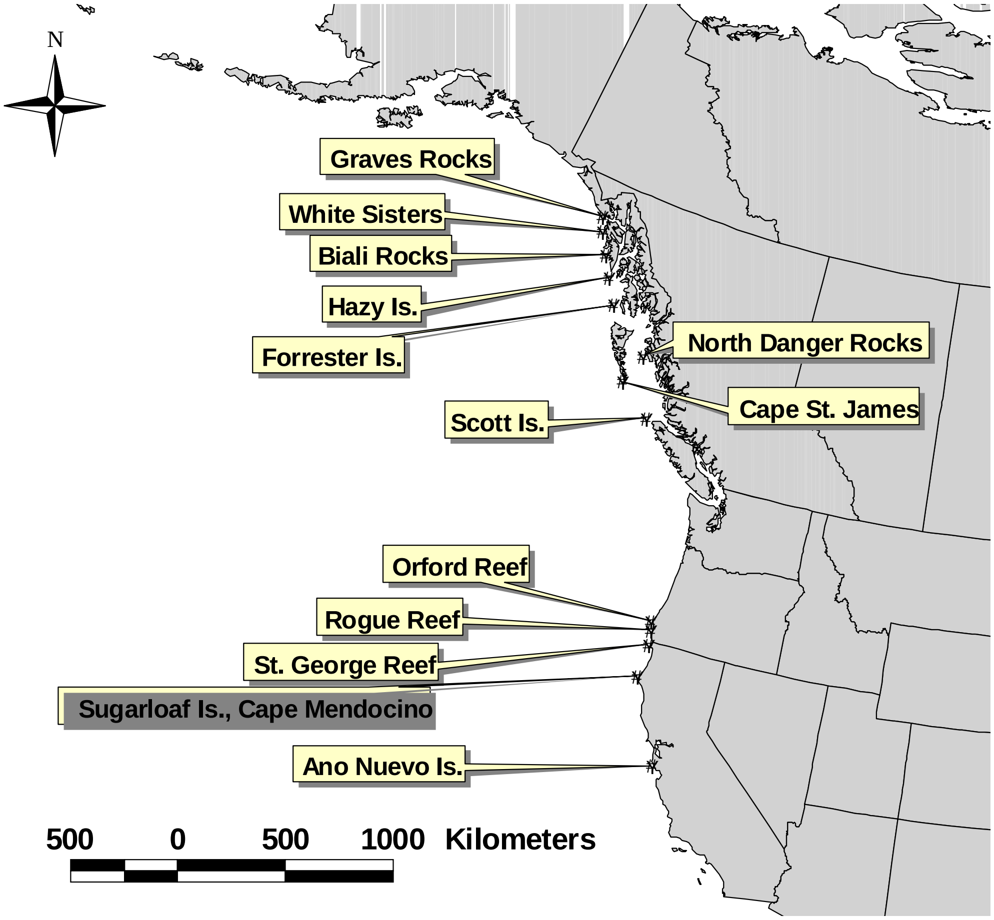


Figure I-6 Geographic range of the eastern DPS of Steller sea lion showing locations of current rookeries (sites where > 50 pups were born)

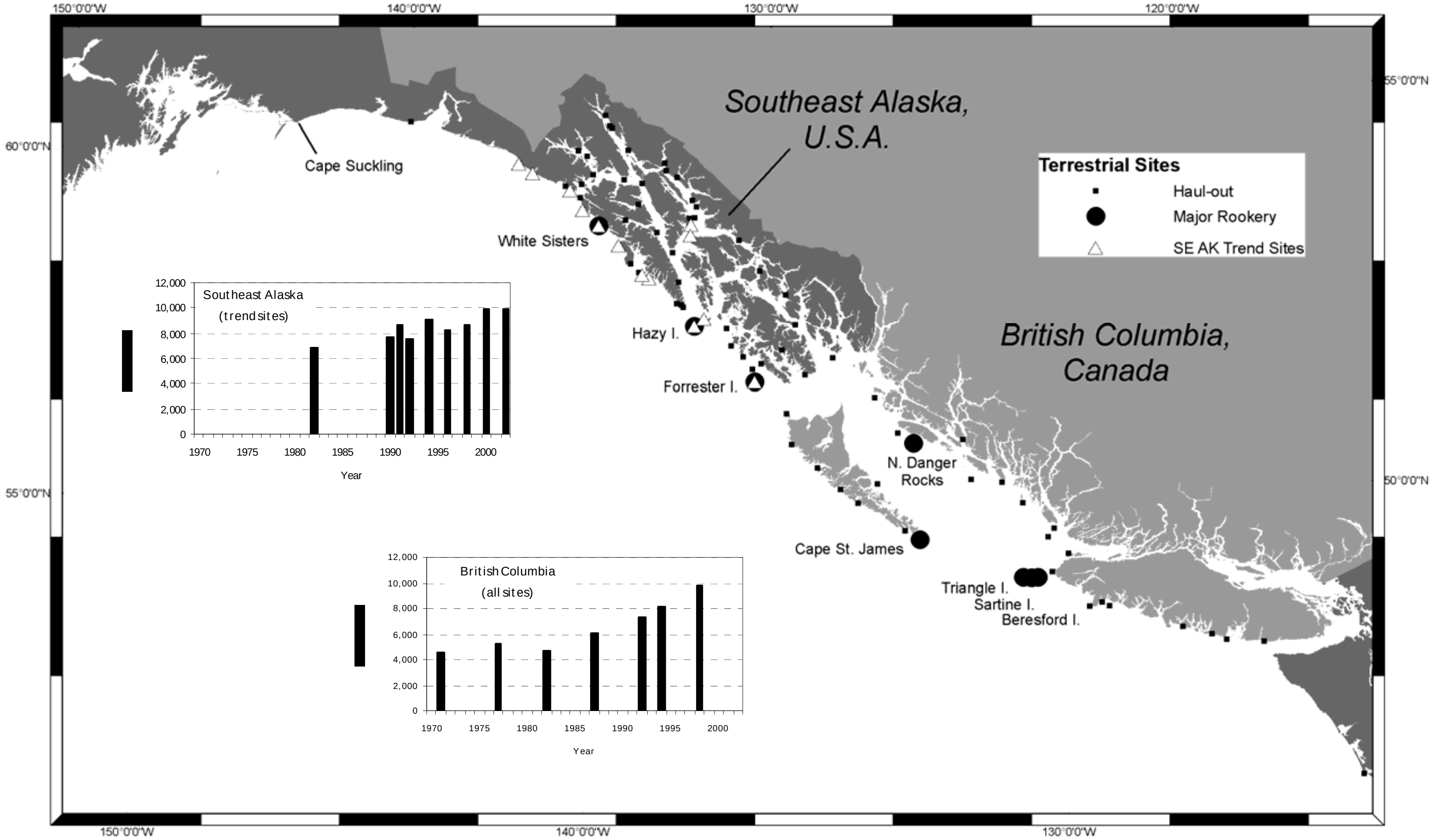


Figure I-7 Counts of adult and juvenile Steller sea lions (non-pups) on eastern DPS terrestrial sites in Southeast Alaska (SE AK; trend sites) and British Columbia (all sites), 1971-2002. Major rookeries are named in both sub-areas (Olesiuk et al. ADF&G references), as is the boundary between the eastern and western distinct population segments (Cape Suckling).

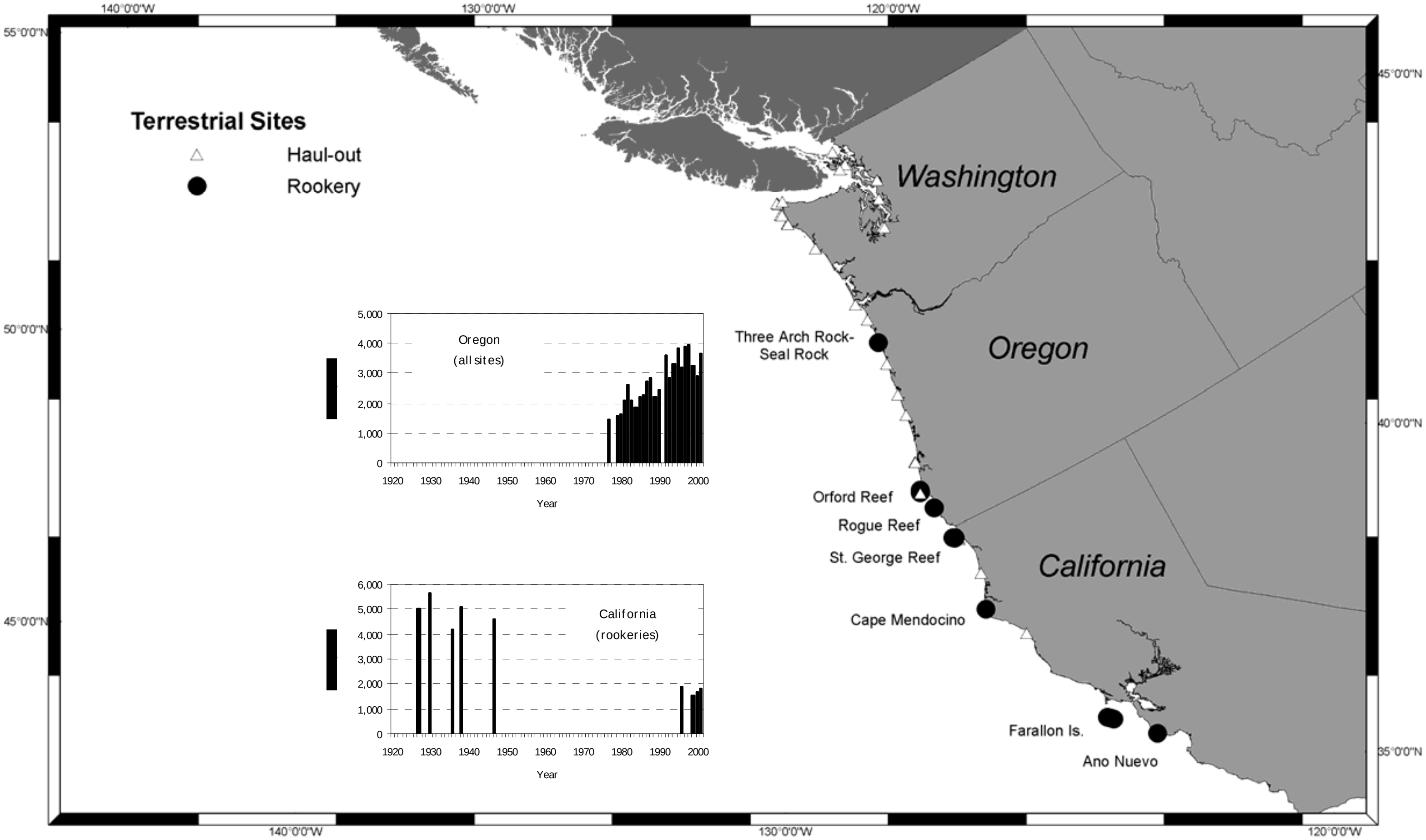


Figure I-8 Counts of adult and juvenile Steller sea lions (non-pups) on eastern DPS terrestrial sites in Oregon (all sites) and California (rookeries), 1927-2001. Major rookeries are named in Oregon and California; there are no rookeries in Washington.

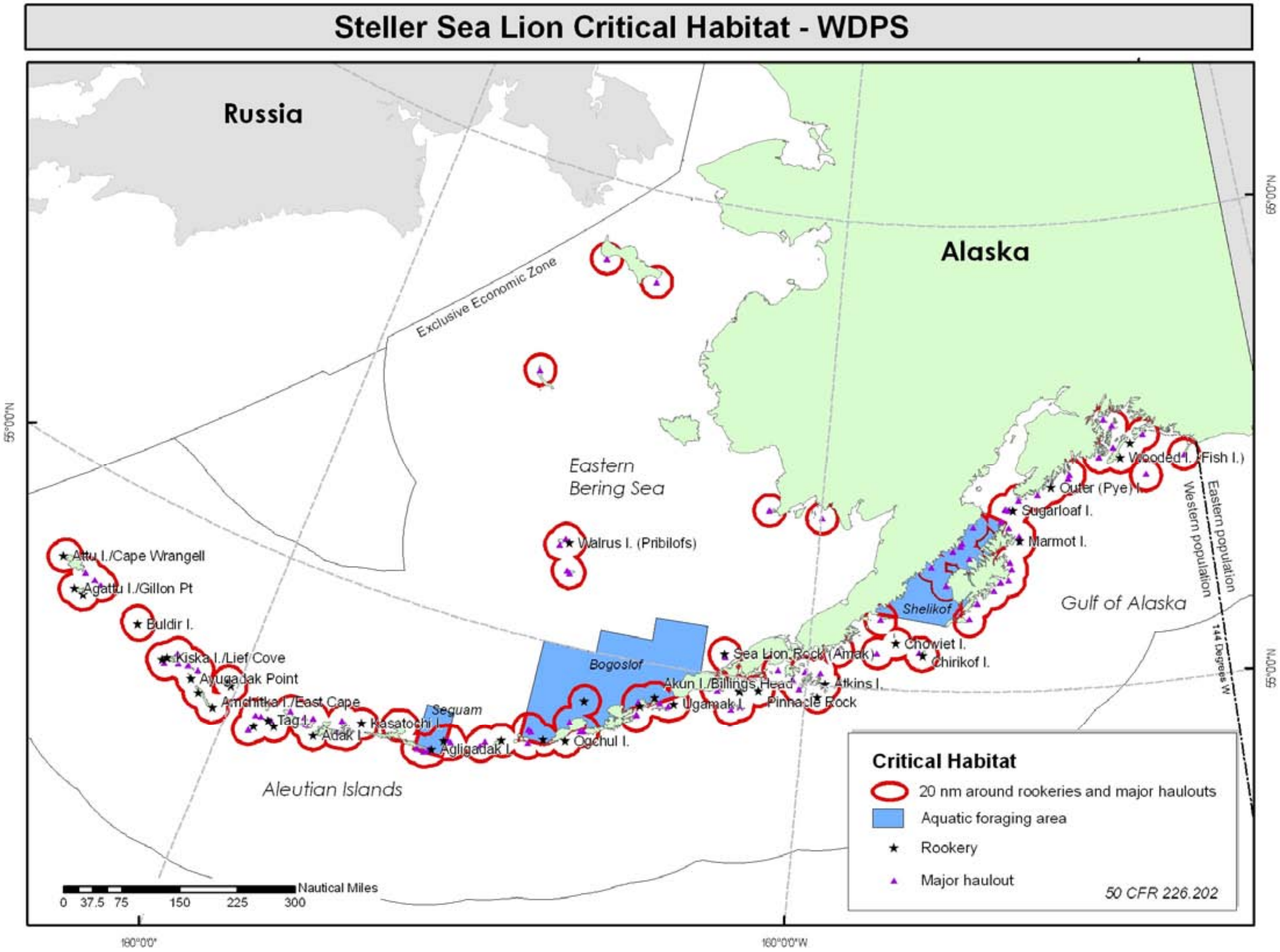


Figure I-9 Designated critical habitat for the western DPS of Steller sea lion in Alaska. 50 CFR 226.202

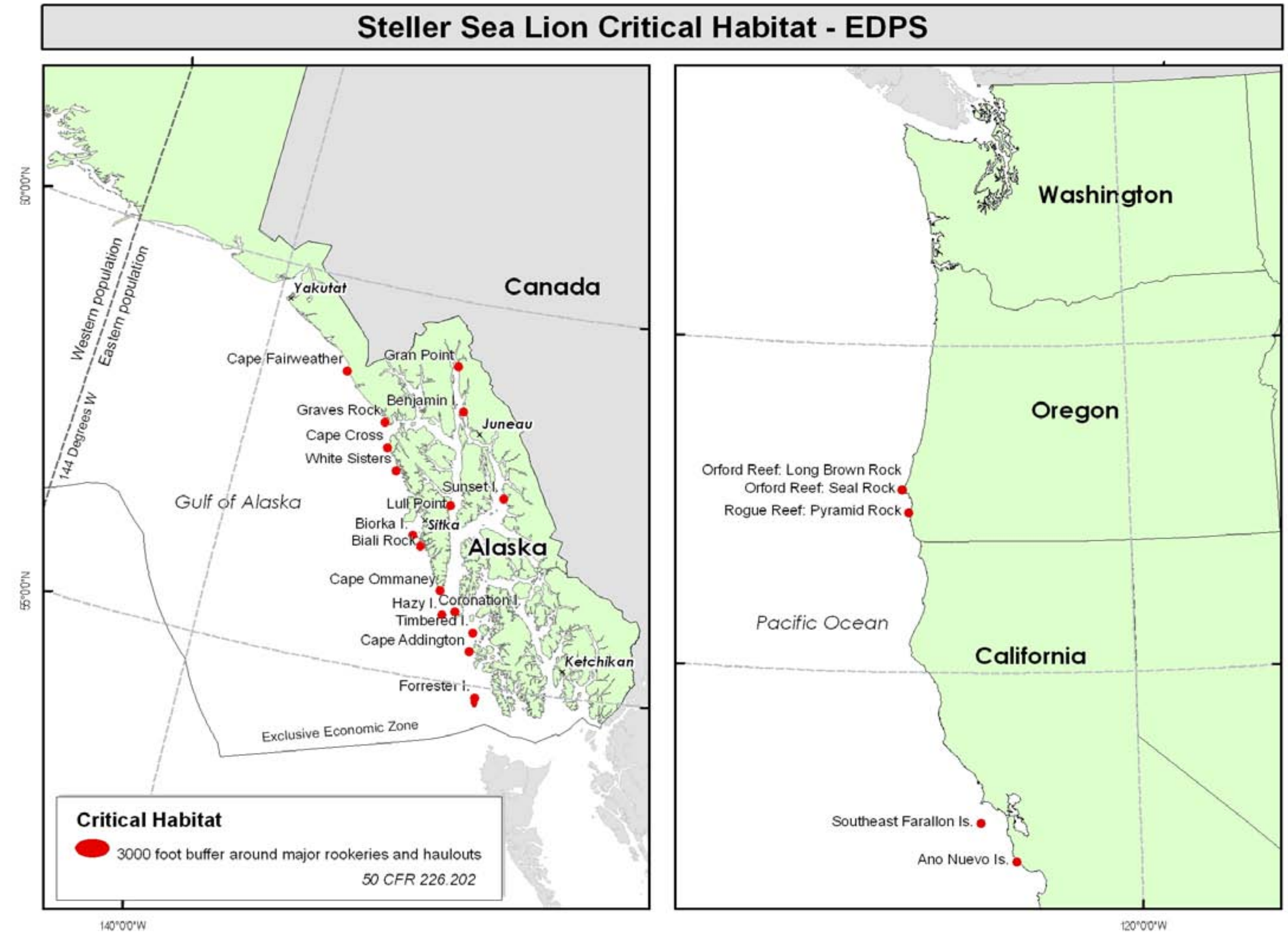


Figure I-10 Designated critical habitat for the eastern DPS of Steller sea lion. 50 CFR 226.202

II. CONSERVTION MEASURES

The objective of this section is to describe the conservation measures that have been undertaken to reduce threats. Not all threats have been addressed with conservation actions. For example, predation has been identified as a known cause of sea lion mortality and as a potential threat to recovery but is not discussed here because no actions have been taken or are proposed to reduce the threat. This section is primarily focused on the western DPS because most actions have been taken there. Conservation measures are organized by threat.

The primary conservation effort for the eastern DPS has been the prohibition of shooting. Historically, shooting destroyed many animals and extirpated rookeries and haulouts in the mid-1900s. In the western DPS, incidental take in fishing gear and the shooting of sea lions by fisherman and others were factors in the decline during the 1970s and 1980s. However, by the early 1990s, laws implemented under the Marine Mammal Protection Act (MMPA), ESA, and the Magnuson-Stevens Fishery Conservation and Management Act reduced the level of intentional take to a negligible amount.

From the mid-1990s to the present, conservation efforts in the western DPS have focused largely on federal fishery restrictions, disturbance issues, and subsistence harvests. Although available data indicate that actions to reduce intentional take have been effective, it is unknown whether fishery conservation measures have been effective in reducing threats to Steller sea lions. However, the reduced rate of decline and period of modest increase that followed the implementation of fishery measures suggest the possibility of a positive effect. Although these changes in overall population trajectories are correlated with fishery conservation measures taken since the 1990s, it is unknown whether the relationship is causal (Hennen 2006). Unlike the direct take of a species, indirect take through competitive interactions is nearly impossible to either prove or disprove.

A. Intentional and Illegal Killing

Prior to 1972, approximately 45,000 Steller sea lions were intentionally killed in Alaska during state-sanctioned commercial harvest and predator control programs (Merrick *et al.* 1987). A large but unknown number of Steller sea lions are believed to have been shot throughout the state between 1972 and 1990 (Trites and Larkin 1992).

These sources of direct intentional killing of Steller sea lions were banned following passage of the MMPA in 1972. A provision under section 118 of the MMPA, however, allowed fishermen to lethally deter Steller sea lions from interfering with commercial fishing operations. The provision allowing lethal deterrence was eliminated in 1990 when sea lions were listed as threatened under the ESA. Following this protection, both NOAA and fishing industry representatives supported a “Don’t Shoot Sea Lions” campaign and two cases of illegal shootings were successfully prosecuted in 1998. Increased public scrutiny and the threat of fishery closures curbed illegal killings, and the current level of illegal shooting is believed to be minimal (Angliss and Outlaw 2002).

Because it is illegal, intentional killing of Steller sea lions is rarely observed, and no formal reports of lethal deterrence in commercial fisheries have been recorded by fishermen or observers since the practice was banned. The two convictions cited above however resulted from confidential voluntary reports from commercial fishermen who witnessed and reported the violations to NMFS Enforcement agents.

Systematic surveys of shorelines have successfully located carcasses of gunshot Steller sea lions (Wynne 1990). In areas where subsistence hunting occurs, it is impossible to determine whether the gunshot sea lions were shot illegally or legally, in a subsistence harvest, and subsequently lost.

B. Incidental Takes in Commercial Fishing

Steller sea lions have been incidentally caught in a variety of commercial fishing gear including gillnets (Wynne 1990), trawls (Loughlin and DeLong 1983), and longlines (Angliss and Outlaw 2005). Steller sea lions may also ingest baited hooks set for salmon by commercial or recreational trollers (Angliss and Outlaw 2005). The frequency of lethal entanglements varies annually, by gear type and method, but the minimum estimate between 1996 and 2000 averaged 29.5 animals a year (Angliss and Outlaw 2005) and was 30.5 and 3.6 in 2005 for the western and eastern DPSs respectively (Angliss and Outlaw 2005).

The MMPA authorized the incidental take (serious injury and death) of marine mammals in the course of commercial fishing operations while striving to reduce that mortality to an insignificant level. The MMPA was amended in 1988 to better monitor the cumulative effects of fishery-specific incidental takes. As a result, each U.S. fishery is designated as being in one of three categories based on its frequency of marine mammal interaction; this "List of Fisheries" is reviewed annually. Vessel owners in Category I or II fisheries (frequent or occasional interactions) are required to register with the NMFS Marine Mammal Authorization Program and to record all lethal marine mammal interactions in a logbook (originally) or self-reports (currently). The 1988 amendments also required the Secretary to implement emergency regulations to prevent further taking of Steller sea lions if more than 1,350 were taken during a calendar year.

In addition, NMFS may place observers on Category I and II vessels to 1) obtain reliable estimates of incidental serious injury and mortality of marine mammals; 2) determine the reliability of reports submitted by vessel owners and operators; 3) identify changes in fishing methods or technology that may decrease incidental serious injury or mortality if necessary; 4) collect biological samples that may otherwise be unobtainable for scientific studies; and 5) record data on bycatch and discard levels of all species.

The 1994 amendments to the MMPA presented a new means of identifying and weighing the cumulative anthropogenic threats to each marine mammal stock and a process for reducing fishery-specific impacts. For each stock, a Potential Biological Removal (PBR) level is calculated that represents the annual human-induced mortality the stock can sustain, based on conservative estimates of minimum population level and net productivity and then reduced by a scaled recovery factor (Angliss and Outlaw 2005). Total annual human-related mortality is then compared to PBR to determine "Strategic Stocks" and identify those fisheries for which incidental take must be reduced. If incidental mortality of a stock in commercial fisheries exceeds PBR, NMFS is required to convene a Take Reduction Team and develop a Take Reduction Plan to reduce the level of incidental fishing-related mortality. Although the western stock of Steller sea lions is considered a "strategic stock," because of its depleted status under the MMPA , the current annual level of incidental take is lower than the calculated PBR; no Take Reduction Team has been convened for either stock.

Observer programs already collecting catch data under provisions of the Magnuson-Stevens Fishery Conservation and Management Act in Category I fisheries were assigned the additional task of reporting incidental marine mammal take in those fisheries. Under this program, incidental take of Steller sea lions is monitored by NMFS observers of groundfish trawl vessels fishing in Alaska, Washington, Oregon, and California (see Angliss and Outlaw 2005 for information on observer data from Alaska fisheries).

For Category I and II state fisheries, NMFS developed a Marine Mammal Observer Program under the MMPA mandates. The Alaska Marine Mammal Observer Program has monitored the incidental take of Steller sea lions and other marine mammals and birds in state-managed set and drift gillnet fisheries for salmon occurring in Prince William Sound, S. Alaska Peninsula, Cook Inlet, and Kodiak. Observers continue to document the incidental take of Steller sea lions from the eastern DPS occurring in the California and Oregon thresher shark and swordfish drift gillnet and the Northern Washington set gillnet fishery. Updated information on incidental fishing-related mortality is incorporated into annual NMFS reviews of the status of marine mammal stocks, including Steller sea lions[7].

C. Subsistence Takes

Alaska Natives were exempted from the 1972 MMPA and ESA ban on taking marine mammals. This exemption allows Alaska Natives to continue taking marine mammals for subsistence or handicraft purposes. The mean annual harvest of Steller sea lions (including struck and lost, which are those animals presumed killed but not recovered) by Alaska Natives for 2000 - 2004 was estimated by the subsistence division of the ADF&G to be 190.4 (Angliss and Outlaw 2005).

In 1994, section 119 of the MMPA was amended to allow for the co-management of marine mammal stocks used for subsistence purposes by Alaska Natives. Co-management provides a mechanism for NMFS to work with Alaskan Native Organizations (ANOs) to manage use of marine mammal species listed under the ESA and to participate in research efforts. For example, the Tribal Government of St. Paul, the Aleut Community of St. George located in the Pribilof Islands, and the Aleut Marine Mammal Commission covering the Aleutian Islands have each signed co-management agreements with NMFS for Steller sea lions. NMFS may enter into additional co-management agreements for sea lion conservation with other ANOs in the future.

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[7] The last observed take of a Steller sea lion in the CA/OR drift gillnet fishery was in 1994. The Offshore Cetacean Take Reduction Plan for this fishery was implemented through regulations in 1997. Under this plan, skippers are required to use a minimum extender length (36 feet), place pingers on their nets, and attend skipper workshops, when scheduled. These requirements have likely reduced the incidental take of Steller sea lions.

The Tribal Governments of St. Paul and of St. George monitor sea lion subsistence harvest as a function of their co-management agreements and provide harvest information to NMFS. The Tribal Government of St. Paul has implemented a real-time harvest monitoring method to increase the accuracy in reporting. This method was also adopted by St. George in 2005. Harvest monitoring in other areas is conducted by the Alaska Department of Fish and Game Department of Subsistence.

D. Research-related mortality

Intentional lethal sampling of Steller sea lions was a primary means of collecting reproductive, morphometric, dietary, and histologic samples for scientific research in the 1960s and 1970s. This sampling method was strictly regulated after passage of the MMPA and was discontinued once the species was listed as Threatened under the ESA.

Activities authorized under the MMPA and ESA are highly regulated and closely monitored and may include the incidental taking or harassment of Steller sea lions in the course of bonafide research. These research activities, including counting, capturing, and handling animals, may result in inadvertent or indirect Steller sea lion mortality.

Efforts are underway to reduce the amount of disturbance on rookeries caused by the presence of researchers for the purpose of counting. Aerial surveys may serve as an alternative to some of the work currently necessitating human presence.

The NMFS Permit office reviews permit applications, which are also reviewed by the Marine Mammal Commission and made available for public review through notice in the federal Register. Researchers are required to submit annual plans and reports of research activities and real-time reports of research-related mortality. Cumulative impacts of multiple projects are monitored by a Regional Coordinator, and all research may be curtailed if incidental mortalities reach a pre-determined cap.

E. Pollution, Contaminants, and Entanglement in Marine Debris

Steller sea lions are exposed to local and system-wide contaminants and pollutants as they traverse the North Pacific basin. Effects on other pinnipeds have included acute mortality, reduced pregnancy rates, immuno-suppression, and reduced survival of first born pups (see Section III), but there have been no published reports of contaminants or pollutants representing a mortality source for Steller sea lions.

Steller sea lions have been observed with packing bands, discarded netting, and other debris around their necks. Such debris can be lethal if the debris is not degradable. Annex V of the MARPOL Treaty bans the dumping of plastic trash in the ocean or navigable waters of the U.S. (outside 3 nm from shore). Information and education combined with voluntary community-based efforts have resulted in the retrieval, recovery, and disposal of discarded nets and gear in several fishing areas (e.g. Oregon, St. Paul Island, Puget Sound).

Researchers record the frequency and type of debris observed on Steller sea lions during resight surveys and, infrequently, the relative amount and type of debris seen on haulouts and rookeries they visit.

F. Disturbance on Terrestrial Sites and Critical Habitat

Disturbance of Steller sea lion haulouts and rookeries can potentially cause disruption of reproduction, stampeding, or increased exposure to predation by marine predators. Terrestrial habitat has been protected throughout the range by a variety of agencies, and by the fact that sea lions generally inhabit remote, unpopulated areas. Many haulouts and rookeries used by the western DPS are afforded protection from disturbance because they are located on land whose access is regulated by the Alaska Maritime National Wildlife Refuge and other agencies.

No transit zones for vessels within 3 nm of listed rookeries were implemented under the ESA during the initial listing of the species as threatened under the ESA in 1990. These 3 nm buffer zones around all Steller sea lion rookeries west of 150°W were designed to prevent shooting of sea lions at rookeries. Today, these measures are important in protecting sensitive rookeries in the western DPS from disturbance from vessel traffic. In addition, NMFS has provided "Guidelines for Approaching Marine Mammals" that discourage approaching any closer than 100 yards to sea lion haulouts.

Since the listing of Steller sea lions in 1990, NMFS has commented on hundreds of federal actions through the informal consultation process. NMFS commonly consults informally with the U.S. Forest Service on logging projects, with the EPA on discharge permits, and with the Minerals Management Service on oil and gas lease sites. NMFS comments on actions that may take place in sensitive Steller sea lion critical habitat and suggests means to avoid the most sensitive areas or minimize the likelihood of having adverse impacts.

In 2002, NMFS implemented the North Pacific Fishery Management Council's recommendation to require a Vessel Monitoring System (VMS) on federally licensed groundfish vessels involved in pollock, cod and Atka mackerel fisheries. The VMS tracks fishing vessels, providing real-time information on vessel location and violation of no-transit and no-trawl areas.

G. Reduced Prey Availability due to Fisheries

Steller sea lions prey upon some fish species that are also harvested by commercial, subsistence, and recreational fisheries (e.g. pollock, Pacific cod, Atka mackerel, salmon, and herring). Evaluation of catch versus estimated biomass of selected prey species, as well as the level of catch in designated critical habitat, indicates that fishery removals have the potential to reduce the availability of these species to sea lions at a variety of spatial and temporal scales (Figures II-1, 2, and 3). Reduced prey availability can represent an acute or chronic threat to sea lion populations. Acute prey shortages may lead to starvation while chronic (or sub-lethal) prey shortages have been shown in other mammals to reduce reproductive fitness, increase offspring mortality, and increase the susceptibility to disease and predation.

Immediately after listing in the early to mid-1990s, NMFS implemented a number of conservation measures intended to ensure that commercial harvests of pollock, Pacific cod, and Atka mackerel would not limit the recovery of Steller sea lions (Ferrero *et al.* 1994, Fritz *et al.* 1995). In addition to those direct actions, many other fishery management measures recommended by the North Pacific Fishery Management Council and implemented by NMFS may have indirectly contributed to Steller sea lion conservation efforts.

In the late 1990s and early 2000s, NMFS reviewed federally managed groundfish fisheries in a series of consultations under section 7 of the ESA. Two of those consultations resulted in a determination that the commercial fisheries were likely to jeopardize the continued existence of the western DPS of Steller sea lion and adversely modify its critical habitat. Therefore, as required under the ESA, additional conservation measures were implemented to avoid jeopardy and adverse modification (NMFS 1998a, NMFS 2000). The expectation was that these measures would promote the recovery of Steller sea lions in areas where potential competition from commercial fisheries may have contributed to the population decline.

A suite of fishery conservation measures was implemented in 2002 after being reviewed under a subsequent ESA section 7 consultation (NMFS 2001). These measures are described in detail in the 2001 Biological Opinion (NMFS 2001) and its Supplement (NMFS 2003). New information highlighted the importance of near-shore areas to juveniles. Thus, the measures were intended to reduce fishing in near-shore critical habitat, reduce seasonal competition for prey during critical winter months, and disperse fisheries spatially and temporally to avoid local depletions of prey.

Steller sea lion protection measures for the groundfish fishery currently include (1) global harvest controls for Steller sea lion prey species (pollock, Pacific cod, and Atka mackerel); (2) spatial harvest controls specific to prey species, gear type, and proximity to rookery, haulout, or forage areas to limit prey species removal in an area; (3) temporal harvest controls for pollock, Pacific cod, and Atka mackerel, including seasonal apportionments to limit prey species removal during certain times of the year; and (4) a vessel monitoring system requirement for all vessels (except vessels using jig gear) fishing for pollock, Pacific cod, or Atka mackerel.

The 2002 measures provided increased protection for near-shore critical habitat areas based on an analysis that closely examined satellite telemetry data from juveniles, and on information of foraging behavior, diet, nutritional stress, and population distribution. The analysis placed increased importance on near-shore critical habitat, specifically identifying those areas within 0-10 nm of listed haulouts and rookeries as more important for foraging sea lions than waters from 10-20 nm offshore.

NMFS (2003), re-evaluated each of the conservation measures after they had been implemented in 2002 and concluded that despite various levels of effectiveness in achieving specific goals, the conservation measures were, in aggregate, successful in avoiding jeopardy and adverse modification of critical habitat. A summary table of the effectiveness of each of the actions can be found in Table IV-1 of NMFS (2003). NMFS (2003) provides an in-depth review of each of the conservation measures, a review of the satellite telemetry data, and an analysis of the important foraging areas for sea lions based on those data. Further, the federal fishery management measures that may have affected Steller sea lions are summarized in North Pacific Fishery Management Council fishery management plans for groundfish. (North Pacific Fishery Management Council 2005a, 2005b).

Additional fine-scale analyses (similar to NMFS 2003), on the adequacy and effectiveness of fishery regulations are needed to develop a robust recovery program for sea lions (see recovery action 2.6.6). In general, more frequent analyses need to be conducted at the spatial and temporal scales essential to foraging sea lions to examine the potential for fisheries to reduce prey at the local, regional, and global scale. Such an approach will enhance assessments of how regulations designed to disperse (both spatially and temporally) and limit the catch are reducing potential competition, and help NMFS determine what modifications are appropriate. Beginning in 1999, NMFS has conducted a series of investigations of fishery effects on local fish populations. These studies, commonly called the FIT studies (after the NMFS Fishery Interaction Team that conducted them), were conducted on Atka mackerel on several local populations in the Aleutian Islands, Pacific cod on the spawning aggregation north of Unimak Pass, and pollock in neighboring gullies south of Kodiak Island. We provide additional detail about these studies in Section II.B.11 below.

The implementation of conservation measures in the 1990s and early 2000s are correlated with a reduction in the rate of decline of the western DPS. However, the information necessary to determine if the conservation measures actually contributed to the reduced rate of decline is not currently available. NMFS (2003) assessed the effectiveness of each of the measures and generally found that they met their intended effect in dispersing and limiting catch. Various reviews have highlighted the need to further assess the efficacy of these measures in order to determine whether changes are needed. However, it is not the role of this plan to evaluate fishery management at this scale; the issues are extremely complex resulting in extensive and controversial biological opinions. The approach taken here is to highlight the important areas of need, provide clear recovery criteria, and to outline a strategy for eliminating uncertainty. The in-depth assessment of particular fishery measures in relation to the recovery goals will be evaluated during section 7 consultations. The major difficulty with this approach is that fisheries authorized by the State of Alaska are not subject to independent consultation under section 7 of the ESA. For this reason the Plan calls for further review of these fisheries through habitat conservation planning (section 10 of the ESA).

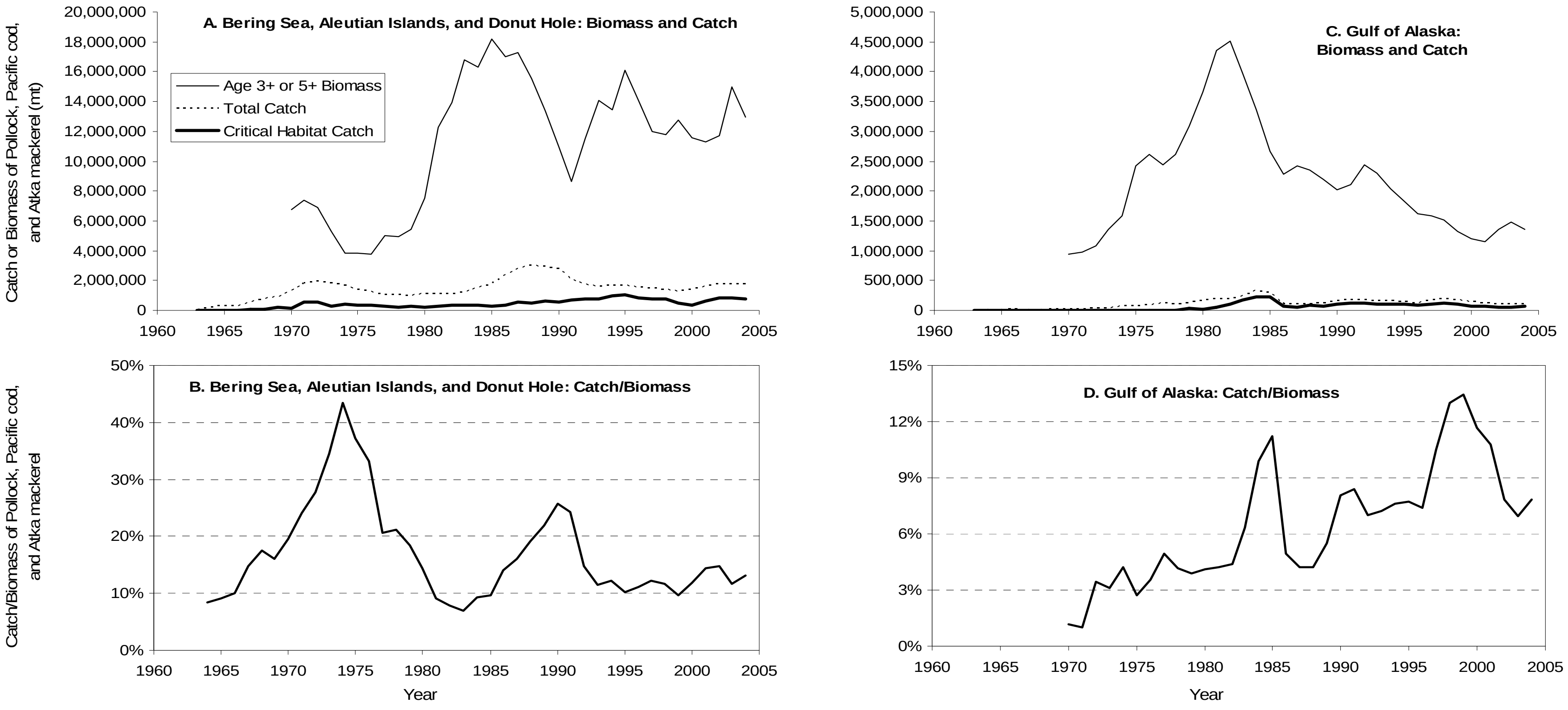


Figure II-1. (A) Catch and estimated age 3+ biomass of walleye pollock, Pacific cod and Atka mackerel in the eastern Bering Sea, Aleutian Islands and "Donut Hole" (international waters of the central Bering Sea) from 1964-2004. Estimated biomass is from stock assessments and includes Bogoslof pollock biomass (Ianelli et al. 2005, Lowe et al. 2005, Thompson et al. 2005). (B) Annual harvest rates calculated from panel (A). (C). Catch and estimated age 3+ biomass of walleye pollock and Pacific cod in the Gulf of Alaska from 1964-2004. Estimated biomass is from stock assessments (Dorn et al. 2005, Thompson et al. 2005). Total catch as well as that portion removed from Steller sea lion critical habitat are shown. (D) Annual harvest rates for the GOA fisheries from panel (C).

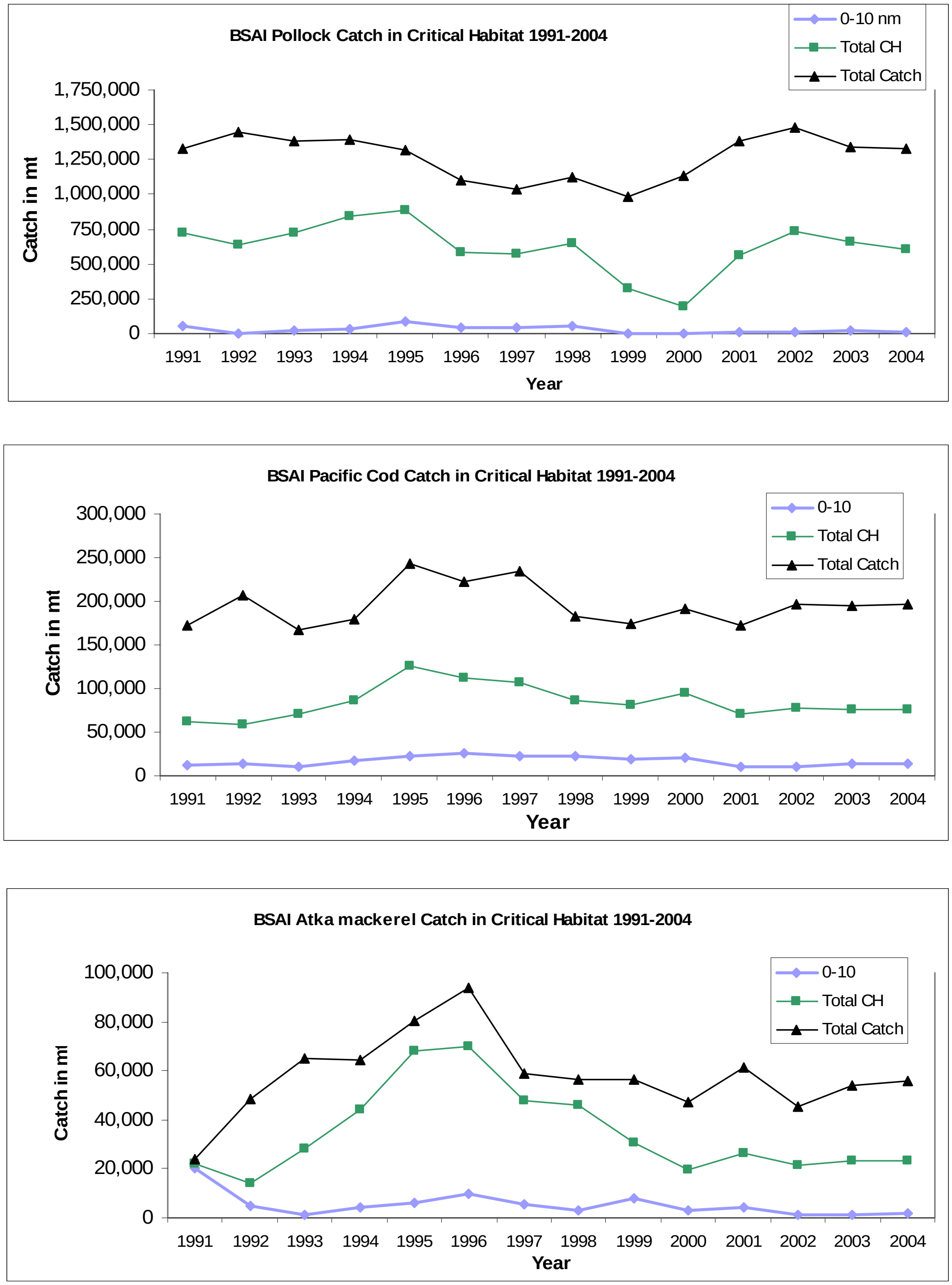
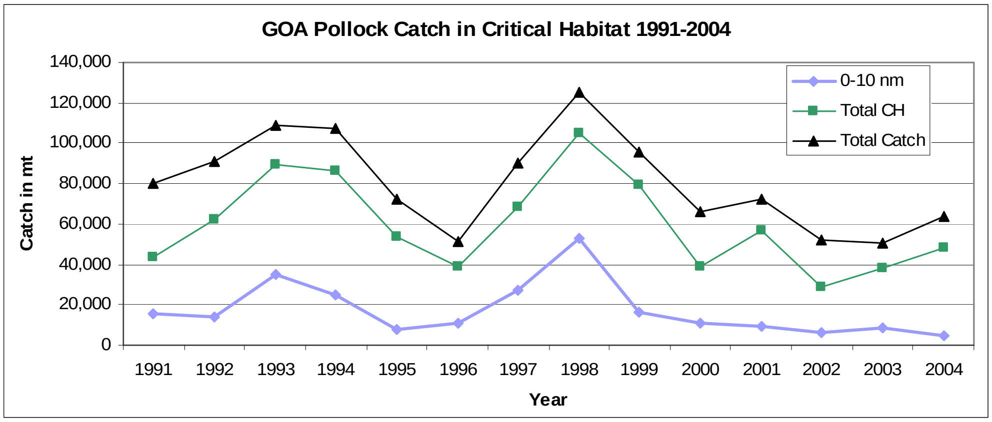


Figure II-2. Catch of Pollock, Pacific cod, and Atka mackerel in critical habitat in the Bering Sea Aleutian Islands (BSAI) from 1991-2004.



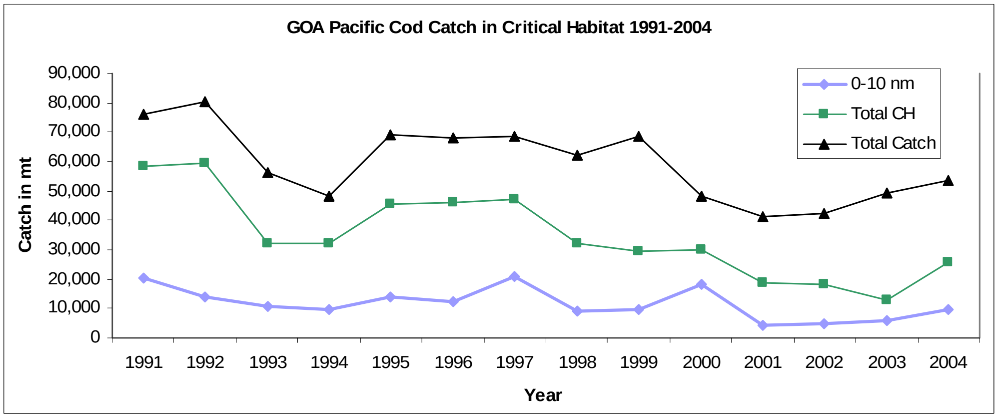


Figure II-3. Catch of Pollock and Pacific cod in critical habitat in the Gulf of Alaska (GOA) from 1991-2004.

III. FACTORS POTENTIALLY INFLUENCING THE WESTERN POPULATION

Differences in the timing and magnitude of the regional population trajectories in the 1970s, 1980s, and 1990s suggest that the overall western DPS decline was not caused by a single factor, but rather by the cumulative effect of multiple factors that had different relative spatial and temporal magnitudes. Indeed, the marked change in the rate of the decline since 1990 suggests that the factors that contributed to the more rapid prior declines may not be the most significant factors operating today (Bowen *et al.* 2001, NRC 2003); in addition, there may have been density-dependent responses at lower population levels. The following discussion provides an overview of those factors, describes how they may continue to be a threat to sea lions, and includes a discussion of data gaps. Section IV (Threats Assessment), will evaluate the information provided here in order to determine the magnitude and nature of the threat to the recovery of Steller sea lions.

We have only a limited or qualitative understanding of how multiple factors interact to create an overall cumulative effect on sea lion populations. In addition, data are insufficient to show what the natural dynamics of Steller sea lion populations have been. Such dynamics would be driven primarily by changes in the North Pacific ecosystem that affect carrying capacity (e.g., prey abundance), but would also be affected by changes in rates of predation and disease. Increased knowledge of both natural ecosystem dynamics and how human activities influence those dynamics is required before their respective impacts on sea lions can be delineated with certainty (NRC 1996, NMFS, 2001, NRC 2003). Yet, a number of theories attempting to explain the decline in sea lions and apparent changes in the structure of North Pacific ecosystems since the 1970s have been developed, and these involve top-down (e.g., direct), bottom-up (e.g., indirect) or a combination of both types of forces (NRC 1996, Anderson and Piatt 1999, Merrick 1997, Orensanz *et al.* 1998, Estes *et al.* 1998, Francis *et al.* 1998, Trites *et al.* 1999, NMFS 1998a, NMFS 2000, Jackson *et al.* 2001, Hunt *et al.* 2002, NRC 2003, Springer *et al.* 2003, Guénette *et al.* 2006, Trites *et al.* 2007b). Depending on the emphasis placed within each individual theory, trophic cascades and systemic modifications were triggered alone or in various combinations by whaling, fishing, predation, or atmospheric and oceanographic changes.

A. Food Web Interactions

The purpose of this section is to provide a general description of how food web interactions via top-down and bottom-up threats may have acted to substantially reduce the Steller sea lion population, and how their aggregate impacts may affect sea lions now and in the near future. Humans are a part of this food web; they may change food web dynamics through direct take of sea lions, removal of sea lions' preferred prey, the removal of large whales, or likely a combination of the above and many other factors (NRC 2003, Wilmers *et al.* 2006).

1. Direct and indirect factors

Factors affecting sea lions can be either bottom-up or top-down, and operate through either a direct or indirect mechanism. Direct threats are generally top-down and involve the instantaneous mortality of the animal (e.g., incidental take in fishing gear, killer whale predation, commercial harvest, subsistence harvest). Indirect threats are generally bottom-up and can lead to decreased rates of reproduction or survivorship (e.g., by making them more vulnerable to direct threats, such as predation) by reducing individual condition or fitness. Diseases, parasites, and toxic substances (contaminants) can indirectly threaten sea lion populations through reduced fitness, inability to obtain prey, and greater vulnerability to predation. Likewise, environmental variability and competition with fisheries have the potential to cause similar indirect effects due to nutritional stress through reductions in prey biomass, availability or quality. Indirect effects of fishing in the BSAI and GOA are unlikely to be trivial (especially if catch is concentrated in time or space), yet we have limited ability to discuss these scenarios and even less ability to assess the impacts on recovery (NRC 2003).

2. Top-down

Several factors act as top-down sources of sea lion mortality; i.e., commercial harvest, intentional shooting, entanglements or incidental catch by fishing gear, disturbance, and predation. Top-down sources of mortality were significant contributors to the sea lion population declines observed prior to the 1990s, when there were relatively large reductions in juvenile survival rates, and smaller reductions for adults (Pasqual and Adkison 1994, York 1994, Holmes and York 2003, Fay 2004). Since 1990, rates of mortality from harvests, shooting, and entanglement and incidental catch have been much less, and that has been reflected in a rebound in both juvenile and adult survival rates (Holmes and York 2003, Fay 2004). Subsistence harvests of Steller sea lions continue but have declined substantially. As previously described, predation by killer whales has the potential to be a significant additional top-down source of mortality (Williams *et al.* 2004). Springer *et al.* (2003) proposed a hypothesis in which killer whales shifted their diet from large whales (following extensive commercial whaling in the 1950s and 1960s) to pinnipeds, resulting in sequential collapses of northern fur seals, harbor seals, and Steller sea lions, and culminating in the collapse of sea otter populations (see also Estes *et al.* 1998). This "Sequential Megafaunal Collapse" hypothesis has been called into question because of inconsistencies with data on large whale catches, killer whale diets, and the spatial-temporal patterns of pinniped declines (Barrett-Lennard *et al.* 1995, Trites *et al.* 2007c, DeMaster *et al.* 2006, and Wade *et al. 2007*). We provide a discussion of this hypothesis and refuting studies in the section entitled "Sequential Megafaunal Collapse" hypothesis" of this Plan. If one or more top-down threats were major impediments to recovery for the western DPS, continued low rates of juvenile and/or adult survivorship would be expected or observed, potentially with little or no change in natality, birth rates or condition. Current estimates of sea lion vital rates do not follow these expected trends.

3. Bottom-up

Evidence that bottom-up factors may have contributed to the decline observed from the mid-1970s through the late 1990s include reductions in size at age (Calkins and Goodwin 1988), possible depressed late-term pregnancy rates (Pitcher *et al.* 1998), and a decline in per capita natality of female sea lions at some rookeries (Holmes and York 2003, Fay 2004, Winship and Trites 2006). These responses by sea lions are opposite to those predicted by top-down factors (Bowen *et al.* 2001, NRC 2003), as body condition, growth rates, and natality should increase or remain the same when population abundance is reduced. Therefore, bottom-up factor(s) appeared to be affecting sea lions at least as early as the 1970s, at the same time that large numbers of sea lions were also killed directly. The combination of reduced population abundance and poor body condition indices is consistent with a substantial reduction in carrying capacity (Gerrodette and DeMaster 1990).

The two bottom-up factors hypothesized to have contributed most to the decline are reductions in prey biomass and quality resulting in nutritional stress (proximate cause) that subsequently decreases vital rates (Trites *et al.* 2006a). However, there are two hypotheses about the ultimate causes of nutritional stress. In one, nutritional stress stems from climate induced changes in the species composition, distribution or nutritional quality of the sea lion prey community (see review by Trites and Donnelly 2003, Trites *et al.* 2006a, and Trites *et al.* 2007b). In the other, fishery-induced reductions in localized or overall prey abundance cause nutritional stress (Braham *et al.* 1980, NMFS 1998a, 2000). Both climate shift and fisheries induced changes in prey communities may have affected the condition of Steller sea lions over the last 40 years, but the relative importance of each is a matter of considerable debate.

The carrying capacity of the North Pacific for Steller sea lions likely fluctuates in response to changes in the environment (Hare *et al.* 1999, Overland *et al.* 1999, Stabeno *et al.* 2001, Benson and Trites 2002, Hunt *et al.* 2002, Shima *et al.* 2002, Trites and Donnelly 2003, Trites *et al.* 2006a), yet what may have been unusual about the decline in sea lions observed through 2000 is the introduction of large-scale commercial fisheries on sea lion prey. While large-scale groundfish fisheries began in the 1960s, their potential for competitive overlap with Steller sea lions (e.g., catches within what would be designated as critical habitat) increased markedly in the 1980s. Overall and localized fisheries removals of prey could have exacerbated natural changes in carrying capacity, possibly in non-linear and unpredictable ways (Goodman *et al.* 2002). Reductions in carrying capacity may have contributed to declines in natality that are believed to have occurred at some rookeries through at least 2002 (Holmes and York 2003, Fay 2004, Winship and Trites 2006, Holmes *et. al. in review*) despite shifts to potentially more favorable environmental conditions that may have occurred in 1989 and 1998 (Hare and Mantua 2000, Bond *et al.* 2003).

B. Factors affecting Steller sea lions

1. Killer whale predation

Killer whale ecology

In the North Pacific Ocean three ecotypes of killer whales have been recognized by their genetics, morphology, acoustics, association patterns, and feeding ecology, including their prey (Bigg *et al.* 1987, Frost *et al.* 1992, Ford *et al.* 1998, Ford *et al.* 2000, Ford and Ellis 1999, Barrett-Lennard 2000, Hoelzel *et al.* 1998, Matkin *et al.* 2006). Differences in the movement patterns among killer whale ecotypes have led, in part, to their names; i.e., “resident”, “transient”, and “offshore.” Specifically, residents have the smallest home range and typically return each year to predictable locations, transients have larger home ranges and have less predictable movements as they transit through local areas quickly, and offshores have the largest home ranges that include areas farther offshore.

Resident killer whales are known to be fish-eaters, in contrast to transients that feed on marine mammals. For offshores, relatively few feeding observations are available, and the limited data indicate these whales appear to prey primarily on fishes, including sharks. However, until the diet of offshores is better understood, the possibility exists that offshores may kill other marine mammals, including Steller sea lions, at least in some regions or seasons. As the currently available information indicates that transient killer whales are the only ecotype that influences the abundance of sea lions, the remaining information on abundance, movements, and diet pertains primarily to transients.

Limited information is available to assess the population structure of transient killer whales within the range of the western DPS, and less information is available for Russian waters. Currently two stocks of transient killer whales have been recognized: (1) the AT1 stock, which occurs from Prince William Sound west through the Kenai Fjords, and (2) the Gulf of Alaska, Aleutian Islands, and Bering Sea (GOA/AI/BS) stock (2004 SAR's). The abundance and stock structure of the AT1 stock have been well documented, and the abundance of this isolated population has declined from 22 whales in 1989 to only 8 whales in 2004 (Matkin *et al.* 1999, Angliss and Outlaw 2005). In contrast, relatively little data exist for the GOA/AI/BS transient stock, particularly for waters west of Kodiak Island.

Surveys conducted by NMFS in the western Gulf of Alaska, Aleutian Islands, and Bering Sea since 2001 have documented that all three ecotypes use these western Alaska waters. Preliminary analyses of photo-id and genetic data from within the coastal survey area from Kenai Fjords to Tanaga Pass provide insights on possible movements of transient killer whales, and also suggest that there may be some finer scale population structuring of transients. Specifically, no movements of individual transients have been documented by photo-id between the central and eastern Aleutians (NMFS unpublished data), with a preliminary dividing line at Samalga Pass (170 degrees West longitude). Preliminary analysis of mtDNA sequences supports this inference, as different haplotypes have been sampled on either side of this possible structural boundary (NMFS unpublished data). However, both genetic and photographic sample sizes are low for the central Aleutians. Similarly, whales from the Gulf of Alaska and those from the Aleutian Islands do not generally appear to overlap in distribution, with a gap in distribution between the Shumagin Islands and Kodiak (NMFS unpublished data). However, there have been a small number of photographic matches of individual whales from the Unimak Pass area in the eastern Aleutians to the northeast side of Kodiak Island (NMFS and North Gulf Oceanic Society, unpublished data). Further samples and analyses are needed to assess the significance of these preliminary findings.

Abundance and diet of killer whales

The abundance of transient killer whales has recently been estimated through (1) line transect surveys, which provide an estimate of the number of whales present, during the sampling period, in the region surveyed and (2) mark-recapture analyses based on whales identified through photo-id, which provide an estimate of the total number of individual killer whales in the region surveyed across the entire survey period. Analysis of line transect survey data collected between 2001 and 2003 indicate that the abundance of transients in the coastal waters between the Kenai Fjords in south-central Alaska and Tanaga Pass in the central Aleutians is approximately 251 whales (95% CI 97-644) during the summer months (Zerbini *et al.* 2006). The density of transients appears to vary regionally, with higher densities from the Shumagin Islands through the eastern Aleutians. However, the minimum count of transients in this area from the combined NMFS and North Gulf Oceanic Society (NGOS) photo-id catalogues is currently 314 whales (Angliss and Outlaw 2005), and preliminary mark-recapture estimates for transients based on photo-identification data are also higher than the line transect estimates (NMFS unpublished data). Current abundance estimates and photo-id cataloguing only refer to coastal waters within approximately 30nm of the Aleutian Islands and adjacent coasts of southwestern Alaska. The abundance and population structure of transient killer whales in offshore areas in the Pacific and Bering Sea are still relatively unknown. Thus, the minimum number of transient killer whales in the U.S. portion of the western DPS is 314, and the estimated abundance will increase when analyses are completed and survey effort increases.

The diet of AT1 transients is relatively well understood. Based on more than 20 years of field observations, these whales are thought to feed primarily on harbor seals and Dall's porpoise (Saulitis *et al.* 2000, Heise *et al.* 2003, Maniscalco *et al.* 2007). The feeding habits of GOA/AI/BS transients are less well known in general and essentially unknown during the period from fall to spring (Maniscalco *et al.* 2007). NMFS is also aware of the tags from 14 Steller sea lions found in the stomach of a stranded killer whale (Heise *et al.* 2003). Heise *et al.* (2003) also report on the examination of the stomach contents of 11 other stranded transient killer whales and found Steller sea lion remains in only 1 other animal (for a total proportion of 2 of 12 with Steller sea lion remains). Observations of feeding by GOA/AI/BS transients have been limited to date, but observed prey include fur seals, gray whales, minke whales, and Steller sea lions (Maniscalco *et al.* 2007, NMFS unpublished data). The analysis by Herman *et al.* (2005) of blubber biopsy samples from eastern North Pacific killer whales indicate that profiles for fatty acids, carbon and nitrogen stable isotopes, and organochlorine contaminants were consistent with previously reported dietary preferences; i.e., fish for resident whales and marine mammals for transients. Regional stable isotope ratios varied considerably, indicating prey preferences may be region-specific, in addition to ecotype-specific. Thus, some groups of GOA/AI/BS transients may specialize on certain prey species, including sea lions (Matkin *et al.* 2005), as AT1 transients specialize on harbor seals and Dall's porpoise. The measured stable isotope values, which reflected diet for the mid-April through mid-July period, for all three killer whale ecotypes were consistent with published dietary preferences based on visual observations. For example, measured stable isotope values for AT1 transients were very similar to modeled stable isotope values, which were based on visual observations (i.e., primarily harbor seals (56%), Dall's porpoises (38%) and harbor porpoises (6%); Herman *et al.* 2005). Measured stable isotope values for GOA, AI, and BS transients indicated the primary prey items were dominated by animals at lower trophic levels than Steller sea lions and harbor seals (Herman *et al.* 2005). Preliminary analysis of blubber samples taken from GOA transients indicates isotope levels similar to local sea lions (NMFS unpublished data[8]).

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[8] There have been two samples taken from GOA transients (e.g., the "kodiak killers") that have been analyzed (collected by Craig Matkin). These samples contained nitrogen isotope values of ~18. Steller sea

Matkin *et al.* (2005) studied killer whales in southeast Alaska and in the Kenai Fjords area from 2002 to 2004. They identified 23 transients, of which 13 were Gulf of Alaska transients (GAT). This was in sharp contrast to their results from southeast Alaska where they identified 100 transients in an area approximately equal in size; indicating about four times the density of transients in southeast Alaska (Matkin *et al.* 2005). Despite the higher density of transients, the southeast Alaska Steller sea lion population has been increasing overall (with some rookeries showing stability), and likely has been increasing for many years. Adult female Steller sea lions seem unaffected by killer whale predation at Chiswell Island although their pups suffered substantial losses over the time period observed by Matkin *et al.* (2005).

In a follow-up paper, Maniscalco *et al.* (2007) describe their observations of GOA transient killer whale[9] predation in the Kenai Fjords area from 2000 to 2005. In many ways this paper represents an evolution of their thinking and an attempt to compare field estimates of predation rates to the model predictions described in Williams *et al.* (2004). Maniscalco *et al.* (2007) observed 9 predation events and an additional 16 which were inferred from remote video monitoring; all were sea lions. However, based on estimates from Williams *et al.* (2004), they would have expected to have observed 103 predation events. They found that unlike killer whales from other regions, GOA transients in the Kenai Fjords rested about 43% of the time. They propose this may be a strategy to conserve energy and suggest that estimates of caloric demand from Williams *et al.* (2004) may be too high.

It has also been proposed that GOA transients may target pups at rookeries during the summer, potentially having a dramatic effect on survival. Maniscalco *et al.* (2007) observed predation of pups at Chiswell Island either by single adult killer whales (one whale in particular) and when adults were teaching calves how to hunt. In British Columbia, a similar study positively correlated transient killer whale group size to pinniped prey size (Ford *et al.* 1998). Preying on pups may be a common strategy for lone, sick, or old transient killer whales (Maniscalco *et al.* 2007, Vos *et al.* 2006, Heise *et al.* 2003).

Hypotheses and modeling attempts

To explore the potential impact of killer whale predation on Steller sea lions, Barrett-Lennard *et al.* (1995) constructed a simulation model. A range of values for transient killer whale abundance, killer whale energy requirements, and killer whale prey selection parameters was explored because of the substantial uncertainty in the current empirical data for these parameters. Steller sea lion parameters in the model include initial population abundance, sex and age distributions, age specific vulnerability to predation, and a density dependent growth rate. The model assumes an unknown 'baseline' level of sea lion mortality due to killer whale predation in a stable sea lion population. Simulations examine changes in sea lion abundance, due to mortality completely

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lions in the GOA have nitrogen isotope values of around 17.5-18.0 themselves (but a low sample size). If killer whales were eating only Steller sea lions, they should have nitrogen isotope values up around 20.0-20.5, which they do not. Limited sample sizes, but the data available do not support the idea that all GOA transients eat sea lions exclusively.

[9] The current estimate for GOA transients is 93 (Angliss and Outlaw 2005).

additive to baseline mortality, from an increase in either killer whale abundance or the percentage of sea lions in the diet of killer whales. Based on parameter values consistent with current empirical data from the range of the western DPS of sea lions, simulation results suggest that: 1) killer whale predation did not initiate the decline of the sea lion population; 2) killer whale predation could cause a continued decline in sea lion numbers in western Alaska based on the estimated abundance of sea lions in 2000; and 3) killer whale predation is not likely to drive the sea lion population to extinction (Barrett-Lennard *et al.* 1995). Further, when the abundance of sea lions declined to 100,000-150,000, the additional mortality (above baseline mortality) from killer whale predation could have been sufficient to drive the decline. Sensitivity analyses indicate changes in sea lion abundance were influenced primarily, and equally, by the number of transient killer whales and the proportion of their diet provided by sea lions, followed by sea lion age-specific vulnerability to predation. When the estimated abundance of sea lions and killer whales in the range of the eastern DPS is used with the parameter combinations that cause a moderate impact on the western DPS of sea lions, the model predicts that killer whale predation would result in a fairly rapid decline of eastern DPS sea lions.

A comparative bioenergetics and demographic model was used by Williams *et al.* (2004) to assess the potential impacts of killer whales on Steller sea lion populations in the Aleutian Islands. Four types of energetic information were measured or estimated: 1) the caloric needs of individual killer whales, taking into account differences in body mass and reproductive status; 2) the caloric value of individual prey including adult sea lions and pups; 3) the digestive efficiency of killer whales, which determined the ability of the animal to utilize energy in prey tissue; and 4) the likely or possible prey preferences of individual killer whales. This information on individual bioenergetics was then compared to population-level estimates of the number of killer whales (NMFS unpublished data), the abundance of sea lions before and during the decline (see Section I.B), and the demographic rates governing the sea lion population (York 1994). The population-wide losses to predation needed to generate the observed changes in the Steller sea lion population, if all losses occurred from predation, were then estimated. Based on their calculations, Williams *et al.* (2004) reported that an average adult killer whale would require 2 - 3 sea lion pups per day or approximately 840 pups per year when feeding exclusively on young Steller sea lions. In comparison, only one third to one half of an adult female sea lion per day (approximately 160 per year) would be needed to satisfy the killer whale's metabolic needs. Williams *et al.* (2004:3379) concluded that "The potential impact of mammal-eating killer whales on …sea lion populations is marked…"Nearly 1,200 Steller sea lions would be eaten per year to meet the caloric requirements of one killer whale pod consisting of 5 individuals, assuming 16% pups and 84% juvenile and adult sea lions consumed, based on the life table for sea lions (York 1994). If the sea lion-eating killer whales in their model prey indiscriminately across all age classes, their modeling results indicated that the annual number of sea lions eaten increased to 39,644 for an estimated population of 170 transient killer whales, approximately three times the highest annual removal rate needed to drive the observed sea lion declines in the 1980s. Based on their calculations, the number of sea lions that would be eaten to satisfy a pod of five killer whales only goes up if pups and juveniles are much more vulnerable to killer whales than adults. If pups are four times more vulnerable, and juveniles three times more vulnerable to killer whale predators than are adults, these same hypothetical sea lion-specialist transient killer whales would consume 64,838 sea lions per year. Despite the conservative estimates of energetic needs and the underestimates of abundance of transient killer whales, the model calculations demonstrated that relatively minor changes in killer whale feeding habits could account for the decline of Steller sea lion populations observed for the Aleutian Islands. "The energy demands of as few as 27 male or 40 female killer whales…" (minimally 23% of transients) "…could account for the estimated 10,885-11,575 Steller sea lions lost per year at the height of the decline. Furthermore, predation losses to a single pod of five killer whales could prevent the present Steller sea lion population from recovering" (Williams *et al.* 2004:3380).

Recently Guénette *et al*. (2007) constructed ecosystem models to: a) evaluate whether killer whale predation might explain the Steller sea lion decline in the central and western Aleutians; and b) understand why sea lions in Southeast Alaska have increased, despite killer whale presence. The results of their simulations suggest that while killer whale predation contributed to sea lion declines in the western and central Aleutians, it was not a primary cause of that decline. However, the authors concluded that killer whale predation could have become a significant source of mortality when sea lion populations were already depressed in both the studied area in the western portion of the range and in Southeast Alaska. Guénette *et al.* (2007:150) summarized that "…large populations of Steller sea lions can withstand predation, but that small populations are vulnerable…".

The studies based on energetics arguments, such as those discussed above, are informative as to the question: Could killer whales cause decline in Steller sea lions if certain assumptions are met? They do not answer the question as to whether killer whales currently are affecting sea lions at a population level or whether they did in the past.

"Sequential Megafaunal Collapse" hypothesis

Examining the potential impact of killer whale predation on Steller sea lions on a broad ecosystem basis, Springer *et al.* (2003) presented a hypothesis that predation was paramount among top-down forces contributing to the sea lion decline. Their "Sequential Megafaunal Collapse" hypothesis is based on the premise that post-World War II industrial whaling depleted large whale populations in the North Pacific, depriving killer whales of an important prey resource. Killer whales thus began feeding more intensively on smaller marine mammals, and this predation resulted in the sequential decline of harbor seals, northern fur seals, Steller sea lions, and northern sea otters in the northern North Pacific Ocean and southern Bering Sea. Due to the acknowledged lack of direct evidence that killer whale predation drove the pinniped declines, Springer *et al.* (2003) explain the declines based on a logical interpretation of known patterns and feasibility analyses of the hypothesized causal process. They suggest current predator prey dynamics are unlikely to provide evidence for the sequential pinniped declines, because prey populations are relatively smaller and comparatively stable, and the abundance of killer whales also may be much reduced.

The Sequential Megafaunal Collapse hypothesis has generated considerable interest and debate concerning the role of killer whale predation in the ecosystem dynamics of the North Pacific. Several studies examined the hypothesis, and indicate alternative interpretations of available data:

Trites et al. (1999, 2004) used ecosystem models to test the effect of whaling in the Bering Sea and the Antarctic Ocean but did not detect a link between the commercial removal of whales and changes in species abundance.

DeMaster *et al.* (2006) concluded that the available data do not support the assumption that some species of large whales were important prey for killer whales, and the available qualitative data indicate that although the biomass of some large whale species likely declined in abundance, those declines were offset by increasing abundances of other large whale species in the 1960s and 1970s. Further, DeMaster *et al.* concluded that statistical tests do not support the assumption that the pinniped declines were sequential.

Mizroch and Rice (2006) show that there was actually a several year lag between the decline in whale catches and the start of the decline of pinnipeds. Because of the extraordinary whale biomass removals in the mid-1960s, any whaling-related prey shifting should have started by 1968, not the mid-1970s as suggested by Springer *et al.* (2003). Mizroch and Rice (2006) also examined data on the contents of killer whale stomachs, and observational records of killer whale interactions with large whales, and refute the Springer *et al.* (2003) assumption that North Pacific killer whales depended on large whales as prey either prior to or concurrent with the whaling era.

Trites *et al.* (2007c) showed that populations of seals, sea lions and sea otters increased in British Columbia following commercial whaling, unlike the declines noted in the Gulf of Alaska and Aleutian Islands. They argue that a more likely explanation than the Springer *et al.* (2003) hypothesis for the seal and sea lion declines and other ecosystem changes in Alaska stems from a major oceanic regime shift that occurred in 1977. Trites *et al.* (2007c:751) concluded that "The hypothesis that commercial whaling caused a sequential megafaunal collapse in the North Pacific Ocean by forcing killer whales to eat progressively smaller species of marine mammals is not supported by what is known about the biology of large whales, the ecology of killer whales, and the patterns of ecosystem change that took place in Alaska, British Columbia, and elsewhere in the world following whaling." They additionally stated that killer whales are unquestionably a significant predator of seals, sea lions and sea otters - but not because of commercial whaling.

Finally, Wade *et al.* (*2007*) argued that available data do not support the Springer *et al.* (2003) hypothesis. A summary of their arguments follows. Large whale biomass in the Bering Sea did not decline as much as suggested by Springer *et al.* (2003), and much of the reduction occurred 50-100 years ago, well before the declines of pinnipeds and sea otters began; thus the need to switch prey starting in the 1970s is doubtful. With the sole exception that the sea otter decline followed the decline of pinnipeds, the reported declines were not in fact sequential. Given this, it is unlikely that a sequential megafaunal collapse from whales to sea otters occurred. The spatial and temporal patterns of pinniped and sea otter population trends are more complex than Springer *et al.* (2003) suggest, and often inconsistent with their hypothesis. Populations remained stable or increased in many areas, despite extensive historical whaling and high killer whale abundance. Furthermore, observed killer whale predation has largely involved pinnipeds and small cetaceans; there is little evidence that large whales were ever a major prey item in high latitudes.

Kenney (2007:515), in a review of the collection of papers in Estes *et al*. (2006) summarized that "…recently published papers…have pointed out several weaknesses in the [Sequential Megafaunal Collapse] hypothesis, such as the finding that killer whales are not important predators of great whales, with the exception of gray whale calves and minke whales; that there is a mismatch in the timing of the major pulse of whaling in the northern North Pacific and the observed declines in pinnipeds and sea otters; and that there have not been similar declines in Dall's porpoise and other small cetaceans that are known to be killer whale prey."

Direct Impact of Killer Whales on Steller Sea Lion

Historical accounts of killer whale predation on marine mammals in the northern North Pacific, though somewhat limited in number, are roughly consistent with recent observations that killer whales prey on a variety of species of marine mammals, particularly pinnipeds and small cetaceans (Mizroch and Rice 2006, Wade *et al* 2007), with specialization likely in GOA transient killer whales (Maniscalco *et al.* 2007). The estimated abundance of mammal-eating killer whales throughout most of the range of the western DPS of Steller sea lion (Kenai Peninsula to Tanaga Pass in the central Aleutian Islands) is 251 (95% C.I. 97-644) (Zerbini *et al.* 2006) for the years 2001-03. Mammal-eating killer whales were found to be more abundant from the Shumagin Islands to the west (226) than they were east of the Shumagins through Kenai Peninsula (27). Mammal-eating killer whales were found to be at their highest density in summer in the eastern Aleutian Islands, stretching from Umnak Island to the west to the Shumagin Islands to the east. Angliss and Outlaw (2005) estimate the number of GOA transient killer whales at approximately 93 individuals. Maniscalco *et al.* (2007) estimated a minimum of 19 and maximum of 39 GOA transients in their Kenai Fjords area study. Matkin *et al.* (2005) found nearly 4 times that number of transient killer whales (100 transients) in a similar area in southeast Alaska.

As presented in more detail above, Williams *et al.* (2004) concluded that "[t]he energy demands of as few as 27 male or 40 female killer whales…" (minimally 23% of transients; minimally especially because the estimates about killer whale population abundance used by Williams *et al.* (2004) were lower than those based on more recent data) "…could account for the estimated 10,885-11,575 Steller sea lions lost per year at the height of the decline. Furthermore, predation losses to a single pod of five killer whales could prevent the present Steller sea lion population from recovering" (Williams *et al.* 2004:3380). A critical assumption of these modeling efforts was that the killer whales would have preyed exclusively on sea lions. Subsequent studies have tested this assumption, and do not suggest that mammal-eating killer whales prey exclusively on Steller sea lions. Wade *et al.* (2007) reviewed observations of killer whale predation on marine mammals since 1950. The percent of kills by mammal-eating killer whales that were Steller sea lions was ~6% in the BSAI, and ~22% in the GOA, with most of those observations from the summer (Table 1 in Wade *et al.* 2007). Since 2001, observations by NMML/NOAA of sea lion kills in summer were 11% in the BSAI. Matkin *et al.* (*in press*) report 4% in spring/summer in the BSAI. Estimates for the BSAI therefore range from 4% to 11%. Analysis of contaminant concentrations and fatty acids confirms that, as suspected, mammal-eating killer whales have a chemical signal in the blubber that is entirely consistent with an exclusive diet on marine mammals (Herman *et al.* 2005, Krahn *et al.2007*). Analysis of stable isotope concentrations in mammal-eating killer whale skin from the BSAI results in values that are in close agreement with values predicted from observations of predation of ~4 to 11% Steller sea lions (Krahn *et al. in review,* NMFS unpublished data), suggesting the visual observations do provide an accurate assessment of killer whale predation during that time of year.

It is important to consider the effects of killer whale predation in terms of sea lion natural mortality rates within the western DPS. For a stable population of Steller sea lions in the western DPS in Alaska, the average annual natural mortality rate is about 20% – this reflects about 10,000 animals dying each year due to trampling, senescence, disease, killer whales, etc. This important fact is often overlooked, and killer whale predation is added on top of this background natural mortality rate (e.g., Barrett-Lennard *et al.* 1995). Maniscalco *et al.* (2007), estimated the average predation by killer whales in Kenai Fjords accounted for 3% (their field metabolic estimates) to 7% (based on Williams *et al.* 2004 estimates) of the local summer seasonal population of sea lions each year. Maniscalco *et al.* (2007), conclude that although these killer whales were observed to eat sea lions exclusively (in their limited study area and time), killer whale predation accounted for only about a quarter of the annual natural mortality and is probably not hindering recovery.

If it is assumed that Steller sea lions (and other pinnipeds) have always been prey of mammal-eating killer whales, then killer whale predation has been, and likely still is, a component of that natural mortality. The current percentages of killer whale predation calculated above are lower than the 20% natural mortality rate. This indicates that this level of predation could have been a component of natural mortality, and therefore not responsible for the decline of the western stock of Steller sea lions. Additionally, if the population of killer whales is assumed to have been the same size historically as it is now, that level of predation would represent a smaller fraction of the sea lion population before its decline, and thus a smaller proportion of natural mortality.

Finally, life-history changes in the western stock of Steller sea lions through time argue against the hypothesis that killer whale predation alone was responsible for the decline. Density-dependent responses seen in the western Steller sea lion population included lower growth and pregnancy rates in the 1980s than the 1970s (Calkins *et al.* 1998, Pitcher *et al.* 1998). This indicates carrying capacity for sea lions likely declined over this period. This apparently continued through the 1990s as evidenced by a possible decline in natality (Holmes and York 2003). These shifts in life history parameters during the declines argue against killer whale predation as a main cause of the decline because there is no direct reason why increased killer whale predation would lead to a decline in natality. In addition, the eastern DPS has increased at approximately 3% per year for at least 20 years while co-existing with a similar population (though higher density) of transient killer whales in an environment historically exposed to commercial whaling and environmental change.

Data gaps

Obtaining the following information would substantially reduce the uncertainty about the potential impact of killer whale predation on Steller sea lions: (1) Diet preferences of GOA/AI/BS transients on a seasonal, multi-annual, and geographical basis, as well as data on prey specializations by individual groups of whales; (2) Improved estimates of the field metabolic rate for transients, the variance in the field metabolic rate among individuals whales, and the factors that may influence that variance; e.g., age, sex, season, reproductive status; (3) Trends in the abundance and distribution of transients in offshore waters of the Gulf of Alaska, Pacific Ocean and Bering Sea, as well as around the central and western Aleutians; (4) Population structure of transients in the North Pacific: relating the Bering Sea, offshore Pacific and offshore Gulf of Alaska to coastal areas of Aleutians and Gulf of Alaska used by Steller sea lions; and (5) Data on the patterns of occupancy (e.g., short and long term, seasonal) of transients within sea lion haul-out and foraging areas, and data on movement patterns to and from these areas. Further, the draft research plan for assessing the role of transient killer whales in North Pacific Ecosystems prepared by the Marine Mammal Commission identifies additional information that should be obtained.

2. Shark predation

Sharks represent another potential predator that may attack Steller sea lions. Although white shark predation on North Pacific pinnipeds has been well documented (LeBoeuf *et al.* 1982, Ainley *et al.* 1985, Long *et al.* 1996), these sharks occur rarely, if at all, in the range of the western Steller sea lion population. Although salmon shark populations have increased since 1990, they are considered piscivorous and have not been reported to prey on Steller sea lions. Another species of large shark, the Pacific sleeper shark (*Somniosus pacificus*), is common in the Gulf of Alaska, Aleutian Islands, and Bering Sea (Orlov 1999). Current indices to sleeper shark relative abundance are based on a recent analysis of sleeper shark bycatch from sablefish longline surveys conducted on the upper continental slope and deepwater gullies of the continental shelf in the eastern Bering Sea, Aleutian Islands, and Gulf of Alaska from 1979-2000 (Courtney and Sigler unpublished data, also see Mueter and Norcross 2002). This analysis indicates sleeper sharks are substantially (about 10x) more abundant in the Gulf of Alaska than in the BSAI region. Further, a significant increase in the relative abundance of sleeper sharks occurred during 1989-2000 in the central Gulf of Alaska, driven largely by the increase of sharks in Shelikof Trough during 1992 and 1993. Most Pacific sleeper shark stomachs that have been examined contained remains of fish and invertebrates (Yang and Page 1998, Orlov 1999), but the remains of harbor seals and porpoises have also been reported (Bright 1959). A recent analysis of sleeper shark stomachs (n=198) collected in the GOA near sea lion rookeries when pups may be most vulnerable to predation (i.e., first water entrance and weaning) found that teleost fishes and cephalopods were the dominant prey (Sigler *et al.* 2006). Tissues of marine mammals were found in 15% of the shark stomachs, but no sea lion tissue was detected. Overall, the study concluded that sea lions are unlikely prey of sleeper sharks, harbor seals are infrequent prey and may be consumed alive, and that cetaceans are a frequent diet item for larger sleeper sharks and commonly scavenged. A companion study documented that shark and sea lion home ranges overlapped (Hulbert *et al.* 2006), and thus the results of these two studies, and others, indicate no scientific evidence presently exists to suggest that sleeper sharks actively prey on Steller sea lions.

Data Gaps

The available evidence does not indicate sharks are a threat to sea lion recovery. However, the diet of sleeper sharks should continue to be estimated through stomach remains, fatty acids, and stable isotopes.

3. Commercial harvest

Currently, no commercial harvest for Steller sea lions exists in the United States, but sea lions were commercially harvested prior to 1973. A total of 616 adult males and 45,178 pups of both sexes were harvested in the eastern Aleutian Islands and Gulf of Alaska between 1959 and 1972 (Thorsteinson and Lensink 1962, Havens 1965, Merrick *et al.* 1987). The pup harvests, which sometimes reached 50% of the total pup production from a rookery, could have depressed recruitment in the short term and may partially explain the declines at some sites through the mid-1970s. However, these harvests do not explain why numbers declined in regions where no harvest occurred, or why in some regions declines occurred approximately 20 years after harvests ceased (Merrick *et al.* 1987). A comparative analysis of the ecology and population status of four species of pinnipeds in similar environments (Steller sea lions in the Gulf of Alaska, Cape fur seals in the Benguela Current, harp seals in the Barents Sea, and California sea lions in the California Current) indicates that directed commercial harvest was not a major factor in the Gulf of Alaska Steller sea lion decline (Shima *et al.* 2000).

Steller sea lions are hunted in Hokkaido, Japan to reduce interaction with local fisheries. An average of 631 animals were killed per year during 1958-1993 (Takahashi and Wada 1998). The animals killed had probably migrated southward from the Kuril Islands. Demographic modeling shows that kills were sufficient to deplete the Kuril population, especially in combination with incidental catches in fisheries (Takahashi and Wada 1998). More current information on the level of kill since 1993 is not available.

Data Gaps

There are no data gaps related to commercial harvest of Steller sea lions within the United States, as such harvests are no longer conducted. As noted above, recent and current data are not available about the level of harvests occurring in Japan.

4. Subsistence harvest

Both the ESA and the MMPA contain provisions that allow coastal Alaska Natives to harvest endangered, threatened, or depleted species for subsistence purposes. Prior to 1992, no comprehensive program estimated the level of subsistence harvest of sea lions in Alaska. However, available information indicates that sea lions were being harvested at several villages on the Bering Sea, in the Aleutian Islands, and on the Gulf of Alaska (Haynes and Mishler 1991). During 1992-2004, harvest data were collected through systematic retrospective interviews with hunters in at least 60 coastal communities throughout the range of sea lions in Alaska (Wolfe *et al.* 2005). Results show the annual take (i.e., harvest plus struck and loss) decreasing substantially from about 550 sea lions in 1992 to about 200 in 1996 followed by annual takes between 165 and 215 from 1997 to 2004 (Table III-1).

In the early 1990s, juveniles were harvested at least twice as much as adults, yet that ratio declined beginning in 1996, and during 2000 to 2004 the ratio of juveniles to adults in the harvest ranged from 0.5 to 1.0. The ratio of males to females harvested in 2004 was 1.8, below the 5-year average of 4.1 during the previous five years. In 2004, 24 adult females were harvested, representing about 20% of the total harvest of known sex and age. During 1992-1995, the greatest numbers of sea lions harvested were in the Pribilof Islands, whereas during 1996-2004 the harvest was greatest in the Aleutian Islands. The surveys that produced these estimates covered all Alaskan communities that regularly hunt Steller sea lions, but a few additional animals are taken occasionally at other locations (Coffing *et al.* 1998, ADF&G unpublished data).

In 1998, the Tribal Government of St. Paul's Ecosystem Conservation Office implemented a real-time data collection program to estimate the take of sea lions, due to concerns by hunters and the local community in the uncertainty of harvest results based on retrospective surveys. Results of the real-time harvest monitoring indicated a sea lion take of about 25-35 per year from 1998-2002, followed by a reduced take of 18 sea lions in both 2003 and 2004 (Zavadil *et al.* 2005). In 2005, 22 Steller sea lions were taken on St. Paul Island (Zavadil *et al*. 2006). The Tribal government also implemented a new subsistence harvest management scheme that likely may have resulted in fewer animals taken. Factors that may be responsible for this decreased take include fewer hunters, fewer animals to hunt in the communities' hunting areas, and voluntary restraint from hunting because of perceived problems with the sea lion population (Wolfe and Hutchinson-Scarbrough 1999).

Information on the harvest of Steller sea lions in Russia is fragmentary. In 1932 and 1933, newborn pups were harvested on Iony Island in the Sea of Okhotsk (1,198 and 805 respectively), and in 1935 about 30 pups were taken on the Shipunsky Cape (Kamchatka) rookery (Nikulin 1937). In 1974, an experimental harvest was conducted on Brat Chirpoev rookery in the Kuril Islands that took 296 pups (Perlov 1975). During the period when the government of the Soviet Union conducted commercial sealing (1960-1990), sea lions were not a target species, but they were taken occasionally with annual harvests ranging from 37 to 650 animals (Perlov 1996). During the 1950s to 1980s, a subsistence harvest was conducted on the Commander Islands and Kamchatka that usually took fewer than 100 animals a year, but this harvest has stopped completely in the late 1980s (Burkanov personal communication). Some sea lions are taken in Chukotka by native hunters, but the number killed is unknown.

Data Gaps

Available evidence does not indicate that, by itself, the current level of subsistence harvests of Steller sea lions is sufficient to significantly reduce the expected recovery rate of Steller sea lions DPS (Atkinson *et al*. in press). However, much of the available information on subsistence harvests of Steller sea lions is based on retrospective surveys of households in which respondents are asked to recall information about their use of marine mammals over the previous year. Takes of surveyed hunters are expanded to those of unsurveyed hunters within the same stratum (Wolfe *et al.* 2003). As noted above, concerns about the level of uncertainty associated with such retrospective surveys have been raised within some portions of the Alaska Native sea lion-hunting community. Current efforts to monitor subsistence harvests should be continued and efforts made to identify, evaluate, and as possible, to reduce sources of uncertainty in subsistence harvest data.

5. Incidental take by fisheries

Many Steller sea lions have been killed incidental to commercial fishing operations in the Bering Sea and North Pacific Ocean. The total estimated incidental catch of Steller sea lions during 1966-1988 in foreign and joint-venture trawl fisheries operating off Alaska was over 20,000 animals (Perez and Loughlin 1991). A particularly high level of take occurred in the 1982 Shelikof Strait walleye pollock joint venture fishery when U.S. trawlers killed an estimated 958 to 1,436 sea lions (Loughlin and Nelson 1986). The estimated take in this fishery declined to fewer than 400 animals per season in 1983 and 1984, probably due to changes in fishing techniques and in the area and times fished. Most of the animals taken were sexually mature females. Fewer than 100 per year were estimated to have been taken during 1985-1987. The level of incidental mortality has continued to decline. The minimum estimated mortality rate incidental to commercial fisheries in 2002 was 29.5 sea lions per year, based on observer data (24.1) and self-reported fisheries information (5.2) or stranded data (0.2) where observer data were not available (Angliss and Outlaw 2005).

During October-December 2002, observers recorded the incidental take of sea lions during a herring trawl fishery in the western Bering Sea. Preliminary estimates of the total number of sea lions caught were 35-60, with 32-50 killed (Burkanov and Trukhin unpublished). The genetic analysis of skin samples from sea lions caught in this trawl fishery will provide insight on which regions the sea lions may be from (i.e., Aleutian, Commander, and Kuril Islands, and Kamchatka). The majority, if not all, of these sea lions were subadult males.

Amendments to the MMPA in 1988 and 1994 required observer programs to monitor marine mammal incidental take in some domestic fisheries. Observers monitored the Prince William Sound drift gillnet fishery in 1990 and 1991 and estimated a mean annual kill of 14.5 Steller sea lions (Wynne *et al.* 1992). Hill and DeMaster (1999) provide observer-based estimates of average annual Steller sea lion incidental mortality for fisheries operating in the range of the western DPS between 1993 and 1997 as follows: 6.8 animals in the Bering Sea groundfish trawl fishery; 1.2 animals in the Gulf of Alaska groundfish trawl fishery; 0.2 animals in the Bering Sea groundfish longline fishery; and 1.0 animals in the Gulf of Alaska groundfish longline fishery. These numbers are minimum estimates of the incidental kill and serious injury in fisheries, because not all fisheries that might take sea lions are covered by observer programs.

Nikulin and Burkanov (2000) documented marine mammal bycatch in Japanese salmon driftnet fishing in the Russian exclusive economic zone of the southwestern Bering Sea. Catch of only one Steller sea lion was observed during 1992-1999, and it was released alive.

Data Gaps

Angliss and Outlaw (2006) noted that the current estimate of annual mortality of Steller sea lions from the western DPS incidental to U.S. commercial fisheries is a minimum estimate because observers have not been assigned to several fisheries that are known to interact with this population. Thus, mortality estimates are incomplete or, for some fisheries, based on data over a decade old. Quantitative information on sea lion incidental catch in other fisheries that occur in

Russian and Japanese waters is not available, but it is possible that some animals have been killed in trawl fisheries for herring and pollock.

6. Illegal shooting

In some areas Steller sea lions are known to have been shot deliberately by fishermen (and perhaps other people), but it is unclear how such mortality may affect the population because the overall magnitude of the take is unknown (Alverson 1992). One of the few estimates of shooting mortality was reported by Matkin and Fay (1980), who calculated that 305 Steller sea lions were shot and killed while interfering with fishing operations in the spring 1978 Copper River Delta salmon gillnet fishery. Data from a 1988-1989 study of the Copper River salmon gillnet fishery indicated that the level of directed kill of sea lions was significantly less than during 1978 (Wynne 1990). During the 1960s, sea lions were sometimes killed and used as bait by crab fishermen (Alverson 1992). Such killing may have had a significant effect in local regions and might have caused animals to move away from certain rookeries and haulout sites (Loughlin and Nelson 1986, Merrick *et al.* 1987, NRC 2003). In 1990, a regulation was implemented to prohibit fishermen from discharging firearms near Steller sea lions, but nonetheless some shooting, resulting in an unknown level of mortality, likely occurs (NMFS 2001, Loughlin and York 2000, NRC 2003).

Simulation modeling suggests that a combination of commercial harvests, subsistence harvests, and intentional and incidental take in fisheries may explain a large portion of the western Steller sea lion population decline that occurred through 1980 (Trites and Larkin 1992). However, the annual decline since 1990 has been much greater than can be accounted for by such direct causes (Loughlin and York 2000).

Data gaps

The primary data gaps are estimates of the magnitude of illegal shooting and incidental take in fisheries for which observer data are not available. Knowledge of the age and gender of such sea lion mortality, and the seasons in which it occurs, would help reduce the uncertainty associated with these factors. Available data suggests that the possible impact of this mortality on the population is equivocal. At a lower priority is the documentation of native subsistence harvest at sites other than the Pribilof Islands. The harvest management scheme implemented on St. Paul and St. George islands, or one like it, should be explored by other Alaska Native Organizations to provide documentation of the level of mortality and the age and sex of collected sea lions. This, along with an improved reporting system, will enhance the information available to better assess the sources of sea lion mortality state-wide.

7. Entanglement in Marine Debris

Steller sea lions may become entangled in lost and discarded fishing gear and other marine debris, including items such as closed packing bands and net material (Calkins 1985). A study conducted in the Aleutian Islands during June-July 1985 to investigate the rate of entanglement found that a very low percentage (approximately 0.07%) of observed sea lions were entangled in net or twine; none were entangled in packing bands (Loughlin *et al.* 1986). A follow-up study was conducted during November 1986 to assess the possibility that sea lion pups were becoming entangled in debris. Researchers saw no entangled pups and only one entangled juvenile out of a total of 3,847 sea lions examined (Loughlin *et al.* 1986). However, these observational studies cannot fully evaluate the frequency of entanglement because entangled animals may die at sea and thus not be observed on land.

Data Gaps

The level of mortality from entanglement is unknown. More current estimates of the frequency and extent of entanglement are needed, including the probability of mortality for entangled sea lions. A more accurate estimate of this mortality is required to determine the potential benefits of cleaning beaches of debris and by intensifying enforcement of regulations governing debris discard.

8. Disease and Parasitism

Disease may include any pathogen of viral, bacterial, protozoan, or fungal origins, which are either known to Steller sea lions and related species or are unknown to Steller sea lions but zoonotic. Whereas disease occurs naturally in all animal populations, there are two ways in which disease could have negatively impacted Steller sea lion populations. Firstly, the occurrence of a contagious pathogen to a naive population could lead to a mass or unusual mortality event. Such events have been documented in other pinniped populations and selected instances are mentioned below. Secondly, several pathogens are known to result in reproductive loss, either through spontaneous abortions, embryonic or fetal resorption, or through rendering the female infertile. Both of these disease impacts have the ability to result in population level decreases, yet neither has been observed or measured in Steller sea lions. In addition, one of the reasons that diseases, or at least the diseases that have been examined, do not appear to be a major factor in the population decline is that the same diseases occurred at about the same rate in both the western and eastern DPSs. Nevertheless, with the current low abundance of sea lions, the potential for disease to impede recovery should not be ignored.

Disease is a natural process, and the mechanism by which many animals die. The important question for sea lions is whether disease agents currently have the potential to reduce population growth through increased mortality or decreased reproductive output. Examples from disease outbreaks in other pinniped populations indicate that this potential clearly exists. For example, an epidemic of phocine distemper virus was responsible for the deaths of thousands of seals in the northwest Atlantic during the 1980s (Heide-Jørgensen *et al.* 1992). The New Zealand Department of Conservation summarized that the New Zealand sea lion has been affected by three epidemics over the past seven years. In the Auckland Islands, over the 1997-98 summer breeding season, more than half of the season's New Zealand sea lion pups died as a result of a mysterious ailment, possibly a bacterial infection and hundreds of adults were also estimated to have died.

Investigators have not seen large numbers of dying or dead Steller sea lions, although sick individuals are found on rare occasions. For example, two sick Steller sea lions that stranded in northern California and were brought in for treatment later died of acute bronchopneumonia of unknown etiology (Morgan *et al.* 1996).

There are two basic methods for determining the prevalence of disease in animal populations: either by necropsy of dead animals and analysis of their tissues to identify pathologies, or by serologic screening of blood samples taken from live or dead animals. Serologic screening requires that a specific disease be identified and antigenic agents developed for testing to detect the presence of an antibody to the disease in the animal's blood. In most cases antigen agents have not been developed specific to Steller sea lions but have been adapted from those developed for other species. This may introduce some inaccuracy in test results, but such tests probably are adequate to indicate presence or absence of a particular disease at some time in the subject's life. The presence of antibodies detected in a serological test shows that an animal has been exposed to a disease agent, but does not indicate that it is unhealthy, or that the pathogen is currently in the animal.

Analysis of components of blood can give insight into the general health of animals. Bishop and Morado (1995) examined blood characteristics of Steller sea lions pups captured live on rookeries in Southeast Alaska and the Gulf of Alaska. White blood cell counts suggested mild physiological stress responses that were perhaps due to capture and handling. Red blood cell counts were suggestive of anemia, especially in animals sampled in the Gulf of Alaska. Zenteno-Savin *et al.* (1997) found higher levels of haptoglobin in Steller sea lion blood in the Aleutian Islands than in Southeast Alaska sea lions. In other animals, elevated haptoglobin levels are known to be associated with stress (e.g., trauma, infection), but no explanation was suggested for the results in Steller sea lions.

As noted in Section I.G, decreased reproduction may occur at a relatively high rate in sea lions. Several of the disease agents that sea lions have been exposed to are known to affect reproduction in other species. Alaskan Steller sea lions have been exposed to two types of bacteria, *Leptospira* and *Chlamydia* (Calkins and Goodwin 1988, Sheffield and Zarnke 1997, Burek *et al.* 2003), and one virus, the San Miguel sea lion virus, that have caused reproductive problems in other species. Specifically, San Miguel Sea Lion Virus and *Leptospira* have been associated with reproductive failures or neonatal deaths in California sea lions and northern fur seals (Smith *et al.* 1974, Gilmartin *et al.* 1976). Virtually nothing is known about the possible effects of *Chlamydia* on pinnipeds, but in other animals *Chlamydia* is known to cause abortion, stillbirths, and production of weak young (Shewen 1980).

In addition to the three disease agents listed above, other disease agents that Steller sea lions have been tested for include phocid herpesvirus, phocine and canine distemper viruses (Barlough *et al.* 1987, Zarnke *et al.* 1997, Sheffield and Zarnke 1997), morbilliviruses, canine parvovirus, *Brucella, Toxoplasma,* and influenza A (Sheffield and Zarnke 1997). There is no convincing evidence for significant exposure to influenza A, morbilliviruses, *Brucella*, canine parvovirus, or *Leptospira* (Burek *et al.* 2003, 2005). Examination and necropsy of dead Steller sea lions has shown some occurrences of hepatitis, *Chlamydia,* myocarditis, endometritis, tumors, and pneumonia (Gerber *et al.* 1993).

Burek *et al.* (2003, 2005) concluded that available serologic evidence does not support the possibility that a disease epidemic occurred during the sea lion decline of the late 1970s and 1980s. They noted, however, that due to sampling limitations the possibility could not be excluded completely. Further, although sea lions have been exposed to several endemic disease agents that could potentially impede recovery, the only available data are the prevalence of antibodies to the disease agents, and the potential for those agents to cause disease among Steller sea lions has not been documented (Burek *et al.* 2005).

Parasites of Steller sea lions include intestinal cestodes; trematodes in the intestine and bile duct of the liver; nematodes in the stomach, intestine, and lungs; acanthocephalans in the intestine; acarian mites in the nasopharynx and lungs; and an anopluran skin louse (Dailey and Hill 1970, Dailey and Brownell 1972, Fay and Furman 1982, Shults 1986, Gerber *et al.* 1993). The potential for parasitism to have a population level affect on sea lions is largely unknown. Whereas parasites may have little impact on otherwise healthy animals, effects could become significant if combined with other stresses (Haebler and Moeller 1993). Available information does not suggest that the sea lion decline was caused by parasitic infections, although there has not been adequate research to assess the relative nature and magnitude of parasitism in sea lion populations. Investigations of parasites require necropsy of carcasses that only occurs on a sporadic basis on beach cast animals.

Data Gaps

To assess the potential for disease and parasitism to impede the recovery of sea lions, systematic protocols need to be implemented to collect and analyze samples obtained from live and dead animals. Further, remaining available samples should be analyzed to establish solid baselines for comparison with future studies.

9. Toxic Substances

Toxic substances can impact animals in two major ways. First, the acute toxicity caused by a major point source of a pollutant (such as an oil spill or hazardous waste) can lead to acute mortality or moribund animals with a variety of neurological, digestive and reproductive problems. Second, toxic substances can impair animal populations through complex biochemical pathways that suppress immune functions and disrupt the endocrine balance of the body, causing poor growth, development, reproduction and reduced fitness. Toxic substances come in numerous forms, with the most-recognized being the organochlorines (OCs; mainly PCBs and DDTs), heavy metals and polycyclic aromatic hydrocarbons (PAHs). There are also a number of “emerging” contaminants, e.g., flame retardant polybrominated diphenyl ethers (PBDEs), which could also be impacting marine mammals (de Wit *et al.* 2002).

Aside from the Exxon Valdez Oil Spill in 1989, which occurred well after the Steller sea lion decline was underway, no other events have been recorded that support the possibility of acute toxicity leading to substantial mortality of Steller sea lions (Calkins *et al.* 1994). However, results from several studies, both published and still being conducted, do not permit the complete rejection of toxic substances as a factor that may currently impact sea lion vital rates. These studies have been conducted on both Steller sea lions and other pinniped species and are briefly reviewed below by toxic category.

Sea lions exposed to oil spills may become contaminated with PAHs through inhalation, dermal contact and absorption, direct ingestion, or by ingestion of contaminated prey (Albers and Loughlin 2003). After the Exxon Valdez oil spill, Calkins *et al.* (1994) recovered 12 Steller sea lion carcasses from the beaches of Prince William Sound and collected 16 additional Steller sea lions from haul out sites in the vicinity of Prince William Sound, the Kenai coast, and the Barren Islands. The highest levels of PAHs were in animals found dead following the oil spill in PWS. Furthermore, sea lion bile samples collected seven months after the spill had levels of PAH metabolites consistent with exposure to PAH compounds (Calkins *et al.* 1994). However, histological examinations found no lesions that could be attributed to hydrocarbon contamination and, hence, no evidence of damage due to oil toxicity (Calkins *et al.* 1994).

OC contaminant exposure in marine mammals and other wildlife has been associated with reproductive failures (Helle *et al.* 1976, Reijnders 1986), population declines (Martineau *et al.* 1987), carcinomas (Martineau *et al.* 1999, Ylitalo *et al.* 2005), and immune suppression (de Swart *et al.* 1994, Ross *et al.* 1996; Beckmen *et al.* 2003). No toxicological studies have been published on Steller sea lions to determine possible effects of OC contaminants. However, OCs that cause health impacts in other species have been measured in subsets of Steller sea lion populations from Japan, the Russian far east, Aleutian Islands, Pribilof Islands, Gulf of Alaska and Southeast Alaska (Lee *et al.* 1996, Varanasi *et al.* 1992, Hoshino *et al.* 2006, Hong *et al.* 2005, Myers 2005). PCB congener levels in 4 out of 10 sea lions near Hokkaido and 2 out of 12 near Olyutorsky Bay (Kamchatka) (Hoshino *et al.* 2006) exceeded the levels in ribbon seals with decreased circulating thyroid hormones (Chiba *et al.* 2001). However, no sea lions in a study by Hoshino *et al.* (2006) exceeded the levels of PCBs in harbor seals that experienced immune suppression (de Swart *et al.* 1996). Furthermore, Steller sea lions may not have the same sensitivity to toxic PCBs as found for ribbon seals. Thus, the ultimate effect of PCB toxicity on sea lion fitness is unknown. Heintz *et al.* (2006) investigated OCs in a primary Steller sea lion prey item (pollock) through much of the range of Steller sea lions in Alaska. They found higher concentrations of OCs in pollock in southeast Alaska, within the range of the eastern DPS, but also found OCs to be ubiquitous throughout their sampling area. Given that the eastern DPS has been increasing (e.g., recovering) while consuming prey with higher OC concentrations, OCs may not be the primary factor for recent declines in population and natality observed in the western DPS.

Most of these studies measured OCs in the blubber or blood of Steller sea lions, although Krahn *et al.* (2001) used feces as the medium to measure OCs. Overall, the studies suggest a decline in OC concentrations over time, which is consistent with that reported for other wildlife species. OC concentrations have been significantly different among Steller sea lions in some regions (Myers and Atkinson 2005, Hoshino *et al.* 2006), although not consistently so throughout all studies (Hong *et al.* 2005). Typically a few individuals with particularly high concentrations will skew the mean results, giving high standard deviations that result in non-significant or inconclusive statistical results. The studies that measured more than one OC generally found that the PCB congeners and DDT metabolites were the most prevalent OCs measured in Steller sea lions. No studies have been published that report any PBDE congeners, however this is likely to change in coming years.

Studies of the effects of known OCs have not been conducted on marine mammals in the U.S. However, studies from Europe have provided threshold levels of OCs above which immunosuppression or reproductive problems can be expected (de Swart *et al.* 1994, Ross *et al.* 1996). Although these studies were conducted on harbor seals, the thresholds are often used for related species such as Steller sea lions. Several individual California sea lions (*Zalophus* *californianus*) have been sampled that had high concentrations of DDTs and PDBs (Kannan *et al.* 2004), which were linked to physiological impairments (Debier *et al.* 2005) and cancer associated mortality (Ylitalo *et al.* 2005). Likewise, a threshold for reproductive failures (i.e., spontaneous abortions) has been estimated, based on a mass toxicity event of California sea lions from the 1970s (DeLong *et al.* 1973, Gilmartin *et al.* 1976). No recent samples from Steller sea lions have approached this threshold, indicating a mass mortality from an acute toxicity event was not likely the cause of the sea lion decline.

Although publications on the effects of emerging contaminants are few (Barron *et al.* 2003), one class of "emerging" environmental contaminants, the PBDEs, are quickly gaining the attention of regulatory agencies (de Wit *et al.* 2002). These compounds are added to plastics, textiles, clothing, electronic circuit boards and other materials as flame retardants. PBDEs are known to enter the environment through urban runoff and sewage outfalls and have been shown to bioaccumulate in marine animals (de Wit *et al.* 2002). A number of studies have shown that some PBDE congeners may induce toxicological effects in laboratory animals, including immune dysfunction, liver toxicity, thyroid disruption and possibly cancer (de Wit *et al.* 2002, MacDonald 2002). Some data are available on the levels of PBDEs in marine mammals from North America (Ikonomou *et al.* 2002, She *et al.* 2002, Stapleton *et al.* 2006). Another study reported that PBDEs have increased 10- to 100-fold in blubber of harbor seals collected near San Francisco Bay over the last decade (She *et al.* 2002). Because these compounds continue to be used in the U.S. and other regions of the world, the levels measured in marine environmental samples are expected to increase. Ikonomou *et al.* (2002) reported that PBDEs may become the most prevalent persistent organic pollutant in arctic ringed seals in the next 50 years. However, few studies have looked at PBDE exposure and associated health effects in marine mammals. Thus, the potential for Steller sea lion exposure to unknown contaminants, such as PBDEs, many of which are increasing, is a significant gap in our understanding of impacts of pollutants on Steller sea lions (Barron *et al.* 2003).

Heavy metals are also contaminants of concern. Heavy metal concentrations measured in Steller sea lion livers were generally much lower than in northern fur seals (Noda *et al.* 1995). For example, mercury levels in the hair of young Steller sea lions from both the western and eastern DPSs were lower than for northern fur seals (Beckmen *et al.* 2002), yet concerns remained about possible effects on fetal development and interactive effects with other contaminants. Castellini (1999) found that zinc, copper, and metallothionien (a chelating compound) levels were comparable between sea lion pups sampled from both the western and eastern DPSs, and were lower than for captive sea lions. Kim *et al.* (1996) reported on the accumulation of butyltin in the liver of Steller sea lions from Alaska and Japan and found much lower levels in the Alaska samples than in those from Japan. These authors also suggested that butyltin degrades rapidly in sea lions and does not bioaccumulate. Although these studies are not comprehensive, they indicate that heavy metals were not likely a significant factor in the decline of the Steller sea lions.

In summary, contaminant risks are largely unknown in Steller sea lions and are little understood in pinnipeds in general (Barron *et al.* 2003). Definitive studies that have causally linked contaminant exposures and adverse effects in pinnipeds have been limited to laboratory studies with PCBs and Hg in dietary studies with captive seals. Field studies with pinnipeds have been confounded with other factors and cannot be unambiguously linked to contaminant caused impacts. The sensitivity of pinnipeds to contaminants relative to the sensitivity of other species is largely unknown. Thus, adverse effect levels of contaminants in Steller sea lions must be inferred from studies in other species (Barron *et al.* 2003). As a result, the primary data gap is an understanding of what levels of contaminants affect sea lion health, and subsequently also affect vital rates, especially reproduction. Further, the possible effects on reproduction from chronic exposure to relatively low concentrations of toxic substances and the potential for reactive metabolites to cause damage to target tissues must be understood to be able to relate observed toxin levels to population effects in the western DPS of Steller sea lion.

Data Gaps

The primary data gap is an understanding of what levels of contaminants affect sea lion health, and subsequently vital rates, especially reproduction. Further, the possible influence on reproduction from chronic exposure to relatively low concentrations of toxic substances and the potential for reactive metabolites to cause damage to target tissues is needed.

10. Disturbance

The possible impacts of various types of disturbance on Steller sea lions have not been well studied, yet the response by sea lions to disturbance will likely depend on season, and their stage in the reproductive cycle (Kucey and Trites 2006). Close approach by humans, boats, or aircraft caused hauled out sea lions to go into the water, and caused some animals to move to other haulouts during a study in Southeast Alaska (Kucey 2005). The discharge of firearms at or near hauled out animals may have a particularly dramatic effect. Vessels that approach rookeries and haulouts at slow speed, in a manner that sea lions can observe the approach have less effect than fast approaches and a sudden appearance. Sea lions may become accustomed to repeated slow vessel approaches, resulting in minimal response. Although low levels of occasional disturbance may have little long-term effect, areas subjected to repeated disturbance may be permanently abandoned (Kenyon 1962). When humans set foot on a rookery or haulout, the response by sea lions is typically much greater, often resulting in stampedes that may cause trampling or abandonment of pups (Calkins and Pitcher 1982, Kucey 2005, Lewis 1987,). In British Columbia, harassment and killing that occurred prior to 1970 resulted in the abandonment of one major rookery, although it is now used as a haulout (Bigg 1988). Repeated disturbances that result in abandonment or reduced use of rookeries by lactating females could negatively affect body condition and survival of pups through interruption of normal nursing cycles.

Since Steller sea lions were afforded ESA protection in 1990, regulations have been in place to minimize disturbance of animals by humans, especially on rookeries. An unknown level of disturbance still occurs with current regulations. Disturbance by research activities includes aerial surveys, capturing of animals for branding, tagging, and sample collection, and close vessel approaches to rookeries and haulouts to observe branded animals. Sea lions are occasionally killed accidentally in the course of such scientific research activities, estimated at about three animals per year (Loughlin and York 2000).

Data Gaps

As noted, the current level of disturbance of sea lions at rookeries and haulouts is not known. Additionally, there is insufficient information about the potential sub-lethal effects of repeated disturbance on the health, reproduction and survival of Steller sea lions, especially females and pups. There is insufficient information about seasonal and environmental factors that might modulate physiological and behavioral reaction to disturbance and the resulting biological significance of any effects.

11. Nutritional Stress Due to Reduced Prey Biomass and/or Quality

The survival of large predatory mammals such as Steller sea lions is dependent on the availability of abundant, high quality prey (Stephens and Krebs 1986). Due to the high energetic demands of Steller sea lions relative to terrestrial mammals (Section I.H) and the large number of sea lions seasonally concentrated on rookeries, this species may be especially vulnerable to reduced prey biomass and quality. As a result, natural and anthropogenic factors that substantially influence prey availability, particularly during critical life history stages (e.g., pregnant females with a nursing pup, or recently weaned juveniles), have the potential to affect Steller sea lion vital rates and impede their recovery.

A reduction in prey resources may result in (1) a reduction in population growth rate, and (2) a lower carrying capacity. Specifically, reduced prey biomass and quality can lead to physiological responses by sea lions that directly (e.g., reduced natality) or indirectly (e.g., increased mortality from predators due to increased foraging) reduces their population growth. A sustained reduction of prey resources across a broad geographic region (i.e., ecosystem) would thus reduce the carrying capacity of sea lions.

As noted in Section I.H.1, Steller sea lions eat a wide variety of marine fish and cephalopods, some of which are densely schooled in spawning, migratory, or feeding aggregations (Sinclair and Zeppelin 2002). The abundances of many of the primary prey species of Steller sea lions have undergone substantial changes during the past 30 years (NRC 1996, 2003, North Pacific Fishery Management Council 2005a, b). Thus, during the period of decline of Steller sea lion populations in the western DPS, many primary prey species increased in abundance, while others decreased or remained relatively stable. Several factors have been implicated in these changes in prey biomass for Steller sea lions: 1) natural or environmental variability, 2) anthropogenic (fisheries) affects, and 3) ecosystem disruption resulting in interspecific competition (Anderson and Piatt 1999, Trites *et al.* 1999, Benson and Trites 2002). These factors may act individually or collectively to affect the availability of prey for Steller sea lions.

Environmental variability and nutritional stress

Periodic shifts in oceanic and atmospheric conditions appear to have had major effects on the productivity and structure of North Pacific ecosystems (Francis and Hare 1994, Francis *et al.* 1998, Hunt *et al.* 2002, Mackas *et al.* 1998, Anderson and Piatt 1999, Trites *et al.* 2006a) with cascading effects on some prey fish populations (Quinn and Niebauer 1995, Hollowed and Wooster 1992, 1995). For example, the size of available habitat for pollock, one of the primary prey species of Steller sea lions, reportedly increased with changes in the mixed layer depth in the Gulf of Alaska associated with climatic changes during the 1980s (Shima *et al.* 2000). Increases in pollock and other gadids (e.g. Pacific cod) in the Gulf of Alaska and Bering Sea (Alverson 1992), and their relatively low nutritional quality (Alverson 1992, Rosen and Trites 2000a) led to the "junk food hypothesis" for the decline of the western DPS of Steller sea lion.

In the original form of the "junk food hypothesis", the quantity of prey available to the declining population of Steller sea lions was thought to be high overall, but the prey community switched from one dominated by high energy prey (e.g., herring and osmerids) to low energy species (e.g., gadids and flatfish; Alverson 1992, Rosen and Trites 2000a). As originally articulated by Alverson (1992), pollock and other gadids were presumed to be equally poor foods for all age classes of sea lions (i.e., both juveniles and adults), resulting in reduced survivorship (of both age classes) and fecundity (Atkinson *et al.,* in press). However, results of subsequent feeding experiments, mathematical models, and field observations suggested that adult sea lion growth and condition should be relatively unaffected by the low energy content of gadids (Rosen and Trites 2000b, 2004, Trites 2003, Trites *et al.* 2006a, Malavear 2002). Instead, low energy prey may detrimentally affect juvenile Steller sea lions more than mature individuals due to their relative inexperience at foraging (Merrick and Loughlin 1997), their higher relative energy requirements (Winship *et al.* 2002), an upper limitation on the amount of food that a sea lion can physically digest to meet its daily energy requirements (Rosen and Trites 2004), or the availability of sufficient prey (Malavear 2002).

Fritz and Hinckley (2005) concluded that patterns and time series of fish abundance, fish recruitment, and sea lion food habits did not support the hypothesis that the 1976-77 regime shift triggered changes in the prey community that would have been deleterious to Steller sea lions. In addition, unpublished feeding experiments at the Alaska SeaLife Center are believed to show no negative consequences to juvenile sea lions fed only pollock (Calkins and Trites unpublished data). This is consistent with published studies showing that there are no different effects between high-lipid and low-lipid (or low-protein and high-protein) prey on sea lion body composition when animals are able to consume sufficient prey to meet their energy demands (Rosen and Trites 2004, 2005).

Although there were no scientific surveys or collections from 100 years ago that are directly comparable with those of the last 25 to 30 years, the observations and conclusions of some of the early naturalists in Alaska are worth reviewing (Nelson 1887, Jordon *et al.* 1896, 1898, Alexander 1898a, b, Jordon and Evermann 1902). These observations should be read with caution as they represent anecdotal information (and unpublished works) and generally were not part of a rigorous scientific study. However, they do provide a sense of the potential variability, as well as stability in the ecosystem.

Nelson (1887) reported that sea lions were scarce in the Aleutian Islands in the 1880s, but were abundant in the Pribilof Islands (about 35,000 animals), and during the early 1800s had once numbered several hundred thousand animals on St. George Island alone (but were extirpated upon direction of the Russians). Dixon (1986) investigated middens on Kodiak Islands and found Steller sea lions to be the most common fauna identified. Causey *et al.* (2005) concluded, based on zooarchaeology of early human sites in the Aleutian Islands from *c.* 3500 yr ago, that seabirds have fluctuated with temperature and precipitation. Populations of marine mammals may have also fluctuated (in abundance or availability to Aleuts) based in part on climate and hunting by Aleuts (Dixon 1986).

Turner (1886) indicated that Pacific cod and Atka mackerel were apparently rarely encountered at Attu Island prior to 1873, but were abundant there in 1878–81. At Attu Island, capelin were said to be very abundant every third year, as may have been the case at Atka Island. At Atka Island, capelin were also abundant when Turner visited (1878–81), and "dead fish [capelin, post-spawning] were so thick on the beach that it was impossible to walk without stepping on hundreds of them" (Turner 1886, p. 102).

Jordan and Evermann (1902) stated that "Alaskan pollock [were] found in the Bering Sea and neighboring waters south to Sitka and the Kurils. It is excessively abundant throughout the Bering Sea, swimming near the surface and furnishing the great part of the food of the fur seal. It reaches a length of 3 feet and is doubtless a good food-fish." and that "Likewise, cod is very abundant in the Bering Sea", and Atka mackerel is described as being abundant in the Aleutian Islands as it is today. Arrowtooth flounder, Greenland turbot, and Pacific halibut were all common." Jordan *et al.* (1896, 1898) in their fur seal accounts state that "In Bering Sea, in August and September, the Alaskan pollock seems to form by far the most important part of the seal's diet" and that "the cod, halibut, and Atka-fish are very abundant."

Alexander (1898a,b) stated that "Cod were abundant…Their abundance may have been the cause of the seals being plentiful in this region." and that "For several days, seals had been observed chasing some kind of fish…2 seals were speared. The fish proved to be Alaskan pollock. Both seals were large males." Kenyon and Wilke (1952) found "Evidence from the food remains on the Pribilof rookeries is that fur seals depend to a large degree on the…family Gadidae during their stay in the Bering Sea." Fiscus, Baines and Wilke (1962) found "Theragra, Mallotus and squid have consistently been the principal food of seals in the Bering Sea" since observations began in the 19th century (N=thousands).

Competition with Fisheries

In addition to environmental factors, fishing can affect the availability of prey on localized and ecosystem-wide scales (Trites *et al.* 2006d), which is of concern for the stability and recovery of Steller sea lion populations (Lowry *et al.* 1982). As summarized by Atkinson *et al.* (in press) in order to compete with a predator for food, commercial fishing has to either reduce abundance, nutritional quality, or distribution of prey resources available to the predator to the point where there are adverse effects on that predator. Fisheries in Alaska are some of the largest in the world. In 2003, approximately 2 million metric tons of groundfish were caught in the BSAI, which is equivalent to a harvest rate of approximately 10% (NMFS). Fishing has the potential to affect Steller sea lion recovery in several ways, including overall ecosystem-wide reductions in prey biomass, local and temporal depletions of prey, and reduced quality (size, age and caloric value) of individual prey by selective removal of larger, older individuals (Goodman *et al.* 2002, *Trites et al.* 2006d).

Many fisheries in the North Pacific are managed using a maximum sustainable yield (MSY) single-species strategy. MSY is based on the assumption that production of fish recruits, on average, is in excess of the level needed for replacement and that fisheries can remove the surplus of adults without jeopardizing future stock recruitment. Fishing mortality rates (F) set using single-species, MSY methodologies are designed to maximize yield (weight of catch) before it is lost to natural mortality (M) and minimize the likelihood of overfishing the target stock. Fishing at $F_{40\%}$ will, over the long run, reduce the average spawning stock size to 40% and total biomass to approximately 50% of their theoretical pristine levels.

While single species catch quotas are set for relatively large management areas (e.g., the Gulf of Alaska), individual vessels that fish for these species work in discrete areas. The potential for fisheries to reduce local abundances of fish was shown for Atka mackerel (Lowe and Fritz 1997) and Pacific cod fisheries (Fritz and Brown 2005), where local, short-term harvest rates were much greater than the annual target harvest rates on the stocks as a whole. Many of the areas fished by the Atka mackerel fishery in the Aleutian Islands and all of the Pacific cod fishery data analyzed by Fritz and Brown (2005) were collected within designated sea lion critical habitat. Statistical and correlative analyses of fishery effort/catch with trends in local sea lion populations have yielded equivocal results, some indicating a positive and some a negative relationship between catch and sea lion population trends (Loughlin and Merrick 1989, Ferrero and Fritz 1994, Dillingham *et al.* 2006). The utility of these analyses is diminished by issues of temporal and spatial scale mismatch between the treatment (magnitude of fish catch around a rookery) and response (population trend at that rookery), since animals breeding at a particular rookery range much farther during the year than the area encompassed by the catch data. One study (Hennen 2006), found significant positive relationships between several metrics of fishing and the steep rates of population decline in the 1980s. This relationship vanished in the 1990s, leading to the conclusion by Hennen (2006) that measures taken in the early 1990s (e.g., trawl exclusion zones, spatial-temporal management, shooting ban, reduction in incidental catch) may have been effective in slowing the decline.

Fisheries generally target larger, older individuals. As a result, a fished population will be composed of smaller, younger individuals, and have a smaller average size and age than an unfished population of the same species (NMFS 2000, Walsh *et al.* 2005, Trites *et al.* 2006d). These fishery-related changes may have two consequences for foraging sea lions. First, the distribution of fish within the water column and geographically, which often correlates with age (Ianelli *et al.* 2005), will be altered in a way that potentially affects availability to foraging sea lions. Second, a reduction in the average size of individual fish will reduce the per capita energy content and may necessitate increased foraging effort by sea lions to obtain the equivalent amount of energy in a larger number of small fish (Calkins and Goodwin 1988, NMFS 2000).

Beginning in 1999, NMFS has conducted a series of investigations of fishery effects on local fish populations. These studies, commonly called the FIT studies (after the NMFS Fishery Interaction Team that conducted them), were conducted on Atka mackerel on several local populations in the Aleutian Islands, Pacific cod on the spawning aggregation north of Unimak Pass, and pollock in neighboring gullies south of Kodiak Island.

1) Atka mackerel: Tagging studies conducted by NMFS since 1999 show large concentrations of fish at Seguam and Kiska, and smaller concentrations at Tanaga and Amchitka. These results are similar to those of Lowe and Fritz (1997), who analyzed changes in in-season fishery catch per unit effort to estimate initial biomass and fishery harvest rates. The trawl exclusion zone around Seguam Island may meet its objective because of a natural separation of suitable habitat for Atka mackerel inside and outside of the zone. By contrast, other trawl exclusion zones (e.g., near Amchitka Pass) may be less effective because the zone bisects habitat allowing fish inside to be vulnerable to fishing outside. NMFS concluded that the results of these tagging studies (e.g., McDermott et al. 2005) support the initial rationale for Atka mackerel fishery management: that the Atka mackerel fishery could create localized depletions of fish within sea lion critical habitat, and management measures (e.g., exclusion zones, time and area allocations of TAC) are necessary to address these effects and allow for sea lion recovery.

2) Pacific cod: There are two recent studies aimed at evaluating fisheries effects on Pacific cod. One was the FIT field study conducted by NMFS (Connors *et al.* 2004; Connors *et al*. in press) near Unimak Pass and the other was an analysis of fishery and survey data presented by Fritz and Brown (2005). These studies came to differing conclusions about the potential for fisheries to cause localized depletion of cod. There are two significant differences between the two studies that are important to the interpretation of their findings. First, the spatial scale (area surveyed) of the FIT study was smaller than that of the data analyzed by Fritz and Brown (2005), which was similar in scale to the cod aggregation being fished. In addition, the temporal scale of the FIT study (replicate surveys separated by several months) may have been too coarse to detect changes in cod density (whether related to the fishery or not) that could potentially affect the foraging efficiency of Steller sea lions. Second, the field experiment was designed to detect the efficacy of the trawl exclusion zone around Cape Sarichef in isolating the effects of the fishery on prey populations outside from those inside. Finding no difference between the changes in the density of fish inside and outside before and after the fishery, Connors et al. concluded that the trawl exclusion zone had no effect and that there was no fishery related localized depletion. While the trawl exclusion zone around Cape Sarichef may be inappropriately scaled relative to the size and movement of the prey aggregation on the SE Bering Sea shelf, these field results do not show that there is no potential for localized depletion related to fishing that could affect foraging success of Steller sea lions. As noted by both Connors *et al* and Fritz and Brown (2005), cod immigrate and emigrate from the aggregation north of Unimak Island during the winter. However, the much smaller study area considered by Connors *et al*. greatly increases the likelihood that fish movement would affect their results. Fritz and Brown (2005) analyzed fishery effort in 2001 outside of the (large) cod spawning area and found no evidence that large numbers emigrated. Consequently, during a 1.5 month fishing period in 2001, cod abundance (as reflected by changes in fishery catch per unit effort) north of Unimak Island declined significantly and this apparently was not related to large-scale emigration. As Fritz and Brown (2005) note, this pattern may be different depending on the

pattern of immigration, emigration, and fishing each year. Based on all the available evidence, it is not possible to definitively conclude that the fishery north of Unimak Island does not affect foraging efficiency of Steller sea lions within their critical habitat by reducing densities of Pacific cod during winter (when the frequency of occurrence of cod in their diet is the highest).

3) Pollock:

a. NMFS conducted three controlled experiments to test the effects of a pollock fishery south of Kodiak Island on the abundance and distribution of pollock, a common prey item of Steller sea lions. The results of the first experiment suggested that the fishery had no effect, the second did not have enough fishing effort to make an assessment, and the third found that the post-fishery biomass in the fished area was lower and that prey schools were more fragmented than in unfished area.

b. NMFS has also analyzed data on pollock density and distribution collected by fishing vessels in the winter in the SE Bering Sea. Barbeaux and Dorn (2003) found that the distribution of fishing effort on pollock north of Unimak Island did not always match the distribution of fish. This could lead to differences in fishing intensity in local areas. Shen *et al.* (2007) found that there was less pollock biomass, more school fragmentation, and a more dispersed prey field at the end of the pollock winter fishing season than during the beginning in the SE Bering Sea. These changes could negatively affect Steller sea lion foraging efficiency.

c. Based on these results, NMFS could not conclude that there was little or no potential for fishery effects on the availability of pollock (as prey) for Steller sea lions.

The objective of fisheries management measures implemented in 2002 (NMFS 2001, 2003) was to remove or mitigate the effects of fisheries that could lead to adverse impacts on Steller sea lions. These measures were intended to address both ecosystem-level effects (e.g., biomass reduction) as well as the temporal and spatial effects of fishing by raising minimum fish stock size thresholds, reducing fishing in near-shore portions of critical habitat, reducing seasonal competition for prey and reducing the likelihood of fishery-related localized prey depletions. The spatial-temporal fishery management measures were based largely on an analysis of the at-sea distribution of sea lions recorded by satellite linked time-depth recorders. The analysis led to the development of a "zonal approach" to management for the 2002 measures (NMFS 2001), in which near-shore portions of critical habitat were considered more important to foraging sea lions than offshore areas. However, most of the data used in the telemetry analysis was collected from juvenile sea lions less than 2 years of age, many of which were likely not completely weaned. As a consequence, the foraging habitat of adult animals, particularly females, is underrepresented in the telemetry data that formed the basis of the management measures implemented in 2002.

Direct impacts of fisheries on Steller sea lion foraging will depend on spatial, temporal, and targeted species overlap for which little data is currently available (Baraff and Loughlin 2000). The potential for competition between fisheries and Steller sea lions, as indicated by energetic models, differs for each prey species considered. For instance, the estimated consumption of gadids based on the energetic demands and diets of wild sea lions was 179,000 ± 36,700 t in all regions of Alaska in 1998. This represented approximately 12% of the total commercial catch (Winship and Trites 2003). In the same study, it was estimated that Steller sea lions consumed a total of 104,000 ± 20,600 t of Atka mackerel in 1998, but this was equivalent to 181% of the fishery catches off Alaska. At this level, Steller sea lion predation would have also accounted for a large proportion of the total natural Atka mackerel mortality.

Management measures to address potential fishery effects on Steller sea lions were first promulgated in 1991-1993, and then extensively modified in 2002 (Fritz *et al.* 1995, NMFS 2003). These included spatial and temporal allocations of harvest quotas to reduce the likelihood of localized depletions of groundfish prey, fishery exclusion zones to limit spatial overlap between fisheries and sea lions, and modified harvest control rules to reduce the likelihood of overall prey abundance being reduced to less than 20% of pristine levels (see Section II). NMFS (2001, 2003) concluded that the current suite of management measures avoids jeopardy to the species and adverse modification of critical habitat.

Inter-specific competition

Piscivorous fish consume many of the same species and sizes of prey as Steller sea lions. The strength of these food-web interactions has likely changed during the past 30 years in response to both natural and anthropogenic factors. For instance, annual differences in the size and distribution of young-of-the-year as well as adult pollock affect annual levels of cannibalism (Livingston 1991, Wespestad *et al.* 2000). Differential rates of fishing within the groundfish community may have also indirectly contributed to increases in arrowtooth flounder populations, a species with considerable diet overlap with Steller sea lions (NMFS 2000, 2001). How these changes as well as substantial increases in the population of Pacific halibut since the 1980s (Hollowed *et al.* 2000, IPHC 2000, Wilderbuer and Sample 2000, Trites *et al.* 1999) affect the prey field and foraging patterns of Steller sea lions or relate to population level impacts remain to be determined.

The diets and distribution of many marine mammals and birds also overlap those of the western DPS of Steller sea lions. As consumers of common prey resources, the dynamics and concomitant prey biomass removed by these sympatric piscivore populations may therefore affect the quantity and quality of prey available to Steller sea lions. As such, recovery of Steller sea lions may be affected by changes in the abundance, distribution, and prey removal by other apex predators. Whales are considered significant consumers in many marine systems and models estimate that prey consumption by cetaceans approaches or exceeds removals by commercial fisheries (Laws 1977, Laevastu and Larkins 1981, Bax 1991, Markussen *et al.* 1992, Kenney *et al.* 1997, Trites *et al.* 1997, Witteveen *et al.* 2006). Such high levels of consumption can have significant effects on the distribution and abundance of prey species and the structure of marine communities (Perez and McAlister 1993, Kenney *et al.* 1997). Likewise, removals and recovery of cetacean populations may affect marine ecosystems through complex trophic cascades (Laws 1985, NRC 1996, Merrick 1997, Trites *et al.* 1997, Springer *et al.* 2003, Witteveen *et al.* 2006). Shore-based and pelagic whaling in the 1900s significantly reduced the number of large whales in the North Pacific, reducing their consumption (biomass removal) of certain fish, cephalopods, and zooplankton within marine ecosystems (Rice 1978) and effectively increasing prey available to other consumers in the system (Springer *et al.* 2006). Following decades of international protection, the abundance of some whale stocks has increased, including an apparent 10% increase in central North Pacific humpbacks between the early 1980s and early 1990s (Baker and Herman 1987, Calambokidis *et al.* 1997). It has been hypothesized that whale stock resurgence may have reduced prey availability and contributed to declines of piscivorous pinnipeds and birds in the Gulf of Alaska and Bering Sea ecosystems (Merrick 1995, 1997, NRC 1996, Trites *et al.* 1999).

Several large piscivorous whales are migratory and fulfill their annual consumption needs on high latitude feeding ground, including waters found within Critical Habitat of the western DPS of the Steller sea lion. Substantial seasonal feeding aggregations of humpback (Waite *et al.* 1998, Witteveen 2003), fin, and gray whales occur within the Gulf of Alaska and Bering Sea. Their diets include large zooplankton species and a variety of schooling fish (Thompson 1947, Nemoto 1957, Moore *et al.* 2000, Tamura and Ohsumi 2000) that are also consumed by Steller sea lions (capelin, herring, sandlance, smelts, small pollock) (Pitcher 1981, Sinclair and Zeppelin 2002) or by the prey of sea lions (pollock, cod, arrowtooth flounder) (Livingston 1993). As such, piscivorous whales have the potential to compete with Steller sea lions both directly when feeding on common prey and indirectly when consuming zooplankton and forage fish upon which other sea lion prey species feed. As populations of piscivorous cetaceans recover, this potential would be expected to increase.

Key Challenges in Evaluating Existence and Potential Causes of Nutritional Stress

A critical research challenge for Steller sea lion researchers is demonstrating the mechanistic links between prey availability, nutritional stress of the individual, and changes in survival and reproductive rates that would lead to population level effects. Table I-15 illustrates the myriad potential biological effects that could occur in immature and adult Steller sea lions if individuals were nutritionally limited. The effects range from morphological, physiological, and behavioral changes to alterations in vital rates that would affect population trends. A comparison of how these effects may have changed across the 1980s, 1990s, and 2000-2004 identifies many of the data gaps that need to be filled to assess current nutritional status for the western DPS of Steller sea lions. For most categories, available data sets are of such limited geographical and temporal scope that evaluating the role of nutritional stress in the decline of Steller sea lion populations or in its recovery has been hampered. For example, other than numbers of individuals from population counts, no measurements have been made for adult Steller sea lions in the Alaska portion of the western DPS since the 1990s. Consequently, changes in body condition, reproductive success or foraging parameters that would be direct indicators of acute or chronic nutritional stress are currently unknown for adults, except for those estimated by demographic models (Holmes and York 2003, Fay 2004, Fay and Punt 2006, Holmes *et al.* in review).

To date, the focus of nutritional research has been on the effects of nutritional status on individual sea lion behavior, health, and physiology. Proximate dietary mechanisms under investigation include: 1) decreased energy intake due to changes in the availability or energy content of prey, 2) changes in the energy requirements of the predator, 3) deficiency of other nutrients (i.e., protein or specific amino acids) or essential elements, 4) physiology of metabolic homeostasis, and 5) assessment of nutritional stress responses for different age classes. Part of the difficulty in assessing chronic nutritional stress lies in determining the temporal or spatial scale of study: i.e., how does system wide or localized availability of prey affect Steller sea lion foraging ecology?

The evaluation of body condition in Steller sea lions remains problematic due to the inability to safely capture large animals, difficulty of working in remote locations, and poor knowledge of natural variation in body condition that occurs between seasons, geographical region, age, and gender. Indices of body condition include body mass, standard length, axillary girth and additional girth rings, and percent body fat. There are difficulties associated with the criterion of body mass in a sexually dimorphic species. The sexes must be examined separately in each geographic area, and longitudinal data (e.g., mean growth rates of branded pups recaptured as juveniles) should be examined. There are also problems peculiar to each of the methods used to measure blubber reserves (direct measure, ultrasound, skinfold calipers, isotope dilution, and bioelectrical impedance analysis).

Data Gaps

Small and DeMaster (2006:527) recommended that "studies designed to determine if fisheries…deplete or disturb the prey field, to the extent that foraging success is reduced, need to be completed." They specified that studies which assess and compare the biomass of prey before, during, and after fishing in a treatment area with the prey biomass in an unfished control area need to continue.

A series of critical data gaps exist regarding the determination of 1) whether rates of natality have indeed continued to decline, 2) whether any effect on natality is due to reduced prey biomass, abundance, and nutritional stress, and 3) how females respond to nutritional stress in their relative energy expenditures on lactation, pregnancy and their own maintenance. Declines in natality estimated in the 1990s at a few rookeries were significant (about 30%; Holmes and York 2003, Winship and Trites 2006a), but the mechanisms involved (e.g., nutritional stress, disease, contaminants) are unknown.

12. Global Climate Change

Climate change has received considerable attention in recent years, with growing concerns about global warming and the recognition of natural climatic oscillations on varying time scales. Global air and ocean temperatures during this century and before are warming (IPCC 2007, see http://www.ipcc.ch), and evidence suggests that the productivity of the North Pacific is affected by changes in the environment (Quinn and Niebauer 1995, Mackas *et al.* 1998).

Increases in global temperatures are expected to have profound impacts on arctic and sub-arctic ecosystems, and some of these impacts have been documented over the last several decades. Specifically, (1) winter temperatures in Alaska and western Canada have increased as much as 3-4 ˚C over the past half century, (2) precipitation, mostly in the form of rain, has increased primarily in winter resulting in faster snowmelt, (3) sea ice extent has decreased about 8% over the past 30 years, with a loss of 15 to 20% of the late-summer ice coverage in the arctic, and (4) glacial retreat, particularly in Alaska, has accelerated contributing to sea level rise (ACIA 2004). These impacts, and others, are projected to accelerate during this century.

The effects of these changes to the marine ecosystems of the Bering Sea, Aleutian Islands, and the Gulf of Alaska, and how they may specifically affect Steller sea lions are uncertain. Warmer waters could favor productivity of certain species of forage fish, but the impact on recruitment dynamics of fish of importance to sea lions is unpredictable. Recruitment of large year-classes of gadids (e.g., pollock) and herring has occurred more often in warm than cool years, while the distribution (with respect to foraging sea lions) and recruitment of other fish (e.g., osmerids) could be negatively affected. Whether these patterns will continue as overall temperatures increase is uncertain, as are the effects on the duration and strength of atmospheric and oceanographic regimes (Trenburth and Hurrell 1994, Hare and Mantua 2000).

Climate-driven changes in productivity and community structure due to warming oceans may already be underway in the northern portion of the Bering Sea and Bering Strait, where sea ice plays a major role in structuring the food web and the ecosystem is particularly vulnerable to rapid system reorganization under global warming. Reduced seasonal sea ice cover, changing hydrographic conditions, and reduced primary production in the northern Bering Sea may be associated with apparent declines in ice-associated benthic species of mollusks and amphipods since the 1990s (Grebmeier *et al.* 2006). Benthic-feeding walrus, bearded seals, gray whales and diving sea-ducks such as Spectacled eider are all threatened by these changes, as are Arctic Native communities whose traditional subsistence culture has relied on these ice-associated mammals and birds for thousands of years. This ecosystem has short, simplified food chains; thus the potential for trophic cascades is higher. Warming seawater in the north could expand the range of groundfish from the south, putting more pressure on the benthic prey base. The northern Bering Sea may be poised for the sort of trophic cascade and system reorganization anticipated by the U.S. GLOBEC (Global Ocean Ecosystems) research program as a consequence of global warming at high latitudes (Grebmeier *et al.* 2006).

Warmer temperatures could shift the distribution of sea lions northward. The eastern DPS increased in size at a rate of approximately 3% per year from the early 1980s through 2004, despite a decline in the size of the breeding population at the southern extent of its range in California. All of the increase in the eastern DPS occurred north of California, and new rookeries established in the 1990s (White Sisters and Hazy Island) were near its northernmost extent in southeast Alaska.

As temperatures warm and global ice coverage decreases, sea levels will rise. This will directly affect terrestrial rookery and haulout sites currently used by Steller sea lions as well as those that may be used by a recovering population. Presumably, sea lions using terrestrial sites will simply move upslope as sea levels rise, assuming that the terrain at the site is suitable. However, sites on some islands with low relief (e.g., Agligadak Island) may be submerged. The net effect of a rise in sea level on overall terrestrial sea lion habitat amount or availability is uncertain, but at the projected rate it is unlikely to have a significant effect for many years.

Fluctuations or cycles in physical and biological characteristics of marine ecosystems may not necessarily affect higher trophic levels because of strategies for survival they have evolved to buffer them against environmental uncertainty. Based on their analyses of possible causes of the sea lion decline, Pascual and Adkison (1994) concluded that environmental cycles were unlikely to have caused declines of the magnitude and duration observed. Shima *et al.* (2000)

did a comparative analysis of population dynamics of four species of pinnipeds in similar variable environments (Steller sea lions in the Gulf of Alaska, Cape fur seals in the Benguela Current, harp seals in the Barents Sea, and California sea lions in the California Current) and found a major decline only for Gulf of Alaska Steller sea lions. They concluded that the success of the other populations suggests that pinnipeds in general have the ability to adapt to environmentally driven changes in prey resources, and that other factors were involved in the decline of Steller sea lions.

Data gaps

More research is necessary to describe linkages between changes in the environment and the dynamics of apex predators such as Steller sea lions. Distinguishing between anthropogenic and environmentally-driven changes in the abundance and distribution of prey resources has eluded scientists and managers, but is necessary in order to understand the forces underlying change in population size and demographics. Furthermore, the direct effects of temperature increases on sea lion metabolic rates, foraging efficiencies, and disease transmission are unknown.

Table III-1. Estimated takes of western DPS Steller sea lions by Alaska Natives in five regions (1992 – 2004). Values include both retrieved harvest and reported struck and lost (Wolfe *et al.* 2005).

Year	**Prince William Sound-Cook Inlet**	**Kodiak & Alaska Peninsula**	**Aleutian Islands**	**Pribilof Islands**	**Bristol Bay**	**Total**
1992	40	60	135	297	9	541
1993	46	64	124	245	6	485
1994	27	67	122	193	1	410
1995	31	144	96	68	0	339
1996	17	65	58	46	0	186
1997	6	46	52	56	4	164
1998	28	27	37	78	0	170
2000	17	32	76	43	0	168
2001	16	47	98	38	0	199
2002	6	24	105	43	0	178
2003	25	41	107	32	0	205
2004	54	21	96	32	1	204
Averages						
1992-95	36	84	119	201	4	444
1996-04	21	38	79	46	1	184

IV. THREATS ASSESSMENT FOR THE WESTERN POPULATION

In Section III, we discussed the various factors that may affect Steller sea lion health and population numbers. The intent of this section is to focus the recovery plan on those factors that represent a threat to sea lion recovery[10] (Table IV-1). A threat is defined as any factor (natural or human related) which represents a substantial impediment to recovery. This definition is somewhat subjective – in some recovery plans all factors are listed as threats while other recovery plans only discuss human-related, or mitigable, factors. Our approach was a compromise which includes natural factors while focusing on the mitigable or new aspect to that threat that needs attention when planning for recovery. For example, killer whale predation may be a natural phenomenon for Steller sea lions, yet given the substantial decline in sea lion numbers, predation may now have a greater proportional impact on survival. Thus, it is important to consider this threat in our recovery planning, acknowledging there may be little we can do about it, and that the impact of that threat may decrease as the sea lion population grows.

A "weight of evidence" approach was used to assess the relative impact of each threat (factor) identified in Section III. This qualitative assessment approach was selected rather than a quantitative approach because of the substantial uncertainty in the understanding of each threat's influence on sea lion population dynamics. By using this approach, NMFS sought to accommodate divergent opinions and to ensure appropriate examination of risk factors on which there was considerable disagreement or uncertainty. sing the extensive scientific and professional expertise of the recovery team, we were able to identify three relative threat levels, Low, Medium, and High. The assessment allowed the development of a recovery strategy and prioritization of recovery actions. It is expected that in future plans that these threats will be more finely assessed. Although the rankings were developed to be relative to each other, a conceptual definition of each level is provided below:

High: a threat with substantial impacts to recovery requiring mitigation and/or further research to identify impacts
Medium: a threat with moderate impacts which, if mitigated, could increase the likelihood of recovery but, in and of itself, has limited impact on population trajectories
Low: a source of mortality that likely has little impact on population trajectory

NMFS used the following parameters to describe each threat: (1) the mechanism by which each threat operates (bottom-up, or top-down, or both); (2) the age-class most vulnerable to the threat; (3) the relative frequency that the threat occurs; (4) the uncertainty in the evidence used to determine the relative impact; and (5) the feasibility of mitigation.

NMFS had difficulty determining the relative impact of two threats on the recovery of the western DPS of Steller sea lion: environmental variability, and competition with fisheries.

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[10] Recovery is defined here as the point in which a species has improved to the point that it can be removed from the federal list of endangered and threatened wildlife and plants.

Individual Team members ranked the threat HIGH or LOW based on various weighting of available scientific evidence. To address the high uncertainty about these threats and the competing hypotheses, the term "potentially high" was assigned. This qualified ranking reflects NMFS' intention of adopting a precautionary approach and highlighting research needs that could reduce critical uncertainties. The Team had also ranked killer whale predation as a "potentially high" threat. After public review and comment, and as additional scientific information became available (e.g., Maniscalco *et al.* 2007), NMFS concluded that the threat was unlikely to be high and assigned a medium ranking to this potential threat in the May 2007 version of the draft revised Plan. However, based on comments and review of the 2007 draft of the Plan which further highlighted the degree of scientific disagreement, controversy, and polarization on this issue, and in keeping with the aforementioned precautionary approach in assessment of threats to the recovery of this endangered DPS, NMFS has now categorized the killer whale predation threat as "potentially high".

The hypotheses proposed to explain the decline of the western stock fall into two categories. The first category, the bottom-up hypotheses, includes potential causes that would affect the physical condition of sea lions such as large-scale fishery removals that reduce the availability or quality of prey species, a climate/regime shift in the late 1970s that changed the abundance or distribution of prey species, nonlethal disease that reduced the foraging efficiency of sea lions, and pollutants concentrated through the food web that contaminated fish eaten by sea lions, possibly reducing their fecundity or increasing mortality. The second category, the top-down hypotheses, encompasses factors that kill sea lions independently of the capacity of the environment to support the sea lion population. These include predators such as killer whales (or possibly sharks) that switched their prey preference to sea lions, incidental takes of sea lions through capture or entanglement in fishing gear that increased as a result of the expansion of commercial fisheries, takes of sea lions in the subsistence harvest that were higher than estimated, shootings of sea lions that were underestimated in the past and present, and pollution or disease that increased mortality independently of effects on nutrition.

The mechanism by which each threat operates is considered "direct" if it kills individual sea lions directly and reduces survival rates (e.g., commercial harvest, intentional shooting, entanglements or incidental catch by fishing gear, disturbance, and predation) and "indirect" if it reduces sea lion body condition and leads to reduced rates of reproduction and survival (e.g., competition with fisheries, environmental variability).

A. Threats

1. Environmental variability – POTENTIALLY HIGH

The potential impact of environmental variability, through a reduction in the biomass and quality of sea lion prey species, has recently received substantial attention and study within the scientific community. Periodic shifts in oceanic and atmospheric conditions may have major effects on the productivity and structure of North Pacific ecosystems, with cascading effects on some prey fish populations. The manner and mechanism by which such "regime shifts" and altered fish populations would affect marine mammals, including Steller sea lions, is poorly understood and remains unresolved. In assessing the relative impact of environmental variability on sea lions, the Team determined, and NMFS adopted this determination, that adult females and juveniles are likely the most vulnerable age-classes, the threat occurs at a high frequency, there is a high level of uncertainty associated with the available evidence, and there is a low feasibility of mitigation. However, the Team did not reach consensus on the relative impact on the recovery of the western DPS due to environmental variability, and the threat was ranked potentially high; the rationale for a high and low rank is described below.

The threat posed by environmental variability to sea lion recovery was ranked high by some members of the team based on the assertion that changes in the North Pacific fish community structure stemming from the regime shift in 1976-77 were substantial enough to alter the quality and availability of prey for Steller sea lions, resulting in a dominance of pollock and other gadids, which resulted in the nutritional stress of juvenile sea lions. The 1976-77 regime shift is hypothesized to have changed the recruitment dynamics or distributions (or both) of multiple fish species across the North Pacific Ocean, resulting in increases in low energy prey (e.g. gadids) and decreases in high energy prey (e.g., herring), which in turn reduced sea lion vital rates. In this scenario, the magnitude of the change to the North Pacific ecosystem caused by the 1976-77 regime shift is thought to be larger than previously experienced by sea lions during the 1900s. If it were within the normal range and sea lions have a high likelihood of occasionally declining more than 80%, modeling suggests that they would have likely gone extinct given their life history characteristics. Fishing could have exacerbated the regime shift related impacts through relatively high local harvest rates of sea lion prey species, increasing their foraging costs. These changes would decrease carrying capacity, yet their impact on sea lions should decrease as their population declined. However, a threat to recovery will persist until the environment, and associated fish distributions and populations, change again to favor sea lions, increasing their carrying capacity and subsequently sea lion survivorship and birth rates.

The threat posed by environmental variability to sea lion recovery was ranked low by some members of the team based on the assertion that although oceanographic and atmospheric conditions have changed over the last several decades, those changes have not been outside the range of natural fluctuation previously experienced by sea lions. Gadids have been and will continue to be a principal component of the diet of sea lions, and there are not likely to be significant consequences to Steller sea lion health or vital rates from such a diet. Further, available evidence indicates that the current fish community structure is very similar to that just prior to the 1976-77 regime shift, and changes in sea lion diets between regimes were unremarkable, thus the potential impact of environmental variability on recovery in the near term is minimal.

Based on the high uncertainty regarding this threat (competing hypotheses which support a high or low ranking), a precautionary approach is to describe this as a potentially high threat.

2. Competition with Fisheries – POTENTIALLY HIGH

The potential impact of competition with fisheries, through a reduction in the biomass and quality of sea lion prey species, has caused considerable debate among the scientific community. The primary issue of contention is whether fisheries reduce sea lion prey biomass and quality at both the local and regional spatial scales that may lead to a reduction in sea lion survival and reproduction, and if sustained, their carrying capacity. The effect of fisheries on the distribution, abundance, and age structure of the sea lion prey field, at the spatial scale of foraging sea lions and over short and long temporal scales, is largely unknown. Further, uncertainty in the available information, including the efficacy of management regulations implemented to mitigate possible adverse effects of fisheries, permits disparate interpretations and inferences. In assessing the relative impact of competition with fisheries (i.e., all fisheries including commercial and sport) on sea lions, the Team determined adult females and juveniles to be the most vulnerable age-classes, fishing activities occur at a high frequency, there is a high level of uncertainty associated with the available evidence, and there is a high feasibility of mitigation. However, the Team did not reach consensus on the relative impact on the recovery of the western DPS of Steller sea lion due to competition with fisheries, and the threat was ranked potentially high; the rationale for both a high and low rank is described below.

The threat posed by competition with fisheries was ranked high by some members of the team based on the potential for fisheries to out-compete sea lions for similar prey (e.g., walleye pollock, Atka mackerel, Pacific cod) subsequently leading to lower sea lion carrying capacity. A decreased carrying capacity could result from the combined effects of seasonally compressed fishing in sea lion foraging areas, the long term impacts of exploitation of sea lion prey since the 1960s, and the indirect effects of fishing on the ecosystem. Fishing may have contributed to changes in the location, density, distribution, availability, quality, and energy value of the sea lion prey field. Population declines may have been driven by reductions in the quality and quantity of available sea lion prey initially caused by the development of groundfish fisheries in the mid-1960s, and then intensified as fishing effort for several prey species increased within sea lion foraging habitats in the 1970s and 1980s. The 1976-77 regime shift could have exacerbated fishing-related impacts by reducing the availability of alternative, non-commercial prey; e.g., osmerids. While a number of conservation measures for Steller sea lions have been enacted by the North Pacific Fishery Management Council and NMFS to mitigate these impacts, the primary objective of these measures was to avoid jeopardy and adverse modification of critical habitat. However, significant uncertainties remain in our understanding of how the sea lion measures and the overall management of North Pacific groundfish fisheries promote sea lion recovery. For instance, it is not known how the needs of a recovering, increasing population of sea lions are accounted for when the long-term equilibrium spawning biomasses of multiple prey species are reduced by as much as 60%. There is also concern about the potential effects of concentrated fishing activities in sea lion foraging areas derived from the localized depletion, even if temporary, of fish stocks. Thus, the current and expected threat of the competitive effects of fishing on the recovery of the western DPS of Steller sea lion was ranked high.

The threat to recovery of the western DPS due to competition with fisheries was ranked low by some members of the team based on the potential that competition occurs only if prey is limited. Even large human harvests of a known prey species may have limited effect on availability to Steller sea lions if a large biomass remains unharvested and accessible. In addition, it can be argued that commercial harvest of adult pollock and other piscivorous fish species might reduce the threat to Steller sea lion recovery posed by these potential competitors. Therefore, commercial removals of these species from critical habitat may reduce their potential for competing with Steller sea lions for smaller forage species. The potential for localized depletion of prey by commercial fisheries could be minimal because sea lions are opportunists, feed on a diverse diet, and are known to exploit seasonally available prey, suggesting prey switching may be a natural foraging strategy. The physiological consequences of switching to alternate prey in response to localized depletion of a target species (whether a natural or fishery-induced diet change) are not necessarily detrimental. In addition, the effects on sea lion foraging efficiency due to the mechanical disruption of large fish aggregations by trawls and other commercial fishing gear are equivocal. The species-specific duration and degree of fish school disruption due to commercial harvest are unknown and worthy of further fish behavior studies. Disruption of large aggregations into multiple smaller groups could enhance Steller sea lion foraging success by increasing the surface-to-volume ratios of the prey aggregations. Therefore, the potential threat to sea lions by disrupting large prey aggregations is uncertain but relatively low.

Threats that could be confounded with fishing include other indirect factors such as environmental variability and increased inter-specific competition. Based on the high uncertainty regarding this threat (competing hypotheses which support a high or low ranking), a precautionary approach is to describe this as a potentially high threat. Specific to the threat that fisheries may pose to Steller sea lions, there is an especially high need to conduct research to reduce uncertainties and to evaluate the efficacy of fishery conservation measures for Steller sea lions. Reviews should be conducted at the scale appropriate to sea lion foraging patterns.

3. Predation by killer whales – POTENTIALLY HIGH

The potential impact of predation by transient killer whales on the dynamics of Steller sea lions and the North Pacific marine ecosystem over the last several decades has recently received substantial attention and study within the scientific community. Information on killer whale abundance, diet, and movements has increased, and new hypotheses have been developed within the scientific community on how predation by killer whales has influenced marine mammal populations, including sea lions. Major limitations in the available data result in substantial uncertainty, especially when trying to determine the historic impacts of killer whale predation. In assessing the relative impact of predation by killer whales on sea lions, the Team determined that pups and juveniles are likely the most vulnerable age-class, that the threat occurs at a high frequency, that there is a high level of uncertainty associated with the available evidence, and that there is a low feasibility of mitigation. The Team did not reach consensus on the relative impact of killer whale predation on the recovery of the western DPS.

Evidence indicates that killer whale predation has been, and likely still is, a source of natural mortality in sea lions. The threat posed by killer whale predation was ranked potentially high due primarily to the large amount of disagreement within the scientific and stakeholder communities about the level of that predation throughout the range, the potential for killer whales to affect Steller sea lion populations given specific assumptions about their feeding habits, and the substantial amount of mortality potentially associated with predation across the range. Available evidence from isotope analyses and field observations indicate that the primary prey species of killer whales are at lower trophic levels than sea lions, yet some groups of killer whales may specialize on sea lions. Recent field studies of predation rates by GOA killer whales, a group known to specialize on sea lions, revealed a sea lion mortality rate below the annual mortality rate. Other studies, within these same areas, have shown increasing juvenile and adult sea lion survivorship through the 1990s and 2000s (data are limited primarily to the Central GOA). Several studies have also determined that available data are not consistent with the major assumptions of the Sequential Megafaunal Collapse Hypothesis. The hypothesis is also not supported by the fact that the eastern DPS has increased at approximately 3% per year for at least 20 years while co-existing with a larger population of transient killer whales in an environment exposed to similar commercial whaling and environmental impacts. Combined, this evidence indicates that the predator-prey dynamics between killer whales and sea lions is unlikely by itself to limit sea lion recovery in the near future. While NMFS ranked the killer whale predation threat as medium after consideration of public comments on the May 2006 version and new scientific findings not available to the Team, comments received from the scientific and stakeholder communities on the 2007 draft of the revised plan further emphasized the high level of disagreement and controversy within both communities on this issue. Thus, as noted previously, due to the uncertainty and the need to be precautionary in our assessment of possible threats to the recovery of this endangered DPS, NMFS has categorized the relative potential impact of this threat as "potentially high". Due to competing theories about the likely importance of killer whale predation on Steller sea lions, but limited data, the potentially high ranking was assigned to ensure we further evaluate the potential for predation to affect sea lion recovery.

4. Toxic substances - MEDIUM

Relatively low levels of toxic substances, including heavy metals, have been documented in Steller sea lions (with some striking exceptions), and these substances are not believed to have caused high levels of mortality or reproductive failure. However, there are no studies on the effects of toxic substances at the population level to determine their impact on vital rates and population trends. Chronic exposure to toxic substances may result in reactive metabolites that could cause damage to DNA, RNA, and cellular proteins. Sea lions exposed to oil spills may become contaminated with polycyclic aromatic hydrocarbons (PAHs) through inhalation, dermal contact and absorption, direct ingestion, or by ingestion of contaminated prey. Newer contaminants such as PBDEs have not been measured in Steller sea lions. Thus, overall, there is still some concern that toxic substances may have indirect impacts on individual vital rates, including reproductive potential.

Adult females and pups are likely the age-classes most vulnerable to toxic substances, the threat occurs at a high frequency (i.e., toxins are commonly found in the North Pacific), and there is a high level of uncertainty associated with the evidence described above. Thus, the relative impact on the recovery of the western DPS of Steller sea lion due to toxic substances is ranked medium, with a medium feasibility of mitigation.

5. Incidental take due to interactions with active fishing gear - LOW

The primary documented sources of information used to assess the impact of incidental take are estimates of mortality based on fishery observer data, self-reported fisheries data, and data on stranded animals. The mean number of lethal entanglements in active U.S. fishing gear was estimated at 31 sea lions per year between 1990 and 2001. Information is not available from several fisheries known to interact with sea lions, thus 31 is a minimum estimate. These estimates were made using a variety of data sources, some being relatively old depending upon the fishery. In the most recent stock assessment for the western stock, Angliss and Outlaw (2006) reported that the minimum estimated mortality rate incidental to U. S. commercial fisheries is 24.6 sea lions per year, based on observer data (24.2) and stranding data (0.4) where observer data were not available.

As many as 50 sea lions were killed incidental to herring fishery operations in Russia, but this is based solely on observer records during a single 3-month period in 2002. Nikulin and Burkanov (2000) documented marine mammal bycatch in Japanese salmon driftnet fishing in the Russian exclusive economic zone of the southwestern Bering Sea. Catch of only one Steller sea lion was observed during 1992-1999, and it was released alive. Additional information on incidental take is available from observations of sea lions suffering from injuries caused by both commercial and sport fishing operations. However, there are large segments of the fishing industry (e.g., longline fleet for Pacific cod, halibut and sablefish, salmon and herring fisheries) that have either no or limited observer coverage, possibly leading to underestimates of incidental take.

NMFS recognizes that there are some fisheries in Alaska that have small or no observer programs, and that there are unobserved fisheries in Russian waters that could have incidental catches of Steller sea lions from the Alaskan portion of the western stock. NMFS based its low threat ranking for fisheries' incidental take on the relatively small number of estimated kills recently from this source in Alaska, which are based in large part on robust observer programs for many fisheries.

Juvenile sea lions are the age-class most vulnerable to incidental take, the threat occurs at a medium frequency, and there is a medium level of uncertainty associated with the evidence above. Because the threat primarily affects juveniles and is probably less than 100 animals across the entire western DPS, the relative impact on recovery is ranked low, with a medium feasibility of mitigation. The primary area of uncertainty is the take of sea lions in Russia. Incidental take within the U.S. portion of the western DPS is very low and it is unlikely that this could be reduced without extraordinary measures.

6. Alaska native subsistence harvest – LOW

Both the ESA and the MMPA contain provisions that allow coastal Alaska Natives to harvest endangered, threatened, or depleted species for subsistence purposes. Based on retrospective surveys, the annual subsistence harvest (including struck and loss) decreased substantially from about 550 sea lions in 1992 to about 200 in 1996 followed by annual takes between 165 and 215 from 1997 to 2004. In the early 1990s, juveniles were harvested at least twice as much as adults, yet that ratio declined beginning in 1996, and between 2000 and 2004, the ratio of juveniles to adults in the harvest ranged from 0.5 to 1.0. The ratio of males to females harvested in 2004 was 1.8, below the 5-year average of 4.1 during the previous five years. In 2004, 24 adult females were harvested, representing about 20% of the total harvest of known sex and age. The greatest numbers of sea lions harvested were in the Pribilof Islands and the Aleutian Islands. The surveys that produced these estimates covered all Alaskan communities that regularly hunt Steller sea lions, but a few additional animals are taken occasionally at other locations as well. Native hunters may currently take some sea lions in Chukotka, but the number killed is unknown. No other subsistence harvests are currently conducted.

Males, both juvenile and adult, are the age-class most vulnerable to subsistence harvests, the threat occurs at a medium frequency, and there is a low level of uncertainty associated with the evidence above. Thus, the relative impact on the recovery of the western DPS of Steller sea lion due to the Alaska Native subsistence harvest is ranked low, with a high feasibility of mitigation.

7. Illegal shooting – LOW

Historical accounts document substantial mortality due to illegal shooting (Alverson 1992, Matkin and Fay, 1980), whereas one study conducted in 1988-1989 in the Copper River Delta indicates that the frequency of occurrence may have declined substantially (Wynne 1990). Such killing may have had a significant effect in local areas and may have caused animals to move away from certain rookeries and haulout sites. Amendments to the MMPA in 1988 and 1994, and sea lion specific regulations in 1990, increasingly prohibited fishermen from discharging firearms near Steller sea lions and likely reduced the impact from this threat substantially. However, some shooting, resulting in an unknown level of mortality, still occurs (NMFS 2001, Loughlin and York 2000, NRC 2003). While legal, Steller sea lions are hunted in Hokkaido, Japan to reduce damage to local fisheries, with an average of 631 animals killed per year during 1958-1993. Animals killed are mostly from the Kuril Islands and Sea of Okhotsk, and overwinter in Japanese waters.

Non-pups are the age-class most vulnerable to illegal shooting, the threat occurs at a low frequency, and there is a medium level of uncertainty associated with the evidence above. Thus, the relative impact on the recovery of the western DPS due to illegal shooting is ranked low, with a medium feasibility of mitigation.

8. Entanglement in marine debris – LOW

Observational studies cannot fully evaluate the potential for entanglement because many entangled animals may die at sea and thus not be seen. A study conducted in the Aleutian

Islands during June-July 1985 found that a very low percentage of observed sea lions were entangled in net or twine; none were entangled in packing bands. A follow-up study was conducted during November 1986 and researchers saw no entangled pups and only one entangled juvenile out of a total of 3,847 sea lions examined. Currently, there is incomplete information available on the number of animals lost at sea, and marine debris will likely continue to be common.

Juveniles are likely the age-class most vulnerable to entanglement in marine debris, the threat occurs at a medium frequency, and there is a medium level of uncertainty associated with the evidence above. Thus, the relative impact on the recovery of the western DPS of Steller sea lion due to entanglement in marine debris is ranked low, with a medium feasibility of mitigation.

9. Disease and parasites - LOW

Available serologic evidence does not support the possibility that a disease epidemic occurred during the sea lion decline of the late 1970s and 1980s; however, due to sampling limitations the possibility can not be excluded completely. Although sea lions have recently been exposed to several endemic disease agents that could potentially impede recovery, the only available data are the prevalence of antibodies to the disease agents, and the potential for those agents to cause disease among Steller sea lions has not been documented. The potential for parasitism to have a population level affect on sea lions is largely unknown. Although parasites may have little impact on otherwise healthy animals, effects could become significant if combined with other stresses. Available information does not suggest that the sea lion decline was caused by parasitic infections, although there has not been adequate research to assess the current relative nature and magnitude of parasitism in sea lion populations and to what exent this may have influenced reproductive capacity.

Adult females and pups are likely the age-classes most vulnerable to disease and parasitism, the threat occurs at a high frequency, and there is a medium level of uncertainty associated with the evidence above. Thus, the relative impact on the recovery of the western DPS of Steller sea lion due to disease and parasitism is ranked low, with a low feasibility of mitigation.

10. Disturbance from vessel traffic and tourism – LOW

The possible impacts of various types of disturbance on Steller sea lions have not been well studied, yet the response by sea lions to disturbance will likely depend on season and their stage in the reproductive cycle. Close approach by humans, boats, or aircraft can cause hauled out sea lions to go into the water, and the discharge of firearms at or near hauled out animals may have a particularly dramatic effect. Vessels that approach rookeries and haulouts at slow speed, in a manner that allows sea lions to observe the approach, should have less effect than vessels that appear suddenly and approach quickly. Sea lions may become accustomed to repeated slow vessel approaches, resulting in minimal response. Although low levels of occasional disturbance may have little long-term effect, areas subjected to repeated disturbance may be permanently abandoned. In comparison, when humans set foot on a rookery or haulout, the response by sea lions is typically much greater, often resulting in stampedes that may cause trampling or abandonment of pups. Repeated disturbances that result in abandonment or reduced use of rookeries by lactating females could negatively affect body condition and survival of pups through interruption of normal nursing cycles.

Pups are the age-class most vulnerable to disturbance from vessel traffic and tourism, the threat occurs at a medium frequency, and there is a medium level of uncertainty associated with the evidence above. Thus, the relative impact on the recovery of the western DPS of Steller sea lion due to disturbance from vessel traffic and tourism activities is ranked low, with a high feasibility of mitigation.

11. Disturbance due to research activities - LOW

Research activities result in disturbance but are closely monitored and evaluated in an attempt to minimize any impacts of research necessary to recover sea lions. Research activities result in the mortality of 1-3 sea lions annually, often due to accidental death during anesthesia or suffocation when animals are herded. However, the potential exists for additional unobserved mortality to occur following the completion of research activities.

Pups are the age-class most vulnerable to disturbance from research activities. The threat occurs at a medium frequency, and there is a low level of uncertainty associated with the evidence above. Thus, the relative impact of disturbance from research activities on the recovery of the western DPS of Steller sea lion is ranked as low, with a high feasibility of mitigation.

B. Synthesis and Discussion of Threats

Those factors identified as threats to the recovery of the western DPS of Steller sea lions are provided in Table IV-1. Both top-down and bottom-up threats can affect Steller sea lion population growth and vital rates. In addition, both types of threats can operate simultaneously and in different ways (i.e., direct or indirect). Steller sea lions have been affected by environmental variability (e.g., regime shifts), diseases, parasites, and predation for their entire existence, and humans have hunted them for food and for other uses for thousands of years. The impact of each of these factors has likely varied over time in response to marine ecosystem dynamics and predator abundance (e.g., killer whales and humans), as well as in response to the size of the sea lion population itself. Steller sea lions persisted in the North Pacific despite the adverse impact of these threats, and they did so without a significant loss of genetic diversity: there is evidence that previous population declines were not severe enough to cause a “genetic bottleneck”. Therefore, for tens of thousands of years prior to the 1970s, Steller sea lions had adapted to and accommodated fluctuations in their carrying capacity due to natural variability, disease and parasitism, killer whale predation, low levels of human-related kills, and apparently maintained, on average, a relatively large population size. Commerical hunting and changes in human technology and mobility changed the ability for direct kills by humans to have population-level effects. In the last several decades, several new threats have developed; i.e., contaminants, global climate change, and both top-down (e.g., incidental take) and bottom-up (reduced prey biomass and quality) effects of fisheries.

It is likely that both top-down and bottom-up threats affected sea lions to varying degrees and at different times to cause the sea lion population declines (e.g., Small and DeMaster, 2006, Atkinson et al. in press). Increases in both types of threats were necessary to account for the rapid rate of decline in the 1980s, which resulted in declines in juvenile survivorship, body size and birth rate. It is plausible that high rates of direct mortality from humans (e.g., legal and illegal shooting, incidental take, subsistence hunting) and mortality from killer whales were augmented by declines in carrying capacity associated with regime shifts, increased interspecific competition, and fishing. A reduction in the rate of population decline in the 1990s suggests that the effect of one or more threats also declined, possibly through density dependence. This coincides with the listing of Steller sea lions under the ESA and the prohibition on shooting at or near Steller sea lions, as well as a change in oceanographic conditions. Thus, there was likely a reduction in direct mortality from humans (top-down). The lower rate of population decline since about 1990 and the associated improvement in survivorship, but possible continued erosion in birth rate, suggest the sustained effect of bottom-up threats and a reduction in the magnitude of top-down threats. This is in contrast to the conclusions of the NRC (2003) which favored top-down controls as the primary factor. Much of the evidence considered here was not available to the NRC in 2002 when they were preparing their report.

In the threat assessment above, threats were delineated as either top-down or bottom-up to examine evidence for expected responses in sea lion vital rates. Trends in survivorship and natality can be used to infer the current relative magnitude of threats. For instance, evidence of a decline in survivorship, without evidence of a decline in body condition, growth, or health, would suggest that top-down threats such as predation or incidental catch in fisheries were the largest impediments to recovery. Alternatively, bottom-up threats may be implicated if reductions in body condition, health, fitness, or natality were observed. Bowen *et al.* (2001; their Table 1) considered the primary hypotheses explaining the sea lion decline and summarized the likely direction of change in sea lion response variables (e.g., birth mass, pup growth rate, foraging effort, body condition) under those hypotheses. The NRC (2003; their Table 6.2) modified the Bowen *et al.* table by (1) reducing the number of response variables, (2) assigning hypotheses as acting under either bottom-up or top-down forcing, and (3) adding the observed direction of the response variables to subsequently assess if the expected and observed responses matched. The expected direction of sea lion responses differed between the Bowen *et al.* and NRC tables for several threats, illustrating the uncertainty associated with the understanding of how sea lions respond under various hypotheses. Acknowledging that uncertainty, both tables show that sea lions are expected to exhibit the same response to several hypotheses. For example, under effects on food (prey) from fisheries and climate/regime shifts, sea lion birth mass, pup growth rate, and body condition will decrease, whereas foraging effort would increase. This particular response overlap is best explained by considering that both climate/regime shifts and fisheries would affect the prey available to sea lions, which could, in turn, affect reproductive success or failure, body condition or starvation, and morbidity.

The assessment of the threats to the recovery of the western DPS of Steller sea lion described above provides the basis for the recovery criteria and recovery actions in Section V. Certainly, our limited understanding of the threats and their dynamics within the environment where Steller sea lions live must be improved. Further, increased knowledge of how sea lions respond to threats is required, which is problematic due to the similar responses exhibited to several different threats. This response overlap makes it difficult to distinguish which threats have the greatest impact on sea lions and which are the largest impediments to recovery.

Table IV-1. Factors identified as threats to the recovery of the western DPS of Steller sea lion in the next 5 years (2006-2010), and the information used as evidence to rank the relative impact of those threats as High, Medium, or Low. The mechanism of each threat either top-down (threats that kill sea lions independent of the capacity of the environment to support them) or bottom-up (threats that affect the physical condition of sea lions due to the inability of their environment to support them); each of these can be either direct (directly reduces survivorship of individual sea lions) or indirect (indirectly reduces body condition and subsequently reproduction and survival) (see NRC 2003 for a detailed discussion of food webs and top-down/bottom-up control).

Threat	Mechanism	Most Vulnerable Age-Class	Frequency of Threat Occurring	Uncertainty	Relative Impact to Recovery	Feasibility of Mitigation
Environmental Variability	Bottom-up	Adult Females & Juveniles	High	High	Potentially High	Low
Competition with Fisheries	Bottom-up	Adult Females & Juveniles	High	High	Potentially High	High
Killer Whales	Top-down	Pups & Juveniles	High	High	Potentially High	Low
Toxic Substances	Top-down or Bottom-up	Adult Females & Pups	High	High	Medium	Medium
Incidental Take by Fisheries	Top-down	Juveniles	Medium	Medium	Low	Medium
Alaska Native Subsistence Harvest	Top-down	Adult & Juvenile Males	Medium	Low	Low	High
Illegal Shooting	Top-down	Non-pups	Low	Medium	Low	Medium
Entanglement in Marine Debris	Top-down	Juveniles	Medium	Medium	Low	Medium
Disease and Parasitism	Top-down or Bottom-up	Adult Females & Pups	High	Medium	Low	Low
Disturbance from Vessel Traffic and Tourism	Top-down or Bottom-up	Pups	Medium	Medium	Low	High

V. RECOVERY PLAN FOR THE WESTERN POPULATION

When a species is listed as threatened or endangered under the ESA, NMFS is required to develop and implement a recovery plan for the conservation and survival of that species. The three specific statutory requirements, set forth in section 4(f)(1)(B) of the ESA, are that each plan incorporates the following:

1. A description of site-specific management actions necessary to achieve the plan's goal for the conservation and survival of the species;
2. Objective, measurable criteria which, when met, would result in a determination, in accordance with the provisions of this section, that the species be removed from the list; and
3. Estimates of the time required and cost to carry out those measures needed to achieve the plan's goal and to achieve intermediate steps toward that goal.

These statutory requirements, along with additional guidelines regarding the prioritization of actions, and a strategy for recovery are discussed below.

A. Definition of Recovery

The ESA does not specifically define "recover", and the term "recovery" is used only with respect to recovery plans "...for the conservation and survival..." of listed species. According to NMFS policy (U.S. Fish and Wildlife Service and NMFS 1990), recovery is "the process by which the decline of an endangered or threatened species is arrested or reversed, and threats to its survival are neutralized, so that its long-term survival in nature can be ensured. The goal of this process is the maintenance of secure, self-sustaining wild populations of species with the minimum necessary investment of resources." The ESA's implementing regulations (50 CFR § 402.02) further define recovery as "improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in section 4(a)(1) of the Act."

The definition provided for recovery in the implementing regulations and the definition provided for conserve in the ESA have essentially the same meaning. Section 3(3) of the ESA states: "The terms "conserve," "conserving," and "conservation" mean to use and the use of all methods and procedures which are necessary to bring any endangered species or threatened species to the point at which the measures provided pursuant to this Act are no longer necessary." Hence, recovery and conserve both mean to bring a species to the point at which it no longer needs the protection of the ESA, because the species is no longer in danger of extinction throughout all or a significant portion of its range. This definition of recovery falls far short of requiring that a species must be restored to its historic range and abundance before it can be considered recovered or delisted. It also falls short of requiring the restoration of a species to all the remaining suitable habitat, unless this is necessary to sufficiently reduce the species' susceptibility to threats to a level at which the species is no longer threatened or endangered.

The phrase "throughout all or a significant portion of its range" (SPOIR) is used in both definitions of endangered and threatened. Neither "significant" nor "range" are defined in the ESA or implementing regulations. Hence, the ESA provides NMFS with latitude to use its discretion, based on the best scientific information available, to develop recovery goals and implement recovery plans designed to conserve and recover species. The ESA clearly does not use the term significant in a statistical sense. Significance cannot be reliably and safely applied in any strictly quantitative framework, because of the great variety of organisms, habitats, and threats that must be evaluated for protection under the ESA.

Given that the ESA is intended to avoid species extinction, we avoid the pitfalls of a purely quantitative approach by instead viewing "significant" in the context of a species' long term survival needs. The term becomes logical, meaningful, and useful if applied in this context. A significant portion of the range is that area that is important or necessary for maintaining a viable, self-sustaining, and evolving population or populations, in order for a taxon to persist into the foreseeable future. That "significant portion" may constitute a large portion of the historic range of a species or a relatively small portion of the historic range. Other parts of a species' range (regardless of whether it is historical, current, or potential range) may not be significant to its long-term survival, regardless of its geographic extent. Therefore, a species extirpated from such areas does not necessarily mean it is threatened or endangered, regardless of the geographic extent of those areas.

Implicit in the ESA definitions of threatened and endangered and in the principles of conservation biology is the need to consider genetics, demographics, population redundancy, and threats (as identified by the listing factors). The ESA is mandated to recover species to the point that they are "not likely" to be in danger of extinction for the foreseeable future throughout all or a significant portion of their range. NMFS believes that the "not likely" standard represents a minimum threshold of risk, and that recovery should also involve maintenance of multiple widespread populations that are independently viable because it is less likely that future singular threats will endanger widely separated multiple populations than a single population with the same abundance. Viable populations have sufficient numbers of individuals to counter the effects of deleterious gene mutations as a result of inbreeding, and to counter the effects of deaths exceeding births and recruitment failure for periods of time. Thus, the conservation biology principle of redundancy is satisfied by the required multiple genetically and demographically viable, self-sustaining populations. Furthermore, the principle of resiliency is satisfied with sufficiently large populations to persist through normal population variations, as well as through unexpected catastrophic events.

The principles of recovery and conservation as defined in the ESA, implementing regulations, and NMFS policy demonstrate the strong relationship between the delisting criteria used for recovery and the five listing factors in Section 4(a)(1) of the ESA. These five listing factors must be addressed in any reclassification of a species (ESA Section 4(c)(2)(B)), and are:

> *"(A) The present or threatened destruction, modification, or curtailment of its habitat or range;*
> *(B) overutilization for commercial, recreational, scientific, or educational purposes;*

> *(C) disease or predation;*
> *(D) the inadequacy of existing regulatory mechanisms; and*
> *(E) other natural or anthropogenic factors affecting its continued existence."*

Recovery is based on reduction or removal of threats and improvement of the status of a species during the period in which it is listed, and not just from the time a listed species is proposed for reclassification. Environmental conditions and the structure of populations change over time, and threats recognized at listing or in subsequent recovery plans may no longer be directly applicable when reclassification is considered. Management actions and tasks conducted by recovery or conservation programs for listed species are expected to minimize or remove threats and improve the species' status.

When delisting a species, NMFS must determine that the five listing factors (recovery factors) no longer apply, e.g., the habitat is no longer threatened with destruction or modification, the current abundance and range is adequate, and the habitat needed to sustain recovered populations is present. Therefore, the recovery goals include management actions and tasks, presented by "recovery factor", as well as downlisting and delisting criteria. These recovery factors were derived from the five listing factors and state the conditions under which threats are minimized or removed.

Recovery is achieved when management actions and associated tasks have been implemented and/or completed to allow genetically and demographically viable, self-sustaining populations to thrive under minimal ongoing management and investment of resources. Achievement of recovery does not mandate returning a species to all or a significant portion of its historic range, nor does it mandate establishing populations in all possible habitats, or everywhere the species can be established or reestablished. Removing a species from protection of the ESA remands the primary management responsibility of that species to the states and to applicable federal agencies acting under the authority of other federal laws (e.g., MMPA), who may choose to further expand its range and populations.

B. Goals

The ultimate goal of this recovery plan is to promote the recovery of the western DPS (and the eastern DPS) of Steller sea lion, and its ecosystem, to a level sufficient to warrant its removal from the federal List of Endangered and Threatened Wildlife and Plants (List) under the ESA. The intermediate goal is to reclassify the wDPS from endangered to threatened.

C. Recovery Strategy

The worldwide population of Steller sea lions was listed as threatened under the ESA in 1990 after severe declines in the 1980s. The first Recovery Plan was published in 1992 and contained 61 discrete recovery actions. In 1997, after continued declines in the core of the range, the western DPS of Steller sea lion was reclassified as endangered. Several factors were identified in the 1997 listing document that could potentially have a negative impact on the western DPS of Steller sea lion, including changes in the availability or quality of prey (due to commercial fisheries or natural phenomena), disturbance, illegal shooting, incidental take in fisheries, subsistence harvest, and contaminants.

Beginning in the 1990s and continuing extensively in the 2000s, a research program has been conducted on Steller sea lions, a species that presents extensive logistical challenges in often severe environmental conditions. Despite these difficulties, these efforts have achieved significant advances in understanding sea lion biology and the factors that influence their population dynamics. Further, knowledge of the complex ecosystem of the North Pacific Ocean has provided insight into the interactions between sea lions and their environment.

By the early 1990s, a series of management actions were successful in reducing the previously high levels of incidental take in commercial fisheries and shooting (legal and illegal) to negligible levels. Since then, conservation efforts have focused on reducing the potential impact of commercial fisheries on sea lion prey through the implementation of fishery management actions. Additional conservation measures have been implemented to reduce the impact of disturbance and subsistence harvest on sea lions.

The documented variation in the rate of population decline across decades and among regions within the western DPS demonstrates the need to employ a recovery strategy that accounts for spatial and temporal differences. These spatial and temporal differences in population dynamics also suggest that the causes of the decline did not act uniformly across the range. A substantial effort has been devoted toward understanding the causes of the decline and some potential does exist to gain some insight into those factors. However, the recovery strategy must focus on those factors that are currently impeding recovery of Steller sea lions and the actions necessary to promote recovery.

As described in the threats assessment (Section IV; Table IV-1), substantial uncertainty affects our ability to determine the relative impact of the primary threats to sea lion recovery or how to mitigate them. Unfortunately, fishery management actions which were implemented to reduce competitive interactions between fisheries and sea lions did not include a monitoring program to assess the efficacy of those measures. Because of this, it is impossible to answer the efficacy questions raised about these measures. Although the conservation measures have been in place for a substantial amount of time and the trend of the western DPS appears to have changed, we cannot infer whether those measures resulted in those population changes or were just coincidental. Thus, a reasonable approach to recovery is to implement the following:

Continue population monitoring and research on the key threats potentially impeding sea lion recovery (Action 1.1.1; Priority 1)

Estimates of population abundance (the only priority 1 action), trend, distribution, health, and essential habitat characteristics are fundamental to

Steller sea lion management and recovery. Further, current information on the primary threats is insufficient to assess their impact on recovery. Focused research is needed on how these threats impact sea lion population growth (especially reproduction) and how they may be mitigated in order to facilitate recovery. In addition to studies on individual threats, the dynamics between threats needs to be better understood to assess the cumulative effects on sea lions.

Maintain current or equivalent fishery conservation measures until change is warranted (Action 2.6.6; Priority 2a)

After a long term decline, the western DPS may be stabilizing. The first slowing of the decline began in the 1990s, suggesting that the management measures implemented in the early 1990s may have been effective in reducing anthropogenic effects (e.g., shooting, harassment, and incidental take). The apparent population stability observed in the last 6 years is correlated with comprehensive fishery management measures implemented since the late 1990s. The current suite of management actions (or their equivalent protection) should be maintained until substantive evidence demonstrates that these measures can be reduced without limiting recovery.

Design and implement an adaptive management program to evaluate fishery conservation measures (Action 2.6.8)

Due to the uncertainty in how fisheries affect Steller sea lions and their habitat, and the difficulty in extrapolating from individual scientific experiments, a properly designed adaptive management program should be explored and implemented if feasible. This type of program has the potential to assess the relative impact of commercial fisheries and to better distinguish the impacts of other threats (including killer whale predation). This program will require a robust experimental design with replication at the proper temporal and spatial scales with the appropriate levels of commercial fishing as experimental treatments. It will be a challenge to construct an adaptive management plan that meets the requirements of the ESA, is statistically sufficient, and can be implemented by the commercial fisheries. Acknowledging these hurdles, we must make a significant effort to determine the feasibility of such a program. Other options to test the efficacy of management measures are possible and should be continued (e.g., modeling using existing data from current management measures), but none offer the robustness and flexibility of an adaptive management program aimed at testing the effectiveness of fishery conservation measures.

Develop an implementation plan (Action 1.5)

An implementation plan will be developed that includes a comprehensive ecological and conceptual framework that integrates and further prioritizes the numerous recovery actions provided in this plan. The implementation plan will contain a synthesis of and establish priorities among the individual actions, coordinate their implementation in a cohesive strategy. Several components will be integrated in the conceptual framework of the implementation plan: (1) the complex dynamics of the North Pacific marine ecosystem, (2) multiple causation in those systems, (3) the need for long-term research, (4) the monitoring required to assess the effectiveness of management regulations, and (5) the development of a modeling approach that examines possibly effects of multiple threats on sea lion population dynamics to evaluate the strength of the evidence for different hypotheses.

The Recovery Criteria and Recovery Actions in the following sections describe how the Recovery Strategy should be implemented to reduce the uncertainty associated with the relative impact of both the natural and anthropogenic threats to Steller sea lions and their marine ecosystem. Information gained from these actions will increase the effectiveness of management measures implemented to recover Steller sea lions.

D. Development of Recovery Criteria

The primary intent of the ESA is to recover listed species and the ecosystems on which they depend, such that the protections of the ESA are no longer necessary (see ESA Section 2(b)). To determine when we have reached this point, the ESA requires that recovery plans, to the maximum extent practicable, incorporate objective, measurable criteria which, when met, would result in a determination that the species be removed from the federal List of Endangered and Threatened Wildlife and Plants (List) (50 CFR 17.11 and 17.12)[11]. The recovery criteria form the basis from which to gauge a species' risk of extinction and comprise the core standards upon which a decision to delist will be based.

Recovery criteria must include measures of demographic health (biological criteria) as well as measures that indicate the elimination of threats to the species (i.e., recovery factor criteria). Both the biological and recovery factor criteria must be met for the western DPS of Steller sea lion to be reclassified or delisted. The biological criteria require evidence that the population status has improved in response to the reduction of threats, while the recovery factor criteria require evidence that the threats have been eliminated or controlled and are not likely to recur. Any new factors or threats identified since listing must also be addressed in this analysis to ensure that the species no longer requires protection as an endangered species.

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[11] In accordance with the provisions of the ESA.

The ESA requires that two sets of criteria be developed for each of the recovery goals for the Steller sea lion:

- To reclassify the western DPS of Steller sea lion as threatened, NMFS must determine that the species' abundance, survival, and distribution, taken together with the five potential sources of threats (recovery factors), no longer render the species "in danger of extinction throughout all or a significant portion of its range."

- To remove the western DPS of Steller sea lion from the List, NMFS must determine that the species is not likely to "become an endangered species in the foreseeable future throughout all or a significant portion of its range."

1. Development of the Biological Recovery Criteria

Recovery criteria have traditionally been derived from a weight of evidence approach without the use of population viability analyses. This approach requires a thorough review and synthesis of all the available biological and ecological information regarding the species (or surrogate species), and the determination of essential demographic parameters (e.g., population abundance and trend, birth rates, age ratios, etc.) and other factors (e.g., habitat needs) that would reflect that the species is no longer at high risk of extinction.

However, in part because of the paucity of available life history data, the Team began its development of recovery criteria by exploring a population viability analysis in an attempt to derive a quantitative biological recovery criterion for the western DPS based on explicit choices for risk and the incorporation of uncertainty (see Appendix). The PVA proved very helpful in determining the factors important to consider when estimating sea lion extinction risk. However, due to the limitations of the model and the assumptions that went into it (e.g., lack of available data on meta-population structure, dispersal rates, density dependence, historic human-related mortality, etc.), the Team ultimately chose a weight of evidence approach which incorporated knowledge gained from the PVA. In essence, the PVA acted as a sensitivity analysis which tested the Team's assumptions and provided feedback on the risk associated with various choices (e.g., time to recovery, likely impacts of previous human-related threats, role of density dependence, potential for unexplained threats to recur, level of uncertainty, etc.).

This discussion provides an overview of the PVA development process, a critique of the PVA and its shortfalls, a description of the weight of evidence approach, and a description of how the PVA results were integrated into the criteria. It is expected that the PVA developed for this recovery plan will be modified and improved from forthcoming research such that criteria could be modified in future revisions to reflect explicit risk choices as developed by a PVA.

2. Extinction risk and population viability

All populations face varying probabilities of extinction due to stochastic events that affect survival and productivity. At given average rates of survival and productivity, and variability around these averages, large populations have lower probabilities of extinction than small ones. Population viability analysis is a form of risk analysis applied to the issue of population extinction. It is a structured and systematic analysis of the interacting factors, including abundance, rates of survival and productivity, demographic and environmental stochasticity, and catastrophes, that determine a population's risk of extinction. In recent years, PVAs have been used as tools in establishing recovery goals for threatened and endangered species such as the northern spotted owl, the desert tortoise, and Atlantic Coast piping plover. Each PVA is individually developed for a target population or species, and consequently, each PVA is unique. The larger goal in mind when conducting a PVA is to ensure that the population of a species is self-sustaining over the long term.

Population viability analysis is likely to be an important tool in developing recovery criteria. The use of PVA in developing demographic criteria has been discussed in peer reviewed literature (Goodman 2002, Reed *et al.* 2002, Clark *et al.* 2002). Although recovery criteria are required under the ESA to be both measurable and objective, the ESA does not provide an explicit standard for those criteria beyond general descriptions (see above). PVA offers the potential to standardize criteria across species by integrating explicit risk choices into the PVA and the results (Goodman 2002). The selection of the extinction risk is a policy decision based on the acceptance of risk; there is no accepted agency policy regarding extinction risk choices. Substantial efforts have recently been directed towards developing rationale for determining appropriate extinction standards. Specifically, NMFS conducted a workshop to consider recovery criteria for large whales (Angliss *et al.* 2002) and a NMFS Quantitative Working Group has proposed guidelines on ESA listing criteria (DeMaster *et al.* 2004). A probability (chance) of extinction over some period of time, i.e., 1% probability of extinction in 100 years, was recommended as the quantitative standard for a species to be considered in high risk of extinction (i.e., endangered) (Angliss *et al.* 2002, DeMaster *et al.* 2004).

As discussed in Chapter I.G.3, several demographic models have been used to explore the population dynamics and persistence (i.e., extinction risk) of Steller sea lions (York *et al.* 1996, Gerber and VanBlaricom 2001, Winship and Trites 2006). There is some degree of consistency between the predictions of all three PVA models (York *et al.* 1996, Gerber and VanBlaricom 2001, Winship and Trites 2006) due in large part to their use of some of the same base population data and to the fundamental assumption of all PVA models that populations will continue to behave as they have in the past after correction for factors that will be different in the future. As such, sea lion populations (i.e., individual rookeries, clusters of rookeries, or the entire western DPS) that declined at fast rates were predicted to go extinct sooner than populations that had declined slowly. Results from the three PVAs conducted indicate that the western DPS has a high probability of declining to a low level if they are considered as a single homogeneous population (by combining all rookery counts and assuming an overarching population trend). However, the prognosis for the species is considerably more optimistic if each of the 33 rookeries is considered as a distinct, independent population with its own probability of persistence, and assuming that differing environmental factors around the respective rookeries remain stationary for the long term (as opposed to the possibility of rolling declines). Under this scenario, PVA models at a spatial scale smaller than the DPS predict that many rookeries will be extirpated, but that the species will persist on the time frame considered, especially if density dependence is assumed to play a positive role.

3. Development of the PVA and results

Each model has its limitations and arguable assumptions. Although previous PVA results were helpful in understanding the viability of the western DPS, the Team decided that a more focused PVA was necessary to evaluate extinction risk and to provide specific recovery criteria. Unlike previous exercises, this PVA (Appendix) is unique because it was developed specifically for the Team based on the explicit guidance and input from the Team with the goal of generating recovery criteria. To accomplish this, decisions were made on important input parameters that were not obligatory in other models. Differences between this PVA and other models is discussed below in Chapter IV.C.4.

Although neither NMFS nor the FWS have adopted specific guidelines for criteria, the recommendations from Angliss *et al.* (2002) and DeMaster *et al.* (2004) were used as the starting point for discussion in the development of the PVA[12] (see Appendix). This is a brief overview of the quantitative approach using a PVA which employed a specific probability of extinction.

The general principles that the Team used were:

- A probabilistic threshold is appropriate to describe the risk of extinction in the criteria;
- A long lived species shall no longer be considered endangered when, given current and projected conditions, the probability of quasi extinction is less than 1% in 100 years;
- A long lived species shall no longer be considered threatened when, given current and projected conditions, the probability of becoming endangered is less than 10% in 20 years;
- A minimum viable population (MVP) approach will be used; MVP is defined as a population that is sufficiently abundant and well adapted to its environment for long-term persistence without significant artificial demographic or genetic manipulations[13].
- A quasi-extinction level using a "genetic effective population size" (N_e) of 1,000 was selected to maintain long term genetic viability[14] (equates to a total

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[12] The PVA was developed by Dr. Dan Goodman (Montanna State University) under contract from NMFS.

[13] Meffe and Carroll (1994) define an MVP as "*the smallest isolated population size that has a specified percent chance of remaining extant for a specified period of time in the face of foreseeable demographic, genetic, and environemental stochasticities, plus natural catastrophes.*"

[14] N_e is the number of individuals contributing genes to the next generation (Wright 1931). An N_e of 50 adults avoids inbreeding depression in the short-term; an N_e of 500 is needed to avoid serious long-term genetic drift; an N_e of 1,000 provides a conservative estimate beyond which significant additional genetic variation is not expected (Allendorf and Ryman 2002).

population size of 4,743 Steller sea lions because of their polygynous, harem-based reproductive strategy);

- Current threats to the species, as well as those that brought the species to the point of listing, must be addressed in the recovery factor criteria;

The underlying model (see Appendix for details on model structure) uses a Bayesian framework that allows for the specification of uncertain inputs as distributions, quantifies uncertainty associated with the estimation mechanism itself, and tracks the propagation of uncertainty through all the steps of the estimation and prediction. The model estimates intrinsic mean growth rates and associated variance for a number of historic time periods, and then probabilistically predicts future population trajectories.

A critical assumption in the PVA is that the recorded history (roughly the last 50 years) of the western DPS is a combined result of natural variation and extraneous influences (i.e., incidental mortality to fisheries, illegal shooting and subsistence harvest, and reduced prey biomass and quality from fisheries). Based on available sea lion population counts, five time periods were selected to represent the population trajectory; 1958-77, 1977-85, 1985-1989, 1989-2000, and 2000-2004. However, these individual population trajectories are affected by human related impacts (e.g., shooting, harvest, fisheries). Because successful management measures have been implemented in the recent past, it is inappropriate to predict future population trajectories based on historical conditions which are likely to be pessimistic (i.e., some of the previous sea lion mortality is unlikely to occur in the future due to the implementation of management measures such as a prohibition on shooting). Thus, the Team estimated the mitigated mortality attributed to some of the extraneous influences using best available empirical evidence and expert opinion.

In the model, the population trajectory of previous time periods was modified to reflect the mitigation measures currently in place. This was done for the four time periods prior to the current time period (2000-2004) which was not modified because this represents the base case of the current suite of mitigation measures. It was assumed that any human-related impacts that occurred in the 2000-2004 time period would continue into the future – and would be the case unless new mitigation measures were implemented immediately to further reduce human-related threats (e.g., larger fishing closures, reduced incidental take, reduction in disturbance). Therefore, the future population projections were based on the recorded sea lion population trend over the last 50 years, as modified by the effect of current mitigation measures. For example, we know that incidental take was a substantial mortality factor during the steep declines in the 1985-1989 time period. Based on current fishery management measures and restrictions on take, it is extremely unlikely that these high levels of take would occur again in the future. Thus, the trajectory in the model for the 1985-89 period was modified to remove the amount of incidental take that the Team considered extremely unlikely to occur again. The mortality that remained during the 1985-89 period (still a substantial rate of decline) was due to other factors that the Team could not identify with any quantitative assessment. The effect of making these changes is to decrease the future extinction risk, and decrease the potential time to recovery (e.g., to reach the MVP as determined by extinction risk choices).

Complications peculiar to the western DPS presented substantial challenges in fitting the PVA. First, the periods of sustained (1970s-2000s) and rapid (1985-89) population declines cannot be fully explained[15]. Second, the North Pacific ecosystem is subject to large-scale, natural, physical and biological variation that is likely to affect long-term (on the order of many decades) sea lion population dynamics (Trites et al. 2007b). However, this may not be consistent among stocks as regime-like forcing is not evident in the recent (30+ year) history of the eastern DPS. Third, human influences on Steller sea lion populations in particular and on the North Pacific ecosystem in general have been substantial; Steller sea lions have been hunted by humans for thousands of years, and the region supports some of the nation's largest fisheries, which developed at the same time the sea lion population was declining. If natural changes (point two) were the driver of the large declines of the 1980s, then not considering their potential impacts in future scenarios would be highly risk-prone. Thus, modeling results of Winship and Trites (2006) which assumed a constant carrying capacity into the future and a density-dependent response by sea lions, are at odds with hypotheses posed by Trites *et al.* (2007b) who posited that environmental changes in the 1970s caused sea lions to decline. On the other hand, if anthropogenic adverse impacts from factors which are currently minimal, such as shooting and incidental catch, were major contributors to the 1980s declines, then it is appropriate to disregard these in future scenarios if we believe that these activities have been, and will continue to be, mitigated.

The estimated magnitude of the extraneous factors (mitigation) did not account for much of the period of rapid decline in the late 1980s (see Appendix for modified trajectories). Not unexpectedly, the initial model projections resulted in a relatively high probability of extinction in the next 100 years. Acknowledging the possibility that factors beyond the extraneous influences may have affected sea lion population dynamics, alternative scenarios and factors were then considered[16]. For example, there is the chance that density-dependence will act at some point to slow the decline and substantially reduce the likelihood of extinction. These alternative scenarios and factors represent how the cumulative and synergistic impact of various factors could have caused the past population decline or could act to avoid extinction. These alternative factors are plausible, yet the available evidence did not allow the Team to determine their likelihood. Thus, based on the expert opinion of the Team, a probability of 0.2 (it must be between 0 and 1) was assigned to *all* the alternative factors. In essence, this probability represents the likelihood that the assumptions of the PVA are incorrect in such a way as to exaggerate the likelihood of extinction. The 0.2 probability was factored into the model results to determine extinction risk; this functionally acted to reduce the likelihood of extinction over a given time period. This approach allowed the team to integrate uncertainty into the model relating to the potential impacts of density-dependence and other alternative hypotheses for the decline.

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[15] Winship and Trites (2006) dismiss the rapid declines of 85-89 without explanation, and model future projections on the more optimistic vital rates observed in the 1990s.

[16] Alternative scenarios included killer whale predation as a driving force, fisheries, ecosystem change, and a multi-threat scenario.

Once the model was completed, the Team tested the PVA using the current population size as a starting point. Perhaps NMFS might be able to downlist to threatened if the current population was to stabilize; this seemed an appropriate starting point to test. It was obvious in the Team's first few runs that the current population level would, even if stable in the near future, have a substantial probability of extinction within the next 100 years. Thus, it was clear to the Team that the western DPS must increase to a substantially larger population size to avoid significant extinction risk.

Knowing that the population needed to increase, the Team was then asked to specify a rate of increase (I) in which to make predictions, such that after a period of X years in which the population increased at I and reached Y population size, the risk of extinction in 100 years would be reduced to 1% (this would define the line between being listed as endangered and threatened). Given that the western DPS had increased at 2.813% between 2000 and 2004, it seemed appropriate to test the assumption that this increase would continue (assuming current mitigation measures remained in place). A secondary consideration is the fact that the eastern DPS had maintained an extended 3% increase over nearly 30 years; this does not mean that the western DPS must have this increase but the evidence suggests that a 3% increase is achievable (observed both in the western and eastern DPSs). This scenario provided two important components to the model; first, it increased the population size such that it would buffer future losses, and second, it added additional time periods (positive growth rates) to the model in which to draw from in future projections – this added time periods of population growth to a relatively pessimistic historic distribution (in which the model would randomly draw). The model could have simply been started from 2004 with a higher population size, but without the additional time periods of positive growth the outcome would be a much higher risk of extinction (i.e., because the model would randomly draw from a series of more pessimistic choices).

Model projections using the modified trajectories (mitigation measures), a 3% increasing population, and the alternative scenarios (probability of 0.2) still resulted in a high probability of extinction. For a hypothesized future assessment in 2034 after 3 decades of 3% annual growth to a population size of 110,434 sea lions, the probability of quasi-extinction within 100 years is 9.71%. Review of the PVA revealed that the probability of extinction was most sensitive to the period of rapid population decline (about 15% per year) observed in the 1985 to 1989 period. Considering the possibility that the 1985-1989 period of decline was unlikely to occur in the future, another model run was completed with the 1985-89 time interval removed from the time series (similar to the assumption of Winship and Trites 2006). In those projections, after 3% growth until 2024 (20 years), the western DPS would have a 1.79% probability of extinction in 100 years. This is still a higher risk than was chosen as an appropriate level. Thus, it would take the western DPS between 20 to 30 years of 3% growth to recover, and considerably longer if conditions similar to 1985-89 had even a small probability of reoccurrence.

Two questions raised during the public review were: Why did the Team choose a 3% increase for the PVA? and What if the population doesn't attain this growth rate? First, the model predicts that a population that performs less well, e.g., increases at only 1%, would have to sustain that increase for a much longer time period before reducing the risk of extinction to the 1% chance in 100 years. Although the assumed future growth rate can be changed to any reasonable value, we cannot predict what the exact growth rate will be. However, judging from the eastern DPS, western sea lions appear to be physiologically capable of a 3% increase in a similar environment, and can sustain this growth for long time periods. Given that a 3% increase has been observed in the western DPS since 2000, it is a reasonable scenario to consider. This does not mean that the only possibility for downlisting or delisting would require this level of population growth, but that if the population grew at a slower rate, then the time period over which this increase was observed would have to be substantially longer in order to change the listing status[17]. Ultimately, the use of a PVA in developing criteria must be an iterative process to test expectations with reality, then to re-evaluate the criteria in future plans (Boyd 2006). For example, if this population began to increase near the theoretical maximum of 8-12% per year, the extinction risk would decline much quicker, and new delisting criteria could be considered with a shorter recovery period (and higher growth rate). The extinction risk and uncertainty would be equal, but growth rate would be higher and time to recovery would be less. Since we cannot consider infinite possibilities for growth rates, the Team chose an objective, measurable rate that had been observed in the eastern DPS, for a short time period by the western DPS, and is likely achievable within the recovery program.

4. Limitations of the PVA

The Team reviewed the overall PVA structure, assumptions, and parameter values, and decided not to develop criteria based exclusively on the model. Numerous limitations described above, and issues pointed out during public and peer review cast doubt on the utility of the PVA alone in determining specific criteria. Specific concerns included the following:

- The uncertainty associated with the estimates of historic extraneous mortality from some sources is high. While there were data on mortalities associated with fisheries incidental catch and historical pup harvests, there were little or no consistent data available for shooting and the competitive effects of fisheries. This could have a large influence on the results (extinction risk).
- Discounting the 1985-1989 time period as a catastrophe that is unlikely to be repeated is inconsistent with a precautionary approach. There is no evidence to support removing this time period from the PVA (beyond the extraneous mortality adjustments made for actions that were unlikely to be repeated). Although Winship and Trites (2006) excluded this time period, they provided no rationale.
- Density-dependence was not explicitly considered in the PVA. The uncertainty in the strength of a density-dependent response by sea lions is an important

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[17] The ESA requires NMFS to develop criteria, that when met, would result in an action to delist or downlist. This does not mean that other scenarios are not possible or that they would not be potential points for agency action. Unlimited scenarios exist in which the population could achieve a high probability of persistence (e.g., 8% population increase for the next 15 years). The important consideration now is that we choose a plausible scenario for the criteria and update those criteria as we learn more about the species recovery. For a species such as sea lions, with multiple threats, substantial declines, and a relatively long recovery period, the process will be iterative.

consideration. Although the alternative scenarios provide for a potential density dependence response, there is no evidence that such a response has occurred for sea lions, and the scientific literature provides little evidence to support that assumption. On the other hand, this could potentially be a factor and recent leveling off of the population decline may represent some sort of density dependent response.

- The PVA was based upon a single population and did not consider sub-population/meta-population dynamics which could be an important influence on persistence (e.g., Winship and Trites 2006). The PVA does not capture the more complex sub-area population changes observed in the 1980s and 1990s.
- The PVA considered a relatively short time period which primarily consisted of declining trends. Thus, the results of the PVA (e.g., prediction of high extinction risk) are unremarkable.
- The PVA did not consider options for time periods, auto-correlation between decline rates of different time periods, different splits in historic time periods, or weighting of periods.
- The PVA used a conservative effective population size (N_e) as the quasi-extinction value which equates to a relatively large corresponding total population size (4,743 animals). There is no guarantee that a population of sea lions reduced to the quasi-extinction value would be doomed to extinction. The estimate is generated by a recognized requirement for genetic diversity rather than a target population. Other mammals have persisted at numbers below this amount, but may have been exposed to long-term genetic difficulties. Thus, examples of short-term survival may not be appropriate when considering long-term extinction risk.
- The ability of the sea lion population to recover from historic declines may be compromised due to disproportionate loss of highly adapted individuals (e.g., productive females killed on or near rookeries); these types of issues were not considered in the PVA, and thus time to recovery may be longer (or different) than expected.

Although each of these issues might in themselves make the PVA more or less conservative, the end result is to reduce our confidence that the criteria would represent a probabilistic description of extinction risk (e.g., 1% in 100 years), which was the intent of the exercise. However, the PVA did act as a sensitivity analysis; it provided insight into the important issues for the Team to consider, and helped develop the weight of evidence approach for both the demographic and recovery factor criteria.

5. Weight of evidence approach to the criteria

The recovery criteria were developed with the consideration of the following issues:

Reasons for listing: Steller sea lions were originally listed in 1990 due to a sharp rate of decline, especially in the mid to late-1980s when annual declines reached 16%. In 1997, when the species was split into an eastern and western DPS, the western DPS was uplisted to endanger due to a continued decline and the lack of evidence that the threats to the species had been substantially reduced or eliminated. The population was not listed due to low numbers, but the expectation that continued declines over a relatively long time period would, within the foreseeable future (e.g., 100 years), result in a high risk of extinction[18]. Thus, to eliminate the reasons for listing the population should cease to decline, and increase for some period to show that it can be self-sustaining. This in turn would indicate that whatever threat(s) to the population or conditions that caused the decline had been removed, mitigated, or changed.

Currently, there is the possibility that the population could decline at 16% per year, as it did from 1985-89, at some point in the future; this is a very important consideration when determining criteria. Certainly other species throughout the country are endangered and are at extremely low numbers (e.g. Northern Right Whale). Steller sea lions are not in this category; although their numbers are relatively high (compared to species such as Colorado Pikeminnow, Florida panther, wolves) their risk of extinction is still high due to long periods of decline and a lack of understanding of the threats. If NMFS had waited to list sea lions until they reached 1,000 animals in the western DPS, the prospects for recovery may have been virtually zero. If in 1997, NMFS had been able to adequately describe the threats and show that they had been removed, NMFS might have avoided uplisting the western DPS to endangered. The combination of knowledge of the threats, how they are acting, assurance that the threats have been ameliorated, and the past trajectories of the population have greater influence on likelihood of extinction and listing decisions than the minimum estimate of current abundance. A review of recovery plans around the country will show a wide variety of views on recovery criteria and total population numbers needed to de-list; this is based primarily on the types of threats, habitat, and potential for recovery. For example, a population of Colorado Pikeminnow may be recovered and relatively viable at a few thousand adults whereas the western DPS of Steller sea lions needs to be much larger to have a similar likelihood of persistence.

Population growth: In developing the recovery plan, it became evident that the eastern DPS has been recovering for nearly 30 years, increasing at about 3% per year throughout much of its range (see Section VII). In effect, the response of the eastern DPS to large reductions in population numbers during the mid-1900s and the subsequent recovery over the last 30 years has provided a possible recovery scenario for the western DPS. Population trends in the eastern DPS indicate that Steller sea lions in the North Pacific can recover from relatively low numbers at a rate of 3% per year and sustain this for many years. Between 2000 and 2004, the western DPS also increased at about 3% per year. Therefore a reasonable recovery scenario for the western DPS is a 3% population increase over some time period. Given that the western DPS declined for many years, a leveling off in the trend would represent a substantial reduction in extinction risk. Thus, the Team developed biological criteria while considering the reasons for listing (primarily the relatively high rate of decline) and the most likely scenario for recovery (an annual population increase of 3%). This does not eliminate the possibility that other scenarios might in fact occur - those scenarios will of course be reviewed by NMFS and considered at that time.

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[18] Although some people now question this decision, modeling efforts at the time predicted a high extinction risk. The ESA doesn't provide thresholds for numbers of individuals; rather it requires the consideration of risk over relatively long time periods.

The PVA results indicated that the population, even if stable over a few decades, would still have a high likelihood of extinction due largely to uncertainty in what caused the decline. To reach a substantially lower level of extinction risk this population needs to grow over several decades to insure that the key threats to the species recovery have been addressed.

Time to recovery: In general, recovery criteria deal with an expected performance of a population over a specified period of time, and the reduction of threats which are likely to be limiting recovery either now or potentially in the future. If we start with a possible recovery scenario of an increasing population at about 3% per year, we can then calculate what the appropriate time period may be. The PVA was an important source for determining what time periods would be necessary. Depending on the assumptions made it would take approximately 30 years of a robust recovery (3%) to reduce the extinction risk to approximately the 1% chance in 100 years. Coincidentally, IUCN criteria for listing endangered species considers population declines of 50% over 3 generations. For sea lions, a generation (i.e., mean age of reproducing females) is about 10 years. Thus, 30 years represents three generations, and should span a period of time sufficient to see environmental changes (e.g., oceanographic regime shifts). The environmental change component was important to the team when considering recovery periods and delisting. Because natural environmental change is listed as a potentially high threat to recovery, the recovery period needed to be long enough to insure the population could sustain itself in at least two different regimes. Population growth for 15 years would reflect sustained growth by two generations of sea lions during two environmental regimes. Such growth provides assurance the population is recovering and not experiencing the unsustainable conditions of the past 30-40 years.

Indications threats have abated: A major driving force in our inability to capture the extinction risk of sea lions is the uncertainty about the threats and their impacts. This uncertainty is important and cannot be dismissed. Without further understanding of the threats, or proof that the threats are no longer occurring, the population will retain the potential of 16% annual decline rates as observed in the late 1980s. Ultimately, the only way for the sea lion population to demonstrate that threats are reduced is to grow over an extended time period.

Sub-population concerns: It is important to consider sub-population declines and recovery. This element was not included as part of the PVA due to a lack of data on exchange rates between populations. Because all parts of the range are currently occupied, it would be wise to maintain those populations as viable entities, with some fluctuations in population numbers expected. Because the previous decline started in one area and spread to other areas, a substantial decline of any two adjacent sub-areas would indicate an active threat that was not predicted. Thus, significant declines over large areas (two subareas or more) could indicate that extinction risk may still be high and that further research would be needed to understand the threats before delisting.

Biological concerns: In general, NMFS expects to see that both juvenile survival and pup production (natality) have increased to the point that the population is not only able to sustain itself, but is able to grow at a modest rate. One feature of the North Pacific, decadal scale climate change, appears to have ecosystem -scale ramifications and may potentially influence the recovery of Steller sea lions. Therefore, the choice of time period length is influenced not only by the need to be confident that juvenile survival and natality have increased and are supporting the population growth rate, but that the recovery scenario has been maintained long enough to have a reasonable likelihood of occurring over multiple regimes. This is not a guarantee that the sea lion population can increase in all regimes, but it does lend further evidence that this population is robust enough to either downlist or delist (depending upon the criteria obtained; see below for specific criteria for downlisting to threatened and delisting).

E. Reclassification to threatened

Downlisting criteria are based upon the current estimated population abundance of approximately 45,000 animals for the U.S. portion of the western DPS, trends in population abundance, the continued human-caused threats to the species, and natural environmental conditions and variability. When determining whether the western DPS should be reclassified to threatened, NMFS will first assess whether the demographic criteria have been met, and then will consider the listing factor criteria.

When considering if demographic criteria are met, NMFS will use available information on the population ecology and vital rates in the U.S. region to ensure that they support the trends observed in the western DPS as a whole or in each sub-area. Certain vital rates are required in order to allow for long term growth. Available information on pup counts, production (natality), juvenile survival rates, population age structure, gender ratios, and other observations will be examined to determine whether they are indicative of the observed DPS-wide or sub-area trend rates.

6. Demographic Criteria: threatened

The western DPS of Steller sea lion will be considered for reclassification to threatened when all of the following conditions are met:

- The population for the U.S. region has increased (statistically significant) for 15 years on average, based on counts of non-pups (i.e., juveniles and adults). Based on an estimated population size of roughly 42,500 animals in 2000 and assuming a consistent but slow (e.g. 1.5%) increasing trend, this would represent approximately 53,100 animals in 2015.
- The trends in non-pups in at least 5 of the 7 sub-regions are consistent with the trend observed under criterion #1. The population trend in any two adjacent sub-regions can not be declining significantly. The 7 sub-regions are:
    a. Eastern Gulf of Alaska (US)
    b. Central Gulf of Alaska (US)
    c. Western Gulf of Alaska (US)
    d. Eastern Aleutian Islands (including the eastern Bering Sea) (US)
    e. Central Aleutian Islands (US)
    f. Western Aleutian Islands (US)

g. Russia/Asia

2. Recovery Factor Criteria: threatened

Eliminating or controlling the threats to the western DPS of Steller sea lion is imperative prior to downlisting to threatened, including all threats identified at the time of listing and any new threats identified after listing. An inclusive list of the threats to recovery is found in Section IV of this Plan. The Plan describes the lack of recovery of Steller sea lions due to changes in the environment, predation, direct takes by humans, and competition for prey resources with fisheries.

In order to reclassify the western DPS of Steller sea lion the following threats-based criteria should be achieved in such a way that the threats do not re-emerge. The best available information indicates that achieving the following threats criteria is necessary in order to recover Steller sea lions. Yet, it is possible that current perceived threats become insignificant in the future due to changes in the natural environment, or changes in the way that the threats affect the entire life cycle of Steller sea lions. When the biological criteria for downlisting are met, NMFS will evaluate and review the criteria under these listing factors to determine their relevance under the current conditions.

Factor A: The present or threatened destruction, modification, or curtailment of a species' habitat or range

Modification of the foraging habitat of the western DPS of Steller sea lion, through both natural and anthropogenic sources, likely resulted in decreased survival and reproduction and may currently limit recovery. Reducing the threats to sea lion foraging habitat will be accomplished through a broad application of recovery actions that protect and improve their habitat. To provide assurance that reclassification is warranted for the western DPS of Steller sea lion, threats to its habitat should be reduced as specified under this factor:

1. Knowledge of the foraging ecology of Steller sea lions and the impacts of fisheries on sea lion prey is sufficient to determine whether fisheries are likely to limit recovery.
2. Federal and state fishery management measures, or their equivalent, especially for pollock, Pacific cod, and Atka mackerel fisheries, are maintained in order to allow for the recovery of Steller sea lions. Modification of the conservation measures is based on the foraging requirements of Steller sea lions.
3. State of Alaska fishery management is reviewed, and those state fisheries that adversely affect Steller sea lions or their critical habitat should be authorized under the MMPA and ESA; habitat conservation plan under section 10 of the ESA or through section 7 consultations.
4. The designation of sea lion critical habitat is adequate to allow for recovery.

Factor B: Overutilization for commercial, recreational, or educational purposes
Human caused mortality for Steller sea lions results from incidental takes in fisheries, illegal shooting, harassment from tourism related activities, and take during scientific research. To provide assurance that reclassification is warranted for the western DPS of Steller sea lion, any overutilization for commercial, recreational, scientific, or educational purposes that threatens its continued existence should be reduced as specified under this factor:

1. Incidental takes are limited in commercial and recreational fisheries such that the effect of the take does not appreciably increase the time to recovery.
2. The occurrence of illegal shooting of sea lions remains low through awareness of regulations and enforcement.
3. Methods are developed and utilized to minimize the impacts of the research program, and those impacts do not limit the time to recovery of the population.

Factor C: Disease or predation
Throughout their range Steller sea lions are prey for killer whales. The impact of predation may be greater when sea lion abundance is reduced or when other factors increase their susceptibility to predation (e.g., reduced prey availability may increase foraging time thereby increasing vulnerability to predation). Currently, disease is considered to have a relatively minor impact on sea lions, but may present greater risks if population abundance declines further. To provide assurance that reclassification is warranted for the western DPS of Steller sea lion, any disease or predation that threatens its continued existence should be reduced as specified under this factor:

1. Methods have been developed and utilized to test sea lions for health related illness that may be limiting recovery and that information is adequate to conclude that disease is not limiting recovery.
2. Knowledge of the impacts of killer whale predation on sea lions is sufficient to determine that predation is not limiting recovery.

Factor D: The inadequacy of existing regulatory mechanisms
To provide assurance that reclassification is warranted for the western DPS of Steller sea lion, any inadequacy of existing regulatory mechanisms that threatens its continued existence should be reduced as specified under this factor:

1. Continue to implement fisheries regulations in 50 CFR part 679, following threats criterion A.1.
2. Update critical habitat by correcting erroneous locations for major rookery and haulout sites listed in 50 CFR parts 223 and 226.
3. Pursue international agreements and develop cooperative recovery programs with Russia and Japan.

Factor E: Other natural or anthropogenic factors affecting its continued existence

To provide assurance that reclassification is warranted for the western DPS of Steller sea lion, several natural and **anthropogenic** threats to its continued existence including subsistence harvest, pollution, toxins, and management should be reduced as specified under this factor:

1. Co-management agreements are in place with Alaska Native Organizations (ANOs) and a working relationship between the ANOs and NMFS results in an accurate accounting of the subsistence harvest, and the harvest levels do not likely limit sea lion recovery.
2. Sources of potential pollution, including offshore oil and gas development, are known and they are not likely to pose significant health risks to the sea lion population.
3. The influence of global climate change and oceanographic variability is examined, including in combination with other human influenced factors, and is determined unlikely to limit recovery.
4. An Alaska stranding network is in place and functional.
5. There is an outreach program to educate the public, commercial fishermen, and others to the continued need to conserve and protect Steller sea lions, including avoidance of rookery and haulout sites and the no-feeding rule around boats and harbors.
6. Catch and effort statistics of state and federal commercial fisheries for Steller sea lion prey species within designated critical habitat are collected and described annually.

E. Delisting

These criteria are based upon the estimate of population abundance in 2004 (the year of the last complete non-pup survey) of about 45,000 animals for the U.S. portion of the western DPS, trends in population abundance, the continued human-caused threats to the species, and natural environmental conditions and variability. When determining whether the western DPS should be delisted, NMFS will first assess whether the biological criteria have been met, and then will consider the listing factor criteria.

When considering if demographic criteria are met, NMFS will use available information on the population ecology and vital rates in the U.S. region to ensure that they support the trends observed in the western DPS as a whole or in each sub-area. Certain vital rates are required in order to allow for long term growth. Available information on pup counts, production (natality), juvenile survival rates, population age structure, gender ratios, and other observations will be examined to determine whether they are indicative of the observed DPS-wide or sub-area trend rates.

1. Demographic criteria: delisting

The western DPS of Steller sea lion will be considered for removal from the List when the likelihood of its becoming endangered in the foreseeable future has been eliminated by achieving the following biological criteria:

- The population for the U.S. region of this DPS has increased (statistically significant) for 30 years (at an average annual growth rate of 3%), based on counts of non-pups (i.e., juveniles and adults). Based on an estimated population size of about 42,500 animals in 2000, this would represent approximately 103,000 animals in 2030.
- The trends in non-pups in at least 5 of the 7 sub-regions are stable or increasing, consistent with the trend observed under criterion #1. The population trend in any two adjacent sub-regions can not be declining significantly. The population trend in any sub-region can not have declined by more than 50%. The 7 sub-regions are:
    a. Eastern Gulf of Alaska (US)
    b. Central Gulf of Alaska (US)
    c. Western Gulf of Alaska (US)
    d. Eastern Aleutian Islands (including the eastern Bering Sea) (US)
    e. Central Aleutian Islands (US)
    f. Western Aleutian Islands (US)
    g. Russia/Asia

2. Recovery Factor Criteria: delisting

Eliminating or controlling the threats to the western DPS of Steller sea lion is imperative prior to delisting, including all threats identified at the time of listing and any new threats identified after listing. An inclusive list of the threats to recovery is found in Section IV of the Plan. The Plan describes the lack of recovery of Steller sea lions due to changes in the environment, predation, direct takes by humans, and competition for prey resources with fisheries.

In order to delist the western DPS of Steller sea lion the following threats-based criteria should be achieved in such a way that the threats do not re-emerge. The best available information indicates that achieving the following threats criteria is necessary in order to recover Steller sea lions. Yet, it is possible that current perceived threats become insignificant in the future due to changes in the natural environment, or changes in the way that the threats affect the entire life cycle of Steller sea lions. When the biological delisting criteria are met, NMFS will evaluate and review the criteria under these listing factors to determine their relevance under the current conditions.

Factor A: The present or threatened destruction, modification, or curtailment of a species' habitat or range

Modification of the habitat of the western DPS of Steller sea lion, through both natural and anthropogenic sources, likely resulted in decreased survival and reproduction and may currently limit recovery. Reducing the threats to sea lion habitat will be accomplished through a broad application of recovery actions that protect and improve their habitat. To provide assurance that delisting is warranted for the western DPS of Steller sea lion, threats to its habitat should be reduced as specified under this factor:

1. Aquatic habitats are protected through appropriate management measures, to provide a prey base adequate to support recovered populations of Steller Sea lions. Conservation measures are based on the foraging requirements of Steller sea lions.
2. Rookery and haulout sites are adequately protected (through state, federal, or private measures) to insure the continued use of these sites for pupping, breeding, attending young, and resting. Research and monitoring plans are in place for all projects that have a high probability of negatively impacting sea lions.

Factor B: Overutilization for commercial, recreational, or educational purposes

Human caused mortality for Steller sea lions results from incidental takes in fisheries, illegal shooting, harassment from tourism related activities, and take during scientific research. To provide assurance that delisting is warranted for the western DPS of Steller sea lion, any overutilization for commercial, recreational, scientific, or educational purposes that threatens its continued existence should be reduced as specified under this factor:

1. A mechanism (e.g., fishery management) is and will remain in place that ensures incidental take is less than 10% of PBR (as defined under the MMPA).
2. Marine Mammal Protection Act (MMPA) enforcement is adequate to restrict illegal shooting to negligible levels.
3. Methods have been implemented and will remain in place to minimize the negative impacts of research.
4. Recreation fisheries, tourism, and other types of disturbance are controlled sufficiently to minimize negative impacts on recovered populations.

Factor C: Disease or predation

Throughout their range Steller sea lions are prey for killer whales. The impact of predation may be greater when sea lion abundance is reduced or when other factors increase their susceptibility to predation (e.g., reduced prey availability may increase foraging time thereby increasing vulnerability to predation). Currently, disease is considered to have a relatively minor impact on sea lions, but may present greater risks if population abundance declines further. To provide assurance that delisting is warranted for the western DPS of Steller sea lion, any disease or predation that threatens its continued existence should be reduced as specified under this factor:

1. Information is adequate to conclude that disease is not compromising the recovered status of sea lions.
2. Knowledge of the impacts of killer whale predation on sea lions is sufficient to determine that predation is not a threat to the recovered status.

Factor D: The inadequacy of existing regulatory mechanisms

To provide assurance that reclassification is warranted for the western DPS of Steller sea lion, any inadequacy of existing regulatory mechanisms that threatens its continued existence should be reduced as specified under this factor:

1. Continue to implement fisheries regulations in 50 CFR part 679, following threats criterion A.1.
2. Adequate international agreements with Russia and Japan are in place to maintain recovered status.

Factor E: Other natural or anthropogenic factors affecting its continued existence

To provide assurance that delisting is warranted for the western DPS of Steller sea lion, several natural and man-made threats to its continued existence including subsistence harvest, pollution, and management should be reduced as specified under this factor:

1. Co-management agreements are in place with Alaska Native Organizations (ANOs) and a working relationship between the ANOs and NMFS have resulted in an accurate accounting of the subsistence harvest, and the harvest levels do not compromise recovered status.
2. Sources of toxins, including offshore oil and gas development, and other pollution and contaminants are known, and they do not pose significant health risks to the recovered sea lion population.
3. Knowledge about impacts of climate change and oceanographic variability is sufficient to determine that their impacts will not likely threaten the recovered status of sea lions.
4. An Alaska stranding network is in place and functional.
5. An outreach program is established to educate the public, fishermen, and others to the continued need to conserve and protect Steller sea lions.
6. An agreement is established with the State of Alaska, which describes their fishery management plan, minimizes the take of Steller sea lions, and describes how future actions taken by the State will comport with the MMPA.
7. Adequate monitoring programs are in place to detect population changes in the future.

F. Recovery Action Outline and Narrative

The recovery actions outlined below reflect the best scientific and commercial information currently available. Estimated time and cost required, task priority and those responsible for carrying out each recovery action are identified in the Implementation Schedule (see Section V.E.).

Following the approval of the recovery plan, NMFS will work with its partners to implement this plan and monitor recovery action implementation. Recovery action 1.5 calls for the development of an implementation plan which will provide a broader and more systematic approach to the research program and other conservation actions. This plan defines the individual actions necessary for recovery but does not provide the further refinement in priorities and timing of events necessary for an effective research program of this large scale. The effectiveness of various recovery measures will be assessed and appropriate modifications implemented to accelerate progress towards the recovery goal. While many factors can confound efforts to evaluate the effects of discrete actions on wild populations, carefully designed monitoring is the key to assessing and improving the effectiveness of recovery actions. Results of this type of monitoring will be considered during biennial reviews of recovery plan implementation (under Action 1.5) to assure timely adjustment of ongoing efforts and priorities. All recommended recovery actions should incorporate monitoring and evaluation to assess their effectiveness in furthering the recovery of the western DPS. The results of research tasks described below will be used to evaluate and refine other recovery actions. The response of populations to recovery measures will be used to revise research priorities.

The plan calls for 78 Actions in 5 categories:

- Baseline Population Monitoring: 11 actions; $3.1 M
- Insure Adequate Habitat and Range for Recovery: 23 actions; $10.1 M
- Protect from Over-Utilization for Commercial, Recreational, Scientific, or Educational Purposes: 10 actions; $1.7 M
- Protect from Diseases, Contaminants, and Predation: 18 actions; $2.9 M
- Protect from Other Natural or Anthropogenic Actions and Administer the Recovery Program: 16 actions; $2.3 M

Recovery Action Outline

1 **BASELINE POPULATION MONITORING**

- 1.1 Continue to estimate population trends for pups and non-pups
  - 1.1.1 Estimate trends for pups and non-pups via aerial surveys
  - 1.1.2 Continue to monitor population trends on Pribilof Islands (particularly the Walrus Island rookery) via aerial surveys or land-based pup counts
- 1.2 Estimate vital rates
  - 1.2.1 Continue to estimate survival, natality, and immigration/emigration rates through a branding/resight program
  - 1.2.2 Promote cooperative pup branding/resight programs in Russia
  - 1.2.3 Develop an age-structured population model using medium format photos from aerial surveys
  - 1.2.4 Develop methods and determine reproductive rates including pregnancy and parturition rates
- 1.3 Monitor health, body condition, and reproductive status
  - 1.3.1 Examine the effects of season, age, and sex on body condition
  - 1.3.2 Develop improved indices of health, body condition, and reproductive status using chemical methods (e.g., hematology serum chemistries, and endocrine monitoring)
- 1.4 Develop and implement live capture methods and non-lethal sampling techniques
  - 1.4.1 Develop improved live capture techniques for general research needs
  - 1.4.2 Develop improved non-lethal sampling techniques to assess health
- 1.5 Develop an implementation plan and update it periodically

2 **INSURE ADEQUATE HABITAT AND RANGE FOR RECOVERY**

- 2.1 Maintain, and modify as needed, critical habitat designations
- 2.2 Redefine and catalog rookery and haulout sites and ensure their protection
- 2.3 Estimate prey consumption and essential characteristics of marine habitat
  - 2.3.1 Collect and analyze scat samples and stomach contents to determine prey consumption
  - 2.3.2 Develop stable isotope and fatty acid methodologies to assess prey consumption
  - 2.3.3 Deploy instruments to obtain fine scale data on sea lion foraging habitat
  - 2.3.4 Evaluate all information on sea lion foraging areas and develop a description of foraging needs

- 2.4 Determine the environmental factors influencing sea lion foraging and survival
  - 2.4.1 Assess the relationships between oceanographic profiles or features and sea lion foraging ecology
  - 2.4.2 Examine the influence of ecosystem variability on non-commercial prey species as an index to sea lion carrying capacity
  - 2.4.3 Distinguish how natural and anthropogenic factors influence marine ecosystem dynamics and subsequently sea lion population dynamics
- 2.5 Investigate sea lion bioenergetics
  - 2.5.1 Determine the physiological diving capabilities and evaluate how this limits the ability to forage successfully
  - 2.5.2 Determine the energetic costs to foraging sea lions
  - 2.5.3 Assess the nutritional value of prey by species, season, and area including digestibility and overall value to sea lions
  - 2.5.4 Develop an energetics model to investigate the interrelationships between prey availability and sea lion growth, condition, and vital rates
- 2.6 Assess and protect important prey resources for sea lions
  - 2.6.1 Improve groundfish stock assessment surveys to determine seasonal and inter-annual patterns of prey abundance, distribution, and movement at scales relevant to sea lions
  - 2.6.2 Assess competition for prey with sympatric consumers (e.g., gadids and flatfish, fur seals, harbor seals, other marine mammals, and seabirds)
  - 2.6.3 Utilize groundfish fishery observer data to assess the spatial-temporal distribution of the fishery
  - 2.6.4 Assess effectiveness of sea lion closure zones around rookeries and haulouts using small-scale experiments
  - 2.6.5 Assess the response of sea lions to changes in prey distribution and availability
  - 2.6.6 Evaluate and implement current or equivalent fishery regulations to protect foraging habitat and prey resources for sea lions
  - 2.6.7 Explore the use of ecosystem based (multi-species) stock assessment models to set fishery catch limits to ensure adequate prey resources for a recovered sea lion population
  - 2.6.8 Design and implement an adaptive management program for fisheries, climate change, and predation
  - 2.6.9 Prepare a habitat conservation plan under section 10 of the ESA for fisheries authorized by the State of Alaska
  - 2.6.10 Consider and implement conservation measures in herring and salmon fisheries in Alaska as appropriate

3 PROTECT FROM OVER-UTILIZATION FOR COMMERCIAL, RECREATIONAL, SCIENTIFIC, OR EDUCATIONAL PURPOSES

- 3.1 Minimize threat of incidental take in fisheries

- 3.1.1 Monitor and evaluate incidental take in commercial and recreational fisheries through observer and self-reporting programs
- 3.1.2 Monitor and evaluate incidental take in non-commercial fisheries

3.2 Minimize threat of intentional killing in fisheries

- 3.2.1 Monitor intentional take via shoreline surveys for carcasses near suspected conflict "hotspots" and by encouraging reporting of illegal shooting through NMFS's enforcement hotline
- 3.2.2 Reduce threat of illegal shooting by developing and promoting use of non-lethal deterrents for commercial fisherman

3.3 Minimize frequency and severity of sea lion-human interactions in ports and harbors

- 3.3.1 Develop and promote non-lethal means of deterring sea lions from hauling out on docks
- 3.3.2 Continue to publicize "No feeding" regulations in harbor areas and keep active programs for notification and enforcement

3.4 Minimize take by recreational and commercial viewing operations

- 3.4.1 Publicize and enforce existing no-transit areas to minimize vessel and aircraft disturbance of rookery sites
- 3.4.2 Review and revise existing Marine Mammal Approach Guidelines and provide to charter operators and other mariners to minimize disturbance at haulouts

3.5 Evaluate and reduce the direct and indirect impacts of research activities

- 3.5.1 Coordinate research efforts to reduce potential for unnecessary or duplicative research-related takes
- 3.5.2 Monitor and minimize unintentional takes associated with research activities

4 PROTECT FROM DISEASES, CONTAMINANTS, AND PREDATION

4.1 Protect Steller sea lions from disease

- 4.1.1 Conduct epidemiological surveys
- 4.1.2 Develop and implement methods for parasite evaluations
- 4.1.3 Develop and implement methods to test immune system functioning
- 4.1.4 Evaluate causes of mortality by examining dead and live animals of all age and sex classes for disease from various sources across the geographic range and in all seasons
- 4.1.5 Develop disease management plans
- 4.1.6 Develop an unusual mortality event (UME) management plan
- 4.1.7 Develop models to simulate disease impacts based on energetics, physiology abundance and demographics.

4.2 Protect sea lions from contaminants

- 4.2.1 Design a contaminant research and management plan
- 4.2.2 Collect samples from free-ranging sea lions and in environmental 'hotspots'

- 4.2.3 Examine blood and tissue samples for evidence of contaminant-linked endocrine effects including free-ranging and captive work
- 4.2.4 Develop models to simulate contaminant impacts and effects based on energetics, physiology, abundance and demographics

4.3 Predation

- 4.3.1 Understand predator life histories, biology and ecology through studies of free-ranging and captive animals
- 4.3.2 Determine killer whale diets
- 4.3.3 Develop methods to obtain samples from live killer whales
- 4.3.4 Expand the stranding network to increase samples of killer whales available for research
- 4.3.5 Determine killer whale distribution and behavior across the North Pacific
- 4.3.6 Estimate numbers of killer whale ecotypes in time and space
- 4.3.7 Develop models to simulate predation rates based on killer whale energetics and abundance of Steller sea lion demographics

5 PROTECT FROM OTHER NATURAL OR ANTHROPOGENIC FACTORS AND ADMINISTER THE RECOVERY PROGRAM

5.1 Reduce damage to sea lions and their habitat from discharges of pollutants by developing preventive measures

5.2 Reduce the potential for sea lion entanglement by improving and continuing programs aimed at reducing marine debris

- 5.2.1 Reduce discards of debris (e.g., trawl web and packing bands)
- 5.2.2 Cleanup derelict gear and beached debris

5.3 Monitor causes of sea lion mortality and use data to direct management actions

- 5.3.1 Continue and expand the Alaska stranding network to increase coastal coverage and community involvement in monitoring sea lion mortality
- 5.3.2 Survey selected areas for stranded animals
- 5.3.3 Expand tissue sampling efforts to improve the information obtained from dead sea lions
- 5.3.4 Monitor the incidence and impact of entanglement in marine debris

5.4 Effectively administer the Steller sea lion recovery program by continuing to provide a recovery coordinator staff position

5.5 Improve sea lion conservation by consulting with the State of Alaska on actions that are likely to adversely impact Steller sea lions

5.6 Conduct an effective outreach program to inform the public about Steller sea lion biology, habitat utilization, and conservation issues

- 5.6.1 Encourage and facilitate public reporting of sea lion observations
- 5.6.2 Publicize current conservation efforts and protective measures.

5.7 Co-manage Steller sea lion subsistence harvests in Alaska by developing co-management agreements with Alaska tribes and tribally authorized Alaska Native Organizations (ANO)

- 5.7.1 Co-manage subsistence harvests and evaluate the efficacy and accuracy of using retrospective subsistence harvest surveys
- 5.7.2 Support Alaska Native subsistence use information programs
- 5.7.3 Analyze carcasses from subsistence harvest to assess age, body condition, and other relevant information to ensure safety of carcasses for human consumption
- 5.7.4 Document local knowledge and cultural science (Traditional Ecological Knowledge, TEK) pertaining to sea lions to better understand changes in sea lion movement (local and seasonal), feeding patterns and prey, seasonal haulouts, predation and ecosystem dynamics

5.8 Improve the effectiveness of research for Steller sea lion recovery by instituting a “fast track” process for expediting NMFS research permits for Steller sea lions.

Recovery Action Narrative

1 Baseline population monitoring

Baseline population monitoring is necessary to support all of the recovery actions. These actions describe the status and trends of the western DPS of Steller sea lions, their vital rates, and the health and body condition of individuals.

1.1 Continue to estimate population trends for pups and non-pups

1.1.1 Estimate trends for pups and non-pups via aerial surveys

Conduct aerial surveys for pups and non-pups biennially at trend sites, and at least every 4 years at all rookeries and haul outs in the western DPS. Surveys using aerial photography have been conducted biennially since 1994. Medium format photogrammetry began in 2002 and this change now allows for counting pups as well as non-pups. Continue aerial surveys for pups and non-pups, population size and trend using medium format photography. Information from trend sites forms the basis of the stock assessment reports.

1.1.2 Continue to monitor population trends on Pribilof Islands (particularly the Walrus Island rookery) via aerial surveys or land-based pup counts

Population numbers have declined dramatically on the Pribilof Islands and only one rookery remains on Walrus Island. This area is not part of the usual aerial survey route or trend sites but on occasion can be included in the flight plan. In addition, researchers can access Walrus island by small boat.

1.2 Estimate vital rates

1.2.1 Continue to estimate survival, natality, and immigration/emigration rates through a branding/resight program

The current branding/resighting program should be continued. Additional branding at rookeries in the central and western Aleutian Islands would be valuable, but the logistics and costs, particularly of an adequate resighting effort, may make this impractical. The disruptive nature of branding young pups on rookeries and the probability of causing some low level of mortality are recognized. However, the importance of obtaining estimates of vital rates and the lack of alternative methods of obtaining these estimates justify this activity. Less invasive methods of marking should be investigated if new technology becomes available. Annual resighting surveys should be conducted even if branding is reduced in frequency. Resighting surveys should be coordinated among all research groups conducting work and a central database created.

In addition to estimating vital rates, both the handling of pups at branding and the resighting of branded individuals present the opportunity for collection of data on growth, condition, health status, genetics, weaning status, range of movements, attainment of sexual maturity, individual variations in terrestrial site use, diet and many other aspects of natural history. Tissue samples for genetic work should be collected from all handled pups and blood samples for archiving and health and condition studies should be collected whenever possible. This work should be promoted for all Russian branded sea lions as well. In addition, protocols should be distributed that direct the sample collection from stranded animals, subsistence-harvested, and fishery by- catch. Data should be recorded and managed in a manner that these ancillary products can be obtained.

1.2.2 Promote cooperative pup branding/resight programs in Russia

The distribution of Steller sea lions in the North Pacific and Bering Seas extends beyond international boundaries, and previous branding work has shown that sea lions travel in both directions. Therefore, any investigation of sea lion population trends in the U.S., especially the western Aleutian Islands, should incorporate information on what is known of the population trends and life history studies of sea lions in Russia.

1.2.3 Develop an age-structured population model using medium format photos from aerial surveys

The classification of sex and age classes from the medium-format photographic surveys provides an opportunity to examine patterns of natality and recruitment, important in understanding the dynamics of the population. If past 35mm photos are found to provide similar opportunities, a retrospective analysis of population age structure may be possible.

1.2.4 Develop methods and determine reproductive rates including pregnancy and parturition rates

The use of tissue samples collected from feces or from the tissue of a pup to indicate the pregnancy status of its nursing mother should be validated. If appropriate protocols can be established, this may provide a rapid and sensitive indicator of changing environmental conditions. Likewise, any sensor technology, including ultrasonography that may assist in developing the methodology to determine pregnancy status or parturition rates should be investigated.

1.3 Monitor health, body condition, and reproductive status

1.3.1 Examine the effects of season, age, and sex on body condition

Seasonal environmental changes result in fluctuations in available prey resources and energetic demands for Steller sea lions. In the last few years sampling of sea lions during autumn, winter and spring has increased. Sampling at these times provides access to different age classes, and during periods when foraging ranges for older juveniles and adults are greatest. This sampling across seasons and age classes should be continued and expanded to include areas in the central and western Aleutian Islands, and western Gulf of Alaska. Sample collections should be coordinated with groups capturing Steller sea lions.

For various age classes, differences in the rate of growth, specific caloric demands, and foraging (diving) capabilities will alter nutritional needs of Steller sea lions. Consequently, we have learned that different stages in life history may be more susceptible than others to the effects of nutritional limitation, exposure to diseases, or pollutants. Sampling and monitoring schemes for Steller sea lions should be expanded to include all age classes across seasons in order to determine the relative vulnerability of pups, juveniles, and adult male and females at different stages in the reproductive cycle.

1.3.2 Develop improved indices of health, body condition, and reproductive status using chemical methods (e.g., hematology serum chemistries, and endocrine monitoring)

To date most studies of wild Steller sea lions have focused on "snapshots" in the lifecycle rather than long-term monitoring of individuals. Longitudinal monitoring that recaptures individuals at several points throughout their life is needed to understand the relationships between condition indices, true condition, and how indices ultimately reflect growth, metabolism, reproductive output, and survival.

A suite of hematological parameters may provide insight into the general health of an animal as well as exposure to disease or contaminants. Blood sampling schemes should include multiple seasons and age classes (1.3.1) to provide insight into changes in health and condition over seasons and ages. New blood parameters (i.e. stress proteins) and the relationship between parameters in blood panels should be investigated. For sampling on summer rookeries, health and condition data should be evaluated to develop a monitoring plan to detect interannual or regional changes in health or condition among pups, juveniles and adults. Captive sea lions may be used to enhance these efforts, and to test or validate new assays or techniques that may give better insight into health and body condition.

Endocrine studies should be used to monitor reproductive status and condition. Circulating or excreted concentrations of hormones can provide information on the reproductive health of individuals. By expanding sampling of blood, urine, feces, and saliva across seasons and age classes, opportunities will be improved to describe sea lion reproductive status and health among more age classes. If endocrine levels can be related to concentrations found in feces, urine, saliva or other less invasive samples then broad-scale collections may become useful for population monitoring. Continued work with captive sea lions may provide baseline data for these types of studies. The potential to relate endocrine function with contaminants, environmental conditions, immune function and behavior should be investigated.

1.4 Develop and implement live capture methods and non-lethal sampling techniques

1.4.1 Develop improved live capture techniques for general research needs

A great deal of work has been done on techniques for capturing and immobilizing marine mammals. Techniques such as dive captures, floating trap captures, and land captures, as well as chemical immobilization, have been developed in the last decade. However, there is a possibility of an accidental sea lion mortality even with the best methods currently available, so efforts to improve techniques should be continued. Current capture methods have been extremely successful in capturing young animals but largely ineffective at capturing the older ages or larger animals. Thus, a critical gap in understanding sea lion foraging behavior has developed and alternative methods should be explored to safely and effectively capture older animals. Two potential methods include the use of floating structures and large nets to capture sea lions. The most important areas to access include the Aleutian Islands which has proven to be a difficult place to use these alternative techniques due to poor weather conditions and difficult terrain.

Special studies need not be conducted for the specific purpose of improving such techniques. Most of the development and evaluation of new techniques should be possible during research conducted for other purposes. Careful monitoring and documenting of vital signs of animals under anesthesia and their responses to anesthesia are needed to regularly reassess and evaluate current methods and for comparison to new methods as they become available. Likewise, thorough disease screening and quarantine protocols should be used for animals being brought into captivity and upon release back into the wild.

1.4.2 Develop improved non-lethal sampling techniques to assess health

Non-lethal sampling is used to monitor health, disease, body condition and other vital parameters in Steller sea lions. Some techniques such as chemical immobilization may pose a risk of mortality. However, the potential impact of this technique has not been borne in the studies that have used chemical immobilization in pinnipeds, particularly otariids. This potential impact does not appear to exceed the benefit to the species from knowledge gained at this time. The large size of Steller sea lions, which makes them difficult to work on otherwise, and improved quality of data obtained from anesthetized animals warrants continuing to use these methods and validates the use of new chemical immobilizing agents for future studies.

A variety of studies require temporary restraint of animals and this includes those focused on disease and contaminant research. Relatively non-invasive sampling techniques can be used for disease surveys, such as the collection of blood samples, swabs of body fluids (oral, nasal, rectal, vaginal or preputial, or tracheal), and feces and urine. Additionally, biopsies of lesions can be collected when present. Samples should be collected from the relatively few Steller sea lions maintained in captivity, for comparison and longitudinal sampling with free- ranging sea lions, when possible.

1.5 Develop an implementation plan, and update periodically

An implementation plan should be developed that includes a comprehensive ecological and conceptual framework that integrates and further prioritizes the numerous recovery actions provided in this plan. The implementation plan should provide a synthesis of the individual actions, and coordinate their implementation in a cohesive strategy. Clearly lacking in this plan is a finer scale analysis of priorities, a synthesis of how the actions should be implemented and in what order, and which actions are more important to the scientific process of understanding the decline of the western DPS of Steller sea lion.

The conceptual framework of the implementation plan must integrate several components: (1) the complex dynamics of the North Pacific marine ecosystem, (2) multiple causation in those systems, (3) the need for long-term research, (4) the monitoring required to assess the effectiveness of management regulations, and (5) the development of a modeling approach that examines possible effects of multiple threats on sea lion population dynamics to evaluate the strength of the evidence for different hypotheses. This framework will establish priorities among the multitude of recovery actions, distinguishing between those actions that will provide the information key to promoting recovery versus expanding the knowledge of sea lion biology and ecology. For example, to provide insights on those factors limiting recovery, physiological and behavioral indices must be directly associated with a mechanism that can be quantified in changes in survival or reproduction. Focusing the top priorities on actions that will promote recovery should also ensure that the new information gained will assist the consultation process under the ESA.

2 Insure adequate habitat and range for recovery

2.1 Maintain, and modify as needed, critical habitat designations

Critical habitat for Steller sea lions is currently designated at 50 CFR 226.202, and includes both terrestrial and marine habitat. Marine foraging areas were designated based on sea lion distribution and abundance, ship-based observations, movements and dive behavior derived from satellite telemetry, historical records of regional sea lion concentration, and distribution of primary prey species. Terrestrial rookery and haulout sites were designated from historic counts of sea lions on land.

Since designation in 1993, critical habitat has been an essential component of Steller sea lion conservation and appears to have generally been effective. However, new information on the foraging ecology of Steller sea lions, potential changes in habitat use, and the application of new technology has revealed areas for enhancement. For example, improvement in satellite mapping technology has allowed greater accuracy in determining the locations of rookery and haulout sites. Some technical errors have been discovered which should be fixed. As population abundance declined, the distribution of sea lions among haulout and rookery sites has changed substantially. These considerations should be taken into account when considering a revision of designated critical habitat.

A revised critical habitat designation should also consider spatial and temporal variation of essential habitat characteristics. Specifically, in addition to stationary habitat features such as bathymetry and the continental shelf, dynamic features such as seasonal distribution and abundance of prey, and the oceanographic features and parameters that influence those prey species should be considered. More refined information on sea lion foraging ecology and seasonal movement patterns may be obtained through advanced telemetry equipment. Further, advances in the statistical analysis of such data should be applied to those data when the designation of critical habitat is revised.

2.2 Redefine and catalog rookery and haulout sites and ensure their protection

Most rookery and haulout sites are adequately protected, yet new threats (e.g., oil and gas development) may subject some sites to additional risk. A catalog of current and historical rookeries and haulouts, their locations, and range of numbers counted at each site should be compiled. Communication channels between NMFS and the land managers of all rookeries and major haulouts should be formalized to ensure that land managers are aware of all applicable protection measures and that NMFS is notified of any potential action that could affect habitat quality.

2.3 Estimate prey consumption and essential characteristics of marine habitat

2.3.1 Collect and analyze scat samples and stomach contents to determine prey consumption

Scat analysis has been used as a relatively inexpensive and non-invasive technique to estimate spatial and temporal trends in sea lion diet since the late 1980s. Currently, scats are the only technique available for estimating size and age of prey species consumed. However, several known biases exist in scat analysis, primarily related to the digestion-resistance and differential rates of digestion of some hard parts, which has lead to the development of alternative methods to understanding diet in marine mammals (see 2.3.2). In order to estimate long-term trends in diet, in part to assist in the evaluation of the impacts of environmental change and management actions, scat samples should continue to be collected, especially in conjunction with other research activities that require disturbing rookeries and haulouts. Identification and analytical methods should be standardized among researchers so data are directly comparable. Methods to calibrate results to accurately reflect the quantity and composition of prey consumed should be further developed and refined. Likewise, stomach contents from subsistence-harvested animals should also be collected when possible, and stomach samples should be used to help validate diet estimates from scat collections.

2.3.2 Develop stable isotope and fatty acid methodologies to assess prey consumption

Scat analyses provide limited data about the amount and type of prey consumed, and the age-specific diet of sea lions cannot be determined from scats. Indirect methods such as the analysis of stable isotopes and fatty acid (FA) signatures may be useful in determining the diet of sea lions, at both the individual and population level, and thus should be further developed and applied.

The FA profile represents an integration of a sea lion’s diet over several weeks to months and represents all species eaten. Further, when adequate data are available on the FA composition of the predator’s potential prey species, quantitative estimation of diet is possible. In the absence of prey FA information, FA profiles of a predator can detect dietary changes and differences between demographic groups. In addition, because tissue samples are collected directly from an individual sea lion, relationships between phenotypic and demographic characteristics of the animal can be tested. Fatty acids have been successfully used to describe regional and age-related differences in fatty acid profiles, and research is currently underway to develop appropriate statistical models to use this data to quantitatively estimate diet composition.

An additional approach to the evaluation of diet in marine mammals is the use of stable isotopes of carbon and nitrogen to evaluate the trophic level of the diet. Specifically, nitrogen isotope ratios change with trophic level in a predictable manner through a step-wise enrichment within marine food webs. In contrast, carbon isotope ratios are more influenced by geographic location of feeding, and may distinguish inshore versus offshore foraging locations. The isotopic measurement of several tissues from the same individual can provide short-, intermediate-, and long-term dietary information depending upon their rates of metabolic activity. Serum and milk provide a short-term integrated diet signature, whereas relatively metabolically inactive tissues such as whiskers can provide a longer-term integration of the isotopic signature for the period of growth of that tissue.

2.3.3 Deploy instruments to obtain fine scale data on sea lion foraging habitat

A vast amount of data on the movements and diving and haulout behavior of sea lions has been obtained through the use of a variety of telemetry instruments, which forms the basis for the current understanding of sea lion foraging ecology and the essential characteristics of sea lion foraging habitat. Information gained in recent years has been primarily from pups (young of the year) and juveniles, many of which were still nursing and thus not foraging 100% on their own. This information was described in detail, including the important caveats and potential problems with using the data to describe foraging habitat, in NMFS (2001 and 2003) from which current fishery conservation measures were developed. However, information is limited on the foraging characteristics of older juveniles (ages 2-4) and adult females from most geographic areas. In general, juveniles (ages 1-4) tend to travel further offshore, make longer foraging trips, and dive deeper than pups, future tagging efforts should focus on juveniles and adult females. These animals are more susceptible to most threats impeding recovery, and thus information on their foraging ecology and habitat use is essential, and that information should be used when fishery conservation measures are revised.

The precision of most dive behavior and location data obtained from nearly all telemetry studies to date has been relatively poor due to instrument limitations, and subsequently accurate information on the locations and depths where sea lions actually consume prey has not been determined. Technological advances integrated in new telemetry instruments may now permit the acquisition of more precise and accurate data on dive patterns, dive depths, and foraging locations. Useful technologies include GPS, stomach temperature telemetry in conjunction with satellite-linked time-depth recorders, and sonar tracking of sea lions. Further, reliable remote release devices or dependable recapture methods may allow the use of more sophisticated satellite telemetry or other devices to obtain finer scale movement and dive data.

2.3.4 Evaluate all information on sea lion foraging areas and develop a description of foraging needs

Data obtained from telemetry studies requires substantial processing and database management prior to conducting statistical analyses. Such data management needs to be completed promptly and in a manner that will allow the integration of databases generated by different researchers and projects. Recently developed advanced analytical and statistical techniques should be applied, as appropriate, to obtain the most informative results pertaining to foraging ecology and habitat use, at the individual and population level, to further understand the potential impact of those threats that result in a reduction of prey biomass and quality. Available telemetry data should be collected from all agencies and research organizations and synthesized into one database, enabling a more complete description of sea lion foraging ecology and habitat preferences.

2.4 Determine the environmental factors influencing sea lion foraging and survival

The dynamics of the North Pacific marine ecosystem influences the biology and ecology of sea lions, and thus have implications for sea lion recovery. Knowledge of the spatial and temporal patterns of the marine ecosystem is limited, as is understanding of the factors that influence those patterns. Thus, integrated studies at the ecosystem and community ecology level are needed to better assess how sea lion population dynamics and their carrying capacity are influenced by ecosystem variability, and to determine how natural and anthropogenic factors may affect that variability.

2.4.1 Assess the relationships between oceanographic profiles or features and sea lion foraging ecology

Recent studies using satellite and radio telemetry suggest that otariid may cue on various oceanographic features (e.g., eddies and currents) to locate prey and to navigate in the open ocean. Oceanographic features are also critical to prey life stages and likely influence both fish stock recruitment, fish distribution, and abundance. There is a growing recognition of the need to integrate existing and future physical and biological oceanographic data into upper trophic level predator/prey studies and to determine the features that influence the distributions of both predator and prey. Such integration may allow predictions of how environmental perturbations influence upper trophic level predators.

Within the next 2-5 years, review studies should be conducted to integrate data collected on similar spatial and temporal scales by physical and biological oceanographers and sea lion biologists. Specific ecological data on sea lions should include age- and sex-specific foraging behavior, reproductive status, and energetic demands. These integrative efforts should determine if available data permits enhanced understanding of the physical forces that influence sea lion prey distribution, foraging decisions, and subsequently health and condition.

2.4.2 Examine the influence of ecosystem variability on non-commercial prey species as an index to sea lion carrying capacity

Some non-commercial fish species, ‘forage fish’ in particular, may play an important seasonal role in the diet, condition, and population dynamics of sea lions in some regions. Information on the seasonal and annual distribution and abundance of these fish species is very limited because they are not included in assessment studies of commercial species. Understanding how changing environmental conditions affect the distribution and abundance of these species, and subsequently the foraging ecology of sea lions, may be used to project how sea lion health and condition, and ultimately population growth, will be effected under different environmental conditions.

Thus, currently available data should be combined with new studies to improve the understanding of how the distribution and abundance of these important prey species is influenced by environmental variability. Existing NMFS and ADF&G trawl data will provide some guidance on whether current assessment methodology is effective for these species, or if sampling techniques need to be modified. Subsequently, periodic long-term assessment programs should be implemented to provide adequate baseline and monitoring data.

2.4.3 Distinguish how natural and anthropogenic factors influence marine ecosystem dynamics and subsequently sea lion population dynamics

The distribution and abundance of marine mammal populations are influenced by changes within their marine environment, yet the understanding of the factors and mechanisms that drive those changes is quite limited. Further, distinguishing between changes from natural versus anthropogenic factors is needed to determine the feasibility and efficacy of management strategies. Certainly, natural ecosystem dynamics are complex and will confound efforts to assess anthropogenic effects. Integrated ecosystem studies (e.g., FOCI, BEST) need to be continued and expanded to obtain baseline information on the factors that most influence ecosystem components and processes. Such information should improve the understanding of the primary ecosystem characteristics that most influence sea lion prey distribution, and subsequently sea lion foraging and population dynamics. Adaptive management experiments (see 2.6.8) can be designed to examine more specific ecosystem and sea lion attributes and parameters.

Integrative modeling and field studies that link physical and biological oceanography with sea lion foraging and physiology need to be conducted. Expansion of ongoing ecosystem modeling studies (e.g., Ecosim, Ecopath) should occur prior to initiation of new field studies to help guide those studies and to determine where data deficiencies and biological links occur. Field studies then need to be conducted in all seasons and in areas of declining and increasing Steller sea lion populations. These studies should be conducted in association with existing or planned cohesive sea lion prey studies (e.g., those in the Kodiak area and the Shumagin Islands regions) to take advantage of the synergistic benefits of linking related research programs.

2.5 Investigate sea lion bioenergetics

2.5.1 Determine the physiological diving capabilities and evaluate how this limits the ability to forage successfully

The ability of sea lions to exploit various prey resources in an energetically efficient manner is limited by their diving ability. This will almost certainly vary with age, gender, and reproductive status; e.g. a female with a young pup will be limited in the distance she can travel from the rookery in search of food. Adult male and female sea lions have different strategies for storage and utilization of energy and thus may differ in their ability to thrive under differing environmental conditions that affect availability of prey.

2.5.2 Determine the energetic costs to foraging sea lions

The energetic costs of foraging have not been adequately determined for either sex or for any age group or for different life history events such as reproduction and molting and will vary depending on the availability of prey in both a spatial and temporal scale. If prey are difficult to locate and capture, more energy will be required; if that expenditure exceeds available body stores the animal's homeostasis is disrupted, especially for young animals that do not have sufficient energy stores for prolonged periods of fasting.

It has been postulated that young sea lions may be energetically stressed while feeding during winter storm periods if adequate prey are not easily available in a reasonable time period. The result of this postulated stress is reduced overall fitness and higher susceptibility to predation, disease, or other factors increasing mortality.

Methods to measure energetic costs and the physiological diving capabilities of foraging pinnipeds have been tested on sea lions during pilot studies, yet thorough studies have not been conducted. Capture and holding techniques have been developed, and a research program should be implemented that reviews past studies, incorporates modifications of methods previously used, and includes seasonal sampling of both sexes and all age groups from both captive and free ranging sea lions.

2.5.3 Assess the nutritional value of prey by species, season, and area including digestibility and overall value to sea lions

The nutritional value of a particular prey type can be viewed as the net energy obtained by its capture and assimilation. An assessment of net nutritional values requires the integration of all costs associated with foraging combined with information on the quality and quantity of prey consumed during a foraging trip, and subsequently digested and assimilated. The energy density of sea lion prey species varies widely spatially and temporally. Information on the energy density for some sea lion prey species is available, yet additional data is needed for a more comprehensive understanding among prey species across regions and seasons. Such information will increase the understanding of the relative nutritional value of sea lion prey, which in turn will improve efforts to model the bioenergetics of sea lion foraging and food requirements.

2.5.4 Develop an energetics model to investigate the interrelationships between prey availability and sea lion growth, condition, and vital rates

Energetics data should be integrated with population status and vital statistics data to develop a foraging model that can be used to test the relationship between prey availability and population growth rate. This analysis will likely require improved knowledge of the specific energetic requirements of sea lions during different life stages including periods of rapid growth, pregnancy and lactation. The effects of seasonal changes in available prey resources, population composition, and spatial relationships should be included in the models. Specific questions to be addressed include the effects of 1) reproductive status on local foraging requirements, 2) potential competition for specific prey with fisheries, and 3) changes in population size on overall prey requirements.

Simulations can also be used to evaluate disease and contaminant impacts on energetics, physiology, abundance, and demographics. A growing need exists to integrate the biological data with population-scale dynamics.

2.6 Assess and protect important prey resources for sea lions

For the endangered Steller sea lion population to recover, prey availability must be adequate to support a growing population. An adequate level of prey includes the energetic cost of obtaining prey, such that the energy expended in capturing prey does not exceed the energy gained. Because the most important feature of marine sea lion habitat is the prey base, measures must be taken to ensure that prey abundance is adequate to support a growing population. All of the tasks under this section are related to Listing Factor A, which addresses the present or threatened destruction, modification, or curtailment of its habitat or range. These tasks also relate to Listing Factor D, which addresses the adequacy or inadequacy of existing regulatory mechanisms.

Threats related to reduced prey availability and food limitation may include reductions of the prey base or changes in its distribution due to fishing, natural environmental fluctuations in prey biomass or changes in its distribution, and competition from other predators in the ecosystem. Currently, it is not possible to determine the relative contribution of these factors, but fishing is the only factor over which humans have direct control. Existing fishery regulations must be monitored and evaluated on a regular, ongoing basis in order to determine their impact on sea lion foraging areas and prey availability, and additional regulations should be adopted as necessary to ensure that sea lion food supplies are adequate to support a recovered population. Moreover, the ability to distinguish between fishing and non-fishing effects on sea lion prey availability will require an experimental framework of treatment and control areas in which fishing is permitted or prohibited so that researchers can more readily distinguish the relative impacts of fishing and non-fishing threats to sea lion prey availability.

2.6.1 Improve groundfish stock assessment surveys to determine seasonal and inter-annual patterns of prey abundance, distribution, and movement at scales relevant to sea lions

Currently NMFS conducts regular (annual or biennial) groundfish surveys, which provide limited information on abundance, distribution, and movements over broad areas. However, the groundfish surveys are not designed to provide information at the spatial and temporal scales relevant to foraging sea lions. Lack of precise information on the distribution of fish stock biomass at both small and large spatial-temporal scales is a major impediment to quantifying the impact of commercial fisheries on sea lion prey and foraging success. Fish survey information is lacking for most areas outside the summer season, and the surveys are not intended to provide estimates of prey biomass in specific locales within sea lion critical habitat. New surveys should be designed and conducted to determine the abundance, distribution and movement of fish at smaller temporal and spatial scales than done currently, and throughout the year.

2.6.2 Assess competition for prey with sympatric consumers (e.g., gadids and flatfish, fur seals, harbor seals, other marine mammals, and seabirds)

Food habits data has been collected in the North Pacific region and analyzed over many years, notably for species such as the northern fur seal and more recently for sea lions. Data have also been collected for seals, cetaceans, major seabird species, and some fish species. In addition, the fishery observer database provides information on the distribution of catch inside and outside of sea lion foraging areas, as well as estimates of the composition and weight of the catch by species. Including these data in ecosystem food web models should be continued in order to evaluate the potential effects of fisheries removals of other species (e.g., cod or halibut) which may compete directly or indirectly with sea lions for common prey (e.g., pollock).

Food web models can provide valuable insight into potential food web dynamics and identify information gaps for future research, and as such they are useful tools for assessing the potential competitive linkages between Steller sea lions and other consumers in the ecosystem. However, these models are not substitutes for gathering more diet and food habits data. Much of the food habits data collected prior to the 1990s was gathered opportunistically and provides limited insight, yet this information is driving the ecosystem models in use today. Thus there is a need for more and better food habits data and analyses to understand more fully the potential impacts of interspecific competition within the ecosystem. Fishery observers should be tasked with collecting stomach samples from target and incidental catch species to improve the data used in ecosystem food web models. In addition, marine mammal biologists should also continue to collect food habits information either directly (e.g., scat) or indirectly (e.g., biopsy for fatty acid and stable isotope analyses).

2.6.3 Utilize groundfish fishery observer data to assess the spatial-temporal distribution of the fishery

Commercial groundfish fisheries can have significant short and long term effects on prey availability for sea lions, including depletion of the target species in the immediate vicinity of the fished area. The fishery observer database provides valuable information on the distribution of fishing effort inside and outside of critical habitat as well as a description of the amount of fish harvested and the method employed. NMFS has utilized this database extensively in the past to document trends and patterns of fishing within sea lion critical habitat and to evaluate the performance of fisheries regulations. NMFS should continue to compile and update this information on a regular basis in order to evaluate the performance of fisheries regulations and to identify potential concerns. Data on catch in critical habitat (e.g., zones such as 0-10 nm, 10-20 nm, critical habitat foraging areas) should be provided as part of the NMFS's Office of Sustainable Fisheries' catch accounting responsibilities.

2.6.4 Assess effectiveness of sea lion closure zones around rookeries and haulouts using small-scale experiments

Small-scale research experiments have been undertaken by NMFS to evaluate whether fishing has the potential to impact sea lion prey resources within critical habitat and whether subsequent closure zones are effective in mitigating potential effects. Preliminary field experiments have been conducted for pollock, Pacific cod, and Atka mackerel fisheries with mixed results. Some of these efforts should be continued or modified as described below.

For pollock, at least one more year of a field experiment off Kodiak Island in the GOA is needed in order to reach an initial conclusion about the effects of fishing on pollock in this area. Of the two years during which there were sufficient commercial removals, one year showed a pollock response to fishing and one year did not. Field work beyond this minimal additional year is also needed to understand the observed interannual variability in the fishery response.

For Pacific cod, three years of field experimentation off Cape Sarichef in the EBS showed no statistically significant fishery effect. One conclusion is that fish movement through the study area displaced or dispersed a fishery effect. Some of that movement may be linked to spawning behavior. In order to understand the space and time scales that fisheries can impact a fish like Pacific cod, further study is needed to understand their movement patterns and spawning behavior. Movement could be studied through tagging and spawning through the collection of maturity data (primarily through the fishery observer program).

For Atka mackerel, the tag release-recovery studies in the eastern and central Aleutians (Seguam and Tanaga areas) showed high Atka mackerel biomass and low movement across trawl exclusion zone boundaries. These are conditions that would maintain localized quantities of forage for sea lions. However, further west, at Amchitka, tagging data shows much lower biomass and high movement rates across trawl exclusion zone boundaries. In this area, fisheries have a greater potential to impact local abundance of Atka mackerel. This is of concern because pup counts are still showing declines in the western Aleutians (compared to increases in nearly all other areas). Further tag release and recovery work needs to be done in the western Aleutians to determine whether there are regional scale differences in Atka mackerel abundance and local movement patterns.

2.6.5 Assess the response of sea lions to changes in prey distribution and availability

Efforts to protect Steller sea lion prey within critical foraging areas will be enhanced by an improved understanding of how sea lions use that habitat and how they respond to changes in the prey field. This entails further deployments of telemetry tracking instruments on sea lions (2.3.3), as well as improved assessments of prey biomass within critical foraging areas (2.6.1) and continued monitoring and documentation of fishery operations within these areas (2.6.3 and 2.6.4). Further efforts should be made to integrate these activities so that changes in the prey field are monitored and manipulated at the same time Steller sea lions are observed to determine if there are any changes in foraging activity or body condition.

2.6.6 Evaluate and implement current or equivalent fishery regulations to protect foraging habitat and prey resources for sea lions

In the late 1990s and early 2000s NMFS reviewed federally managed groundfish fisheries in a series of consultations under section 7 of the ESA. Two of those consultations resulted in a determination that these commercial fisheries were likely to jeopardize the continued existence of the western DPS of Steller sea lion and adversely modify its critical habitat. Therefore, as required under the ESA, additional conservation measures were implemented to avoid jeopardy and adverse modification. The expectation was that these measures would promote the recovery of Steller sea lions in areas where potential competition from commercial fisheries may have contributed to the population decline.

The increase in Steller sea lion numbers in the early 2000s corresponds to implementation of a suite of fishery conservation measures. Such conservation measures should be maintained until it can be positively determined that reducing those protections for Steller sea lions would not reduce the likelihood for or increase the time to recovery. This includes protections developed to 1) avoid disturbance and competition around rookeries and major haulouts, 2) avoid competition during the early winter season, 3) disperse the fisheries spatially, and 4) disperse the fisheries temporally. New information on the foraging needs of juvenile and adult female sea lions should be integrated into these conservation measures at regular intervals such that they are successful in protecting the important and potentially vulnerable components of the sea lion population.

Fishery management policies and plans must take into account the types and amounts of food needed to support a recovering sea lion population. Sea lion food requirements should be explicitly accounted for in setting acceptable biological catches of groundfish, and the methods used in these calculations should be described.

2.6.7 Explore the use of ecosystem based (multi-species) stock assessment models to set fishery catch limits to ensure adequate prey resources for a recovered sea lion population

Although NMFS supports the development of ecosystem-based stock assessments, all Alaska groundfish stock assessments are based on single species methodologies that do not explicitly consider other consumers in the ecosystem. NMFS should ensure that fisheries for important sea lion prey (e.g., Pacific cod, pollock, Atka mackerel) are not limiting recovery of sea lion populations by exploring ways to explicitly account for the needs of Steller sea lions or other consumers in the ecosystem.

2.6.8 Design and implement an adaptive management program for fisheries, climate change, and predation

The mechanisms by which different threats affect sea lions can be similar, as are the responses that sea lions exhibit to these different threats. This represents a fundamental difficulty in identifying which threats are impeding recovery and which mitigation measures would be effective. A properly designed and implemented adaptive management program is needed to assess the relative impact of fisheries, climate change, and predation (Bowen *et al.* 2001, NRC 2003). This program will require a robust experimental design with replication at the proper temporal and spatial scales, and significant amounts of commercial fishing such that an effect (if it exists) could be detected. Given signs of a recovery in the western DPS, it is important to take this opportunity to implement an adaptive management program to test the underlying hypotheses of the conservation measures. With an increasing population, NMFS will have more latitude under the ESA and MMPA to implement a fisheries program which might result in more take of sea lions in order to attain valuable information about the efficacy of conservation measures and the relative contribution of climate change and predation as threats to recovery.

As noted by Bowen *et al.* (2001), field experiments in the open ocean at this spatial scale have not been attempted before, and any experiment must be carefully designed to address a host of difficult issues regarding the size and numbers of experimental units, the length of time to run the experiment, the response variables to be measured, the means of detecting change in those variables, and the likelihood of distinguishing between fishing and natural factors. Coordination among agencies and organizations involved in the development and design of such an experiment should be provided by the Steller sea lion Recovery Plan Coordinator.

2.6.9 Prepare a habitat conservation plan under section 10 of the ESA for fisheries authorized by the State of Alaska

Near shore fisheries authorized by the State of Alaska interact with Steller sea lions and are likely to result in adverse effects including both sub-lethal and lethal takes. These fisheries (e.g., salmon, herring, and groundfish) are not currently authorized to take Steller sea lions under the ESA. Studies have been funded through the Alaska Department of Fish and Game to look into the competitive overlap between State-managed fisheries and Steller sea lions. These analyses should provide the basis for an ESA habitat conservation plan (section 10) to minimize the take of Steller sea lions while providing the legal authority for incidental take under a section 10(a)(1)(B) permit for commercial and sport fisheries. If adverse impacts are found during the development of the habitat conservation plan, conservation measures should be determined and adopted in order to mitigate fisheries impacts.

2.6.10 Consider and implement conservation measures in herring and salmon fisheries in Alaska as appropriate

Sea lion mitigation measures in state waters have been implemented only for pollock, Pacific cod, and Atka mackerel parallel fisheries. Conservation measures for other fisheries which affect Steller sea lion prey should also be considered (as described above under a section 10 habitat conservation plan). State fisheries for herring and salmon should be further investigated for potential competitive overlap with Steller sea lions. Both herring and salmon are important prey items for sea lions at various locations and times of the year. Relatively high harvest rates in the herring fisheries may also impact sea lions through the local depletion of the herring stocks. The relationship between the herring fishery and foraging sea lions should be investigated further as these ephemeral foraging events on spawning herring, as well as foraging on over-wintering herring, may be critical to sea lion health, condition and reproduction.

3 Protect from over-utilization for commercial, recreational, scientific, or educational purposes

3.1 Minimize threat of incidental take in fisheries

The threat of incidental mortality in fisheries is considered medium both in “Frequency of Occurrence” and “Relative Impact.” Although some Alaska fisheries have extensive observer coverage, some have insufficient observer coverage to estimate or monitor the rate of incidental mortality needed to adequately assess incidental mortality’s impact on the sea lion population. Reducing the potential threat of incidental mortality will require understanding the conditions that result in entanglement, information that can be gathered by fishery observers or cooperative research programs. Knowledge of the nature of entanglement is key to developing mitigation strategies whose implementation should be encouraged through outreach and enforcement.

3.1.1 Monitor and evaluate incidental take in commercial and recreational fisheries through observer and self-reporting programs

Observers have collected data on the incidental take rates of Steller sea lion in several state and federal fisheries. Many Category II and III fisheries under the MMPA, however, have had no dedicated observer programs to assess their potential threat to Steller sea lions. Fishery-specific observers can gather data on the frequency, circumstances, and species caught incidentally in many fisheries. These data can be used to derive mortality estimates and document the conditions in which sea lions are most vulnerable to incidental entanglement, injury, or death.

In fisheries where the rate of incidental mortality is low, deriving statistically reliable mortality estimates may be cost-prohibitive. In such cases, NMFS should seek cost-effective means such as repeated beachcast carcass surveys to detect fishery-related mortality. Dedicated observer effort could subsequently be focused on the fisheries, areas, and seasons most needed.

The NMFS Incidental Marine Mammal Take Report Form allows fishermen to document the circumstances surrounding incidental take in commercial fishing gear. Although the form is available on the web, additional effort should be made to advertise the reporting requirements and provide hard copies to commercial fishermen. Copies of this form should be distributed at commercial fishery trade shows and in port offices. An annual reminder of reporting requirements and at least one copy of the Incidental Marine Mammal Take Reporting form should be sent to each state and federal fishery permit holder, possibly with Marine Mammal Exemption Program registration information. There is no incentive for fishers to report incidental takes and draw attention to their fishery so few have been submitted to NMFS. Cooperative Research Programs could be developed within regions where multiple fisheries are suspected of incidentally taking Steller sea lions in order to collect necessary data on gear interactions and potential mitigation measures while providing a degree of anonymity to fishers experiencing takes.

3.1.2 Monitor and evaluate incidental take in non-commercial fisheries

A number of non-commercial fisheries are conducted in state waters that have the potential for interacting with Steller sea lions, including recreational and personal use fisheries, as well as scientific sampling (test fishing). Steller sea lions may become entangled in gillnets or swallow baited hooks used in recreational and personal-use fisheries, at salmon hatcheries, and in state-managed test fisheries. Currently, no system exists that provides for or encourages standardized reporting of Steller sea lion injury or mortality in these fisheries. NMFS should work with ADF&G to develop a system for documenting the frequency, location, and outcome of reported incidental entanglement. The resulting database should be linked to NMFS's Alaska Marine Mammal Observer Program and Alaska Marine Mammal Stranding Network databases. NMFS should issue an MMPA section 101(a)(5)(E) permit to the State of Alaska to authorize the incidental take of listed marine mammals in state-managed fisheries. This will require the preparation of a National Environmental Policy Act document and completion of an ESA section 7 consultation.

3.2 Minimize threat of intentional killing in fisheries

Although the direct killing of Steller sea lions in the course of commercial fisheries has been banned since 1990, occasional cases of intentional illegal shooting are reported and prosecuted. Some have involved shooting as a lethal means of actively deterring Steller sea lions that are stealing fish from commercial fishing gear. Other cases have involved proactive and lethal efforts to remove Steller sea lions from broad fishing areas. Therefore minimizing the threat of intentional take may involve regionally-oriented education, mitigation, and/or prosecution of offenders.

3.2.1 Monitor intentional take via shoreline surveys for carcasses near suspected conflict "hotspots" and by encouraging reporting of illegal shooting through NMFS's enforcement hotline

Because it is illegal, intentional killing of Steller sea lions is rarely documented by observers and no reports of lethal deterrence in commercial fisheries have been recorded by fishermen or observers since the practice was banned. Therefore, alternative efforts may be needed to detect and monitor the occurrence of illegal, intentional shooting of Steller sea lions (if it occurs). Since 1990, two convictions have resulted from confidential voluntary reports from commercial fishermen who witnessed and reported the violations to NMFS Enforcement agents. NMFS should encourage such confidential reporting by publicizing hotline numbers at fishery trade shows and in Marine Mammal Exemption Program mail-outs. Systematic surveys of shorelines have successfully located gunshot carcasses of Steller sea lions in several regions of Alaska (see 5.3). In addition to using these surveys to focus further enforcement surveillance, forensic examination of carcasses may yield ballistic information with which to convict illegal shooters. All violations should be prosecuted to the fullest degree possible.

3.2.2 Reduce threat of illegal shooting by developing and promoting use of non-lethal deterrents for commercial fisherman

In some areas Steller sea lions are adept predators on fish caught in commercial fishing gear. Shooting at sea lions to deter them from stealing fish and damaging gear has been banned since 1990 but no non-lethal deterrents have been promoted as alternatives. Such alternatives should be developed, tested, and promoted for use by commercial fishermen in areas where conflicts between sea lions and fisheries exist. A viable deterrent would reduce the threat of incidental capture and illegal shooting in several commercial fisheries.

3.3 Minimize frequency and severity of sea lion-human interactions in ports and harbors

Human encounters with Steller sea lions have occurred in several harbors where Steller sea lions have become habituated to human activities. Fishermen and dock workers may both intentionally and inadvertently feed Steller sea lions with fish from boat decks and at processing stations. This attraction to harbors may increase Steller sea lion vulnerability to ship-strikes, coastal pollutants, and potential predation by killer whales. In addition, habituated sea lions have hauled out on public and commercial docks, obstructed harbor structures, or posed a threat to human safety by charging and biting people.

3.3.1 Develop and promote non-lethal means of deterring sea lions from hauling out on docks

Currently, enforcement agents may shoot sea lions that pose an immediate threat to human safety. Although this has not yet occurred, the frequency and severity of human-sea lion encounters may intensify as local Steller sea lion numbers and harbor use increase. Non-lethal deterrent techniques or devices need to be developed (see 3.2.2) and made available to harbor enforcement officials to minimize the possibility that deadly force will be needed when sea lions pose a threat to human safety. Port and harbor personnel should be assisted in identifying and providing alternative haulout opportunities for Steller sea lions in harbors where they currently use public facilities and pose a threat to human users.

3.3.2 Continue to publicize "No feeding" regulations in harbor areas and keep active programs for notification and enforcement

Steller sea lions have been drawn to some Alaskan harbors to scavenge fish remains discarded by sport and commercial fishermen. NMFS should remind fish processors and the public of the potential hazards of and prohibition on feeding Steller sea lions at the beginning of each boating or fishing season. Harbor personnel should be encouraged to provide and maintain fish waste stations for processing sport-caught fish. NMFS should continue posting “Do Not Feed Steller Sea Lions” signs near fish waste stations and processing docks and enforce violations of this prohibited take.

3.4 Minimize take by recreational and commercial viewing operations

The potential exists for recreational and commercial wildlife viewing activities to incidentally take Steller sea lions. Boaters and pilots that approach haulouts and rookeries may disrupt sea lion resting, breeding, and nursing activities or spook animals into the water where they may be more susceptible to predation and ship strike. The potential for disturbance from such approaches varies with the type of approach (vehicle, proximity, speed, duration) as well as seasonal and site-specific Steller sea lion behavior. Although 3-mile no-transit zones are established and enforced around rookeries, restrictions on approaching Steller sea lions on haulouts are poorly defined and publicized in minimum approach guidelines

3.4.1 Publicize and enforce existing no-transit areas to minimize vessel and aircraft disturbance of rookery sites

Commercial and recreational mariners should be reminded annually of the existence and location of the 3 nm No-Transit areas around rookeries via print media and seasonal Notice to Mariners announcements. U.S. Coast Guard and coastal air charter pilots should be reminded annually of the rationale and limits to minimum altitudinal passage over Steller sea lion rookery sites. NMFS Enforcement agents should investigate reported violations and prosecute offenders when possible.

3.4.2 Review and revise existing Marine Mammal Approach Guidelines and provide to charter operators and other mariners to minimize disturbance at haulouts

Mariners in many areas have the opportunity to approach and view Steller sea lions resting on terrestrial haulouts. Although the MMPA and ESA prohibit the harassment of Steller sea lions during such approach, this activity is not clearly restricted or regulated. NMFS has provided guidelines for mariners to stay at least 100 yds from the animals in order to prevent harassment. However, in areas where Steller sea lions are habituated to humans and vessel traffic, approaches closer than 100yds may not elicit a response from animals on haulouts. On more remote sites and under other circumstance, animals may spook at greater distances. NMFS should review approach guidelines specific to Steller sea lions, update or modify them as appropriate, and consider developing guidelines into formal regulations. NMFS should then post guidelines and regulations for viewing Steller sea lions online and make printed versions available to mariners and charter operators at the onset of each boating season.

3.5 Evaluate and reduce the direct and indirect impacts of research activities

Lethal or non-lethal incidental take of Steller sea lions may occur in the course of bona fide research. These activities are authorized under the MMPA and ESA and are regulated through a scientific permit process. Permit applications are reviewed by NMFS Permit office and the Marine Mammal Commission and are available for public review in the Federal Register. Researchers are required to submit annual plans and reports of research activities and real-time reports of research-related mortality. A Regional Coordinator monitors the cumulative impacts of multiple projects and may curtail such research if incidental mortalities reach a permitted cap.

3.5.1 Coordinate research efforts to reduce potential for unnecessary or duplicative research-related takes

The Regional Coordinator should continue to maximize coordination, minimize duplication, and enhance collaboration of Steller sea lion research efforts. NMFS should consider convening periodic regional meetings of authorized Steller sea lion researchers to help coordinate collaborative efforts, discuss emerging technologies, and minimize cumulative effects of permitted research.

3.5.2 Monitor and minimize unintentional takes associated with research activities

When possible, researchers should use new technologies that reduce disturbance, potential mortality, and the need for invasive methods (e.g. safer drugs for anesthesia, photography for census work). Approach and handling methods should be reviewed periodically to minimize the potential for injury or mortality from permitted activities. In addition, studies should be undertaken where possible to evaluate the effects of disturbance caused by particular research activities with the objective of quantifying what if any unintended takes occur and how they could be minimized.

4 Protect from diseases, contaminants, and predation

4.1 Protect Steller sea lions from disease

Disease agents can affect a population directly by affecting survival, growth and body condition; and indirectly by having reproductive effects. Monitoring diseases in a population is important, because diseases rarely occur apart from the influence of other factors such as nutritional status. Therefore, monitoring changes in disease and the appearance of new diseases is another way to monitor populations. Disease agents can affect a population directly by affecting survival, growth, and body condition and indirectly by affecting reproduction.

4.1.1 Conduct epidemiological surveys

Periodic analysis for agents and exposure to agents with potential to affect survival, growth and body condition (e.g. ,nematodes, leptospirosis, viruses) and those with potential to cause reproductive effects (eg. Chlamydia, Toxoplasma gondii, herpesvirus, caliciviruses, Brucella) should be monitored. To date, a collective survey for most of these has been conducted and published (Burek *et al.* 2003, 2005). In addition a survey of young age classes in all seasons has been completed in Southeast Alaska and can serve as baseline data. Continued analysis of blood samples collected in the western DPS should continue to reach similar baseline levels. Blood samples from all handled sea lions should be archived for retrospective analyses if necessary. Monitoring should include analysis for agents and exposure to agents with potential to affect survival, growth, and body condition (eg. nematodes, leptospirosis, viruses) and those with potential to cause reproductive effects (eg. Chlamydia, Toxoplasma gondii, herpesvirus, caliciviruses, Brucella).

4.1.2 Develop and implement methods for parasite evaluations

Information is needed on the prevalence and intensity of parasite infections in Steller sea lions to determine the pathogenicity of these parasites and whether the parasites are causing significant mortality or other effects on the overall condition of the animals. Collecting samples of feces and other bodily fluids from live animals is needed for assessment of the intensity and effects of infestations in dead animals.

Specifically, focused work is needed to assess the prevalence and intensity of hookworm (Uncineria spp.) in Steller sea lions. Hookworm infestation has been documented in this species (Olsen 1958) and has been associated with decreased body condition and increased mortality in other pinniped pups, such as California sea lions (Lyons *et al.* 2001). To gain an understanding of this infection in Steller sea lions, we need to determine the prevalence of infection at the time of branding based upon fecal loop samples, the prevalence and duration of infections based upon fecal samples, the prevalence and duration of infections based upon fecal samples of older pups (2-10 months of age), and the intensity of infections from necropsies.

4.1.3 Develop and implement methods to test immune system functioning

Factors such as infectious disease, diet, endocrine hormones, and stress have effects on the immune system and its response. Methods are needed to assess both humoral and cellular immune function in this species. The development of Steller sea lion specific reagents and diagnostic tests is needed to improve our ability to monitor their health and immune responses. Humoral immune function can be evaluated by measuring isotype levels in serum as part of assessing general health status and by measuring antibodies to assess past and present exposure to pathogens. Lymphocyte function assays can be used to assess cellular immunity. Major histocompatibility complex (MHC) gene analysis (immune response genes) can be performed to identify the different genotypes and phenotypes on the population. Polymorphism of these genes within a population is thought to be needed for immunologic vigor (i.e. a correlation with population health).

4.1.4 Evaluate causes of mortality by examining dead and live animals of all age and sex classes for disease from various sources across the geographic range and in all seasons

A better understanding of the causes of mortality of Steller sea lions is needed. To accomplish this, the number of dead and moribund animals examined annually needs to increase. Necropsy is one of the best tools for determining causes of mortality. This can be accomplished through further development of the stranding network in Alaska, examination of fisheries by-catch animals, subsistence harvested animals and through rehabilitation efforts. Existing organizations and individuals should be informed of Steller sea lion strandings, and the data should be catalogued and maintained by a central agency. Necropsies should be performed in a systematic manner and accompanied by histologic examination and processing of tissues collected from all age classes of animals. To facilitate this, standardized necropsy protocols need to be developed; protocols and sample collection kits should be made available to biologists on capture cruises and at field stations, observers on boats, subsistence hunters and others; and training opportunities should be provided where needed. Samples to be collected should include: tissues for histology, contaminant analysis, future disease studies (archived samples), and DNA analysis (stock identification); stomach contents for diet analysis; female reproductive tracts for monitoring reproductive status; feces; blood; and other bodily fluids.

A priority list for observations, measurements, and specimens to be collected should be made available to state fish and game agencies, federal research and management teams, subsistence hunters, rehabilitation organizations and other appropriate groups throughout the Steller sea lion range.

4.1.5 Develop disease management plans

To monitor changes in health and disease over time, and to compare future work to current data, a central database with baseline data should be developed. Methods for sample collection, laboratory analysis and data synthesis should be standardized. This will help ensure that all data collected by different groups is comparable. Health investigations should focus on monitoring for detection of future events. Disease surveillance studies should focus on expected routes of introduction of disease into the population and should be adapted as new diseases emerge or new research techniques become available. Consideration should be given to designing studies based on geographic and oceanographic features rather than on population-based points of capture.

Due to the difficulties associated with obtaining large numbers of samples from all ages of Steller sea lions, the use of data collected (or that could be collected) and analyzed from sympatric species (California sea lions, northern fur seals, harbor seals) should be considered. A comprehensive disease monitoring plan should include monitoring of changes in specific diseases and detection of new and emerging disease agents.

4.1.6 Develop an unusual mortality event (UME) management plan

Having a UME management plan would facilitate a response to any emergent UMEs that may occur in Steller sea lions. Identifying high risk events resulting from natural or anthropogenic causes (e.g. oil spill, biotoxin, contaminants, disease) and developing specific plans will allow resources (funds, personnel, diagnostic labs, equipment) to be mobilized for immediate response to the emergence of such events. Such contingency plans have been developed for other species and could be used as a template for the Steller sea lion plan. The plan should include: 1) criteria to determine what the triggers should be to initiate such a response, as well as criteria that would lead to increased sampling consideration versus monitoring; 2) protocols for capturing and marking live animals to monitor the event and to determine when it is over; and 3) protocols for debriefing and disinfection of supplies and instruments to prevent any disturbance or spread of disease as a result of the response.

4.1.7 Develop models to simulate disease impacts based on energetics, physiology, abundance and demographics.

Simulations can be used to evaluate disease impacts and effects on the abundance of Steller sea lions at small spatial scales (rookeries) and large (populations). A growing need exists to integrate the biological data with population-scale dynamics. But modeling efforts have lagged behind. This is likely due to the inability of scientists and resource managers to provide the necessary information for mathematicians and modelers to be able to develop meaningful models. As our ability to obtain and provide useful data improves, so should the ability to create useful models.

4.2 Protect sea lions from contaminants

The number of potentially toxic compounds that are being developed is growing rapidly, and these compounds can potentially impact many marine animals, including Steller sea lions, in a myriad of ways.

4.2.1 Design a contaminant research and management plan

To assess any potential health impacts, scientists should develop methods to assess health related to contaminants and collect and analyze samples for potential contaminants, including endocrine disrupting and immunosuppressive chemicals. Contaminant management plans should be developed to prepare for the mitigation of any contaminants that have the potential to be released in the natural habitat of Steller sea lions.

4.2.2 Collect samples from free-ranging sea lions and in environmental "hotspots"

Collect samples from dead and live animals of all age and sex classes from various sources across the geographic range in all seasons. Analyze samples for the presence of organochlorines, other persistent organic compounds, heavy metals, and other potential contaminants, including endocrine disrupting and immunosuppressive chemicals. Compare with environmental samples collected from 'hot spots' that could be used to help identify the source of the contamination and focus mitigation measures. Non-point sources of contamination will be more difficult to identify or mitigate, but any environmental sampling that can assist with the process should be encouraged.

4.2.3 Examine blood and tissue samples for evidence of contaminant-linked endocrine effects including free-ranging and captive work

One of the primary mechanisms by which contaminants affect an animal is through the endocrine system. Although much work has been done on developing baseline endocrine values for many terrestrial mammals, little work has been done on marine mammals. Given that some hormones are highly conserved across species (i.e., sex steroids), the development of baseline concentrations of these hormones for marine mammals, particularly Steller sea lions, should be readily obtained. Peptide hormones, however, tend to be species-specific, and the development of validated assays and subsequent baseline concentrations for these will be more difficult. The mechanism by which endocrine disruption due to contaminants impacts the overall health of the animal can be very complex. In addition to baseline concentrations, alterations in endocrine profiles due to altered physiological states (during gestation, lactation, or fasting), are virtually unknown. Once these profiles are established, the impact of endocrine disruption from contamination can be better evaluated.

4.2.4 Develop models to simulate contaminant impacts and effects based on energetics, physiology, abundance and demographics

Simulations can be used to evaluate the impacts of contaminants and effects on the abundance of sea lions on small spatial scales (rookeries) and large (populations).

4.3 Predation

Predation may be a significant source of mortality of pinnipeds, including Steller sea lions. Predation rates on particular prey species can be affected by the health status of individuals and the availability of alternate prey. The effect of predation on prey populations can increase when those populations are small. Thus, predation as a factor in the dynamics of Steller sea lions must be evaluated.

4.3.1 Understand predator life histories, biology and ecology – captive work

The principal predators of Steller sea lions are killer whales. Of the three ecotypes of killer whales, only transient killer whales are thought to prey on marine mammals. Among transient killer whales, diets vary by region and pod, and pods may travel over long distances. It is necessary to identify transient killer whales within the range of Steller sea lions and determine their diets, distribution, movements, and abundance. Likewise, scientists need to understand the behavior of killer whales when foraging, especially as it may relate to prey switching or seasonal changes, to assess any potential impact on prey populations. The study of captive killer whales to better understand the bioenergetics or physiology of this species should be encouraged. These animals have the ability to support longitudinal studies that would not be feasible in the free-ranging setting.

4.3.2 Determine killer whale diets

Knowledge of diets of transient killer whales is critical to evaluating effects they may have on prey populations. Three primary means of estimating diet are analysis of stomach contents of stranded whales, analyzing ratios of stable isotopes of carbon and nitrogen in bone and soft tissues, and using fatty acids in blubber to infer the composition of prey. While these are established methods, other methods to identify diet should be encouraged. Observations of feeding killer whales at sea can provide additional valuable information on diet.

4.3.3 Develop methods to obtain samples from live killer whales

New and refined methods to obtain samples from live killer whales need to be developed. Biopsy samples used to obtain tissue samples for use in studies of genetics and diet determination are currently the primary method of collection and are limited by range. Continued development of methods would assist in increasing the sample size.

4.3.4 Expand the stranding network to increase samples of killer whales available for research

Killer whales die and wash ashore in various places. Many locations are remote and populated by few people. People who observe beached killer whales in these areas might not report them to scientists. However, samples from stranded killer whales can be of significant value in determining the diet of killer whales. The existing stranding network should be expanded to accommodate killer whale sample collection for diet analysis.

4.3.5 Determine killer whale distribution and behavior across the North Pacific

Killer whales are common in coastal waters and can be viewed by following them in vessels while they forage and by recording them with remote cameras at rookeries and haulouts. Much can be learned about predation behavior and rates, diets, and distributions of transient killer whales by following pods of foraging whales. This method is most successful in summer when weather conditions are favorable, but is needed during other seasons, as well. Cameras deployed at Steller sea lion rookeries may provide important information on predation rates, the age structure and behaviors of animals preyed upon, the numbers of whales involved, the pod composition, and identities of individual killer whales. Other technologies, such as acoustic sensors and satellite-linked UHF tags, should be developed to enhance our knowledge of the long-term movement and foraging ecology of killer whales. Included in this task is the continued development of tags and tag delivery methods to increase deployment duration.

4.3.6 Estimate numbers of killer whale ecotypes in time and space

As only transient killer whales are thought to prey on marine mammals, it is necessary to identify and census them to calculate their proportion of the killer whale population Information on their abundance and distribution in space and time is needed to assess the impact of predation. Killer whale ecotypes can be recognized by their distinctive physical markings and genetic characteristics.

4.3.7 Develop models to simulate predation rates based on killer whale energetics and abundance of Steller sea lion demographics

Simulations can be used to evaluate predation pressure and effects on the abundance of Steller sea lions at small (rookeries) and large spatial scales (populations).

5 Protect from other natural or anthropogenic factors and administer the recovery program

Principal responsibility for implementation of the Steller Sea Lion Recovery Plan lies with the NMFS Office of Protected Resources. Recovery actions will need to be coordinated with the NMFS regional offices as well as with other federal and state resource management agencies and user groups. Moreover, education and public outreach will be vital to full operation of this plan. Estimated costs outlined in this section are those attributed to NMFS; costs that may accrue to other agencies and user groups are not estimated here.

5.1 Reduce damage to sea lions and their habitat from discharges of pollutants by developing preventive measures

The discharge of pollutants poses a threat to sea lions but can be mitigated through the monitoring and evaluation of potential sources of pollution and the implementation of preventative measures by regulatory agencies. Industrial, agricultural, and urban sources of pollution are relatively uncommon in areas of Alaska adjacent to Steller sea lion habitat, but offshore oil and gas leasing and the marine transport of oil pose threats that should be continuously monitored and evaluated. This should be accomplished through the Office of Protected Resources' review of oil and gas lease sales, oil spill contingency plans and other permitting and approval processes.

5.2 Reduce the potential for sea lion entanglement by improving and continuing programs aimed at reducing marine debris

Observations of sea lions at sea and on land indicate that entanglement in debris is continuous if infrequent. Efforts to estimate the extent and type of such entanglement are important to help identify the entangling material and its source, to foster the reduction of such debris when possible. Current programs focused upon the generalized reduction of discard at sea (e.g., MARPOL) and those focused upon the commercial fishing industry need to be continued and expanded.

5.2.1 Reduce discards of debris (e.g., trawl web and packing bands)

Previous efforts should be resurrected to educate fishermen and other mariners on various prohibitions against discarding refuse at sea and on the effects of such materials in strangling and otherwise adversely affecting sea lions, similar to the "Don't teach your trash to swim" campaign of the Pacific States Marine Fisheries Commission.

5.2.2 Cleanup derelict gear and beached debris

In particularly sensitive areas surrounding major and accessible rookeries and haul-outs, beach cleanup initiatives should be promoted in order both to remove marine debris that might affect sea lions and to educate the public on the need to prevent such debris from being discarded in the first place.

5.3 Monitor causes of sea lion mortality and use data to direct management actions

This section focuses on the general need to monitor sea lion mortality and strandings and use that information to promote recovery.

5.3.1 Continue and expand the Alaska stranding network to increase coastal coverage and community involvement in monitoring sea lion mortality

Marine mammal stranding networks have provided valuable data for determining regional, seasonal, and annual patterns of marine mammal mortality. By locating sick and dead animals, samples and data from stranding networks have been used to detect and monitor the incidence of both anthropogenic (e.g. gunshot, fishing gear entanglement) and natural agents of mortality (e.g. episodic outbreaks of leptospirosis, domoic acid toxicity, and San Miguel sea lion virus) in pinniped populations.

Through aggressive public notices, people in coastal communities should be informed that Steller sea lion strandings, regardless of condition, are of particular significance and should be reported to local stranding responders immediately. The number of organizations and individuals trained to respond to strandings and collect Level A stranding data has increased in Alaska with the growing involvement of Alaska Native NGOs and others in the Alaska Marine Mammal Stranding Network. However, efforts should be made to train and support additional stranding responders in remote Alaskan communities and provide them the tools and support needed to contribute viable samples and data collected from stranded Steller sea lions. Continued training, guidelines, and protocols should be provided.

5.3.2 Survey selected areas for stranded animals

Efforts should be made to identify sites where there is a high probability of being able to locate and sample dead animals. This effort will need to be coordinated with activities of stranding networks, aerial survey crews, field enforcement, and other field activities. For example, dead pups should be counted and sampled during pup counts when pups have been separated from adults, but rookeries should not be disturbed solely to recover dead pups. If identified sites are not visited regularly as part of the activities listed above, it may be desirable to conduct special field efforts to examine and sample dead animals.

5.3.3 Expand tissue sampling efforts to improve the information obtained from dead sea lions

Accurately determining the cause of death of sea lions often requires collection of tissue samples. These samples need to be properly stored or transferred to a storage facility soon after death. Local stranding network members should be encouraged to respond immediately to reports of dead or moribund sea lions. In areas where such reports are frequent, these responders should be provided with adequate collection, storage, and shipping support to maximize the analytical value of the samples collected. Such efforts should be coordinated through the regional NMFS Stranding Networks.

5.3.4 Monitor the incidence and impact of entanglement in marine debris

Entanglement of sea lions in marine debris occurs. There needs to be a continued effort to estimate the relative number of entangled animals in stranding data, by observers aboard vessels at sea, and during field research activities such as pup counts and observations at intensive study sites. When possible, the entangling material should be identified as this may provide clues about the circumstances under which sea lions become entangled. Literature surveys should be conducted regularly to update information on related marine mammal and bird entanglement incidents, and surveys of occurrence of entangling materials (packing bands, net fragments, etc.) at sea. Data should be used to support efforts to mitigate mortality.

5.4 Effectively administer the Steller sea lion recovery program by continuing to provide a recovery coordinator staff position

NMFS should maintain a full-time person to coordinate recovery efforts for Steller sea lions. Duties of the sea lion coordinator should include:

a. Coordinate all aspects of NMFS sea lion recovery efforts, such as evaluation and development of regulations, designation of critical habitat, and section 7 consultations;
b. Provide liaison with regional fishery management councils, state fishery managers, FWS wildlife refuge managers, enforcement agencies, researchers, and other interested parties;
c. Publish annual activity reports and work plans;
d. Facilitate and coordinate research activities, including development of scopes of work for contracts;
e. Coordinate data management, and facilitate data analysis and distribution.

5.5 Improve sea lion conservation by consulting with the State of Alaska on actions that are likely to adversely impact Steller sea lions

The State of Alaska exerts sovereignty over many activities within the territorial sea (0-3 miles from shore). To the extent that the management of fisheries, development of oil, gas, and minerals, or other activities might affect the recovery of Steller sea lions, it will be important that active consultation between NMFS and the State of Alaska continue. Direct liaison with the ADF&G and the Alaska Board of Fisheries has been successful and should be continued and expanded (see 2.6.9; 2.6.10). Interaction with the Alaska Department of Natural Resources, Alaska Department of Environmental Conservation, and the Alaska Department of Transportation and Public Facilities, among others, may be necessary on a case-by-case basis.

5.6 Conduct an effective outreach program to inform the public about Steller sea lion biology, habitat utilization, and conservation issues

Many of the regulations put into place to protect Steller sea lions will apply to all members of the public. Some conservation measures broadly affect mariners while others are specific to commercial fishermen. Public affairs personnel in responsible agencies should plan and implement well-rounded public awareness programs that describe the status of sea lions and the protective regulations that are in place. Types of coverage that have proven effective include websites, news releases, mail-outs, signs, public service announcements, interpretive programs, films, and environmental education lesson plans featuring sea lions.

5.6.1 Encourage and facilitate public reporting of sea lion observations

Coastal residents can provide valuable sea lion observational data if made aware of the need and provided a means to do so. The public should be made aware that they can aid in the Steller sea lion recovery effort by reporting observations of sea lion strandings and entanglement (see 5.3.1), branded and tagged animals (see 1.2.1), regulatory violations (see 3.2.1), and predation by killer whales (see 4.3.5). To facilitate public involvement, a local coordinator should encourage public reporting of observations, then coordinate and standardize the collection of resulting data. This coordinator may be a local stranding network member, enforcement officer, extension agent, or researcher. The coordinator should be familiar with sea lion conservation efforts and research needs to ensure that pertinent details are recorded and that appropriate entities receive the information. One avenue for the reporting of real-time observations of strandings and branded animals through the promotion of public outreach would be the creation and maintenance by the Office of Protected Resources of a dedicated website with instructions for the submission of local observations.

5.6.2 Publicize current conservation efforts and protective measures.

Many sea lion protective measures affect the activities of commercial fishermen, so it is particularly important that they be updated on the rationale and details of these measures. Information can be distributed as part of ongoing regulatory programs (e.g., in permit mail-outs, logbooks and regulation books), as well as through media directed specifically at the fishing industry (e.g., trade magazines). Materials and trained personnel should be made available to assist industry in developing its own additional educational programs. Fishermen and their representatives should be encouraged to continue their involvement in the development, evaluation, and implementation of sea lion conservation measures. To minimize disturbance at rookeries, all mariners and pilots should be reminded of no-transit zone boundaries through periodic media updates, Notices to Mariners, and updates to the Coast Pilot. Recreational and charter operators should be presented updates and reminders of 'approach guidelines' through pre-season meetings and/or press releases. Efforts to reduce human-sea lion interactions in harbors should be continued through posting of DO NOT FEED SEA LION signs and providing fish waste disposal options to recreational fishermen.

5.7 Co-manage Steller sea lion subsistence harvests in Alaska by developing co-management agreements as appropriate with Alaska tribes and tribally authorized Alaska Native Organizations (ANO)

Involving indigenous people in resource management has shown substantial benefits to the conservation of the species, and co-management regimes commonly occur throughout the circumpolar north for many different species. In Alaska, co-management agreements have been developed between the federal managing agencies and with tribes or tribally authorized organizations for many marine mammal species taken for subsistence in Alaska. Specifically, these agreements have provided a means for 1) collecting and analyzing data on marine populations, 2) monitoring harvest of marine mammals for subsistence, 3) participating in research, and 4) developing co-management structures.

The tribal governments of St. Paul and St. George currently have co- management agreements in place with NMFS, but these only affect Steller sea lions in proximity to the Pribilof Islands. Co-management agreements should be established between NMFS and Alaska tribal governments and/or tribally authorized ANOs for other areas used by both Steller sea lions and Alaska Natives.

5.7.1 Co-manage subsistence harvests and evaluate the efficacy and accuracy of using retrospective subsistence harvest surveys

Documenting the number, sex and age of Steller sea lions taken by Alaska Natives is critical to understanding the impact of the subsistence harvest on the population. Subsistence harvest monitoring should continue on an annual basis through co-management partnerships with ANOs and with Alaska tribal governments. Sea lion subsistence harvest has traditionally been monitored via retrospective volunteer survey of identified informants in approximately 60 coastal communities. Questions have been raised about the accuracy of the method and if it is the most appropriate monitoring method for an ESA listed species. These monitoring methods should be evaluated in conjunction with Alaska Tribes and tribally authorized ANOs. If necessary, adjustments to the method should be made.

5.7.2 Support Alaska Native subsistence use information programs

These programs may work to promote sound hunting techniques by developing culturally appropriate educational materials that recognize the traditional importance and uses of sea lions and the best hunting techniques and practices, through elder-youth mentoring and through cultural activities such as spirit camps and community celebrations. Alaska Native subsistence use of sea lions is rooted in the Alaska Native culture. Effective local conservation and management actions will be those that minimize conflicts with the local culture. Working with Alaska Natives to develop culturally appropriate educational materials that recognize the importance of sea lion subsistence use can provide a foundation for local management plans and conservation efforts. For example, elder-youth mentorship programs should be developed in the villages to train younger hunters in sound hunting practices, traditional uses, and cultural values such as respect for the resource and conservation. Cultural activities such as spirit camps, community celebrations, cultural revitalization activities including brochures, videos, and other materials may also provide avenues for encouraging responsible hunting.

5.7.3 Analyze carcasses from subsistence harvest to assess age, body condition, and other relevant information to ensure safety of carcasses for human consumption

Most of the marine mammal co-management agreements in Alaska include biological sampling programs whereby hunters and subsistence users are involved in data and sample collection of those animals taken for subsistence. A standardized necropsy protocol is needed from federal, State, and Tribal governments, including ANOs and NGOs. As part of the process, prior sea lion biosampling programs should be evaluated, and if necessary, locals trained in standard protocols for data and tissue collection for continued monitoring. Sampling kits and shipping support should be provided to hunters, along with a list of samples needed.

5.7.4 Document local knowledge and cultural science (Traditional Ecological Knowledge, TEK) pertaining to sea lions to better understand changes in sea lion movement (local and seasonal), feeding patterns and prey, seasonal haulouts, predation and ecosystem dynamics

Local and traditional knowledge studies have provided many insights into marine mammal life history and behavior, including movement, migration pattern, geographic expansion, seasonal and annual prey species, predators, etc. These surveys can provide a longer time perspective than commonly available. Current local and traditional knowledge surveys are underway in select places to document knowledge about movement, seasonal behavior and haulouts, and predator and prey interactions. These surveys should continue, and the geographic scale should be expanded throughout the state.

5.8 Improve the effectiveness of research for Steller sea lion recovery by instituting a "fast track" process for expediting NMFS research permits for Steller sea lions

This recovery plan has noted that scientific research is essential for understanding and mitigating the threats to Steller sea lion recovery. However, delays in the current permitting process by NMFS has slowed research activities and in some cases resulted in lost research opportunities. A new streamlined process should be investigated to reduce the permitting process for Steller sea lion related research to less than 6 months.

G. Recovery Action Implementation Schedule

The Implementation Schedule that follows outlines actions and estimated costs for the recovery program for the western DPS of Steller sea lion, as set forth in this recovery plan. It is a guide for meeting the recovery goals and criteria outlined in this plan. This schedule indicates action priorities, action numbers, action descriptions, duration of actions, the parties potentially responsible for actions (either funding or carrying out), and estimated costs. Parties believed to have authority or responsibility for implementing a specific recovery action are identified in the Implementation Schedule. When more than one party has been identified, the proposed lead party is indicated by an asterisk (*). The listing of a party in the Implementation Schedule does not require the identified party to implement the action(s) or to secure funding for implementing the action(s). Priority numbers are assigned as described below, which follow exactly the descriptions and approach required in the NMFS interim Recovery Planning Guidance. In addition, priority 2 actions are further split to distinguish priority within the extensive list of priority 2 actions: priority 2b actions are ones that follow priority 2a actions or are of secondary importance.

Priority Number

Priority 1 - An action that must be taken to prevent extinction or to prevent the species from declining irreversibly in the foreseeable future.

Priority 2 - An action that must be taken to prevent a significant decline in species population / habitat quality or some other significant impact short of extinction.

- **(a)** Actions that should either be taken first, or are of primary importance.
- **(b)** Actions that follow priority 2a actions, or are of secondary importance.

Priority 3 - All other actions necessary to provide for full recovery of the species.

In the threats column, M=monitoring activity; IT=incidental take by fisheries; SUB=Alaska native subsistence harvest; IS=illegal shooting; E=entanglement in marine debris; D/P=disease and parasitism; T=toxic substances; DVT=disturbance from vessel traffic and tourism; DR=disturbance from research; KW=killer whales; EV=environmental variability; F=competition with fisheries.

Plan Task	Priority	Responsible Parties	Task Duration	Fiscal Year Costs ($K)					Threats
				FY 1	FY 2	FY 3	FY 4	FY 5	
1. BASELINE POPULATION MONITORING									
1.1 Continue to estimate population trends for pups and non-pups									
1.1.1 Estimate trends for pups and non-pups via aerial surveys	1	NMFS	annual	*250*	*250*	*250*	*250*	*250*	M

Plan Task	**Priority**	**Responsible Parties**	**Task Duration**	**Fiscal Year Costs ($K)**					**Threats**
				FY 1	**FY 2**	**FY 3**	**FY 4**	**FY 5**	
1.1.2 Monitor population trends in the Pribilof Islands (particularly the Walrus Island rookery) via aerial surveys or land-based counts	2a	NMFS	annual	*50*	*50*	*50*	*50*	*50*	M
1.2 Estimate vital rates									
1.2.1 Continue to estimate survival, natality, and immigration/emigration rates through a branding/resight program	2a	NMFS, ADF&G	annual	*1,000*	*1,000*	*1,000*	*1,000*	*1,000*	M
1.2.2 Promote cooperative pup branding/resight programs in Russia	2a	NMFS, Russia	annual	*500*	*500*	*500*	*500*	*500*	M
1.2.3 Develop an age-structured population model using medium format photos from aerial surveys	2b	NMFS	1 yr	*20*					M
1.2.4 Determine pregnancy and parturition rates	2a	NMFS	annual	*30*	*30*	*30*	*30*	*30*	M
1.3 Monitor health, body condition, and reproductive success									
1.3.1 Examine the effects of season, age, and sex on body condition	2b	NMFS	annual	*500*	*500*	*500*	*500*	*500*	M,F,EV
1.3.2 Develop improved indices of health, body condition, and reproductive status using chemical methods (e.g., hematology serum chemistries, and endocrine monitoring)	2a	NMFS	10 yrs	*250*	*250*	*250*	*250*	*250*	M,D/P
1.4 Develop and implement live capture methods and non-lethal sampling techniques									
1.4.1 Develop improved live capture techniques for general research needs	2a	NMFS	5 yrs	*250*	*250*	*250*	*250*	*250*	M,D/P
1.4.2 Develop improved non-lethal sampling techniques to assess health	2b	NMFS	5 yrs	*200*	*200*	*200*	*200*	*200*	M,D/P
1.5 Develop an implementation plan	2a	NMFS	1 yr with biennial updates	***50***		***10***		***10***	M
TOTAL – ACTION 1				***3,100***	***3,300***	***3,040***	***3,030***	***3,040***	
2. INSURE ADEQUATE HABITAT AND RANGE FOR RECOVERY									
2.1 Maintain, modify as needed, critical habitat designations	3	NMFS	5 yrs	*100*	*100*	*100*	*100*	*100*	F,EV
2.2 Redefine and catalog rookery and haulout sites	3	NMFS, FWS,	1 yr with	*5*					DVT,IS,DR

Plan Task	**Priority**	**Responsible Parties**	**Task Duration**	**Fiscal Year Costs ($K)**					**Threats**
				FY 1	**FY 2**	**FY 3**	**FY 4**	**FY 5**	
and ensure their protection		BLM, USFS	5 yr updates						
2.3 Estimate prey consumption and essential characteristics of marine habitat									
2.3.1 Collect and analyze scat samples and stomach contents to determine prey consumption	2a	NMFS	annual	*400*	*400*	*400*	*400*	*400*	F,EV
2.3.2 Develop stable isotope and fatty acid methodologies to assess prey consumption	2a	NMFS	annual	*150*	*150*	*150*	*150*	*150*	F,EV
2.3.3 Deploy instruments to obtain finer scale data on sea lion foraging habitat	2a	NMFS	annual	*500*	*500*	*500*	*500*	*500*	F,EV
2.3.4 Evaluate all information on sea lion foraging areas and develop a description of foraging needs	2a	NMFS	2 yrs with updates	*200*	*200*				F,EV
2.4 Determine the environmental factors influencing sea lion foraging and survival									
2.4.1 Assess the relationships between oceanographic features and sea lion foraging ecology	2b	NMFS	2 yrs	*125*	*125*				F,EV
2.4.2 Examine the influence of ecosystem variability on non-commercial prey species as an index to sea lion carrying capacity	3	NMFS	5 yrs	*300*	*300*	*300*	*300*	*300*	F,EV
2.4.3 Distinguish how natural and anthropogenic factors influence marine ecosystem dynamics and subsequently sea lion population dynamics	2b	NMFS	5 yrs	*500*	*500*	*500*	*500*	*500*	F,EV
2.5 Investigate sea lion bioenergetics									
2.5.1 Determine the physiological diving capabilities and evaluate how this limits the ability to forage successfully	3	NMFS	5 yrs	*500*	*500*	*500*	*500*	*500*	F,EV
2.5.2 Determine the energetic costs of foraging to sea lions	2a	NMFS	5 yrs	*1,500*	*1,500*	*1,500*	*1,500*	*1,500*	F,EV
2.5.3 Assess the nutritional value of prey by species, season, and area including digestibility and overall value to sea lions	2a	NMFS	3 yrs	*150*	*150*	*150*			F,EV
2.5.4 Develop an energetics model to investigate the interrelationships…and sea lion growth, condition, and vital rates	2b	NMFS	5 yrs	*100*	*100*	*100*	*100*	*100*	F,EV
2.6 Assess and protect important prey resources for									

Plan Task	Priority	Responsible Parties	Task Duration	Fiscal Year Costs ($K)					Threats
				FY 1	FY 2	FY 3	FY 4	FY 5	
sea lions									
2.6.1 Improve groundfish stock assessment surveys to determine seasonal and inter-annual patterns of prey abundance, distribution, and movement at scales relevant to sea lions	2b	NMFS, ADF&G	annual	*1,500*	*1,500*	*1,500*	*1,500*	*1,500*	F,EV
2.6.2 Assess competition for prey with sympatric consumers (e.g., gadids and flatfish, fur seals, harbor seals, other marine mammals, and seabirds)	3	NMFS	5 yrs	*250*	*250*	*250*	*250*	*250*	F,EV
2.6.3 Utilize groundfish fishery observer data to assess the spatial-temporal distribution of the fishery	2a	NMFS, ADF&G	annual	*20*	*20*	*20*	*20*	*20*	F
2.6.4 Assess effectiveness of sea lion closure zones around rookeries and haulouts using small-scale experiments	2a	NMFS, ADF&G	3 yrs	*750*	*750*	*500*			F, DVT
2.6.5 Assess the response of sea lions to changes in prey distribution and availability	2a	NMFS	5 yrs	*200*	*200*	*200*	*200*	*200*	F,EV
2.6.6 Evaluate and implement current or equivalent fishery regulations to protect foraging habitat and prey resources for sea lions	2a	NMFS, ADF&G	annual	*2000*	*2000*	*2000*	*2000*	*2000*	F
2.6.7 Explore the use of ecosystem based (multi-species) stock assessment models to set fishery catch limits to ensure adequate prey resources for a recovered sea lion population	2b	NMFS, ADF&G	5 yrs	*60*	*60*	*60*	*60*	*60*	F,EV
2.6.8 Design and implement an adaptive management program for fisheries, climate change, and predation	2a	NMFS, ADF&G	3 yrs dev. 10 yrs impl.	*500*	*500*	*500*	*200*	*200*	F,EV,KW
2.6.9 Prepare a habitat conservation plan under section 10 of the ESA for fisheries authorized by the State of Alaska	2a	ADF&G	3 yrs	*100*	*100*	*50*			F
2.6.10 Consider and implement conservation measures in herring and salmon fisheries in Alaska as appropriate	2b	ADF&G	annual	*200*	*200*	*200*	*200*	*200*	F
TOTAL – ACTION 2				***10,110***	***10,105***	***9,480***	***8,780***	***8,780***	
3. PROTECT FROM OVER-UTILIZATION FOR COMMERCIAL, SCIENTIFIC, OR EDUCATIONAL PURPOSES									

Plan Task	Priority	Responsible Parties	Task Duration	Fiscal Year Costs ($K)					Threats
				FY 1	FY 2	FY 3	FY 4	FY 5	
3.1 Minimize threat of incidental take in fisheries									
3.1.1 Monitor and evaluate incidental take in commercial fisheries through observer and self-reporting programs	3	NMFS, ADF&G, USCG	annual	*500*	*500*	*500*	*500*	*500*	IT
3.1.2 Monitor and evaluate incidental take in non-commercial fisheries	3	NMFS, ADF&G, USCG	1 yr	*300*					IT
3.2 Minimize threat of intentional killing in fisheries									
3.2.1 Monitor intentional take via shoreline surveys for carcasses near suspected conflict 'hotspots' and by encouraging reporting of illegal shooting through NMFS's enforcement hotline	3	NMFS, ADF&G, USCG	annual	*250*	*250*	*250*	*250*	*250*	IS
3.2.2 Reduce threat of illegal shooting by developing and promoting use of non-lethal deterrents for commercial fisherman	3	NMFS	2 yrs	*300*	*300*				IS
3.3 Minimize frequency and severity of sea lion-human interactions in ports and harbors									
3.3.1 Develop and promote non-lethal means of deterring sea lions from hauling out on docks	3	NMFS, USCG	2 yrs	*100*	*100*				DVT,IS
3.3.2 Continue to publicize "no feeding" regulations in harbor areas and keep active programs for notification and enforcement	3	NMFS, USCG	annual	*50*	*50*	*50*	*50*	*50*	DVT
3.4 Minimize take by recreational and commercial viewing operations									
3.4.1 Publicize and enforce existing no-transit areas to minimize vessel and aircraft disturbance of rookery sites	3	NMFS, USCG	annual	*20*	*20*	*20*	*20*	*20*	DVT
3.4.2 Review and revise existing Marine Mammal Approach Guidelines and provide to charter operators and other mariners to minimize disturbance at haulouts	3	NMFS	annual	*25*	*25*	*25*	*25*	*25*	DVT
3.5 Evaluate and reduce the direct and indirect impacts of research activities									
3.5.1 Coordinate research efforts to reduce potential for unnecessary or duplicative research-related	3	NMFS	Annual	*25*	*25*	*25*	*25*	*25*	DR

Plan Task	Priority	Responsible Parties	Task Duration	Fiscal Year Costs ($K)					Threats
				FY 1	FY 2	FY 3	FY 4	FY 5	
take									
3.5.2 Monitor and minimize unintentional take	3	NMFS, USCG	5 yrs	*200*	*200*	*200*	*200*	*200*	DR
TOTAL – ACTION 3				***1,770***	***1,470***	***1,070***	***1,070***	***1,070***	
4. PROTECT FROM DISEASE, CONTAMINANTS, AND PREDATION									
4.1 Protect sea lions from disease									
4.1.1 Conduct epidemiological surveys	2b	NMFS	5 yrs	*250*	*250*	*250*	*250*	*250*	D/P
4.1.2 Develop and implement methods for parasite evaluations	2b	NMFS	5 yrs	*50*	*50*	*50*	*50*	*50*	D/P
4.1.3 Develop and implement methods to test immune system functioning	2b	NMFS	5 yrs	*25*	*25*	*25*	*25*	*25*	D/P
4.1.4 Evaluate causes of mortality by examining dead and live animals of all age and sex classes	2a	NMFS	10 yrs	*50*	*50*	*50*	*50*	*50*	All
4.1.5 Develop disease management plans	2b	NMFS	2 yrs	*30*	*30*				D/P
4.1.6 Develop an unusual mortality events (UMEs) management plan	2b	NMFS	2 yrs	*50*	*50*				D/P,DVT,IT
4.1.7 Develop models to simulate disease impacts on energetics, physiology, abundance and demographics	2b	NMFS	5 yrs	*100*	*100*	*100*	*100*	*100*	D/P
4.2 Protect sea lions from contaminants									
4.2.1 Design a contaminant research and management plan	2a	NMFS	2 yrs	*30*	*30*				T
4.2.2 Collect samples from free-ranging sea lions and environmental 'hotspots'	2a	NMFS	5 yrs	*200*	*200*	*200*	*200*	*200*	T
4.2.3 Examine blood and tissue samples for evidence of contaminant-linked endocrine effects	2a	NMFS	5 yrs	*100*	*100*	*100*	*100*	*100*	T
4.2.4 Modeling contaminant impact and effect	2a	NMFS	5 yrs	*100*	*100*	*100*	*100*	*100*	T
4.3 Predation									
4.3.1 Understand predator life histories, biology, and ecology – captive work	2b	NMFS	5 yrs	*400*	*400*	*400*	*400*	*400*	KW
4.3.2 Determine killer whale diets	2a	NMFS	5 yrs	*300*	*300*	*300*	*300*	*300*	KW
4.3.3 Develop methods to obtain samples from live killer whales	2b	NMFS	5 yrs	*100*	*100*	*100*	*100*	*100*	KW

Plan Task	**Priority**	**Responsible Parties**	**Task Duration**	**Fiscal Year Costs ($K)**					**Threats**
				FY 1	**FY 2**	**FY 3**	**FY 4**	**FY 5**	
4.3.4 Expand the stranding network	2b	NMFS	2 yrs	*25*	*25*				KW, M
4.3.5 Determine killer whale distribution and behavior across the North Pacific	2a	NMFS	5 yrs	*500*	*500*	*500*	*500*	*500*	KW
4.3.6 Estimate numbers of killer whale ecotypes in time and space	2a	NMFS	5 yrs	*500*	*500*	*500*	*500*	*500*	KW
4.3.7 Develop models to simulate predation rates based on killer whale energetics and abundance and Steller sea lion demographics	2b	NMFS	5 yrs	*100*	*100*	*50*	*50*	*50*	KW
TOTAL – ACTION 4				***2,910***	***2,910***	***2,725***	***2,725***	***2,725***	
5. PROTECT FROM OTHER NATURAL OR ANTHROPOGENIC FACTORS AND ADMINISTER THE RECOVERY PROGRAM									
5.1 Reduce damage to sea lions and their habitat from discharges of pollutants by developing preventive measures	2b	NMFS, USCG	5 yrs	*25*	*25*	*25*	*25*	*25*	T
5.2 Reduce the potential for sea lion entanglement by improving and continuing programs aimed at reducing marine debris									
5.2.1 Reduce discards of debris (e.g., trawl web, packing bands)	2b	NMFS, USCG	5 yrs	*100*	*100*	*100*	*100*	*100*	E
5.2.2 Cleanup derelict gear and beached debris	3	NMFS	5 yrs	*100*	*100*	*100*	*100*	*100*	E
5.3 Monitor causes of sea lion mortality and use data to direct management actions									
5.3.1 Continue and expand the Alaska stranding network to increase coastal coverage and community involvement in monitoring sea lion mortality	2b	NMFS, ADF&G	5 yrs	*100*	*100*	*100*	*100*	*100*	All
5.3.2 Survey selected areas for dead stranded animals	2b	NMFS	5 yrs	*50*	*50*	*50*	*50*	*50*	All
5.3.3 Expand tissue sampling efforts to improve the information obtained from dead sea lions	2b	NMFs	5 yrs	*100*	*100*	*100*	*100*	*100*	All
5.3.4 Monitor the incidence and impact of entanglement in marine debris	2b	NMFS	5 yrs	*100*	*100*	*100*	*100*	*100*	All

Plan Task	Priority	Responsible Parties	Task Duration	Fiscal Year Costs ($K)					Threats
				FY 1	**FY 2**	**FY 3**	**FY 4**	**FY 5**	
5.4 Effectively administer the Steller sea lion recovery program by continuing to provide a recovery coordinator staff position	2a	NMFS	annual	*850*	*850*	*850*	*850*	*850*	All
5.5 Improve sea lion conservation by consulting with the State of Alaska on actions that are likely to adversely impact Steller sea lions	2a	NMFS, ADF&G	annual	*250*	*250*	*250*	*250*	*250*	F,IT,IS,E,DVT
5.6 Conduct an effective outreach program to inform the public about Steller sea lion biology, habitat utilization, and conservation issues									
5.6.1 Encourage and facilitate public reporting of sea lion observations	3	NMFS, ADF&G	5 yrs	*50*	*50*	*50*	*50*	*50*	M
5.6.2 Publicize current conservation efforts and protective measures	3	NMFS	annual	*50*	*50*	*50*	*50*	*50*	All
5.7 Co-manage Steller sea lion subsistence harvests in Alaska by devloping co-management agreements as appropriate with Alaska tribes and tribally authorized Alaska Native Organizations (ANO)									
5.7.1 Manage subsistence harvests and evaluate the efficacy and accuracy of using retrospective subsistence harvest surveys	2a	NMFS, ADF&G	annual	*150*	*150*	*150*	*150*	*150*	SUB
5.7.2 Support Alaska Native subsistence use information programs	2a	NMFS, ADF&G	annual	*75*	*75*	*75*	*75*	*75*	SUB
5.7.3 Analyze carcasses from subsistence harvest to assess age, body condition, and other relevant information to ensure safety of carcasses for human consumption	2a	NMFS	annual	*100*	*100*	*100*	*100*	*100*	D/P,T
5.7.4 Document local knowledge and cultural science (Traditional Ecological Knowledge - TEK) pertaining to sea lions to better understand changes in sea lion movement (local and seasonal), feeding patterns and prey, seasonal haulouts, predation and ecosystem dynamics	2a	NMFS	2 yrs	*100*	*100*				All
5.8 Improve the effectiveness of research for Steller sea lion recovery by instituting a "fast track" process for expediting NMFS research permits for Steller sea lions.	2a	NMFS	2 yrs	*100*	*100*				All

Plan Task	Priority	Responsible Parties	Task Duration	Fiscal Year Costs ($K)					Threats
				FY 1	FY 2	FY 3	FY 4	FY 5	
TOTAL – ACTION 5				*2,300*	*2,300*	*2,100*	*2,100*	*2,100*	
TOTAL – ALL ACTIONS				***20,190***	***19,815***	***18,415***	***17,705***	***17,715***	***93,840***

VI. FACTORS POTENTIALLY INFLUENCING THE EASTERN POPULATION

With the widespread, long-term decline in the western DPS of Steller sea lion, attention has focused on identifying factors that may have reduced productivity or increased mortality. The situation is much different with the eastern DPS, in which (with the exception of the southern end of the range) the number of sea lions has been increasing. Because of the long-term positive population growth of the eastern DPS, no threats to recovery have been identified. However, we recognize that certain factors are affecting or have the potential to affect the dynamics of the population.

A. Impact of Individual Threats

Section III of the Plan presents a detailed discussion of a number of factors that could influence Steller sea lions in general, with emphasis on data specific to the western DPS. This section of the Plan includes only information on such factors that are specific to the eastern DPS of Steller sea lion.

1. Predation

North Pacific killer whales can be categorized into three distinct eco-types that differ in morphology, ecology, behavior and genetics (Stevens *et al.* 1989, Hoelzel and Dover 1991, Hoelzel *et al.* 1998, Baird 2000, Barrett-Lennard 2000, Ford *et al.* 1998, 2000). Notably, feeding ecology varies among the three forms with resident whales targeting fish and transient whales targeting other marine mammals (Dahlheim and Heyning 1999, Ford *et al.* 1998, 2000). Feeding observations on offshore whales are more limited; but initial observations suggest that fish and shark appear to be targeted.

Three stocks of transient killer whales are currently recognized to use the waters within the range of the Steller sea lion (Angliss and Outlaw 2005). Two transient killer whale stocks occupy the range of the western DPS of Steller sea lion (i.e., AT1 stock and the Gulf of Alaska, Aleutian Islands and Bering Sea transient stock) and one stock termed the “West Coast transients” occupy the range of the eastern DPS of Steller sea lion. The abundance and stock structure of the “West Coast transients” have been well documented and are described below.

“West Coast transients” occur between Southeast Alaska and California (Barrett-Lennard 2000). West Coast transients have significant differences in mtDNA and nuclear (microsatellite) DNA and are considered a discrete population from the two, transient killer whale stocks described in western Alaska (Barrett-Lennard 2000). Based on photo-identification studies, an estimated 323 whales comprise the community of West Coast transients (Black *et al.* 1997, Dahlheim *et al.* 1997, Ford and Ellis 1999). Of these, 105 whales are known to occur in California waters (Black *et al.* 1997). Another 218 whales occur in the waters of Washington State, British Columbia, and Southeast Alaska (Dahlheim *et al.* 1997, Ford and Ellis 1999).

Most of the transient whales identified from Washington State, British Columbia, and Southeast Alaska are known to frequently move among the three regions (e.g., 77% of the whales seen in Southeast Alaska are also seen in British Columbia and Washington State); representing a range of 950 miles [1,759 km] (Dahlheim and White submitted). However, there are some individual transient groups that have a more limited range and have only been seen in either Southeast Alaska or British Columbia or Washington State (Dahlheim *et al.* 1997, Ford and Ellis 1999). Conversely, there are some individuals that have an expanded range. For example, a transient group, well known from Southeast Alaska, was also documented off Coos Bay, Oregon (a distance of approximately 1,150 miles [2,129 km]; Dahlheim and White submitted). Four whales frequently seen off Central California were seen once in Southeast Alaska: a distance of 1,436 miles [2,660 km] (Goley and Straley 1994). This movement was the longest documented for transient whales. Out of the 105 whales described from California, only 14 have been seen in Washington, British Columbia, and Southeast Alaska (Black *et al.* 1997). Given the reduced number of photographic matches between California and the more northern regions, there has been some suggestion that these two groups of transient whales (i.e., California transient whales versus the Washington/British Columbia /Southeast Alaskan transient whales) comprise different communities. In summary, only 13% of the California transient population is seen in northern waters (i.e., Washington, British Columbia, and Southeast Alaska). Conversely, transient whales frequently move between Southeast Alaska, British Columbia, and Washington State waters with 77% of the population observed in all three areas. With the exception of four individuals, movements between Prince William Sound and Southeast Alaska do not occur.

The diet of West Coast transient killer whales includes a variety of marine mammal prey. Based on nearly 30 years of field observations, these whales have been seen to feed on harbor seals, Steller sea lions, elephant seals, gray whales, minke whales, Dall's porpoise, harbor porpoise, Pacific-white sided dolphins, and sea birds (Baird and Dill 1995, 1996, Ford *et al.* 1998, Dahlheim and White submitted). Prey specialization by individual pods of killer whales has been documented for a few groups by Baird and Dill (1995, 1996). However, Ford *et al.* (2000) and Dahlheim and White (submitted) documented individual pods of killer whales preying on a variety of prey items.

Based on the mortality rates used in Loughlin and York (2000), about 5,500-6,200 sea lions will die each year in a stable or increasing population of approximately 40,000 animals (see above). An unknown portion of these mortalities will result from predation by transient killer whales residing in the range of the eastern DPS of Steller sea lion.

Long *et al.* (1996) reported white shark bites on 548 live and dead pinnipeds in central California, of which 53 were Steller sea lions. For the period from 1970 to 1992 the number of shark-bitten pinnipeds shows an overall increase attributable to increases in both the predators and their primary prey (California sea lions and elephant seals). Long and Hanni (1993) speculated that white shark predation could impede recovery of Steller sea lions in California if the number of sea lions declines further and the shark population continues to increase.

2. Harvests, killing, and other human impacts

The United States has not conducted any commercial harvests of Steller sea lions in the range of the eastern DPS. During the period from 1912 through 1968, government control programs killed thousands of Steller sea lions on rookeries and haulouts in British Columbia (Bigg 1985). By 1970, when sea lions were given protection in Canada, the population had been reduced by about 70%, and one rookery had been eliminated (Olesiuk 2001). In the U.S., prior to the MMPA, there were both sanctioned and unsanctioned control efforts and the killing of Steller sea lions by fishermen and others was commonplace.

Both the ESA and the MMPA contain provisions that allow coastal Alaska Natives to harvest endangered, threatened, or depleted species for subsistence purposes. During 1992-1998, harvest data were collected through systematic interviews with hunters in at least 60 coastal communities throughout the range of Steller sea lions in Alaska (e.g., Wolfe and Mishler 1998, Wolfe and Hutchinson-Scarbrough 1999). Only 20 animals were reported taken in the range of the eastern DPS during the entire 1992-1998 period.

Amendments to the MMPA in 1988 and 1994 required observer programs to monitor marine mammal incidental take in some domestic fisheries. Observer programs during 1993-97 provided an estimate of 1.2 Steller sea lions killed per year in the California/Oregon thresher shark and swordfish drift gillnet fishery (Hill and DeMaster 1999). Three mortalities were recorded in the California/Oregon/Washington groundfish trawl fishery in 1994 and 1997, and one was recorded in the northern Washington marine set gillnet fishery. Injuries to Steller sea lions have been reported in logbooks from Southeast Alaska and California/Oregon/Washington salmon troll fisheries (Hill and DeMaster 1999). The numbers above are minimum estimates of the incidental kill and serious injury in fisheries, because not all fisheries that might take sea lions are covered by observer programs and fishers often fail to report injuries or mortalities.

Incidental take in Canadian fishery operations has not been estimated. Some Steller sea lions are killed as part of an aquaculture predator control program in British Columbia. Preliminary values indicate a mean annual kill of 12.4 animals between 1993 and 1997. Trawl fisheries have not occurred in Southeast Alaska since the 1990s, thereby eliminating another historic mortality source.

The level of intentional and incidental killing of Steller sea lions by humans has apparently been relatively small as the population has been increasing for about 30 years. It is quite possible that, if these sources of direct mortality were eliminated or reduced, the eastern DPS would increase at a higher rate.

3. Entanglement in debris

Steller sea lions become entangled in a variety of debris including many types of fishing gear, loops of line, and packing bands. All of these may cause mortality; however, the extent is unknown and may range from a fraction of a percentage to several percent a year. In the Gulf of Alaska and Aleutian Island, Steller sea lions have been seen entangled in lost and discarded fishing gear, including items such as packing bands and net material (Calkins 1985, Loughlin *et al.* 1986). Between 2000 and 2005 ADF&G recorded all entangled sea lions during brand-resighting surveys in Southeast Alaska. These surveys occur in the summer and visit virtually all rookeries and haulouts in Southeast. ADF&G reported that 0.21% of the animals observed had some sort of entanglement including packing bands and netting around the neck or fishing gear hanging from the mouth (ADF&G unpublished data). This is a minimum estimate as not all entanglements are visible (such as swallowed hooks) and on an annual basis some entanglements and deaths occur before and after the survey period. It is not known what percentage of entangled animals die as the severity of the injuries varies. This source of mortality could probably be reduced by cleaning beaches, increasing public awareness, and by aggressive enforcement of regulations governing debris discard.

4. Parasitism and disease

Parasites that have been reported in Steller sea lions include (1) intestinal cestodes; (2) trematodes in the intestine and bile duct; (3) nematodes in the stomach, intestine and lungs; (4) acanthocephalans in the intestine; (5) acarian mites in the nasopharynx and lungs; and (6) an anopluran skin louse (Dailey and Brownell 1972, Dailey and Hill 1970). In California sea lions, a serious infection of nematodes causes ulcers that may lead to massive stomach bleeding and death. However, the number of deaths attributed to this cause is not known. Parasites have been found in Steller sea lions and may cause mortality in malnourished animals. To date, adequate research has not been conducted to assess the relative nature and magnitude of parasitism in sea lion populations; however, the available information does not suggest that parasitic infections are limiting population growth. Investigation and identification of parasites requires either necropsy of carcasses, which only occurs sporadically on beach cast animals, or collection of fresh feces.

As discussed for the western DPS, some preliminary surveys have been done on fecal examinations from live captured animals in both western and eastern DPSs. Similar parasites are generally detected in both DPSs; however, little information is available on the parasite loads or effects on the animals. Hookworms are of particular interest because of their ability to cause morbidity and mortality in other pinnipeds. Some work has been done on hookworm loads in eastern DPS pups. In pups less than 3 months old examined in 2003 and 2004, total intestinal worm burdens ranged from 18 to 3,477 (Burek *et al.* 2003, 2005). These levels can cause mortality due to anemia in northern fur seals (*C. ursinus*) (Olsen 1958).

Diseases have been studied on Steller sea lions in both eastern and western DPSs, mostly as serologic surveys of prevalence and occurrence in the population (Barlough *et al.* 1987, Burek *et al.* 2003, 2005, Zarnke *et al.* 1997). Whereas exposure to many disease agents has been identified in Steller sea lions, little is known about the disease agents themselves or how they may impact the sea lion populations and no evidence has been found of disease limiting population growth

5. Toxic substances

The NMFS Northwest Fishery Science Center examined blubber samples from 24 Steller sea lions from southeast Alaska and reported PCB levels of 630-9,900 ng/g wet weight and DDT levels of 400-8,200 ng/g wet weight (NMFS unpublished data). PCB levels at the upper end of this range have been shown to reduce juvenile survival in sea otters (AMAP 2002), but the consequences for Steller sea lions are not known.

Castellini (1999) found that the levels of zinc, copper, and metallothionein (a chelating compound) were comparable between Steller sea lion pups sampled from the eastern and western DPS, and were lower than for captive sea lions. Castellini also found that circulating zinc and metallothionein levels were elevated in southeast Alaska sea lion pups during the early 1990s, but returned to values comparable to Aleutian Island pups by 1997. Metallothionein levels are one measure of exposure of sea lions to heavy metal contamination. The similarity of levels in both eastern and western DPSs suggests that heavy metal contamination may be having similar effects on both DPSs**.** Sydeman and Jarman (1998) found elevated levels of copper, mercury, and selenium in tissues (type not specified) from eight pups found dead at Ano Nuevo Island and Southeast Farallon Island. Existing studies on Steller sea lions have shown relatively low levels of toxic substances (with few exceptions), as well as heavy metals, and these levels are not believed to have caused high mortality or reproductive failure (Lee *et al.* 1996) and are not considered significant contributors to observed Steller sea lion declines.

6. Global climate change

Characteristics of recent climate change in the North Pacific were discussed in detail in Section III.H. In that section it was noted that some features of the ecosystems of the Pacific Northwest (California to British Columbia and southeast Alaska) and the northern North Pacific (Gulf of Alaska and Bering Sea) are out of phase, including recruitment of Pacific salmon and some groundfish stocks (Hollowed and Wooster 1992, Hare *et al.* 1999), and zooplankton biomass (Brodeur *et al.* 1996, Roemmich and McGowan 1995). Such variability may be due to patterns of transport in the North Pacific Current when it bifurcates off the coast of British Columbia to form the northward-flowing Alaska Current and the southward-flowing California Current (Wickett 1966, Hollowed and Wooster 1992). How such variations may affect organisms at the top of the trophic system, such as Steller sea lions, is unknown.

Sydeman and Allen (1999) investigated correlations between oceanographic features and population dynamics of central California pinnipeds Multiple regression analysis of sea surface temperatures and upwelling index versus abundance found no relationship for Steller sea lions. Additionally, despite documented shifts in climate and oceanographic processes that may have affected productivity at multiple trophic levels, California sea lion pup production along the US west coast has increased at approximately 5% per year since 1975 and the eastern DPS of Steller sea lion has also increased at approximately 3%/year with no apparent variability associated with climatic variation. Thus, although there have been documented and perhaps more frequent oceanographic and climatic changes, the population of Steller sea lions has not responded negatively from a population perspective. The most evident change is that all of the new rookeries in the eastern DPS have been established in Alaska at the northern end of the range suggesting a population shift to the north.

7. Reduced prey biomass and quality

In contrast with the situation with the western DPS, no evidence suggests that Steller sea lions in the eastern DPS were nutritionally limited during the 1970s and 1980s. The potential for competition for prey exists as there are commercial fisheries that target several species that are also prey for sea lion in the eastern DPS. These include fisheries that take Pacific cod, walleye pollock, Pacific hake, salmon, and herring. Although many of these species are also both commercially fished and prey for the western DPS Steller sea lions, the two areas differ significantly both in the magnitude and distribution of the fisheries and in the fish communities themselves. For example, the range of one of the most important prey items of the western DPS, Atka mackerel, does not extend into the eastern DPS (see Mueter 1999).

Pollock, as a species, range throughout Alaska, British Columbia, and the northern portions of the American west coast. However, the bulk of the pollock population is located in the range of the western DPS. Commercial fisheries for pollock in the range of the eastern DPS have been sporadically pursued since the 1970s (primarily by foreign fleets) but have been much smaller in magnitude than those in the range of the western DPS. Since the early 1990s, fishing with trawls has been prohibited in all portions of the EEZ off Southeast Alaska. This effectively precludes a pollock fishery in the range of the eastern DPS, though a small trawl pollock fishery continues off British Columbia each year. Other fisheries in SE Alaska for Pacific cod, salmon, and herring commonly use fixed gear (e.g., hook and line, pots) or mobile, non-trawl gear such as seines and trolling. This reduces the rate at which fish can be caught (for most species and gears), and could reduce the likelihood of fishery-induced local depletions of commercially important prey species.

Along the U.S. West Coast, Pacific hake (Pacific whiting) is the dominant groundfish biomass in the California Current, supporting the largest fishery on the west coast south of Alaska. Pacific hake ranges from southern and central California spawning grounds during January-March to summer and fall feeding grounds off Washington State, British Columbia, and Alaska. Pacific hake is an important prey fish for many marine mammal predators, including California sea lions, Steller sea lions, northern fur seals, northern elephant seals, Pacific white-sided dolphins, Dall's porpoises, killer whales, and sperm whales (Livingston and Bailey 1985). Based on scats collected during summer and early fall, Pacific hake appears to be a major prey item of Steller sea lions along the Oregon and Washington coast, just as pollock is a major prey in Alaska (Gearin *et al.* 1999).

Steller sea lions no longer breed on the Channel Islands and have declined at Ano Nuevo Island and the Farallons Islands since the 1970s, while Steller sea lion numbers at the northern California and southern Oregon rookeries (between 42-43 N. lat., where the summer-fall diet appears dominated by hake) have increased significantly in recent years. Meanwhile the Pacific hake stock (age 3+ fish) has declined steadily from an estimated high of 5.8 million metric tons in 1987 to the lowest levels on record in 2001, only 800,000 tons with no concurrent change in Steller sea lion population trajectory.

Steller sea lion trend counts at the largest rookeries in southern Oregon (clustered between 42-43 N. Lat.) increased nearly 4% per year during the 1990s. Under U.S. regulations, the shore-based fishing season opens March 1 and offshore fishing season opens May 15; at-sea processing and night fishing are prohibited south of 42 N. lat. and fishing effort is typically concentrated between central Oregon and Cape Flattery, Washington (Helser *et al.* 2002). Thus, the typical distribution of hake fishing effort is well north of the major Steller sea lion breeding and pupping grounds in southern Oregon and northern California.

Fishery regulations and the typical distribution of the fishery north of 44 N. Lat. may minimize the potential for competition between foraging Steller sea lions and the hake fishery during the summer months when the Steller sea lion population is concentrated on rookeries in southern Oregon and northern California. However, the migratory summer-fall distribution of the hake stock varies widely depending on annual environmental conditions, extending as far north as the Queen Charlotte Islands and southeast Alaska during the warm El Niño ocean conditions of 1998, but concentrated south of Cape Blanco in southern Oregon (approx. 43 degrees N. lat.) during the cool La Niña conditions of 2001 (Helser *et al.* 2002). It is perhaps noteworthy that the increasing northern California-southern Oregon Steller sea lion breeding populations occupy a midpoint in the migratory distribution of the West Coast hake stock, a geographic position that may serve to buffer those populations from the worst effects of ENSO events on the annual distribution of the hake stock.

8. Disturbance

A recent MS thesis by L. Kucey at the University of British Columbia noted that Stellers sea lions in Southeast Alaska are sensitive to various types of disturbance and respond with temporary movements from the area. In British Columbia, harassment and killing of Steller sea lions prior to 1970 resulted in the abandonment of a rookery, although it is now used as a haulout (Bigg 1988). Animals in the eastern DPS are protected from disturbance by the ESA, and effects from disturbance should be minimal at the present time. However, increased human use of areas adjacent to Steller sea lion haulouts in Southeast Alaska does represent a potential threat in the future but the level of those effects is unknown. Recent applications for development in Southeast Alaska are illustrative of the potential for cumulative effects to become a threat in the future however. In spring 2005, NMFS issued a finding of "not likely to adversely affect" for a proposal known as the Juneau Access Improvements project. This project proposes to build a road on the east side of Lynn canal between Juneau and Skagway that would require construction and ultimately road placement less than 300 feet from the Gran Pt. haulout. In addition, the Kensington Gold Project has also been evaluated for its potential to disturb Steller sea lions in Lynn Canal with the construction and use of two marine terminals. The Biological Evaluation for this project identifies potential impacts such as oil spills, noise disturbance, vessel /marine mammal collisions and effects on spawning prey bases. Individually, the NMFS has determined that these projects do not jeopardize the existence of Steller sea lions. However, the continued pressure of developments in otherwise wilderness areas may ultimately result in the abandonment of haulouts.

B. Cumulative Effects

The potential for cumulative effects in the eastern Steller sea lion population is the same as for the western DPS as described in Section V.C. However, given that sea lion numbers are increasing in Oregon, British Columbia, and southeast Alaska, any individual or combined effects currently occurring in that region are apparently not significant enough to prevent the population's growth in those areas. In contrast, the low numbers of sea lions in central California should trigger some investigation of the possible negative consequences of cumulative and synergistic effects. Sydeman and Allen (1999) speculated that the decline of Steller sea lions in central California may be due to the combined effects of oceanic warming affecting prey availability, competition for prey with California sea lions, contaminants, and disease.

C. Threats Assessment

The purpose of this section is to assess the relative impact of all previously identified threats to the eastern DPS of Steller sea lion. Unlike the western DPS where apparent threats have been identified as potentially limiting the population's recovery, there is no similar evidence of limiting factors in the eastern DPS. For example, although threats listed for the western DPS such as predation, disturbance, incidental take, contaminants etc. are present and could certainly act upon the eastern DPS in similar negative ways, there is no evidence to support that any either individually or collectively are current threats to recovery. In addition to the threats and mechanisms in the western DPS table being present to some degree, the most vulnerable cohorts within the eastern DPS would also likely be the same. Because of the lack of identified threats to recovery a threats table was not created for the eastern DPS.

Similar to the western DPS, there is also uncertainty as to the level of current and historical impact of various threats and whether there have been changes in the magnitude of those threats to the eastern DPS. It is thought the prior threats, primarily in the form of directed human take (shooting), have been adequately addressed. As the breeding range and center of the eastern DPS has moved northward, prior threats associated with the previous southern range extent such as competition with other increasing pinniped populations and activities associated with a high human population density may have been largely alleviated.

VII. RECOVERY PLAN FOR THE EASTERN POPULATION

A. Goals

The ultimate goal of this recovery plan is to promote the recovery of Steller sea lions to a level sufficient to warrant their removal from the List of Endangered and Threatened Wildlife and Plants (List) under the ESA.

B. Recovery Strategy

In 1990, Steller sea lions were designated as one threatened population across the range which stretched from southern California to Canada, Alaska, and across into Russia and northern Japan. The listing was primarily based on a substantial decline of the western DPS as well as contraction of the southern extent of the eastern population range.

In 1997, after continued declines in Alaska and new genetic information which revealed further population structure, NMFS split the population into two DPSs, and reclassified the western DPS as endangered while keeping the eastern DPS as threatened. At the time, few threats were described for the eastern population while population counts indicated an increasing trend throughout much of the population. However, concern over possible interactions between sea lions from the western DPS and the eastern DPS, a contraction in the very southern part of the range of the eastern DPS, and the potential impacts from fisheries led NMFS to maintain the eastern DPS as threatened until a longer time series of population increases could be observed with other positive confirmation that human activities were not a threat to recovery. Protection for the eastern population has been provided primarily by the MMPA, the Magnuson-Stevens Fishery Conservation and Management Act, and the Fisheries Act of Canada. The most important protection has likely been prohibitions on lethal takes. Since 1997, the population has continued to increase and is thought to be relatively large compared to historical levels, particularly in the northern portion of the range.

At present, the most likely threats are development, increased disturbance and habitat destruction, increases in magnitude or distribution of commercial or recreation fisheries, and environmental change. Fortunately rookeries are located at remote sites, normally isolated offshore reefs and islands unsuitable for development. In addition many rookeries are in protected areas such as parks, refuges, wilderness areas, and ecological reserves, where future development is unlikely. Other impacts result from entanglement in fishing gear and other marine debris, which appears to be affecting sea lions in substantial numbers. None of these impacts are likely to affect the recovery of the species, but monitoring is vital to ensuring that important sea lion habitat is sufficiently protected. Both the MMPA and the ESA provide protection for sea lions and their habitat. Further conservation actions should be pursued through federal, state, and local governments to ensure that rookery and haulout sites are adequately protected and to ensure that fisheries and other human actions do not adversely affect sea lion prey resources.

Given the current status and lack of significant threats to recovery, the primary recovery actions provided in this plan are to ensure management agreements are in place with the states, and develop a post-delisting monitoring plan to ensure re-listing is not necessary after removal from the List. We also recommend the initiation of a status review and consideration of whether the eastern DPS should be removed from the List. Provided in Section VII.F below is an outline of the post-delisting monitoring plan, which should be developed prior to any proposed delisting notice.

C. Delisting Criteria

The ESA requires that recovery plans, to the maximum extent practicable, incorporate objective, measurable criteria which, when met, would result in a determination in accordance with the provisions of the ESA that the species be removed from the List (50 CFR 17.11 and 17.12). The recovery criteria comprise the core standards upon which the decision to delist a species will be based.

To remove the eastern DPS of Steller sea lion from the List, NMFS must determine that the species' abundance, survival, and distribution, taken together with the threats (i.e., ESA listing factors), no longer render the species "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." Any new factors identified since listing must also be addressed in this analysis to ensure that the species no longer requires protection under the ESA.

Recovery criteria must include the elimination of threats to the species as well as measures of demographic health. Both sets of criteria serve as checks on one another – one set of criteria requires evidence that the threats to Steller sea lions have been eliminated or controlled and are not likely to recur (listing factor criteria), and the other set of criteria requires evidence that the population status of Steller sea lions has improved in response to the reduction in threats (biological criteria).

1. Development of the criteria

There has been some concern over the performance of rookeries and haulouts at the very southern end of the range in California especially in contrast to the growth observed in southeast Alaska. However, no criteria are provided here for sub-regions within the range. It is not unusual for the geographical limit of a species range to perform less than the core regions. A somewhat similar northward movement in Steller sea lion distribution and the establishment of new breeding sites have been noted in the northwestern Pacific (Burkanov and Loughlin 2005) where the southern range limit in the northwestern Pacific has moved northward by 500-900 km over the past 50 years and several new rookeries were established (Burkanov and Loughlin 2005). The question of whether an area constitutes a "significant portion of the range" relates to the biological importance rather than the geographical extent. Although the population trend at the southern limit of the range has not followed the same trajectory as the Alaska portion there are no data available to suggest it is biologically unique (Ono 1993). However, given the limited genetic studies done in the southern part of the range, the potential for uniqueness cannot be ruled out. This demography does not increase the risk of extinction for this species and therefore, it seems appropriate to evaluate the eastern DPS as a whole when establishing recovery criteria.

Goodman (Appendix) evaluated the risk of extinction for the eastern DPS based on currently available information. For the past 25-30 years, the eastern DPS has been growing steadily at about 3% per year, a rate which is about 1/3 of the rate that would represent the common understanding of the biological potential of a pinniped population under similar circumstances. During this time, the population size has more than doubled and is now estimated to be about 46,000 animals. There has been no evidence of appreciable effects of environmental variation, or of appreciable variation in whatever human–caused influences may be affecting this population (Appendix).

The present size of the California portion of the population, which occupies the southern edge of the species range, is about 20% of that recorded there in the middle of the 20th century, and it is believed that the population may have been larger yet in the 19th century (Appendix). In recent times, consistent California-wide counts began in 1996. During the recent decade of monitoring, pup production in California has trended upward, while non-pup numbers have varied from one census to the next, but with no clear trend (Tables I-9 and I-10).

If current trends continue, the risk of near- or medium-term extinction for the eastern DPS is very low (Appendix). While there is no evidence to the contrary, conclusive information is not available that this hypothesis is true, or that these conditions will continue in the future. Therefore, if current protection measures are maintained, we would expect that this population would remain at low risk of extinction.

When considering if demographic criteria are met, NMFS will use available information on the population ecology and vital rates in the U.S. region to ensure that they support the trend observed in the eastern DPS as a whole. Certain vital rates are required in order to allow for long term growth. Available information on pup counts, production (natality), juvenile survival rates, population age structure, gender ratios, and other observations will be examined to determine whether they are indicative of the observed DPS-wide trend.

2. Demographic Criteria

The eastern DPS of Steller sea lion will be considered for removal from the List when the likelihood of its becoming endangered in the foreseeable future has been eliminated by achieving the following biological criteria:

- The population has increased at an average annual growth rate of 3% per year for 30 years.

In 1997, the listing of the eastern population was based in large part in the uncertainty regarding the population trend, although even at that time it appeared to be increasing. However, NMFS was concerned that not enough years had been observed to assure that the population was in fact increasing. The time period chosen for this criterion reflects three generations such that we would be assured that survival and reproduction were robust. NMFS should use scientifically valid tests for significance to determine whether the observed counts reflect actual increases in the population.

3. ESA Listing Factor Criteria

It is imperative that threats to the species be controlled prior to removal from the List. This includes all threats identified at the time of listing, as well as any new threats identified since listing. An inclusive list of those threats and limiting factors is found in the Plan. The Plan describes the decline of Steller sea lions due to changes in the environment, predation, direct takes by humans, and indirect effects such as competition for prey resources with fisheries and with other increasing populations of pinnipeds.

In order to remove the threatened eastern population of Steller sea lion, the following threats-based criteria should be achieved. In addition, the criteria should be achieved in such a way that the delisting of the eastern population is not likely to result in the re-emergence of the threat. The best available information indicates that achieving the following threats criteria are necessary in order to recover Steller sea lions. Yet, it is possible that current perceived threats become insignificant in the future due to changes in the natural environment, or changes in the way that the threats affect the entire life cycle of Steller sea lions.

When the biological criteria (above) are met, NMFS will evaluate and review the criteria under these listing factors to determine their relevance under the current conditions and whether they have been met.

Factor A: The present or threatened destruction, modification, or curtailment of a species' habitat or range

The decline of the eastern population of Steller sea lions is in large part attributed to direct mortality from predator control programs and shooting by fishermen and others. This intentional killing of sea lions was a generally accepted behavior until recent years. In general, terrestrial habitat for the eastern population has been either protected or not impacted to any large degree based in large part on the remote areas occupied by sea lions. There may be some exceptions along the southern California coast. Prey resources currently appear to be adequate to support recovery. Future fisheries management and other marine resource management should specifically consider sea lion needs in their planning.

To provide assurance that delisting is warranted for the eastern population of Steller sea lion, threats to its habitat should be reduced as specified under this factor:

1. Marine habitats, particularly in regard to prey populations, must be maintained through appropriate fisheries management and control of contaminants.
2. Rookery and haulout sites need to be adequately protected (through state, federal, or private measures) to insure the continued use of these sites for pupping, breeding, attending young, and resting. Research and monitoring plans should be in place for all projects that have a high probability of negatively impacting sea lions in order to make sure that these activities do not result in harm to sea lions or their habitat.

Factor B: Overutilization for commercial, recreational, or educational purposes

Human-caused mortality of Steller sea lions includes subsistence harvest, incidental takes in fisheries, illegal shooting, entanglement in marine debris, and take during scientific research. In general, the MMPA provides adequate protection for sea lions from the eastern population. None of these factors now appear to be preventing recovery, although it would be appropriate to reduce the magnitude of these when possible.

Factor C: Disease or predation

Although Steller sea lions are taken by killer whales throughout their range there is no indication that killer whale predation is outside of normal or background levels expected in this population at this abundance level. The final evaluation is that predation is not limiting recovery. Diseases are known to occur within this population but appear to be limited to those endemic to the population and are unlikely to have population level impacts. Therefore no criteria are necessary to reduce disease and predation.

Factor D: The inadequacy of existing regulatory mechanisms

One potential threat to Steller sea lions is increased human disturbance in previously remote areas. Little is known about the potential impacts from changes to the physical environment, disturbance due to vessel traffic, or tourism related activities. Because of lack of information, it is not possible to quantify these threats. However, the potential threat from increased human disturbance highlights the need to keep regulatory mechanisms such as the MMPA in place to protect sea lions. Research and/or monitoring programs should be put into place to oversee activities that have the potential to negatively impact Steller sea lions. Other actions to protect haulout and pupping areas (as described under factor A) could provide substantial insurance against future impacts from development and anthropogenic disturbance. These actions are:

1. Agreement is reached with the State of Alaska which describes their fishery management plan, minimizes the take of Steller sea lions, and describes how future actions taken by the State will comport with the ESA and MMPA.
2. A Steller sea lion recovery coordinator is on staff at NMFS.

Factor E: Other natural or anthropogenic factors affecting its continued existence

To provide assurance that delisting is warranted for the eastern population of Steller sea lions, several actions are recommended to assure that factors do not develop that would threaten its persistence.

1. An outreach program is established to educate the public, commercial fishermen, and others to the continued need to conserve and protect Steller sea lions.
2. An Alaska stranding network is in place and functional.

D. Recovery Actions Outline and Narrative

At the time of initial listing (1990), it was not recognized that there were two genetically distinct populations of Steller sea lions in North America. What is now recognized as the western DPS was undergoing a major decline while the trend of the eastern DPS was uncertain. It was also of concern that the southern portion of the eastern DPS had contracted and several haulout sites and at least two rookeries were no longer used by Steller sea lions. Additionally, there was apprehension that whatever factor was responsible for the decline in the west, e.g. disease, might spread to the east. In retrospect it is now apparent that the eastern DPS has been consistently increasing at about 3% per year throughout its range for about 25 years, with the exception of central California. The southernmost sites appear to have stabilized, albeit at levels far below their historical maximums. The eastern DPS has increased by about 225% over the last 25 years and four new rookeries have been established in Southeast Alaska. With the exception of the southern portion of the range, the reduced population size in the 1970s was thought to be the result of direct human related mortality, largely in the form of shooting by fisherman and others who viewed sea lions as competitors for fishery resources. With the passage of protective legislation in both the U.S. and Canada and with changing social values, this source of mortality has been substantially reduced. Although there are still a number of factors that negatively impact the dynamics of the eastern DPS, none of these either alone or in combination appear to pose a threat to recovery.

Based upon the lack of threats to the eastern DPS and the population status and trends, the following two recovery actions are provided.

Recovery Action Narrative

1. Monitoring

1.1 Develop a post delisting monitoring plan

A post delisting monitoring plan should be developed (see Section VII.F below) which would guide monitoring activities for 10 years post delisting. The objective should be to ensure that necessary recovery actions remain in place and that it can be confirmed that there are no threats to the population's continued existence.

2. Protect from other natural or anthropogenic factors and administer the recovery program

2.1 Initiate a status review to determine whether to delist the eastern DPS of Steller sea lions.

NMFS should initiate a status review and determine whether the eastern DPS has met the recovery criteria found in this plan and should be removed from the list of threatened species. No threats to recovery have been identified and the population has been increasing for over 25 years, new rookeries have been created, and the population is at historical high levels.

E. Recovery Action Implementation Schedule

NMFS should implement the recovery actions within one year after the completion of this plan. A post delisting monitoring plan is an essential part of the delisting package and should be developed during the status review process. The process should take about one year to complete.

Costs:

1.1	Develop a post delisting monitoring plan; $50,000. Implementation of the monitoring plan would require an annual cost of about $100,000 for population surveys and health monitoring
2.1	Initiate a status review to determine whether to de-list the eastern DPS of Steller sea lions; $100,000

F. Post-Delisting Monitoring Plan

Post-delisting monitoring (PDM) refers to activities undertaken to verify that a species delisted due to recovery remains secure from risk of extinction after it has been removed from the protections of the ESA. The primary goal of PDM is to confirm that the species does not require relisting as threatened or endangered during the period following removal of ESA protections. Section 4(g), added to the ESA in the 1988 reauthorization, requires NMFS to implement a system in cooperation with the states to monitor for not less than five years the status of all species that have recovered and been removed from the lists of threatened and endangered plants and animals (50 CFR 17.11, 17.12, 224.101, and 227.4). Section 4(g) directs NMFS to make prompt use of their emergency listing authorities under section 4(b)(7) to prevent a significant risk to the well-being of any recovered species. While not specifically mentioned in section 4(g), authorities to list species in accordance with the process prescribed in section 4(b)(5) and 4(b)(6) may also be utilized to reinstate species on the list of threatened and endangered plants and animals, if such an action is found to be appropriate.

The ESA does not require the development of a formal PDM "plan." However, for the eastern DPS of Steller sea lions, a 10-year plan is likely to be very helpful to NMFS in ensuring that the species has recovered and that the protections of the ESA are no longer necessary. The key components of the plan should include population trend monitoring (biennial or triennial), monitoring of development near terrestrial habitats, monitoring for unusual mortality events, and monitoring of fishery management plans to ensure they stay consistent with sea lion requirements. A PDM plan should be developed by NMFS in collaboration with the States and Canada before the species is proposed for delisting (if warranted).

Key components of the plan should include:

- Continue to estimate population trends (biennial or triennial) for pups and non-pups
- Closely monitor trend and status of rookeries and haulouts at southern end of range (California)
- Monitor for unusual mortality events via a stranding network including impacts from fishing gear and other human related materials (e.g., plastic bands, discarded fishing nets, flashers)
- Continue to ensure the protection of important terrestrial habitat (rookery and haulout sites)
- Monitor the magnitude and distribution of commercial and recreational fisheries to ensure the continued protection of important sea lion prey resources
- Conduct additional research on the genetic structure of the eastern population
- Monitor incidental takes in fisheries
- Monitor direct takes
- Monitor frequency and severity of Steller sea lion-human interactions in ports and harbors
- Monitor impacts of recreational and commercial viewing operation
- Monitor impacts of research activities
- Monitor for disease and health related to contaminants
- Monitor predation as a significant sources of mortality

- Conduct an effective outreach program to inform the public about Steller sea lion biology, habitat utilization, and conservation issues

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APPENDIX: A PVA Model for Evaluating Recovery Criteria for Steller Sea Lions

Overview from the Steller Sea Lion Recovery Team

The primary intent of the ESA is to recover listed species, and the ecosystems on which they depend, such that the protections of the ESA are no longer necessary. Biological recovery criteria (criteria) form the basis from which to gauge a species' recovery and subsequent risk of extinction, whereas listing factor criteria ensure that the threats have been controlled or eliminated.

Recovery criteria are required under the ESA to be both measurable and objective. However, the ESA does not provide an explicit standard for those criteria, nor is there accepted agency policy regarding choices of risk. A probability (chance) of extinction over some period of time (e.g., 1% probability of extinction in 100 years) has been recommended by some as the quantitative standard for a species to be considered in high risk of extinction (Angliss and *et al.* 2002, DeMaster *et al.* 2004). The integration of such a standard in a modeling approach provides a definitive means to measure the risk of extinction. However, selection of the extinction value is a policy decision based on the acceptance of risk. Alternatively, criteria can be, and has traditionally been derived from a weight of evidence approach. This approach requires a thorough review and synthesis of all the available biological and ecological information regarding the species (or surrogate species), and the determination of essential demographic parameters (e.g., population abundance and trend, birth rates, age ratios, etc.) that would reflect that the species is no longer at high risk of extinction. The Team explored both approaches to develop criteria. This appendix is a brief overview of the quantitative approach using a population viability analysis (PVA) which employed a specific probability of extinction (>1% chance of extinction within 100 years). The PVA was developed by Dr. Dan Goodman (Montana State University) under contract from NMFS. In contrast, the weight of evidence approach is provided in Section V.

Development of the PVA analyses

Substantial efforts have recently been directed towards developing rationale for determining extinction standards. Specifically, NMFS conducted a workshop to consider recovery criteria for large whales (Angliss *et al.* 2002) and a NMFS Quantitative Working Group has proposed guidelines on ESA listing criteria (DeMaster *et al.* 2004). Additionally, the issue has been discussed in peer reviewed literature (Goodman 2002, Reed *et al.* 2002). Although NMFS or the FWS have not adopted specific guidelines for criteria, the recommendations from these efforts were used as the starting point in the development of criteria. The general principles that the Team used were:

A probabilistic threshold is appropriate to describe the risk of extinction in the criteria;a long lived species shall no longer be considered endangered when, given current and projected conditions, the probability of quasi extinction is less than 1% in 100 years; a long lived species shall no longer be considered threatened when, given current and projected conditions, the probability of becoming endangered is less than 10% in 20 years;

Current threats to the species, as well as those that brought the species to the point of listing, must be addressed in the criteria;

The PVA developed for the Team makes the assumption that the recorded history (roughly the last 50 years) of the western DPS is a combined result of natural variation and extraneous influences (i.e., incidental mortality to fisheries, illegal shooting and subsistence harvest, and reduced prey biomass and quality from fisheries). Based on available sea lion population counts, five time periods were selected to represent the population trajectory; 1958-77, 1977-85, 1985-1989, 1989-2000, and 2000-2004. However, these individual population trajectories are affected by human related impacts (e.g., shooting, harvest, fisheries). Because successful management measures have been implemented in the recent past, its is inappropriate to predict future population trajectories based on historical conditions which are likely to be pessimistic (i.e., some of the previous sea lion mortality is unlikely to occur in the future due to the implementation of management measures such as a ban on shooting). Thus, the Team estimated the mitigated mortality attributed to some of the extraneous influences using best available empirical evidence and expert opinion.

In the model, the population trajectory of previous time periods was modified to reflect the mitigation measures currently in place. This was done for the 4 time periods prior to the current time period (2000-2004) which was not modified because this represents the base case of the current suite of mitigation measures. Therefore, the future population projections were based on the recorded sea lion population trend over the last 50 years, as modified by the effect of current mitigation measures. For example, we know that incidental take was a substantial mortality factor during the steep declines in the 1985-1989 time period. Based on current fishery management measures and restrictions on take, it is extremely unlikely that these high levels of take would occur again in the future. Thus, the trajectory in the model for this time period was modified to remove the amount of incidental take that the Team considered extremely unlikely to occur again. The mortality that remained during that time period (still a substantial rate of decline) was due to other factors that the Team could not identify with any quantitative assessment. The effect of making these changes is to decrease the future extinction risk, and decrease the potential time to recovery.

With estimates of intrinsic mean growth rate and associated variance, the model then probabilistically predicts future population trajectories. The Bayesian framework allows for specifying uncertain inputs as distributions, quantifies uncertainty associated with the estimation mechanism itself, and tracks the propagation of uncertainty through all the steps of the estimation and prediction.

Two complications particular to the western DPS presented substantial challenges to the PVA. First, is the sustained population decline (1970s through 2000s) which cannot be explained. Second, the North Pacific ecosystem is subject to naturally large physical and biological variation that likely constitutes the dominant process that drives sea lion population dynamics over long time scales (though this is not evident in the recent history of the eastern DPS).

The estimated magnitude of the extraneous factors (mitigation) did not account for much of the period of rapid decline in the late 1980s. Not unexpectedly, the initial model projections resulted in a relatively high probability of extinction in the next 100 years. Acknowledging the possibility that factors beyond the extraneous influences may have affected sea lion population dynamics, the alternative scenarios described in the threats assessment (see Section IV) were then considered. These alternative scenarios, or hypotheses, represent how the cumulative and synergistic impact of various threats could have caused the past population decline. These alternative scenarios (as well as others not described here) are each scientifically plausible, yet the available evidence does not allow us to determine which of them is more likely. Thus, based on the expert opinion o f the Team, a probability of 0.2 (must be between 0 and 1) was assigned to *all* the alternative scenarios. In essence, this probability represents the likelihood that the assumptions of the PVA are incorrect and the western DPS is not likely to become extinct. The 0.2 probability was then factored into the model results to determine extinction risk.

Model projections using the modified trajectories (mitigation measures) and the alternative scenarios (probability of 0.2) still resulted in a high probability of extinction. For a hypothesized future assessment in 2034 after 3 decades of 3% annual growth to a population size of 110,434 sea lions, the probability of quasi-extinction within 100 years is 9.71%. Review of the PVA revealed that the probability of extinction was most sensitive to the period of rapid population decline (about 15% per year) observed in the 1985 to 1989 period. Considering the possibility that the 1985-1989 period of decline was unlikely to occur in the future, another model run was completed with the 1985-89 time interval deleted from the time series. In those projections, after a period of 3% growth until 2024, the western DPS would have a 1.79% probability of extinction in 100 years.

Approach to the development of biological criteria for Steller sea lions

The Team reviewed the overall model structure, assumptions, and parameter values used in the PVA, and decided to use a weight of evidence approach for the criteria instead the quantitative probability of extinction approach. The rationale for that decision was based on the substantial uncertainty associated with the estimates of mortality due to extraneous influences, the uncertainty in the alternative scenarios, and the weak rationale for removing the 1985-1989 time period from the time series. Discounting the 1985-1989 time period as a catastrophe that is unlikely to be repeated is inconsistent with a precautionary approach. Further, the uncertainty in the strength of a density dependent response by sea lions is an important consideration. Although the alternative scenarios provide for a potential density dependence response, there is no evidence that such a response has occurred for sea lions, and the scientific literature provides little evidence to support that assumption.

Thus, the PVA was instrumental in providing the Team with insights on how the threats need to be addressed in order to develop downlisting and delisting criteria. Those insights were applied in using the weight of evidence approach in selecting the criteria, and especially in developing the listing factor criteria that determine how the threats must be controlled or eliminated.

Discussion of various analytical approaches to estimate Steller sea lion persistence

As discussed in Chapter I, several demographic models have been used to explore the population dynamics and persistence of Steller sea lions. Unlike previous exercises, this PVA is unique because it was developed specifically for the Team based on the explicit guidance and input from the Team with the goal of generating recovery criteria. To accomplish this, decisions were made on important input parameters that were not obligatory in other models. Specifically, the extinction standard of 1% in 100 years was selected to provide a definitive measure of risk, while a quasi-extinction level of an effective population size of 1,000 was selected to maintain long term genetic viability. Other differences in fundamental assumptions were different between the PVA described here and those prepared earlier. For example, Winship and Trites (2006) based future simulations on population trends observed in the 3-4 intervals between 1978 and 2002. By contrast, projections in the Team's PVA were based population trends in 5 intervals over the 1958-2004 time period. Winship and Trites (2006) also examined the risk of extirpation separately for each rookery, assumed in one of three scenarios that density dependence would regulate those individual rookeries about their estimated 1990s carrying capacities, and modeled the 1985-1989 decline as a catastrophe that would not influence future trajectories. These differences in model structure and assumptions are important to recognize when comparing model projections.

A PVA MODEL FOR EVALUATING RECOVERY CRITERIA FOR THE WESTERN STELLER SEA LION POPULATION

Report to the Steller Sea Lion Recovery Team

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ABSTRACT

This is the report of a project to explore the use of a Bayesian population viability analysis, in a decision theory framework, to define recovery criteria that satisfy the requirements of the Endangered Species Act, for the western US distinct population segment of Steller sea lion. The project was undertaken by the author, as a consultant to the Steller Sea Lion Recovery Team that was in the process of writing a draft for a new Recovery Plan.

A subgroup of the Recovery Team provided three crucial inputs as expert opinion. These were: (1) quantification of necessary policy elements that are not fully specified in existing agency guidelines, (2) specification of some uncertain factual elements that were needed for the modeling but could not be fully determined by statistical analysis of hard data, and (3) specification of the probability of the essential correctness of the core assumptions of the PVA model that was used versus alternative hypotheses that greatly discount the risk to the population.

The standard adopted for downlisting from Endangered to Threatened was 99% probability of the population persisting for 100 years without declining below a quasi-extinction threshold. The quasi-extinction threshold adopted was 4743 individuals, corresponding to a genetically effective population size of 1000. The belief in the essential correctness of the model was quantified at 80% probability that the alternative hypotheses discounting the risk to the population are not correct. The subgroup drew up a table representing their opinion of the intensity, during respective past time intervals, of factors responsible for past threats to the population as used in the model.

The central hard data used in the modeling were the 6 available population wide estimates of population size, that span time intervals averaging approximately 10 years in duration. The salient features of the available information, from the standpoint of assessing extinction risk, are the combination of large, but much reduced population size, continuing and volatile decline for many decades until just a few years ago, unexplained dynamics, failure to recover as expected, and a context of very large fisheries operations and large natural ecosystem

variability.

The basic PVA model captures this state of knowledge by assuming that the population is subject to random changes in its growth rate, at random intervals, where the distribution of exponential growth rates is normal, and the distribution of interval length is exponential with a mean duration of 10 years. The dynamics in the model are not density dependent, and a specific analysis was done to elucidate the circumstances under which a population might go through such a wide range of population sizes without displaying density dependence.

The PVA model represents process variation through stochasticity of the changes in population growth rate and stochasticity of the time intervals between changes. The PVA model incorporates parameter uncertainty by representing the parameters of the distribution of population growth rates as a joint distribution of the uncertain mean and uncertain standard deviation. The joint distribution of these uncertain parameters was obtained by Bayesian inference from the past data, as adjusted by the subgroup's expert opinion concerning correction for threat factors that are believed to have had different intensity in the past than they will in the future. The primary reason for belief in these differences is the changes in implementation of regulatory protection for sea lions, and changes in the operation of the fisheries.

The basic model, applied to the data as adjusted by the subroup's inputs, and using the subgroup's policy specifications and appraisal of overall correctness, predicts almost 30% probability of quasi-exinction within 100 yrs from 2004, if the current level of protections is maintained. The population grew at roughly 2.8% per year in the interval 2000-2004. If that growth continues till 2024, the population size will then be 83,352 (roughly doubling the population size from 2004, in a little less than two generations). At that population size, and if the population growth were known to have stayed constant at 2.8% through two intervals, with all the other inputs the same, the assessment becomes almost 13% probability of quasi-exinction within 100 yrs from 2024, if the current level of protections is maintained. Absent knowledge of the rates of growth between 2004 and 2024, the attainment of a population size of at least 83,352 in 2024, with all other inputs the same, the assessment becomes about 19% probability of quasi-exinction within 100 yrs from 2024, if the current level of protections is maintained. All these scenarios fail to meet the risk standard for a downlisting criterion.

Sensitivity analysis shows that the observed, but unaccounted for, steep rate of decline in the period 1985-1989, contributes a very large component of the calculated risk, both through its influence on the inferred mean rate of population growth and its influence on the inferred random variation in growth rate. If this interval were removed from the analysis, a recovery criterion of 83,352 as the population size in 2024, with all other inputs the same nearly meets the standard (achieving 1.43% probability of quasi-exinction within 100 yrs from 2024, if the current level of protections is maintained). The subgroup was not able to justify exclusion of the 1985-1989 decline rate from the analysis, though it is acknowledged that there may have been some peculiarities in the fishery unique to this interval.

The large role of uncertainty in forcing the analysis toward very stringent recovery criteria (high population values), indicates that a new approach to extinction risk assessment, taking account of planned experimentation and a firm commitment to adjust management in response to future monitoring and results of experiments, may be the only way to obtain more readily attained recovery criteria while still satisfying the chosen standards. Reconciling this approach to present interpretations of the legal requirements of the Endangered Species Act may take some careful examination, particularly of the legal nature of commitments to future implementation of a plan with contingencies based on results of future monitoring and experimentation, and the required demonstration that management measures built into the plan really are adequate to contain the risk. The theory of how to technically quantify the total risk of such a plan is known.

INTRODUCTION

Background to this Project

The western US distinct population segment (DPS) of Steller sea lions (SSL) was listed as Threatened under the Endangered Species Act (ESA) in 1990, and uplisted to Endangered in 1997. The current population is still rather numerous compared to most ESA large mammal populations that are listed as Endangered. The very dramatic decline, documented from 1958, from a population size that was an order of magnitude larger, is not accounted for by available quantitative data on factors affecting mortality or reproduction. The rates of decline themselves were highly variable over time, and the variation is not accounted for by available quantitative data on factors affecting mortality or reproduction. The marine system where these sea lions live has been subject to large scale commercial fishing activities, and the nature of the fisheries changed considerably in the era when the SSL decline was documented. The marine ecosystem where the western DPS SSL live is subject to considerable natural variation on a longer than annual time scale. Despite the institution of some protective measures at the time of first listing, the decline continued through the time of uplisting. After institution of more protective measures a few years after uplisting, the population, in the interval 2000 to present, has shown a definitely positive growth rate that is still much smaller than would be expected for a pinniped population that is far below carrying capacity. The eastern population of the same species, occupying a somewhat different ecosystem where different fisheries operate, showed consistent positive population growth over the past three decades, at a rate that is much smaller than would be expected for a pinniped population that is far below carrying capacity.

The combination of large, but much reduced population size, continuing and volatile decline for decades until just a few years ago, unexplained dynamics, failure to recover as expected, and a context of very large fisheries operations and large natural ecosystem variability presents a case that is a bit of a challenge to informal methods for developing decision rules for purposes of ESA regulation. In part this is owing simply to the unfamiliarity and ambiguity of this combination of circumstances. The prominence of uncertainty in this summary of the facts also adds to the challenge, for intuitive, non-quantitative approaches are not notably effective in factoring uncertainty into a decision process.

The decisions required under the ESA are verbally described, in legislation and regulation, in terms which can be re-expressed in the technical framework of statistical decision theory (Goodman, 2002a; Goodman, 2005), and the core quantitative population risk status assessments can be carried out by the methodology of Bayesian population viability analysis (Goodman, 2002b). In principle, a decision theory formulation and quantitative risk assessment approach might promote consistency and transparency, reduce ambiguities, and provide a structured and technically defensible approach for the scientific determinations required by the ESA.

When a new Steller Sea Lion Recovery Team (SSLRT) was convened, and charged with producing a new draft Recovery Plan (SSLRP) for consideration by NOAA, a project was initated in 2002, funded by NOAA, for this author to consult with the new recovery team on development specifically of possible recovery criteria by means of a Bayesian population viability analysis in a decision theory framework.

The new SSLRT team is a large group, with representation of stakeholders as well as biological experts. A subgroup of members of the new recovery team, with quantitative interests and biological expertise and experience with SSL, was formed to serve as an expert panel during the course of this project. They had two primary formal roles in addition to providing a general wealth of knowledge about the Steller sea lion, its environment, and the available hard data. These formal roles were the quantification of policy elements that are not fully specified in existing agency guidelines, and the specification of expert opinion for uncertain factual elements that could not be fully determined by statistical analysis of hard data.

In particular

1. The expert panel provided the quantitative interpretation for the verbal standard corresponding to *not in danger of extinction*, and this was encapsulated in an explicit statement of the required probability for the population persisting above a stipulated numerical threshold for a stipulated period of time.
2. The expert panel provided the expert opinion on what would constitute reasonable assumptions about the quantitative magnitudes of a set of factors (used by the model) that historically influenced the population decline but that had not been monitored well enough to be estimated directly from data.
3. The expert panel provided the expert opinion quantifying the overall reasonabless of the assumptions of the model versus alternative hypotheses that would lead to a conclusion of much lower risk to the population.

These inputs from the expert panel are noted specifically in this report. In this sense, the resulting analysis constitutes a formal, quantitative and logical merging of the available pertinent population data, the subgroup's operationalization of otherwise ambiguous verbal policy constraints, the subgroup's expert opinion on some needed factual inputs about the past intensity of some of the threats to the population, and the subgroup's expert opinion about the suite of applicable hypotheses concerning the causes of the decline and the forces that will be operating in the future. The analysis takes account of both the data and the uncertainties. The analysis is documented in sufficient detail to show the influence of the data, the assumptions, the expert opinion, and the quantification of policy thresholds, on the calculated recovery criterion that results.

Recovery Criteria

A recovery criterion states a set of measurable and objective conditions which, if they actually are achieved, are believed, with high confidence, to indicate that the population will have a high probability of survival thereafter. The set of measurable and objective conditions may include evidence ensuring that specified threats have been controlled, and it may include requirements for having attained a specified population size or having demonstrated some specified demographic rates for some specified period of time.

As a verbal narrative, the justification for a recovery criterion, or list of recovery criteria, will involve some account of how the population came to be endangered, an explanation of how the criteria will indicate that the original causes for the endangered status have been sufficiently controlled, and a description of why it is believed that, with the threats reduced to that level, and with the population having attained the stipulated status, the prospects really are high for continued persistence. Because of the complexity of ecological phenomena, and the incompleteness of our scientific knowledge, the coherence of this narrative may be difficult to judge if it is purely verbal.

A quantitative representation offers the advantage of unambiguous capture of all the details of knowledge, and assumptions, and acknowledged knowledge gaps, in a form that allows precise testing of whether they are indeed consistent with the available facts in accounting for the past, and in meeting the standard of predicting an acceptably high probability of population persistence for the future with a defined set of management measures in place. Further, a quantitative representation provides a means for calculating the probability that the recovery criteria will be attained, from the current state, in a specified amount of time, under a defined management scenario, given the evidence and the assumptions. A Bayesian population viability analysis (PVA) is the general statistical and modeling framework for accomplishing these simultaneous retrospective and prospective analyses.

A PVA for the western US population of the SSL will need to address two particular complications that are somewhat novel for the PVA enterprise, as it has been developed to date. One is that the most striking feature about the dynamics of this population is a very large chronic decline that is largely unexplained. The second is that the population occupies a portion of the North Pacific ecosystem that is now known to be subject to large, natural, long time scale, physical and biological variation, called oceanographic "regimes," so that the dominant process variation in the natural dynamics of the Steller Sea Lion population could conceivably be driven by these regimes on a time scale, roughly, of decades, rather than by interannual variation that is independent from one year to the next. A final major complication for analysis is that the period of well documented fast decline for the western SSL began more or less coincident with a well documented dramatic regime shift indexed by the Pacific Decadal Oscillation (PDO), but fisheries activities also changed greatly at about that time.

A PVA Approach for the Western US Steller Sea Lion

There are no direct recorded observations from which the numbers of the Steller Sea Lion could be estimated previous to the 1950's. It is not in dispute that the western US population of the Steller Sea Lion declined by approximately 80% between the 1950's and the year 2000. But there are several fundamentally different visions of why the population declined, and these portend very different predictions of how the population will fare in the future. Several of these different visions are scientifically plausible, but there is insufficient objective information now available for selecting one to the exclusion of the others.

The SSL population has persisted in substantial numbers in the north Pacific for the roughly 10,000 years since the last ice age, and the species is much older than that. The latest genetic data suggest that, up until the present decline, the population had not previously undergone severe genetic bottlenecks during the time since the last ice age, but it is not known how much this constrains the possible long-past history of population swings. It also is not known how much the forces driving the population dynamics during the past 50 years depart from those that dominated the previous 10,000 years.

It is reasonable, but not certain, to assume that the dynamics of the population in the foreseeable future will be an extension of the recent dynamics, corrected for the effects of possibly unique ecosystem events that occurred during the last 50 years, and corrected for the effects of controllable human activities that, as a matter of policy, assuredly will not be repeated, or at least will be deployed differently in the future than they were in the recent past.

This does not deny a role for oceanographic regime shifts. The reality of large scale oceanographic variation is acknowledged, but forecasting of that variation for the future can only be probabilistic (stochastic), and the translation of the oceanographic variation to an effect on the SSL population can only be estimated from a very limited sample of recent observations.

In essence, this modeling approach treats the recorded history of the population as a combined result of natural variation, including the influences of regime shifts, and extraneous influences that were particular to the period. It attempts to factor out the effects of the extraneous influences, and then estimates the properties of the natural variation in terms of a mean growth rate and a pattern of long-term variance in the population growth rate. With estimates of intrinsic mean growth rate and its variance in hand, the model can then probabilisticly predict future population trajectories under scenarios of specified future extraneous influences (including management and regulation), which may depart from the known or assumed past. The Bayesian framework allows for specifying uncertain inputs as distributions, quantifies uncertainty associated with the estimation mechanism itself from the retrospective analysis (parameter uncertainty), and it tracks the propagation of uncertainty (both process variation and parameter uncertainty) through all the steps of the estimation and prediction.

The apparent dynamics of the Western US SSL differ fundamentally from those of the Eastern DPS. The Eastern population was considerably reduced, presumably primarily by shooting, earlier in the 20’th century, but since the institution of protection, this population, except for its southernmost portion (California), has been recovering fairly steadily at about 3% to 3.5%, with no evidence of decade-scale regime-like variation, and only slight environmental variation at shorter time scales. An analysis of the Eastern population data is given in Appendix A.

AVAILABLE POPULATION INFORMATION

Population Estimates

Six population-wide estimates for the Western US DPS of SSL are available at intervals of from 4 to 18 years, averaging 9.2 years. The estimates were put on a common basis in a draft analysis by a subgroup of the current SSLRT drawing on non-pup population estimates from Loughlin et al. (1992), Loughlin (1997), and Loughlin and York (2000), total population estimates in Fritz and Stinchcomb (2005), and reconciled with expansions for pup to non-pup ratios based on information in Calkins and Pitcher (1983) and Fritz and Stinchcomb (2005). From these census estimates, the realized (net) population growth rates for those five intervals may be calculated.

year of census	**population size estimate**	**realized (net) annual exponential growth rate**
1958	227571	-0.00906
1977	191571	-0.04737
1985	131143	-0.16843
1989	66857	-0.04134
2000	42429	+0.02813
2004	47483	

Table 1.

We will assume that the censuses took place at or near regime phase transitions, so that the 5 growth rate estimates may be treated as independent samples from the distribution of growth rates. If in fact the census intervals straddle some regime shifts, the above assumption will result in an underestimate of the true process variation in growth rates, and will therefore underestimate the extinction risk.

Figure 1 shows the 6 census estimates as dots, plotted against year, and the line connecting the census estimates shows the trajectory corresponding to constant exponential growth within each interval, at the by-interval rates calculated in Table 1. This trajectory manifests the combined effects of uncontrollable ongoing environmental variation operating on the population, possible uncontrollable but unique events that will not recur, and mortality factors attributable to human activities that have been regulated and modified to some extent recently, and potentially can be controlled by regulation in the future.

For purposes of the PVA we will attempt to differentiate, both conceptually and numerically, between the population's underlying dynamics (including effects of natural, possibly regime-like, environmental variation) which we expect to continue to operate in the foreseeable future, and the extraneous factors that affected the dynamics over the period of record, but which we do not expect to operate in the same way in the future.

The very rapid decline in the 5 year 1985-1989 period is a matter of particular concern. A decline of 16% per year will doom any population if the decline continues or occurs too frequently without being balanced by enough episodes of compensating growth. In fact, available quantitative estimates of the magnitude of extraneous mortality factors during 1985-1989 only account for a modest portion of the 16% decline rate, leaving a considerable decline still unaccounted for. There were some peculiarities in the operation of the fisheries during this time that greatly increase the uncertainty about the magnitude of their influence. It is suggestive that the big decline in the 1985-1989 interval followed a short-lived, large winter pollock fishery in Shelikof Strait, Gulf of Alaska, and more or less coincided with the beginning of major winter fishing for pollock in the Bering Sea. Winter pollock fishing had not been pursued in either area to a large extent prior to the early 1980s. Both winter fisheries took place in areas thought to be important for SSL foraging, and there is suspicion that winter foraging may be a critical factor for SSLs, especially for the young and adult females with pups. There was also a roe stripping operation at that time, where the entire stripped carcasses of adult pollock were discarded overboard, attracting SSLs and possibly increasing incidental catch. Subsequently, regulations prohibiting roe-stripping were enacted. The joint venture operations were at their height then, and there were very few observers on the domestic catcher vessels that delivered to foreign-owned motherships and processors. SSLs in the mid-1980s in the Gulf of Alaska may have been nutritionally stressed, as indicated by decreased growth rates and a possible reduction in late-term rates of pregnancy compared to SSLs collected in the mid-1970s (Calkins and Goodwin 1988; Pitcher et al. 1998).

Harvests, Shooting, and Incidental Catch

There is some information bearing on possible magnitudes for some of the mortality factors comprising the extraneous influences during the intervals for which growth rates can be estimated. Incidental catch estimates for the trawl fisheries based on observer data, were calculated by Perez and Loughlin (1990). Available quantitative information bearing on harvests, shooting, and incidental catch was compiled and analyzed by Trites and Larkin (1992). A draft analysis by a subgroup of the SSLRT extended and extrapolated the Trites and Larkin estimates. This resulted in the values in Table 2, where the cell entries are the accumulated number of deaths attributed to each cause over the interval.

	period				
	1958-1977	1977-1985	1985-1989	1989-2000	2000-2004
factor					
Non-subsistence direct harvest	45,178	0	0	0	0
Subsistence harvest	9,995	2,900	850	3,300	750
Non-harvest shooting	12,716	8,277	1,870	2,200	1,000
Incidental catch and gear entanglement	28,191	14,461	2,255	330	150

Table 2.

The historic non-subsistence direct harvest was confined to pups, and took place during a discrete subinterval, 1963-1972, of the period to which it is assigned.

Based on the Calkins and Pitcher (1982) analysis of sex ratio, age distribution, pregnancy rates, and female reproductive history, of the individuals aged 3 and older in their sample of 250 sea lions collected between 1975-1978 from various sites in the Gulf of Alaska, and assuming stable age distribution and zero population growth, an adjusted model life table (Rebecca Taylor, pers. com.) for the SSL is developed in Table 3.

age class	**survival rate**	**fecundity**	**stable age distribution**	**reproductive value**
1	0.756	0.000	0.1650	1.000
2	0.756	0.000	0.1247	1.323
3	0.756	0.000	0.0943	1.749
4	0.867	0.100	0.0713	2.313
5	0.880	0.178	0.0618	2.552
6	0.888	0.257	0.0544	2.698
7	0.893	0.310	0.0483	2.748
8	0.896	0.310	0.0432	2.730
9	0.897	0.310	0.0387	2.700
10	0.898	0.310	0.0347	2.664
11	0.896	0.310	0.0311	2.621
12	0.894	0.310	0.0279	2.579
13	0.894	0.310	0.0250	2.538
14	0.894	0.310	0.0223	2.492
15	0.894	0.310	0.0200	2.440
16	0.894	0.310	0.0179	2.382
17	0.894	0.310	0.0160	2.317
18	0.894	0.310	0.0143	2.245
19	0.894	0.310	0.0128	2.164
20	0.894	0.310	0.0114	2.073
21	0.894	0.310	0.0102	1.972
22	0.894	0.310	0.0091	1.859
23	0.894	0.310	0.0082	1.732
24	0.894	0.310	0.0073	1.591
25	0.894	0.310	0.0065	1.432
26	0.894	0.310	0.0058	1.057
27	0.894	0.310	0.0058	1.255
28	0.894	0.310	0.0047	0.835
29	0.894	0.310	0.0042	0.587
30	0.894	0.310	0.0037	0.310

Table 3.

This model life table embodies more detail than can be supported specifically by the available data. Further, it is based on samples from a particular restricted spatial area and time period. Its only use here will be to correct for the effect of the age selectivity of the non-subsistence pup harvest, getting the pup harvest expressed in units dynamically commensurate with the mortality counts from the non-selective extraneous mortality factors.

In this model life table, the mean per capita reproductive value in the population, at stable age distribution, would have been 1.98893 and the cohort generation time would have been 12.05 (in units of age class). Accordingly, a pup harvest of 1 individual would have been demographically equivalent to an unselective harvest of 1/1.98893 individuals.

With the exception of the non-subsistence pup harvest, we will assume that all the other extraneous mortalities in Table 3 are unselective with respect to age and sex, and operated at constant per capita rates within each respective time period,

Prey Competition from Fisheries

Retrospective consideration of the history of the fisheries during the period 1958-2004 allows reasoned assignment of relative magnitudes of the mortalities that might be attributed to prey-competition fishery effects in the respective intervals. In this fashion, a subgroup of the SSLRT proposed a relative schedule of prey-competition fishery effects, expressed as instantaneous per capita mortality on a per year basis. These are listed below in Table 4. Note that from the history of the fishery it is believed that the prey competition effect was negligible in the first 10 years of the 1958-1977 period.

Start year of period	**Prey competition fishery effect (relative)**
1958	0.0
1968	0.015
1977	0.050
1985	0.060
1989	0.025
2000	—

Table 4.

As of 2000, the protections afforded around the rookeries, in critical habitat, and by the time area closures, under the terms of the Biological Opinion, were thought to have reduced to a low level the fisheries prey competition effect due to local interactions, so all the fishery competition effect then remaining would have been the spatially diffuse effect of the TAC itself. The SSLRT subgroup was not able to put a quantitative estimate on that effect, even relative to the other components of the prey-competition fishery effect.

If it is assumed, as per the 2000 Biological Opinion, that the fishery restrictions imposed by the chosen Reasonable and Prudent Alternative in 2000 achieved a reduction of the prey-competition fishery-related depression of population growth by an amount corresponding to an absolute increment of 2.5% in annual growth rate compared to the prey competition fishery effect operating in the 1989-2000 interval, the assumed by-interval prey competition fishery effects may be scaled absolutely. With this assumption, the fishery prey competition effect, exclusive of the spatially diffuse TAC effect, could be read at face value from Table 4. The SSLRT PVA workgroup did not adopt this assumption, but we shall pursue it as a scenario in the analysis here.

CORRECTING POPULATION GROWTH FOR COUNT OF EXTRANEOUS DEATHS AND FISHING MORTALITY

Our interest is to calculate an underlying natural growth rate r_n for each interval, from the observations on net population growth and estimates of the count of deaths from extraneous factors and assumptions about mortality owing to prey competition from the effects of the fishery.

Consider an interval where the known sum of extraneous deaths q_x was the result of a constant instantaneous per capita extraneous mortality rate ρ_x operating within that interval, and where the assumed mortality rate attributed to the fishery-caused prey competition ρ_f also operated as a constant instantaneous per capita extraneous mortality rate, and where the observed population sizes at the beginning N_{t_b} and end N_{t_e} of the interval resulted from operation of a constant net per capita instantaneous growth rate r_Σ during the interval.

Within the interval, at time t, the population size may be interpolated as

$$N_t = N_{t_b} e^{r_\Sigma (t - t_b)} \quad . \qquad \text{eq[1]}$$

The instantaneous change in population size may be parsed as

$$\frac{dN}{dt} = r_n N_t - \rho_x N_t - \rho_f N_t \quad , \qquad \text{eq[2]}$$

where the first term is the net contribution of the underlying natural growth, the second term is the deaths and suppression of reproduction owing to the extraneous mortality tallied in the counts, and the third term is the prey competition fishery effect.

Then, q_x, the presumed known sum of extraneous deaths during the interval, must be equal to the integral of the deaths owing to ρ_x over the interval

$$\begin{aligned} q_x &= \int_{t_b}^{t_e} \rho_x N_t dt \\ &= \int_{t_b}^{t_e} \rho_x N_{t_b} e^{r_\Sigma(t-t_b} dt \\ &= \rho_x N_{t_b} e^{-r_\Sigma t_b} \int_{t_b}^{t_e} e^{r_\Sigma t} dt \\ &= \frac{\rho_x N_{t_b} e^{-r_\Sigma t_b}}{r_\Sigma}(e^{r_\Sigma t_e} - e^{r_\Sigma t_b}) \\ &= \frac{\rho_x}{r_\Sigma}(N_{t_e} - N_{t_b}) \quad . \quad \text{eq[3]} \end{aligned}$$

From

$$r_\Sigma = r_n - \rho_x - \rho_f \quad , \quad \text{eq[4]}$$

and

$$r_\Sigma = \frac{1}{t_e - t_b} \ln(\frac{N_{t_e}}{N_{t_b}}) \quad , \quad \text{eq[5]}$$

we have, therefore,

$$r_n = \frac{1}{t_e - t_b} \ln(\frac{N_{t_e}}{N_{t_b}})(1 + \frac{q_x}{N_{t_e} - N_{t_b}}) + \rho_f \quad , \quad \text{eq[6]}$$

which may be calculated directly for all intervals in the SSL data series except the first.

The first census interval, 1958-1977, presents some additional complications because the non-subsistence pup harvest was confined to the subinterval 1963-1972, and the fishery prey competition effect is presumed to have operated only for the subinterval 1968-1977. Therefore we have 4 effective subintervals, 1958-1963, 1963-1968, 1968-1972, 1972-1977, with different r_Σ, ρ_x, and ρ_f operating in each, but with an actual census available only for 1958 and 1977. This required numerical solution.

For notational compactness, designate the 4 respective subintervals with the subscripts $\alpha, \beta, \gamma, \nu$, referring also to the start year of the subinterval. Let ρ_x be the presumed constant extraneous mortality rate, exclusive of the pup harvest and the fishery prey competition effect, presumed operating at the same intensity in all 4 subintervals. Designate q_x as the tally of these deaths over all 4 subintervals. Let ρ_p be the presumed constant mortality rate owing to the pup harvest and operating only in subintervals β and γ. Designate q_p as the tally of the pup harvest deaths corrected for the age selectivity. Designate ρ_f as the presumed known growth rate reduction owing to fishery caused prey competition, presumed constant, but operating only in subintervals γ and ν.

The net realized growth rates for the respective subintervals, then, are

$$r_{\Sigma\alpha} = r_n - \rho_x \quad , \quad \text{eq[7]}$$

$$r_{\Sigma\beta} = r_n - \rho_x - \rho_p \quad , \quad \text{eq[8]}$$

$$r_{\Sigma\gamma} = r_n - \rho_x - \rho_p - \rho_f \quad , \quad \text{eq[9]}$$

$$r_{\Sigma\nu} = r_n - \rho_x - \rho_f \quad . \quad \text{eq[10]}$$

With a set of trial values for $(r_n, \rho_x \rho_p)$ these relationships allow a calculation of the associated intermediate population sizes

$$N_\beta = N_{1958} e^{5 r_{\Sigma\alpha}} \quad , \quad \text{eq[11]}$$

$$N_\gamma = N_\beta e^{5 r_{\Sigma\beta}} \quad , \quad \text{eq[12]}$$

$$N_v = N_\gamma e^{4 r_{\Sigma\gamma}} \quad , \quad \text{eq[13]}$$

and an associated "prediction" of N_{1977} that can be compared to the observed for a calculated discrepancy

$$\Delta N_{1977} = |N_{1977} - N_v e^{5 r_{\Sigma v}}| \quad . \quad \text{eq[14]}$$

Similarly, calculations following the logic of eq [3] can be used to "predict" associated values of ρ_x and ρ_p that can be compared to the trial values for calculated discrepancies

$$\Delta\rho_x = |\rho_x - \frac{q_x}{(N_\alpha - N_{1958})/r_{\Sigma\alpha} + (N_\beta - N_\alpha)/r_{\Sigma\beta} + (N_\gamma - N_\beta)/r_{\Sigma\gamma} + (N_{1977} - N_\gamma)/r_{\Sigma v}}| \quad , \quad \text{eq[15]}$$

$$\Delta\rho_p = |\rho_p - \frac{q_p}{(N_\beta - N_\alpha)/r_{\Sigma\beta} + (N_\gamma - N_\beta)/r_{\Sigma\gamma}}| \quad , \quad \text{eq[16]}$$

The quantity of interest is solved by a numeric search for the combination (r_n, ρ_x, ρ_p) that minimizes a weighted sum of the discrepancy terms. To adjust for the difference in scale between the mortality rate terms and the population size term, the latter was divided by 1000 in forming the weighted sum. The best combination found resulted in $7x10^{-4}$ as the weighted sum of discrepancy terms.

The set of calculated values of the by-period underlying natural growth rates, under the set of estimates for the extraneous mortality factors in Tables 2 and 4, in the base scenario taking the estimates in Table 4 at face value as absolute magnitudes, is given in Table 5.

Year of census	**Population size estimate**	**Realized (net) annual influences growth rate**	**Extraneous exponential growth rate**	**Underlying exponential**
1958	227571	-0.00906	0.02603	+0.01697
1977	191571	-0.04737	0.07010	+0.02273
1985	131143	-0.16843	0.07304	-0.09540
1989	66857	-0.04134	0.03487	-0.00647
2000	42429	+0.02813	0.01058	+0.03871
2004	47483			

Table 5.

The values in Tables 2 and 4 are open to question, so there is uncertainty attached to the last column in Table 5. The PVA subgroup of the SSLRT thought the methods used to arrive at estimates for the harvests, shooting, and incidental take, probably gave rise to defensible *minimum* values, but it was not clear what basis was available for expanding these to defensible central estimates. The absolute magnitudes of the values for the fishery prey competition effects are speculative, but their relative magnitudes are indirectly supported by historical knowledge of the fishery.

It is sobering, in this connection, that for 3 of the 5 periods the estimated net effect of the extraneous influences is larger than the absolute magnitude of the estimated underlying growth rate. So the uncertain estimates of the components of the extraneous influences will matter to the analysis, and this uncertainty must be borne in mind in interpreting the results.

Figure 2 shows the 6 census estimates as circles, plotted against year, and the heavy line connecting the census estimates shows the trajectory corresponding to constant exponential growth within each interval, at the by-interval realized rates calculated in Table 1, while the thin trajectory shows a projection of a population initiated at the observed population size in 1958, and growing subsequently according to the calculated underlying growth rates for each respective period, as given in Table 5. This projected trajectory represents what would have happened, in the absence of density dependence, if, from 1958 on, the population had been released from the extraneous influences attributable to human activities as estimated in Tables 2 and 4.

The credibility of the underlying rates calculated to form Figure 2 rests on the estimates of extraneous mortality and reproduction suppression in Tables 2 and 4. The plausibility of the hypothetical trajectory shown in Figure 2, in particular the plausibility of whether the population would have reached the high densities shown, rests on whether density dependence would have kicked in at these high densities. The applicability of the calculated underlying rates at densities such as have been observed in the last 50 years is not connected to the question of density dependence at higher population levels.

ASSUMPTIONS AND ALTERNATIVE HYPOTHESES

Assumptions of the PVA

The PVA model assumes that the population dynamics in the foreseeable future will be governed by growth rates that change randomly at random intervals, where the lengths of the intervals represent environmental phases which have a characteristic average duration, and where the distribution of the growth rates is consistent with the sample observed in the past 50 years, adjusted for known extraneous factors that will not be represented in the future. This random growth is not density dependent. This kind of random walk in the population dynamics leads eventually either to extinction or impossibly large population sizes, so the very long history of the SSL could not have been governed by such dynamics.

In other words, the PVA model makes the two very particular assumptions, that (1) something in the SSL ecosystem changed in recent times and this is responsible for these ultimately unsustainable random density independent dynamics, and that (2) this change will continue into the foreseeable future.

These two assumptions do not commit to the details of what it is that changed. There are several possibilities such as effects of long term climate change, or community restructuring as a result of more than a century of harvesting heavily at certain levels of the marine food web. This could be consistent with some, but not all, of the hypotheses that have been advanced to attempt to explain the recent SSL decline.

From the standpoint of the PVA analysis, there are four main clusterings of pertinent hypotheses. These are

Hypothesis I.

Recently significantly altered ecosystem; the changed ecosystem condition will persist; the dynamics of the SSL observed in the last 50 years (after adjustment for identified extraneous mortalities that will not be repeated) are representative of the dynamics that will be manifested in this new ecosystem state; the dynamics are not significantly stabilized by density dependence in the range of population sizes between the recent levels and the level where Allee effects and loss of genetic diversity could become significant

Hypothesis II.

Recently significantly altered ecosystem; the changed ecosystem condition will persist; the dynamics of the SSL observed in the last 50 years (after adjustment for identified extraneous mortalities that will not be repeated) constitute a transition to a new lower carrying capacity; the future dynamics will exhibit sufficient density dependence around this new lower carrying capacity that, provided the identified extraneous mortalities of the past 50 years are not repeated, the population will not be at risk of declining within a 100 year time frame to a level where Allee effects and loss of genetic diversity could become significant

Hypothesis III.

Recently significantly altered ecosystem; the ecosystem condition will restore itself in the near future and stay in the restored (normal) state; in the restored state the SSL dynamics will themselves return to normal, and provided the identified extraneous mortalities of the past 50 years are not repeated, the population will not be at risk

Hypothesis IV.

The recent ecosystem dynamics are not abnormal in the long run for this population; the dynamics of the SSL observed in the last 50 years (after adjustment for identified extraneous mortalities that will not be repeated) are not abnormal for the population; the SSL population dynamics are sufficiently density dependent at population levels below the recent range that, provided the identified extraneous mortalities of the past 50 years are not repeated, the population will not be at risk of declining within a 100 year time frame to a level where Allee effects and loss of genetic diversity could become significant

Hypothesis **I** is the hypothesis underlying the PVA. Hypothesis **II** is distinguished from it by the interpretation that the recent SSL decline was a one-time transition and the assumption of strong density dependence. Hypothesis **III** is distinguished from the hypothesis underlying the PVA by the assumption that the recent dynamics will not be repeated and that the ecosystem conditions responsible for the recent dynamics will not be repeated. Hypothesis **IV** is distinguished by the assumption of very strong density dependence manifesting itself at lower population levels but not throughout most of the range of population levels experienced by the SSL.

All four hypotheses in the abstract have at least the plausibility of being scientifically possible. Hypothesis **IV** is perhaps undermined by the current genetic evidence probably being inconsistent with a history of such wide population swings, but this interpretation needs to be pursued with explicit genetic modeling. Otherwise, the available evidence is not compelling for selecting among the hypotheses.

The effective strength of density dependence and the carrying capacity that it confers play an important role in three of the hypotheses (**I**, **II** and **IV**). The recent dynamics of the SSL do not display density dependence. Thus, there is no basis in evidence specific to the SSL for justifying the assumption that density dependence is operating with the strength that would need to be assumed under hypotheses **II** and **III** (but modeling can quantify the strength of density dependence that is assumed under each hypothesis). On the other hand, it is generally assumed that normal population dynamics for populations in normal circumstances will be density dependent. So the adoption of hypothesis **I** assumes that the recent (and foreseeable) circumstances of the SSL are not normal, which of course is possible. The plausibility of circumstances that can cause population dynamics to appear density independent is explored theoretically in Appendix B.

We are left then with conclusion that assigning relative probabilities to the four hypothesis clusters will be largely a matter of expert judgment tempered by the precautionary principle which would give greater weight to the more pessimistic Hypothesis **I** (which is the basis of the PVA) and also buttressed by the scientific principle of parsimony which would also give greater weight to Hypothesis **I**, in that Hypothesis **I** extrapolates most directly from the actual evidence, and so introduces fewer new assumptions.

Combining the PVA with the Alternative Hypotheses

Because the defining metric in the PVA result is a probability, analyses can be combined across scenarios by simply averaging the results weighted by probabilities assigned to the scenarios. This extends to consideration of some alternative core models as might be embodied in some of the hypotheses described above.

In particular, the three alternative hypotheses that depart from the assumptions of the PVA model can be distilled in modeling terms to two pertinent alternative conceptualizations: (1) that the population is subject to sufficiently strong density dependence that will over-ride the effects of fisheries and natural environmental variation to confer a very high probability that the population will not decline to or below the quasi-extinction level, or (2) that the recent history of the population was a consequence of freak conditions, not directly related to the local fishery operations, and these conditions have a negligibly small probability of recurring within the next 100 years.

While there is no compelling evidence for adopting either of these conceptualizations with high confidence, these alternatives cannot be entirely dismissed either. A precautionary approach would accord them low, but not necessarily zero, prior probability.

Since each of these alternatives predicts zero probability of quasi-extinction within 100 years, within a wide spectrum of management scenarios, it is a simple matter to average this together with the numerical result of the PVA as described here. Formally, if the PVA, which assumes that both alternatives are not true, yields a predicted probability of quasi-extinction of X%; and the prior probability assigned to either or both of the alternatives being true is Y%; then the overall probability of quasi-extinction is

$$Z = (1 - Y * 100) * X\% \qquad . \qquad \text{eq[17]}$$

The PVA subgroup of the SSLRT adopted the value 20% for Y, to discount the risk estimate from the PVA for the probability that the model assumptions used in the PVA are fundamentally wrong.

MODEL STRUCTURE

Assume that, episodically, the population exponential growth rate *r* is drawn from a normal distribution with unknown, but stationary, mean μ_r and standard deviation σ_r. Assume that this growth rate *r* governs the population dynamics for the duration of the regime phase. Assume that the regime persistence is controlled by a constant Markovian annual persistence probability, so the persistence time will have a geometric distribution, and for annual persistence probability p_1, the mean phase duration $\overline{t}$ will be

$$\overline{t} = \frac{p_1}{1 - p_1} \quad . \qquad \text{eq[18]}$$

At each phase transition, the new value of *r* is drawn from the underlying normal distribution.

This is essentially the Brownian motion model used by Dennis, Munholland and Scott (1991), modified for predominantly "regime-like" variation in growth rates. The Dennis et al model has been widely used (reviewed by Kareiva in an NCEAS session). Of the plausible stochastic PVA approaches, the Dennis et al model makes the least data demands. Because it is highly simplified, it represents a kind of "null" model which is used by default unless data are available to support a more complicated model, or unless available data convincingly contradict the assumptions of the model in the context of its use.

In deference to the more or less decadal time scale of the PDO, the model was run with a specified mean phase persistence time of 10 years. This also corresponds roughly to the temporal resolution of the available population-wide estimates for the US portion of the western DPS of the SSL.

ESTIMATION OF MODEL PARAMETERS

Treat the 5 calculated underlying growth rates, based on the 6 population-wide census estimates and the estimates of extraneous influences, as representing an iid sample of 5 realizations from this process, and carry out a Bayesian inference, with conventional vague priors, on this basis, for the unknown mean and standard deviation of the underlying normal distribution of the exponential growth rates. The prior used for the mean μ_r was a uniform over a broad enough range so that the range did not influence the posterior. The conventional vague prior used for the process standard deviation σ_r had the probability of the variance σ_r^2 proportional to the reciprocal of the variance over a broad enough range so that the range did not influence the posterior.

The joint posterior distribution for (μ_r, σ_r) encompasses the parameter uncertainty pertaining to the distribution from which actual growth rates are drawn. Any particular value of the standard deviation σ_r represents the process variation for the stochastic population model. This estimation approach extends the Dennis et al procedure by incorporating parameter uncertainty, in the spirit of Ludwig (1996).

In the actual event, the available sample size of observed population-wide decadal growth rates is small for the SSL, so the parameter uncertainty in the estimation of μ_r and σ_r is large. Numerical exploration reveals that the uncertainty in μ_r is especially influential on the PVA results.

If the inference is carried out directly on the sample of growth rate values listed in Table 1, the resulting estimate of the parameters of the growth rate distribution will apply to the set of extraneous influences that were operating at the time of those censuses. So, use of these estimates for future projection either assumes that those extraneous influences will continue to operate similarly in the future, or requires adjustment for the different operation of extraneous influences in the future.

If the inference is carried out directly on the calculated sample of underlying growth rate values listed in Table 5, the resulting estimate of the parameters of the growth rate distribution will apply for a future in which all the extraneous influences, estimated in Tables 2 and 4 and accounted for in calculating Table 5, are eliminated. Other scenarios may be created by estimating parameters from the underlying sample (Table 5) for the retrospective analysis, and, for the prospective analysis, adding to these estimates the effects of the assumed values for the future operation of extraneous influences. In particular, analyses to evaluate the adequacy of downlisting or delisting criteria need to be explicit about what level of protections–and therefore what magnitude of extraneous influences–are presumed to be operating *after* the downlisting or delisting, so that these are reflected correctly in the risk assessment.

USE OF THE MODEL FOR ASSESSING PROBABILITY OF PERSISTING FROM PRESENT STATE

The algorithm for use of the model for assessing the probability of persistence is as follows:

1. Conduct a retrospective analysis from the data and estimates of the history of extraneous influences, to obtain a posterior distribution of the parameters (μ_r, σ_r) that define the distribution of the underlying growth rates.
2. Then, for the prospective analysis, compute a sample of population trajectories, each spanning the time horizon of interest (100 years), and each initialized at a specified population size. For each trajectory draw one realization of (μ_r, σ_r) from the posterior distribution obtained by the Bayesian inference, and adjust that value for whatever is assumed about the extraneous influences (and management) operating in the future. Initialize the environmental state with the last growth rate observed in the data series (adjusted as per the assumptions about future extraneous influences). For each time step in the model, project the population according to the prevailing growth rate, and sample a binomial with parameter $(1 - p_1)$ for regime phase transition. At each transition draw a new value of r from the normal with parameters (μ_r, σ_r) as assigned to that trajectory. At each time step test whether the population size has declined below the quasi-extinction level. Record the fraction of trajectories which have declined below the quasi-extinction level at least once before reaching the time horizon. One minus this fraction is the probability that the population will persist till the time horizon without quasi-extinction.

Quasi-extinction Level

From genetic considerations, the quasi-extinction level chosen by the PVA subgroup as the reference point for a standard is an *effective* population size of 1000. This value is in the range of the values currently proposed in the conservation genetics literature for maintaining long term genetic viability (reviewed by Allendorf and Ryman, 2002). Taking into account the fraction adult, fraction reproductive, and sex ratio among reproductive individuals in the population, as estimated in a draft analysis by a subgroup of the SSLRT, an effective population size of 1000 for the SSL corresponds to a total population size of 4743 individuals. This will define quasi-extinction in the analyses conducted here.

Persistence Standard

The PVA subgroup of the SSLRT adopted a standard of less than 1% probability of quasi-extinction within 100 years as the risk threshold for downlisting from ESA Endangered to ESA Threatened. This standard and its rationale are documented in an earlier NMFS workshop considering recovery criteria for large whales (Angliss, et al., 2002).

PVA STANDARD FOR DOWNLISTING FROM ENDANGERED TO THREATENED

PVA standards for persistence generally are stated in terms of a probability for persisting a specified period of time without declining below an extinction, or quasi-extinction, threshold. The standard adopted by the SSLRT PVA subgroup for downlisting from ESA Endangered to Threatened is:

Greater than 99% probability that the population will persist for 100 years without declining below the quasi-extinction level.

That standard will be used in this analysis.

Base Scenario

The base scenario considered here estimates dynamic parameters from the underlying sample (Table 5) for the retrospective analysis, and, for the prospective analysis, adds to these estimates the extraneous influence rates associated with the 2000-2004 period. This assumes that the protections of the RPA adopted under the 2000 BiOp, and the spatially diffuse effects of the current TAC, will continue into the future, even after downlisting and delisting, should these occur.

Time to Quasi-extinction from 2004

Under the base scenario, the inference on the parameters of the distribution of underlying growth rates yields the posterior marginals for (μ_r, σ_r) shown in Figures 3 and 4. The distribution of μ_r is approximately normal (theoretically it is expected to be t-distributed), and the distribution of σ_r is gamma-like, and their posterior correlation is approximately 0, showing that this Bayesian inference has properties rather like a t-test (which would be expected theoretically).

The posterior distribution of the derived quantity, time to quasi-extinction (truncated at 500 yr), is shown in Figure 5, with a blow up for times up to 100 years in Figure 6, and a cumulative for times up to 100 years in Figure 7. The probability of quasi-extinction within

500 years from the date of the assessment, 2004 (the date of the last census in the data series used), is 73.13% (Figure 5). The probability of quasi-extinction within 100 years from the date of the assessment is 37.23% (Figures 6 and 7), so the present state is far from qualifying for the recovery standard.

The distribution of population size at the 100 year time horizon in this analysis is shown in Figure 8. This shows a 83.20% probability of numbers less than 500,000 so the absence of density dependence is allowing some degree of optimistic “escape” very high population size by some modest fraction of the trajectories. If we consider 500,000 to be the plausible ecological limit for the SSL population size, then the model allows 16.8% of the trajectories to escape to unreasonably large population sizes, but even if density dependence held these trajectories below 500,000 very few of them would give rise to quasi-extinctions, because these are predominantly trajectories driven by more favorable values of (μ_r, σ_r).

Diagnosis of the Sources of the Present Risk

Much of the extinction risk emerging from the results of the inference in the retrospective analysis is owing to the presence in the sample of the 1985-1989 period where the estimate of the underlying growth rate is a disconcerting -9.45% after correction for the estimates of the extraneous influences during this period. If this period were omitted from the sample used in the sample for the retrospective analysis, the remaining sample of underlying growth rates would have a definitely positive mean and a much reduced variance. Such a retrospective inference would lead to a more favorable prospective analysis. The effect is dramatic: the assessment in 2004 would be that the probability of quasi-extinction within 100 years would be 3.47%, roughly one tenth the risk calculated when 1985-1989 is included in the sample.

USE OF THE MODEL FOR EVALUATING RECOVERY CRITERIA

Meeting the Standard in the Abstract

The standard generally is understood to be a threshold probability for persisting a specified period of time without dropping below the quasi-extinction level. Therefore, the most generalized statement of a recovery criterion, given a PVA model that calculates the probability of quasi-extinction, would be:

Any set of circumstances which the PVA calculates to have a probability of quasi-extinction that meets the standard, qualifies for a determination of “recovered.”

This approach is certainly logically coherent, and it also has the property of *efficiency*, as will be discussed below, but it does not have the motivating appeal of superficial simplicity and concreteness. In other words, intuitive comfort with a generalized criterion of this sort requires a degree of technical understanding.

Simple concrete criteria, expressed in terms of a threshold population level and/or a period of threshold population growth, can serve as more readily understood targets. Conceivably, such targets could help to guide recovery strategy for a population where there are opportunities for various sorts of interventions with predictable effects on age-specific vital rates.

There are two types of approach to simple concrete criteria. One is a sample list of scenarios, that have been pre-analyzed to determine which do, and which do not, meet the standard. If the list of scenarios is not exhaustive, the list will not constitute a comprehensive decision rule, because circumstances can be encountered which are not categorized by the list of scenarios. Essentially, they function as illustrations, which may suffice to convince an audience about the reasonableness of use of the PVA to test for meeting the standard; and the list may suggest a pattern for guiding recovery strategy.

The second type of approach to concrete criteria collapses the dimensionality of the pertinent description of the population to a single orderable measure, and identifies a critical value such that above that value it is predicted that the standard will be met, and below that value it is predicted that the standard will not be met. For the collapsed description to be intuitively satisfying, the combining formula needs to be very simple, and the inputs need to be very simple. It remains to be determined whether such a collapsed description will actually be an effective predictor. If the predictive power is degraded by the collapsed description, maintaining a constant level of precautionary performance will require a wider safety margin. This will be inefficient.

Consider that a PVA making full use of the information available delivers a distribution of time to extinction as in the thick curve in Figure 9, where the spread of the distribution reflects both the real process variation in the projected stochastic population growth, and the uncertainty about the parameters of that projection. Imagine that this result just meets a particular standard, in that the tail of the distribution to the left of threshold level T_h has area A_c, where the standard is that the probability of extinction by time horizon T_h must not exceed A_c.

Increases in the parameter uncertainty, as would occur through use of some “index” rather than using all the available information, will increase the spread in the distribution, as in the medium curve in Figure 9. Because the tails of the medium curve are extended compared to the thick curve, more area under the medium curve will spill over into the tail to the left of T_h, giving a higher probability A_i of extinction before T_h, thus failing to meet the standard.

Therefore, to meet the standard, the simpler index needs to indicate a more favorable central value, for example a higher population level if the index is based just on population level, shifting the curve to the right till the tail area left of T_h again equals the standard A_c, as in the thin curve in Figure 9. This shift to a more demanding value of the less informative index is the added margin of safety that is the practical cost imposed by abandoning pertinent information.

These theoretical issues are explored in greater detail in Goodman (2002b). Figure 9 was created using the theoretical, Poisson failure process, model from that analysis, where the medium curve represents a scenario that differs from the thick curve only in the length of the observation period, affecting the uncertainty in the “data,” but with the same raw failure event rate in the data, and the mean time to quasi-extinction in the two scenarios is very similar (161 versus 153 years). But the scenario with the greater uncertainty has a considerably larger tail area for times less than 100 years. The thin line represents a scenario with the same length of observation period as the medium line, but with a substantially lower (40% smaller) raw failure event rate, giving about the same tail area below 100 years as the thick curve, and a considerably longer mean time, about 233 years.

Fully Specified Example Scenario Approach

The algorithm for use of the model for assessing the probability of persistence in a fully specified future scenario is as follows:

1. Express the candidate recovery scenario as a concrete population trajectory into the future, in terms of the quantities that the model uses as input. In the case of the Brownian motion model, these consist of (a) a certain number of regime phases with particular growth rates manifested, as inputs to the inference component, and (b) a particular population size reached, as input to the stochastic projection component.
2. Treat the stipulated future trajectory as if it were observed, and append it to the actual data series.
3. Carry out the PVA analysis for a projection from the end of the combined series of actual and stipulated "data," for a duration specified by the chosen standard. Compare the calculated probability of persistence to the standard: if it meets the standard, the candidate scenario can serve as a sufficient recovery criterion. I.e., if that scenario materializes, a determination of recovery would be justified.

But this would not be a necessary criterion, since there would be many possible ways of *not* matching this scenario, involving different population levels and different future growth rates–some might meet the standard, and some might not.

This fully specified example approach allows the postulated "example criterion" to influence the prospective PVA analysis both by setting a starting population size and by adding the stipulated trajectory between the present and that future starting time to the sample of observations that condition the inference on (μ_r, σ_r).

The only period of observed positive net population-wide growth for the Western US DPS showed an annual growth rate of just under 3% per year (2.813%). This is also roughly equal to the long term growth rate observed for the Eastern population, north of California. For these reasons it is reasonable to first explore recovery criteria premised on observing a continuing 2.813% growth for different periods of time.

To explore the adequacy of a recovery criterion based on a stipulated number of decadal "regime phases" of continued growth at 2.813%/yr (the current apparent growth rate), premised on all the present protections represented in the 2000-2004 period continuing into the future (including the 100 year period over which the projection is conducted to assess extinction probability), we add decadal intervals with 2.814% growth to the analysis, and redo the inference on (μ_r, σ_r) conditioned on those future observations, and redo the 100 year stochastic projection based on the new inference and with the new starting population size based on continued 2.813% growth.

This scenario premised on particular future data, yields a future inference with less uncertainty about (μ_r, σ_r) compared to the present, and a larger posterior mean for μ_r and a lower posterior mean for σ_r.

The result for the hypothesized future assessment in 2014, after one decade of growth to a population size of 62,911 is a higher probability of persistence from that time. Compared to the present base scenario, the probability of quasi-extinction within 100 years from the time of the assessment is reduced by roughly one third to 25.01%. For a hypothesized future assessment in 2024 after 2 decades of growth to a population size of 83,352 the probability of quasi-extinction within 100 years from the time of the assessment is reduced roughly by one third again to 15.97%. For a hypothesized future assessment in 2034 after 3 decades of growth to a population size of 110,434 the probability of quasi-extinction within 100 years from the time of the assessment is reduced by roughly one third yet again to 9.71%.

Concrete Criterion Approach

The algorithm for use of the model for assessing the probability of persistence from the time of future assessment conditional on having attained a concrete specified criterion is as follows:

1. Express the candidate concrete specified criterion as a threshold population size at a specified check in time. This is an unambiguous criterion since it partitions the set of all future trajectories into a subset that meet the criterion and the remainder that do not. The probability of falling in each subset, based on current information, can be calculated, as can the probability of meeting the actual standard, subsequently, conditional on falling in the subset that meets the criterion.
2. For the present case, analyze the existing data (through 2004), with the likelihood function modified to assign zero likelihood to all (μ_r, σ_r) combinations that give rise to trajectories that do not meet the specified criterion (assuming that the protections and extraneous influences of the 2000-2004 period continue into the indefinite future), while allowing the fit to the data for the 5 observed intervals to determine the likelihood for all (μ_r, σ_r) combinations that do meet the specified criterion. This is equivalent to inference on the combined data set of the 6 existing censuses assuming that each occurs at a regime transition, assuming the estimates of extraneous factors for those intervals assuming those values are exact, and using the incomplete hypothesized future information that the population exceeds the criterion threshold size at the specified check in time (but without knowledge of the particular population size at check in time, and without knowledge of the regime transitions between 2004 and the check in time or of the intermediate details of the population trajectory). The resulting inference on (μ_r, σ_r) then will be conditional on having met the specified criterion, and so the derived parameter, time to quasi-extinction, will also be conditional on having met the specified criterion.

Comparing the calculated probability of persistence to the standard will reveal the adequacy of the candidate concrete specified criterion.

For the example criterion of a population size of at least 83,352 individuals in 2024 (for comparison to the fully specified scenario of 2 regimes of constant growth at 2.813% to that same population size at that date), the resulting performance is 23.92% probability of quasi-extinction within 100 years of the check in time for the candidate concrete criterion. As expected, this performance is inferior to that of the comparison fully specified scenario, which yields a 15.97% probability of quasi-extinction within 100 years of the assessment in 2024.

For the scenario where the 1985-1989 period is excluded from the retrospective analysis, the performance for a criterion of a population size of at least 83,352 individuals in 2024, the resulting performance is 1.79% probability of quasi-extinction within 100 years of the check in time for the candidate concrete criterion. For the scenario where the 1985-1989 period is excluded from the retrospective analysis, the performance for a criterion of a population size of at least 62,911 individuals in 2014, the resulting performance is 2.55% probability of quasi-extinction within 100 years of the check in time for the candidate concrete criterion.

USE OF THE MODEL FOR ASSESSING THE PROBABILITY OF ATTAINMENT

The algorithm for assessing the probability of attainment of a concrete criterion is as follows:

1. Compute a sample of population trajectories, each spanning the time allotted for the recovery criterion, from present, and each initialized at the present population size. For each trajectory draw one realization of (μ_r, σ_r) from the posterior distribution obtained by the Bayesian inference from just the actual observations. For each time step in the trajectory, sample for regime shift; and at each regime shift draw the governing growth rate for that regime phase from a normal distribution with parameters (μ_r, σ_r), and project the population size accordingly until the next regime shift.
2. For each trajectory test whether the recovery criterion is satisfied within the allotted time. This fraction of trajectories that meet the criterion is the probability of attainment, from the present, of a state satisfying the recovery criteria.

It is worth noting, that this analytical mechanism could also be used for evaluating jeopardy, since jeopardy is defined as reducing the prospects for recovery.

For the example criterion of a population size of at least 83,352 individuals in 2024, the probability of attainment, assessed from 2004, is 40.45%. For the scenario where the 1985-1989 period is excluded from the retrospective analysis, the probability of attainment, assessed from 2004, is 42.86%.

STRATEGIES FOR REDUCING UNCERTAINTIES

The numerical values used for taking account of anthropogenic effects and regulation in the past data have a large influence on the outcome of the analysis. Uncertainties, because they introduce more spread into the distributions in the stochastic projection, increase the probability of quasi-extinction, and therefore force larger margins of safety into the calculated recovery criterion.

The exploratory removal of the underlying growth rate estimate for 1985-1989 from the sample used for this retrospective analysis was shown to be highly influential. Such removal cannot be justified from the present position, without greatly revising the judgment about the probability of alternative hypotheses (i.e., either this is encompassed in the alternative hypothesis of freak conditions that will not be repeated, or it constitutes a new alternative hypothesis about the uncertainty of the extraneous influences estimates for 1985-1989 compared to their uncertainty in the other periods in the data record). Of course, if new information were to come to light about a higher magnitude (than estimated in Tables 2 and 4) for the extraneous influences operating in 1985-1989, this could be used directly to obtain a new estimate for the underlying growth rate in this period, which would increase the estimate of the mean growth rate and decrease the estimate of its variance, leading to a lower calculated risk in the prospective analysis.

There probably is not much that can be done about the uncertainty surrounding past events, such as shooting, which were not actually monitored or reported.

The effects of the 10 and 20 nm trawling closures and the time-area closures are not known. Knowledge of these rates will also be crucial to evaluation of the circumstances under which such protections might be lifted. These rates could be addressed in the future with large scale experiments. The outcomes of those experiments might lead to less restrictive recovery criteria, and more generous regulations.

SUMMARY AND CONCLUSIONS

The representation of environmental variation in this PVA analysis, often downplayed in PVA models, introduces a large source of extinction risk with the parameter values estimated from the data for the western DPS of SSL (but not the eastern DPS). Even taking all the expert opinion estimates at face value, to attempt to account for some components of the observed population decline in the respective periods bracketed by survey estimates, the resulting sample of calculated regimal underlying growth rates has a negative mean and a large variance (sample variance), so the growth rate distribution inferred in the retrospective analysis drives the prospective analysis to high extinction probability. The negative mean and high variance in the sample of estimated underlying growth rates is owing in substantial part to the presence in the sample of the large negative value for the 1985-1989 period.

Assumptions and Sensitivity

The only scenario, encountered in this limited analysis, which offers some prospect for downlisting in the time frame of two decades, sets the downlisting criterion at a population size of 83,352 in 2024; and is subject to some significant side conditions. To obtain a risk estimate near the standard of 1% probability (1.43% is achieved) of quasi-extinction within 100 years, this criterion requires the following:

1. omission of the 1985-1989 estimate of underlying growth rate from the retrospective analysis, (restoring this period to the sample multiplies the calculated risk by a factor of 13.4),
2. acceptance of the harvest, shooting, and incidental catch estimates for the other periods as correct, (if these estimates are 25% too high, the calculated risk is multiplied by a factor of 1.056)
3. acceptance of the prey-competition fishery-effects relative estimates for the other periods as absolute and correct, (if these estimates are 25% too high, the calculated risk is multiplied by a factor of 1.54),
4. acceptance of the combined 20% probability of the alternative hypotheses for the core model, (if the true probability of the alternative hypotheses is 0, the calculated risk is multiplied by a factor of 1.25), and
5. assumption that the extraneous influences in the future will continue at the 2000-2004 level, even after downlisting, (if the sum of extraneous influences operating in the future is increased by a factor of 1.25, the calculated risk is multiplied by a factor of 1.21)

We see, from a sensitivity standpoint, that the key requirement, for the favorable assessment of this downlisting criterion, is the discarding of the data for the 1985-1989 period from the retrospective analysis. But the justification for this manipulation is obscure. The possibility that the natural conditions during 1985-1989 were abnormal (and will not recur) is already represented in one of the "alternative hypotheses," which has already been taken into account, and assigned a probability in the expert opinion of the PVA subgroup.

The remaining possibility is an additional "hypothesis" about the data. This is the possibility that the estimates (in Tables 2 and 4) of extraneous influences and prey competition effects for the 1985-1989 period are wildly unreliable (and grossly under-represent the true values), but the estimates for the other periods are reasonably accurate. This is *possible*, as it is known that the nature of the fishery activity during 1985-1989 was quite different from what it was in the other periods, and there was essentially no monitoring of the effects. On the other hand, there is no basis in evidence to assign a really high probability to this possibility.

Risk Control through Adaptive Policy

We are left then with a superficially attractive recovery criterion whose performance depends on an unsupported, but possible, hypothesis. If this criterion were adopted, and acted upon for downlisting in 2024, with current information, this would not control risk to the level demanded by the standard. A much more stringent criterion would be required to meet the standard, but this would entail a considerably longer wait for downlisting.

The resolution to this impasse is to accept the uncertain downlisting *conditionally*, and build into the plan for the management after downlisting a set of monitoring and protection contingencies that will control risk to a finer level than is attained by the criterion itself. The basic idea is to use the future monitoring to provide a level of certainty about the presently uncertain key assumptions, and to use the commitment for management responses to the results of that monitoring as a guarantee that the actual risk will be kept to an acceptable level.
The risk containment effect would be achieved by the three-fold commitment:

1. that downlisting does not reduce any protections until experimentation and monitoring proves (to a precautionary standard) that this lifting of protections will not raise the risk above the downlisting standard,
2. that there will be adequate monitoring for any indications of a departure from the critical assumptions, most especially a high resolution monitoring to detect any return of a period of rapid unaccounted for decline,
3. and a firm plan for increasing protections by an amount predicted to compensate adequately, should such a decline be detected.

The design, optimization, and rigorous evaluation of such an adaptive system is the next technical challenge to pursue in the development of the mathematics and computational implementation of Bayesian PVA analysis, building on the methods presented here for assessing conditional probabilities of extinction.

In pursuing the potential of adaptive recovery plans, the distinction between verbal and mathematical standards becomes extremely important. The label "adaptive management" has had some currency for over a quarter century, during which time it has suffered from denoting a wide spectrum of actually different enterprises. At one end of this spectrum is the rigorous theory of optimizing the trade-offs among experimentation, risk, and action under uncertainty (Walters and Hilborn, 1976). At the other end of the spectrum is the temptation to call any improvisational approach to management "adaptive." In practice, this latter has not proven helpful (Ludwig, Hilborn and Walters, 1993).

The crucial difference is that genuine adaptive management–far from being improvisational–develops, in advance, a plan that covers all contingencies, and has verified the optimization of the path that will be chosen in response to each possible outcome of the experiments and monitoring, including damage control for the eventuality of experiments with unfavorable outcomes. The Bayesian machinery illustrated in this PVA analysis offers the computational basis for calculating the probabilities that are used for driving the decision rules at each branchpoint in the plan path, and for evaluating whether a given plan, overall, controls risk to the desired level.

For legitimate use in the ESA context, an adaptive plan should be required to demonstrate in advance that the overall plan meets the recovery standard for risk when all possible paths are considered in relation to their respective probabilities, and taking into account the reduction in uncertainty that will be achieved from the results of the experiments (Goodman, 2005).

Grounds for Rejection

The Bayesian PVA approach, as employed in this example with the participation of the SSLRT PVA subgroup, provided a structured framework for orderly discussion of the critical elements necessary for reasoned development of many components of the recovery plan. This went beyond the recovery criteria section, linking it to the threats assessment, and the management and research planning as well.

Bayesian analysis and decision theory, as employed here for operationalizing value preferences and analyzing evidence to make decisions to satisfy the standard, constitutes a system with the property of "coherence" (Berger, 1985). Roughly, this means that the system will generate decisions that are consistent with the synthesis of values, beliefs, and evidence. In other words there is no logic for improving upon this system if a decision rule has been implemented with correct self-awareness of values and expert opinion, and competent use of modeling and statistics on all the available data.

This raises the question of what rationale might justify *not* following through with the results of such an analysis. In the present case, the persistence standard, the quasi-extinction definition, the population-wide census data, the estimates of the extraneous influences, and the judgment that the basic PVA model has an 80% probability of being correct, all were developed and/or reviewed by the same group of experts. Therefore, *this* group could not logically reject the result without exhibiting inconsistency with their own values and judgment and data. More broadly, rejection of the result raises the possibility that the recovery criterion will not be consistent with the threats assessment section of the Recovery Plan, if the PVA subgroup's expert input to this analysis was consistent with the threats assessment.

Some other group, of course, could reject the result if they genuinely disagreed with the initial value judgments (policy) or had a sound basis in defensible expert opinion for disagreeing with the estimates of the extraneous influences or the basic premise of the PVA model. Presumably there is not much scope for scientific disagreement about the objective data used in the analysis–but if new information were to come to light revising the data, the analysis should be revised accordingly.

Notwithstanding the recognized usefulness of the Bayesian PVA framework for structuring the efforts of the SSLRT PVA subgroup, and notwithstanding the degree of consensus achieved about the inputs, there was some discomfort within the group about the result. This bears closer diagnosis. In particular it would be valuable to elicit whether the discomfort primarily reflected disappointment or primarily disbelief.

If the problem is disbelief, this would warrant further review of the PVA to confirm the degree of confidence placed on the technicalities of the modeling and analysis. If the problem is disappointment, this too warrants further discussion. If the disappointment is simply an expression of impatience at how long a time might have to pass until downlisting or delisting were allowed under the strict criterion, it might be worth some discussion to discover why there is such a strong desire for earlier downlisting and delisting, given the premise that the present protections will be continued regardless. If earlier downlisting would remove the discomfort, even if the present protections are continued, the natural resolution is to accept the PVA, and opt for conditional downlisting and adaptive recovery planning as the means to allow earlier downlisting without disavowing the PVA inputs or the PVA result, and without compromising the standard.

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APPENDIX A.
ANALYSIS OF EASTERN POPULATION OF STELLER SEA LIONS

Introduction

The eastern population of Steller sea lions breeds on rookeries from SE Alaska to northern California. These subpopulations were dramatically reduced by human activities, primarily direct take, it is believed, in the period before protection under the MMPA. This reduction was probably at least as severe as that experienced by the western DPS during the same time period. With the institution of protection, the eastern DPS, unlike the western DPS, rapidly showed signs of gradually recovering throughout most, but not all, of its range. The exception is the southernmost portion of the range, in California.

The area occupied by the eastern DPS is a substantially different ecosystem from the area of the western DPS. There are different fisheries. The sea lions, as revealed from scat analysis, eat a different diet. Both areas are subject to oceanographic “regimes” but the regimes express themselves differently in their biological effects. From the standpoint of salmon production, for example, the biological effects appear to be reversed in phase.

The longest monitoring series available for the eastern population are the records of non-pup counts from Oregon, and the pup count series from SE Alaska, starting in 1977 and 1979 respectively. These two portions of the subpopulation area account for the bulk of the eastern DPS. Both time series of counts show consistent exponential growth estimates of about the same magnitude, centered around 3 to 3.5%, SE Alaska being the lower, with similar confidence interval widths, and with no indication of large environmental variation (or varying extraneous influences). This is quite different from the Western Alaskan population. The apparent difference between the SE Alaska and Oregon growth rate estimates is considerably smaller than the respective confidence interval widths.

Oregon Non-pup Counts

The 24 counts of non-pups at Oregon sites span the years 1977-2002, and are annual except for missing 1978 and 1991. The census estimate series was analyzed by Bayes fitting of a simple exponential, treating all the variation as observation (census) error, treated as multiplicative log normal, and assuming an exact underlying exponential trajectory with unknown growth rate, unknown true starting population size and unknown census error standard deviation. The priors on growth rate and starting population size were uniform, and the prior on the log space variance of the census error was proportional to its reciprocal. The posterior mean for the growth rate was 3.64%, the posterior standard deviation was 0.405%, and the posterior mode was 3.68%. The 95% posterior interval was from 2.42% to 4.44%. The posterior probability that the growth rate is less than 2% is about 0.01%. The posterior distribution for the growth rate is shown in Figure A1.

Figure A2 shows the trajectory from the posterior mode on growth rate and starting population, with the census values shown as dots about the trajectory, and with the log space residuals (the log of the multiplicative factor of departure from the exact exponential trajectory) shown below as dots about the $y = 0$ line. The residual analysis shows a slight tendency to cycle with a wave length of around 5 years, but overall the residuals appear to have a very stationary distribution with no systematic lack of fit to a simple exponential. This supports the assumption of essentially no environmental variation.

SE Alaska Pup Counts

The 9 counts of pups at SE Alaska sites span the years 1979-2002, and are generally less frequent than annual. The census estimate series was analyzed by Bayes fitting of a simple exponential, treating all the variation as observation (census) error, treated as multiplicative log normal, and assuming an exact underlying exponential trajectory with unknown growth rate, unknown true starting population size, and unknown census error standard deviation. The priors on growth rate and starting population size were uniform, and the prior on the log space variance of the census error was proportional to its reciprocal. The posterior mean for the growth rate was 3.13%, the posterior standard deviation was 0.413%, and the posterior mode was 3.14%. The 95% posterior interval was from 2.29% to 3.95%. The posterior probability that the growth rate is less than 2% is about 0.7% and the probability of growth rate less than 1.5% is about 0.1%. The posterior distribution for the growth rate is shown in Figure A3.

Figure A4 shows the trajectory from the posterior mode on growth rate and starting population, with the census values shown as dots about the trajectory, and with the log space residual (the log of the multiplicative factor of departure from the exact exponential trajectory) shown below as dots about the $y = 0$ line. The residual analysis shows no systematic lack of fit to a simple exponential, supporting the assumption of essentially no environmental variation.

The Theory of Inference with Census Error and Environmental Variation

For census error with no environmental (process) variation, the elementary model is for an underlying population trajectory that is a simple exponential. For constant multiplicative log normal census error, the log of the censuses will have a normal distribution about the log of the true population size. The census deviations, in the log space, under this model, will be iid, showing no serial correlation and no trend in mean or variance. This is the basis of the Bayesian inference used here for the eastern DPS of US SSL.

For process variation, but with no census error, the elementary model is for an underlying trajectory, in the log space, that is a 1-d random walk (Brownian motion). When viewed at regular time steps, the series of snapshots of population sizes corresponds to by-interval growth that samples a normal distribution of exponential growth rates with a stationary mean and variance. With the modification for regime-like variation, so that the intervals at which the population is observed are also intervals over which the population growth rate really is constant and the interval length itself samples an exponential distribution, this is the basis of the Bayesian inference used here for the western DPS of US SSL.

The elementary model allowing for both census error and process variation superimposes a multiplicative lognormal stochastic observation process on the underlying discrete time population growth process which samples a normal distribution of actual growth rates. For inference with this model, it is assumed that the trajectory of actual population size is unknown, and a likelihood function must be constructed around the series of censuses as observables. This could form the basis of a Bayesian evaluation of whether the underlying process variation really is negligible for a population where it is not known whether there is appreciable process variation and where there is substantial random census error.

The most thorough approach to implementation of this in a likelihood function for inference on the distribution of actual exponential growth rates would be by a state-space model that represents the unknown time series of actual population sizes as parameters that must be inferred jointly along with the mean and standard deviation of the growth rates and the standard deviation of the log space census error. The model would be hierarchical, since some of the unknown parameters (the parameters of the distribution of the growth rate) bear on the distribution of some of the other unknown parameters (the time series of actual population sizes).

Since the interest is in obtaining inference on the mean and standard deviation of the distribution of exponential growth rates, it is more natural to formulate the model in the state-space of the time series of unknown actual realized growth rates. This also has the advantage of clarity in that the hyperparameters specifying the distribution of growth rates directly express a probabilistic relationship among the growth rates which are themselves parameters. The state spaces of the trajectory of population sizes and actual realized growth rates are of course mathematically interconvertible as long as the population size at one point in the trajectory is included among the set of parameters.

Notation for the realized-growth-rate-space model is given in Table A1.

Observables

m	number of observations in the possibly irregular time series of censuses
$\underline{n}$	m-element vector of censuses, in order
$\underline{t}$	m-element vector of times of censuses, in units of time steps so that $t_1 = 1$ and $t_m = k$

Primary Parameters

$N_{t(m)}$	population size at end of census time series
σ_ε	standard deviation of the log of the multiplicative census error
$\underline{r}$	$(k-1)$-element vector of exponential rate of increase by time step
r_j	exponential rate of increase for time step j, iid normal

Hyper-parameters

μ_r	mean of the distribution of r
σ_r	standard deviation of the distribution of r

Derived Parameters

$\underline{N}$	$(k-1)$-element vector of population trajectory, one value per time step, starting at time of first census, derived from $N_{t(m)}$ and $\underline{r}$
N_j	population size at time step j
$\underline{\varepsilon}$	$(k-1)$-element vector of multiplicative census errors, one value per census, starting at time of first census, derived from $N_{t(m)}$ and $\underline{r}$, and $\underline{n}$
ε_j	multiplicative lognormal census error at census j, sampled from distribution with mean of 0 and standard deviation σ_ε in the log space

Table A1

Equivalently, the vector $\underline{r}$ could be designated derived, and the trajectory $\underline{N}$ designated among the primary parameters, giving the population size state-space model.

The dynamics of population growth with this model are

$$N_{j+1} = N_j e^{r_j} \quad , \qquad \text{eq[A1]}$$

$$r_j = \ln(N_{j+1}/N_j) \quad , \qquad \text{eq[A2]}$$

$$r \sim Gaus(\mu_r, \sigma_r) \quad . \qquad \text{eq[A3]}$$

So the corresponding prior component is

$$p(r_j) = \frac{e^{-\frac{1}{2}(\frac{r_j-\mu_r}{\sigma_r})^2}}{\sigma_r\sqrt{2\pi}} \quad . \qquad \text{eq[A4]}$$

The above is prior because it represents a relationship among parameters, and involves no observables.

Censusing and census error then give rise to

$$\ln(n_i) \sim Gaus(\ln(N_{t(i)}), \sigma_\varepsilon) \quad . \qquad \text{eq[A5]}$$

So the corresponding likelihood component is

$$p(n_i) = \frac{e^{-\frac{1}{2}(\frac{\ln(n_i)-\ln(N_{t(i)})}{\sigma_\varepsilon})^2}}{n_i\sigma_\varepsilon\sqrt{2\pi}} \quad . \qquad \text{eq[A6]}$$

The joint likelihood then is

$$p(\underline{n}|\underline{N}, \sigma_\varepsilon) \propto \prod_{j=1}^{m} \frac{e^{-\frac{1}{2}(\frac{\ln(n_j)-\ln(N_{t_j})}{\sigma_\varepsilon})^2}}{n_j\sigma_\varepsilon\sqrt{2\pi}} \quad , \qquad \text{eq[A7]}$$

where

$$N_{t_j} = N_{t_m} \prod_{i=t_j}^{t_m-1} e^{-r_i} \quad , \qquad \text{eq[A8]}$$

$$\ln(N_{t_j}) = \ln(N_{t_m}) - \sum_{i=t_j}^{t_m-1} r_i \quad . \qquad \text{eq[A8]}$$

So

$$\ln(p(\underline{n}|\underline{N}, \sigma_\varepsilon)) \propto -\sum_{j=1}^{m}\left(\frac{1}{2}(\frac{\ln(n_j)-\ln(N_{t_j})}{\sigma_\varepsilon})^2 + \ln(n_j\sigma_\varepsilon\sqrt{2\pi}\,)\right) \quad . \qquad \text{eq[A10]}$$

The model-driven joint prior elements then are

$$p(\underline{r}, \mu_r, \sigma_r) \propto \prod_{i=1}^{k-1} \frac{e^{-\frac{1}{2}(\frac{r_i-\mu_r}{\sigma_r})^2}}{\sigma_r\sqrt{2\pi}} \quad . \qquad \text{eq[A11]}$$

So

$$\ln(p(\underline{n}|\underline{N},\sigma_\varepsilon)) \propto -\sum_{j=1}^{m}\left(\frac{1}{2}(\frac{\ln(n_j)-\ln(N_{t_j})}{\sigma_\varepsilon})^2 + \ln(n_j\sigma_\varepsilon\sqrt{2\pi}\,) \right)$$

. eq[A12]

If we adopt conventional vague priors for the primary parameters, the log of the joint proportional posterior distribution is

$$\begin{aligned}\ln(p(\mu_r,\sigma_r,\sigma_\varepsilon,N_{t(m)},\underline{r}|\underline{n})) \propto & -\frac{1}{2\sigma_\varepsilon^2}\sum_{j=1}^{m}\left((\ln(n_j)-\ln(N_{t_m})-\sum_{i=t_j}^{t_m-1} r_i) \right)^2 \\ & -\sigma_\varepsilon\sum_{j=1}^{m}\ln(n_j) \\ & -\frac{1}{2\sigma_r^2}\sum_{i=1}^{k-1}(r_i-\mu_r)^2 \\ & -k\ln(\sigma_r)-\ln(\sigma_\varepsilon) \quad . \quad \text{eq[A13]}\end{aligned}$$

Numerical implementation of the Bayesian inference may be carried out by simulating the posterior distribution with a MCMC algorithm such as Metropolis. For the best case of a series of *m* censuses (observables) that are entirely consecutive, this model involves 2 necessary hyperparameters and $m + 1$ primary parameters. We see, therefore, that from the standpoint of the number of observations relative to the number of irreducible parameters, this is not a promising system for inference. Accordingly, we are assured that the priors will in fact be influential.

Further, we must consider robustness of the inference to details of the model. The most tenuous appearing aspect of this inference is the resolution of process variation from census error when both are conflated to an extent in the time series of the observable census estimates. The two kinds of variance do give rise to somewhat different looking patterns in the time series of census estimates. Figure A5 shows a typical realization from a simulation of the assumed model with the parameter values from Table A2 to illustrate a census trajectory with census error but no process variation. We see a high frequency variation about a simple exponential trend.

N_{1977}	1600
σ_ε	0.2
μ_r	0.035
σ_r	0.0

Table A2

Figure A6 shows a typical realization from a simulation of the assumed model with parameter values from Table A3 to illustrate a census trajectory with process variation but no census error. We see that the trajectory wanders, on a fairly long wavelength, giving rise to a systematic lack of fit to a simple exponential. This difference in pattern has formed the basis for some statistical attempts to separate the two kinds of variance (Holmes, 2001) by frequentist methods.

N_{1977}	1600
σ_ε	0.0
μ_r	0.035
σ_r	0.1

Table A3

Figure A7 shows a typical realization from a simulation of the assumed model with parameter values from Table A4 to illustrate a census trajectory with both census error and process variation. We note that of the three simulations (Figures A5, A6, A7), the case with no process variation looks most similar to the real data trajectory for the Oregon non-pup counts (Figure A2). But we also note that even this case has a somewhat different pattern from the actual data–the actual data show a slightly more consistent pattern of residual variance over the length of the trajectory, but a longer wavelength to the variation. We will return to a diagnosis of this difference in appearance after a mathematical dissection of the prospects for statistical separation of census error and process variation variances.

N_{1977}	1600
σ_ε	0.2
μ_r	0.035
σ_r	0.1

Table A4

The Mathematics of Distinguishing Census Error from Process Variation

Consider the derived quantity

$$q_j = \ln(\frac{n_{j+1}}{n_j}) \qquad . \qquad \text{eq[A14]}$$

for a pair of consecutive censuses that are separated by a single time step (at whatever time scale is chosen for the analysis). This new quantity is a transformation just of the observables, the census estimates.

Under the model assumed here, the variable q will form a stationary stochastic time series sampling a normal distribution with properties: mean μ_r, variance $\sigma_r^2 + 2\sigma_\varepsilon^2$, lag-1 autocovariance $-\sigma_\varepsilon^2$, and zero autocovariance at all other lags. This gives rise to a joint likelihood for a time series of q, with just the three parameters $(\mu_r, \sigma_r, \sigma_\varepsilon)$, as a multivariate normal with a highly structured covariance matrix. For a sequence $\underline{q}$ of consecutive values, of length s, all separated by one time step, the likelihood is

$$p(\underline{q}|\mu_r,\sigma_r,\sigma_\varepsilon) = \frac{e^{-((\underline{q}-\underline{\mu_r})'\Sigma_q^{-1}(\underline{q}-\underline{\mu_r}))/2}}{|\Sigma_q|(2\pi)^{k/2}} \qquad , \qquad \text{eq[A15]}$$

where Σ_q has, for all i, elements

$$\sigma_{(i,i)} = \sigma_r^2 + 2\sigma_\varepsilon^2 \qquad , \qquad \text{eq[A16]}$$

$$\sigma_{(i,(i+1))} = \sigma_{((i+1),i)} = -\sigma_\varepsilon^2 \qquad , \qquad \text{eq[A17]}$$

and 0 everywhere else. For a sequence of q with some time gaps, the likelihood is a product of such expressions for each segment without gaps.

This formulation makes possible numerical solution for the Bayes posterior distribution by means of a SIR algorithm, which will avoid the convergence issues to be expected with MCMC for the state-space formulation. For a data series with no gaps, the only difference in information provided to the two formulations will be the prior on N_m in the state-space model. For a data series with gaps, the state-space model derives some limited information, not available to the multinormal model, from the possible growth rates bridging each gap. In practice, the difference between the inferences should be slight.

The diagnostic insight provided by the formulation for inference with the data transformed to $\underline{q}$ is that the partitioning of the total variance into process variation and census error depends entirely on the serial correlation structure of $\underline{q}$. This, in turn, depends entirely on the assumed model having no serial correlation in the census error and no serial correlation in the process variation. Any departure from these time series properties of the census error and process variation will give rise to a different actual autocovariance structure for $\underline{q}$, so that a force fit to the structure presumed for the matrix Σ_q will misallocate the variance between σ_r and σ_ε in the inference.

In reality, we generally would *not* expect environmental variation to be free from serial correlation, and we would not be surprised at high serial correlation. The reasonableness of an assumption of no serial correlation in the census error depends on the mechanism of censusing. If the censusing (e.g. “sightability” or “availability”) is affected by age distribution in the population, or by breeding status, or by geographic distribution or behavior that might be affected by environmental variables, the census error can exhibit considerable serial correlation.

Serial Correlation in the Oregon SSL Data

Figure A8 shows the time series of the apparent exponential “growth rate” in the census estimates of non-pups from the Oregon sites. This is the vector of the quantity q calculated from these data. If the data conform to the assumptions of the model, the calculated time series should exhibit no serial correlation at lags greater than 1, and a negative serial correlation at lag 1 corresponding to the ratio of the census error variance to the sum of the process variance and twice the census error variance. The negative serial correlation at lag 1 should give rise to a preponderance of simple alternation between high and low values. Instead we see a high frequency of peaks and troughs that each persist for two years, and also there is considerable representation of intermediate values in the year between a high and a low.

In fact, the calculated time series of q shows a small negative serial correlation at lag 1, a large negative serial correlation at lag 2, and a small positive correlation at lag 3. From the model,we could obtain nominal point estimates of the census error and process variation standard deviations according to

$$\sigma_{\varepsilon} = \sqrt{-\gamma_q(1)} \quad , \qquad \text{eq[A18]}$$

$$\sigma_r = \sqrt{\sigma_q^2 - \sigma_{\varepsilon}^2} \quad , \qquad \text{eq[A19]}$$

where σ_q^2 is the variance in the time series of q, and $\gamma_q(1)$ is the lag-1 autocovariance.

Table A5 shows the sample serial correlation at the 3 lags, along with the total variance calculated as the geometric mean of the variances of the two series formed to compute the serial correlation at each lag, along with the "nominal" point estimates for the census error standard deviation and process variation standard deviation calculated by treating the serial correlation as if it were lag-1 and assuming the data conform to the model (which they definitely do not). [Program SERLAG and STDS]

lag	**serial correlation**	**total variance**	**nominal** σ_{ε}	**nominal** σ_r
1	-0.0981	0.0281	0.0525	0.1502
2	-0.5837	0.0261	0.1234	0.0661 **i**
3	0.1120	0.0269	0.0548 **i**	0.1813

Table A5

The imaginary values for nominal standard deviations arise when the partitioning of the variance according to the model gives rise to a negative point estimate for one of the variances, which might more properly be thought of as zero variance.

We see that if the population had been censused every other year, rather than annually, the data would have appeared to agree with the assumptions of the model, and the conclusion from inference with the model would be that there is little environmental variation, and the residuals from a fit to a simple exponential are almost entirely census error. As it is, the annual censuses show that the data do not conform to the assumptions of the model, for reasons that are undetermined, but there are many possible mechanisms that might be responsible for what almost certainly is serial correlation in the census error.

Risk Evaluation for the Eastern Population

Based on the monitoring of SE Alaska and Oregon subsets of the population, the eastern DPS, north of California, for the past 25 or 30 years, appears to have been growing steadily, but at a modest rate that is perhaps 1/3 of the rate that would represent the common understanding of the biological potential for a pinniped population that is not experiencing crowding effects, impaired habitat, or ongoing harvest or incidental take. During this time, the population size has more than doubled, and the total population size for the DPS is now estimated to be around 46,000. During this time, there has been no evidence of appreciable effects of environmental variation, or of appreciable variation in whatever human-caused extraneous influences may be affecting this population.

The present size size of the California portion of the population, which occupies the southern edge of the species range, is about 20% of that recorded there in the middle of 20th century, and it is believed that the population may have been larger yet in the 19th century. In recent times, consistent California-wide counts began in 1996. During the recent decade of monitoring pup production in California has trended upward, while non-pup numbers have varied from one census to the next, but with no clear trend.

A working hypothesis to account for these observations on the eastern DPS is that:

1. the population is not sensitive to ongoing regime-frequency environmental variation,
2. the depressed, but steady and positive, growth rate north of California is owing to a combination of ecosystem modification and possible incidental take that is stable and sustainable,
3. the carrying capacity is not less than 46,000 total individuals, and
4. the lack recovery of the California portion of the population is owing to a range contraction responding to the warming trend of the past several decades.

If all this is true, and continues to be true, the risk of near- or medium-term extinction for this population is very low. While there is no evidence to the contrary, we do not have conclusive information that this hypothesis complex is true, or that it will continue to hold in the future. Accordingly we could judge this population to be at low risk provided management maintains the current level of protection, keeps human impact at no more than its present level, and monitors to make sure that evidence contrary to the hypothesis complex will be detected and the risk classification and management will be revised as indicated. The most critical contingencies to monitor for would be a northward extension of the region of the shift to a much reduced population density, or a shift to negative apparent census growth for more years than might be attributable to chance census error.

APPENDIX B.
DENSITY DEPENDENCE IN RELATION TO HABITAT, RESOURCE AVAILABILITY AND PREDATION PRESSURE

INTRODUCTION

Classic Density Dependence

A standard example of apparent density dependent population growth from marine mammal dynamics is the case of the Antarctic, where many of the seal populations and almost all of the great whale populations were severely reduced through direct harvest by mid 20’th century. Almost all of these species feed predominantly on krill (Laws, 1977). Upon cessation of the harvests, both the seal and whale populations rebounded at rates at the upper end of what is thought to be the biologically feasible range for their life histories. Sustained fur seal population growth rates from roughly 10% to 17% annually were estimated (Payne, 1977; Boyd, 1993). And even great whales, with their much later age at maturity and longer interval between births, showed high annual growth rates, from 7% for blue whales (Branch, Matsuoka, Miyashita, 2004) to nearly 11% for humpback whales (Bannister, 1994).

Steller seal lion population dynamics certainly do not look like that. Neither the eastern nor western US population has yet to grow at greater than 3.5% annually, though they too were reduced far below their earlier numbers, and the present role of incidental catch and direct take is thought to be small.

Relation to the PVA

The Brownian motion PVA model does not incorporate density dependence. This deserves some scrutiny, since it is known mathematically that very long term persistence requires some degree of density dependence in otherwise random growth dynamics (Royama, 1977); elementary considerations of the effects of crowding and availability of limited resources give rise to population growth models with some degree of density dependence (Lotka, 1937); and some degree of density dependence has been demonstrated empirically in a variety of populations (Dennis and Taper, 1994).

The tactical *choice* to use a model without density dependence may be more or less forced as a practical expedient if the key parameters that quantitatively represent density dependence are not known, and cannot effectively be estimated from the available data, which is most often the case (Ginzburg, Ferson, and Akcakaya, 1990). But these reasons for the choice do not address the question of what bias or unrealism may be introduced by the choice, nor do they really explain what underlying assumptions are implied. Here we review these issues.

Evolutionary ecological history plays out on a time scale of the half-lives of species and the persistence times of large scale configurations of environments, generally on the order of tens of thousands of years, or more. From this perspective, the application of a PVA to project 100 years into the future is actually a *short* term prediction, however much it may strain our scientific predictive capabilities.

For this reason, some of the unreasonable long term features of the Brownian motion model may be reasonable enough in an actual PVA application. In the very long run, the Brownian motion model generates two main categories of trajectories: those that absorb at extinction, and those that go to unlimitedly large population sizes. In the shorter term, for a population that is experiencing problems, the predicted excursions below some threshold level may well be realistic, and the time horizon could be short enough to keep the predicted high excursions within reasonable limits; and if unreasonably high predicted excursions are encountered they should be interpreted simply as escape from the extinction vortex.

When a population comes to our attention as a subject for PVA, this is often because its numbers and/or dynamics do *not* look normal. PVA assessments are usually motivated by conservation concern for a population that has declined to very low levels compared to its evolutionary and ecological history, or because it is exhibiting dynamics that seem unusually volatile. These dynamics are often thought, or hypothesized, to be affected by recent habitat loss or habitat modification or ecosystem disruption that may have dramatically changed the operation of density dependence for the population; or fluctuating exogenous mortality forces may have overwhelmed the presumed stabilizing influence of density dependence.

In these circumstances, there is no strong a priori basis for assuming that density dependence will rescue a population from further decline, either as a continuing trend or as a random walk. The effective carrying capacity may itself have declined to zero or entered a phase where it fluctuates erratically and widely. In such a case, the population dynamics, at least for the time scale of the PVA projection, may behave pretty much as the Brownian motion model represents them. The following mathematical exploration will pay particular attention to the way various factors can greatly change carrying capacity.

Carrying Capacity

The term "carrying capacity" is roughly used to signify the number of animals a particular piece of real estate can support. If this is understood to be strongly context dependent, no harm is done by the rough definition. If, on the other hand, this encourages a belief that "carrying capacity" can be measured simply as an inventory of habitat, the rough definition will be severely misleading. The mathematically precise definition of "carrying capacity," which we will label *K* for purposes of this discussion, is the equilibrium population level, above which the population declines and below which the population grows. The purpose of the analysis developed here is to show how habitat quantity, resource availability and predation pressure interact to determine *K*.

MODEL

General Model for Population Growth

Population size, *N*, is governed by the balance between the inherent resource driven component of per capita growth rate, *r*, and exogenous mortality factors, some of which operate in a constant per capita fashion represented by *u*, and some of which are simply a constant absolute drain on the population represented by *w*,

$$\frac{dN}{dt} = rN - uN - w \qquad . \qquad \text{eq[B1]}$$

For example, a loss to predation that is proportional to the encounter rate would appear in *u*, where a predation loss that is the same regardless of population size would appear in *w*.

Let the inherent resource-driven component of the per capita growth rate be proportional to the surplus of usable resources above the constant per capita maintenance demand of the organism. Let the amount of habitat be *h*, the resource concentration per unit habitat be *s*, the threshold concentration for usability at which foraging intake exceeds foraging costs be *f*, and let the per capita resource demand for maintenance be *m*. Then,

$$r = c(h(s - f) - mN) \qquad , \qquad \text{eq[B2]}$$

where *c* is the proportionality constant converting free resources to growth.
Combining equations [B1] and [B2] gives

$$\begin{aligned}\frac{dN}{dt} &= c(h(s-f)-mN)N-uN-w\\ &= -(cm)N^2+(ch(s-f)-u)N-w\\ &= -\beta N^2+\alpha N-w \qquad , \qquad \text{eq[B3]}\end{aligned}$$

where the Greek letters

$$\alpha = ch(s-f)-u \qquad , \qquad \text{eq[B4]} \qquad \#$$

and

$$\beta = cm \qquad , \qquad \text{eq[B5]}$$

simply collect terms that will be phenomenologically revealing.

In the absence of the effects of w, the per capita instantaneous growth rate at vanishing population size (essentially the logarithm of what is usually called R_{max} would be α and the carrying capacity would be α/β. We note that u figures in α, so even without consideration of the constant drain w or the resource concentration threshold f we have an effect of an exogenous mortality on the expressed carrying capacity.

Graphical Diagnosis of the General Model

In units of net per capita growth, equation [B3] becomes

$$\frac{1}{N}\frac{dN}{dt} = -\beta N+\alpha-\frac{w}{N} \qquad , \qquad \text{eq[B6]} \qquad \#$$

from which we see that the net per capita growth rate, as a function of population size, is a difference between two functions: one is a straight line with slope $-\beta$ and intercept α

$$y = -\beta N+\alpha \qquad , \qquad \text{eq[B7]}$$

and the other is an equilateral hyperbola

$$y = \frac{w}{N} \qquad , \qquad \text{eq[B8]} \qquad \#$$

with vertex at

$$y = N = \sqrt{w} \qquad . \qquad \text{eq[B9]}$$

We may graph these two lines, as in Figure B1, with the geometric reference points labeled. Where the two lines intersect, their difference is zero, so these are points where the net growth rate is zero. They may be solved for directly as the two roots of the quadratic obtained from setting equation [B3] to zero.

The larger of the two roots is at an intersection above which the hyperbola is larger valued than the straight line, so this is a stable equilibrium, which therefore is the mathematical carrying capacity:

$$K = \frac{\alpha+\sqrt{\alpha^2-4\beta w}}{2\beta} \qquad . \qquad \text{eq[B10]} \qquad \#$$

The smaller of the two roots is at an intersection *below* which the hyperbola is larger valued than the straight line, so this is an unstable equilibrium, a repelling point, which therefore is the critical depensation level, Q:

$$Q = \frac{\alpha - \sqrt{\alpha^2 - 4\beta w}}{2\beta} \quad . \qquad \text{eq[B11]}$$

If the population is reduced below Q, it cannot recover under its own dynamics, and it will decline to extinction. Thus Q is the population level where the mathematical Allee effect becomes over-riding.

ANALYSIS

Properties of the Species and Properties of the Environment

From equations [B10], [B4], and [B5] we see that K actually is a function of a long list of parameters, one of which h is a direct inventory of available habitat, one of which s is a direct measure of productivity of the habitat, three of which c, f and m are properties of the species which scale the habitat measures to units of population growth, and two of which u and w are mortality factors which could change with changing conditions, so they are not strictly measures of either habitat or the species properties. As any of the parameters exogenous to the species–namely h, s, u or w–change, K will change. But simply observing a change in K will not reveal which factors were involved in the change. If we had direct information on the various factors, we might parse the exogenous from environmental influences on effective carrying capacity. In the absence of that level of detailed information, carrying capacity itself is phenomenological– it is simply the equilibrium population size, regardless of its components.

In practice, if K fluctuates, the actual changes in K generally will not be observed directly, unless the carrying capacity remains nearly constant long enough for the density dependent approach to carrying capacity to be resolvable from the population censuses. More usually, all we will observe, without very detailed research into mechanisms, are the transient changes in the growth rate of the population.

If we had enough information about the variation in K to construct a stochastic model for it, that would provide an intellectually satisfying basis for a PVA. Absent that information, we build a PVA around the observable variation in population growth rate.

Carrying Capacity as a Function of Conditions

From Figure B1 we see that the carrying capacity, as the right-most of the two intersection points, will vary in position as the two lines move. The straight line will move in response to changes in its slope and intercept, given by β and α, and the hyperbola will move outward from the origin as w increases. The two lines will pull apart with steepening slope of the straight line (larger β), downward translation of the straight line (smaller α), or outward translation of the hyperbola. As the two lines pull apart, the two intersection points draw closer together, so the critical depensation level and K draw closer together.

Just as the two lines separate, Q and K merge, the stable equilibrium point vanishes, and the effective carrying capacity collapses in a mathematical discontinuity. At that collapse, the population will decline to extinction unless conditions change to restore a positive carrying capacity. The approach to collapse of K comes without phenomenological warning, as it can occur at non-zero values of K, and it occurs as a discontinuous response to the driving factor.

From equation [B10], keeping the effective carrying capacity above collapse requires

$$hs > hf + \frac{u + 2\sqrt{mw}}{c} \quad . \qquad \text{eq[B12]}$$

The collapse can occur under conditions where available habitat and resource concentration are well above zero.

In the absence of an effect of w, the response of K to conditions will not exhibit a discontinuity, but K can still go to zero while habitat is available and resource concentration is above zero. For w equal zero, the condition for non-zero carrying capacity is

$$hs > hf + \frac{u}{c} \quad . \qquad \text{eq[B13]}$$

Numerical Illustration

The following is only for illustration. The parameter values are those used to generate Figure B1.

Parameter		**value**
Resource to growth conversion	$c =$	0.00002
Habitat area	$h =$	10,000
Resources per unit habitat area	$s =$	2.5
Foraging cost	$f =$	1.5
Per capita maintenance demand	$m =$	1
exogenous per capita mortality	$u =$	0.05
Fixed mortality drain	$w =$	1000
Results		**Value**
Effective carrying capacity	$K =$	67,604
Critical depensation level	$Q =$	7,396
Maximum growth rate, absent constant mortality drain	$R_{max}\|_{w=0}$	1.16
Intrinsic carrying capacity	$K\|_{u=0,w=0}$	100,000
Carrying capacity absent effect of exogenous per capita mortality	$K\|_{u=0}$	94,721
Carrying capacity absent effect of exogenous constant mortality drain	$K\|_{w=0}$	75,000
Critical level for collapse of *K*		**Value**
Habitat area	$h_{crit} =$	6,972
Resources per unit habitat area	$s_{crit} =$	2.20
exogenous per capita mortality	$u_{crit} =$	0.111
Fixed mortality drain	$w_{crit} =$	2,813

Table B1

It may be tempting to look for direct numeric parallels to the SSL example, but far too many of the parameters in the illustration are completely unknown for the SSL for this to be meaningful. The point simply is to illustrate how the elementary density dependence model with environmental influences can readily give rise to extremely volatile fluctuation in *K*, which, from the perspective of observations only on the population trajectory, could appear (and function) like random population growth without effective density dependence. This could be the case for the recent dynamics of the SSL.

A PVA MODEL FOR EVALUATING RECOVERY CRITERIA FOR THE WESTERN SSL POPULATION

Daniel Goodman
March 30, 2006

FIGURES

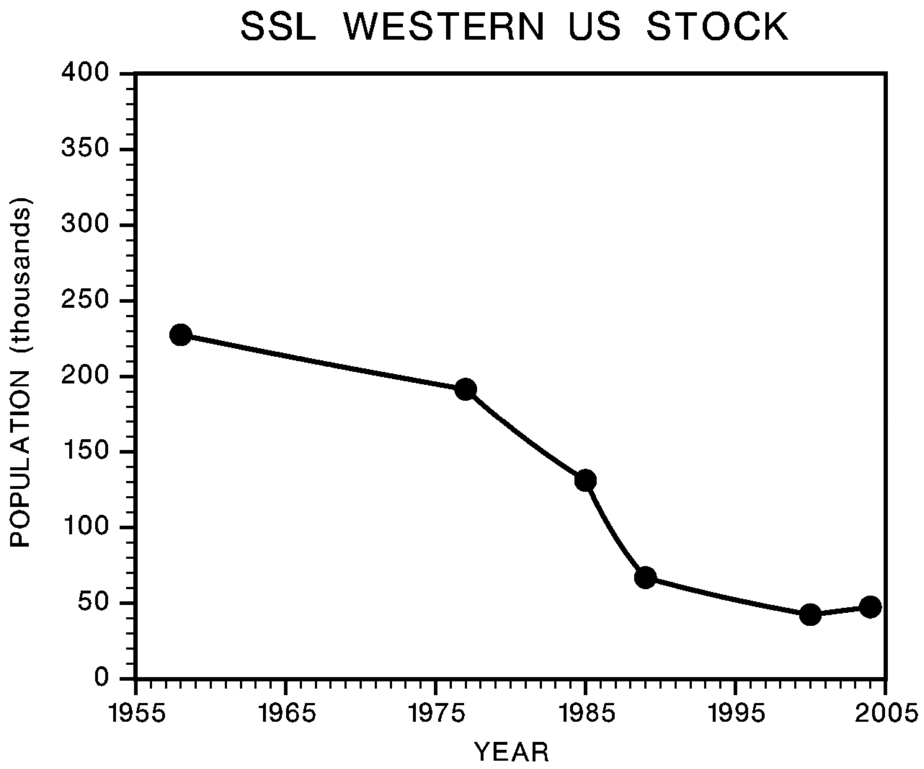
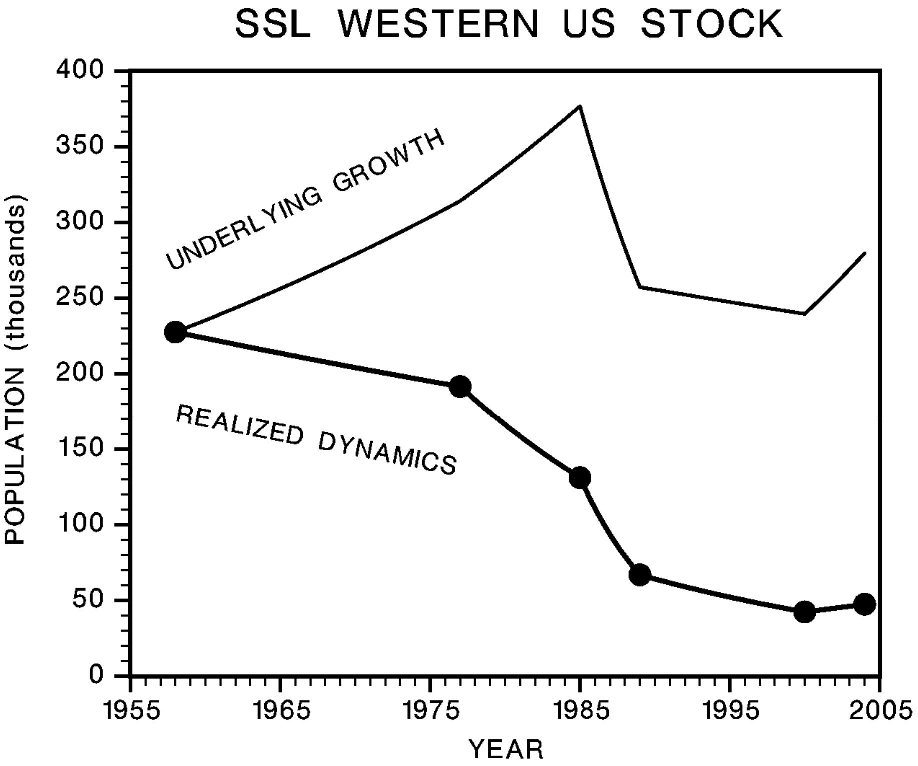


Figure 1



SSL WESTERN US STOCK
POPULATION (thousands)
400
350
300
250
200
150
100
50
0
1955
1965
1975
1985
1995
2005
YEAR
UNDERLYING GROWTH
REALIZED DYNAMICS

Figure 2

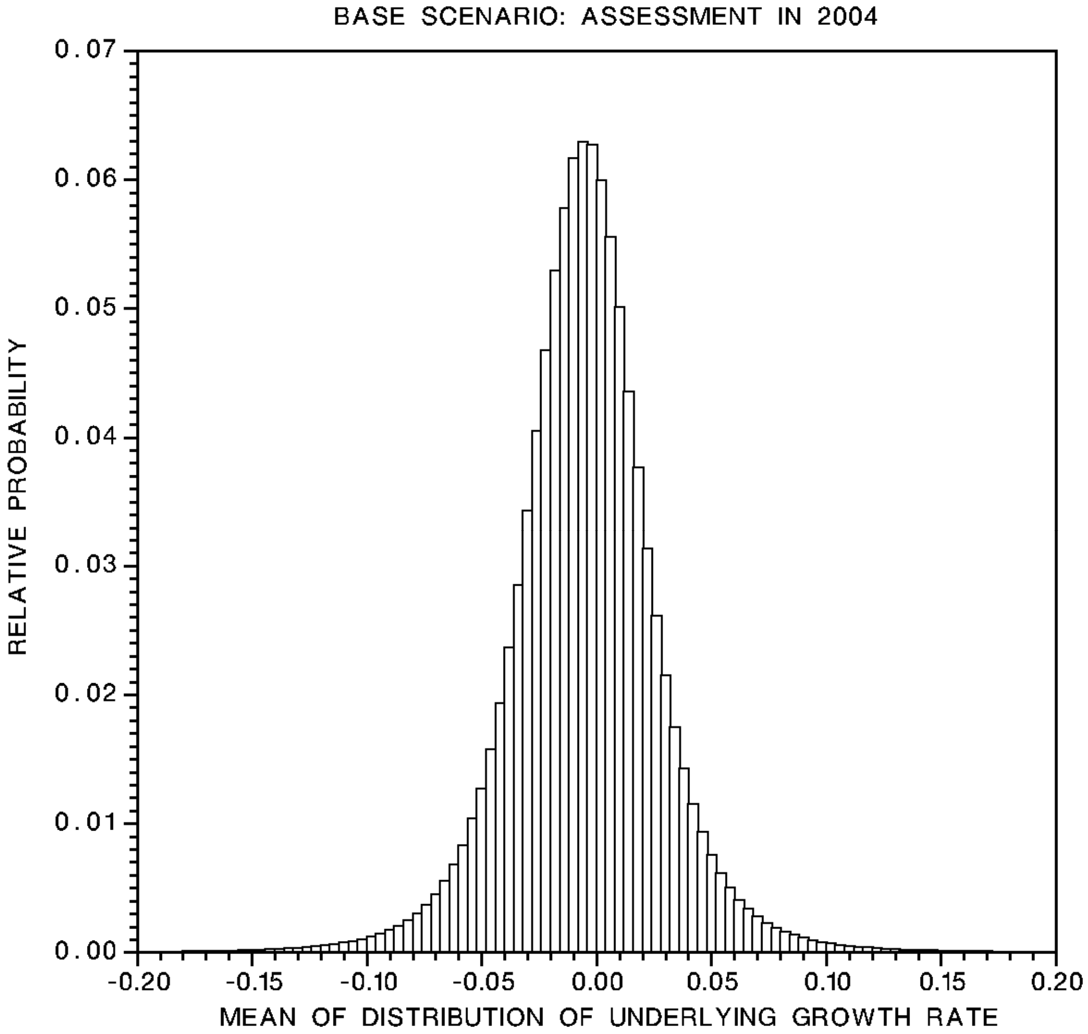

BASE SCENARIO: ASSESSMENT IN 2004
RELATIVE PROBABILITY
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0.00
-0.20
-0.15
-0.10
-0.05
0.00
0.05
0.10
0.15
0.20
MEAN OF DISTRIBUTION OF UNDERLYING GROWTH RATE

Figure 3

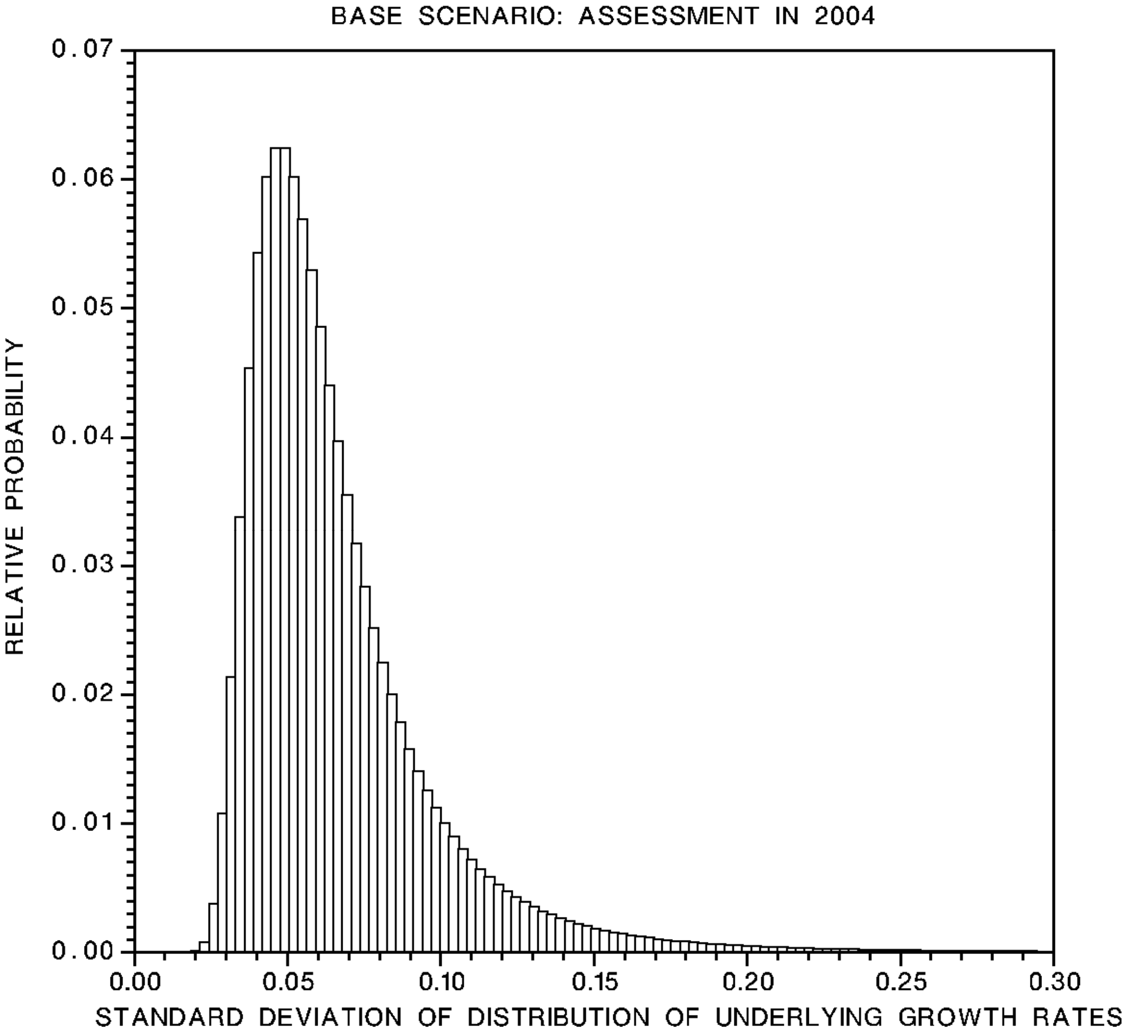

BASE SCENARIO: ASSESSMENT IN 2004
RELATIVE PROBABILITY
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0.00
0.00
0.05
0.10
0.15
0.20
0.25
0.30
STANDARD DEVIATION OF DISTRIBUTION OF UNDERLYING GROWTH RATES

Figure 4

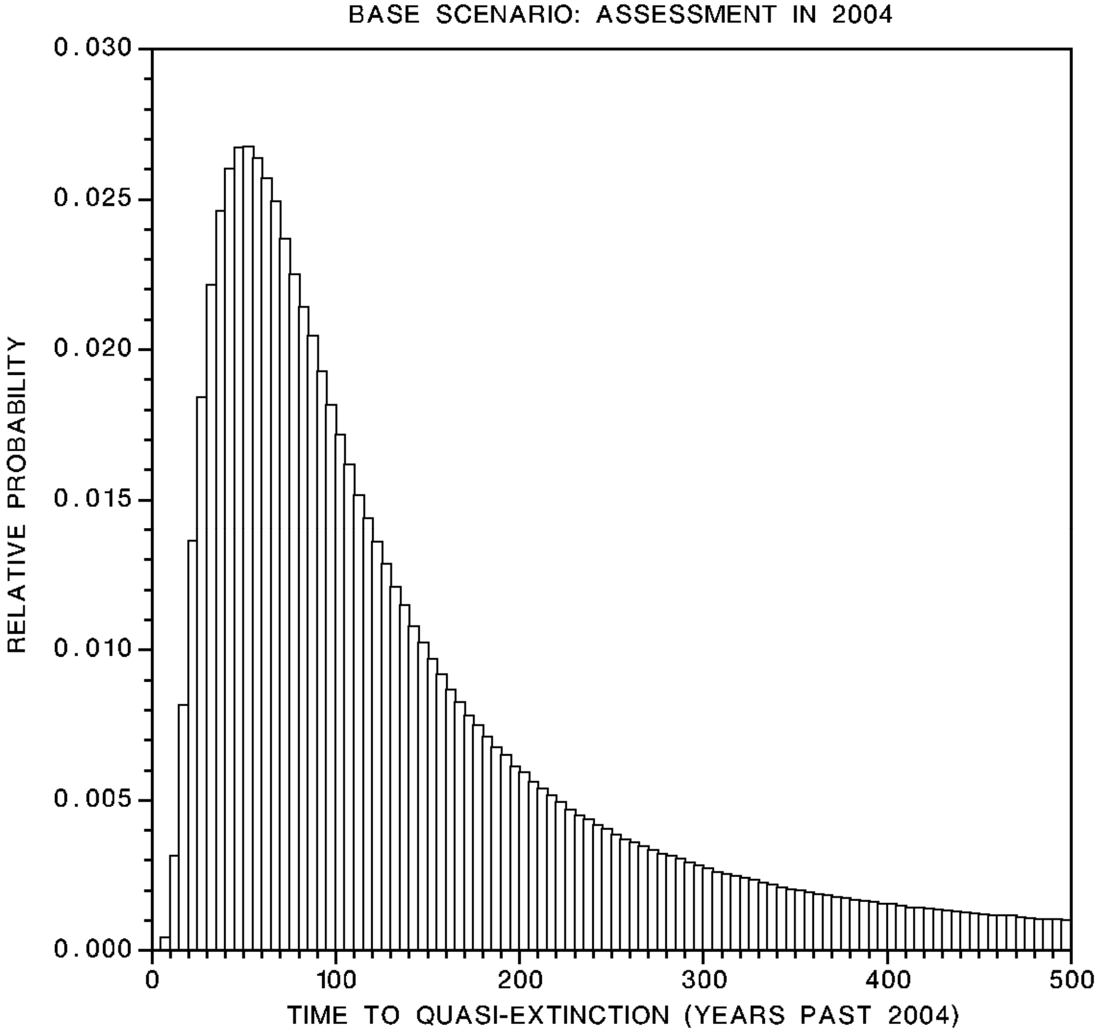

BASE SCENARIO: ASSESSMENT IN 2004
RELATIVE PROBABILITY
0.030
0.025
0.020
0.015
0.010
0.005
0.000
0
100
200
300
400
500
TIME TO QUASI-EXTINCTION (YEARS PAST 2004)

Figure 5

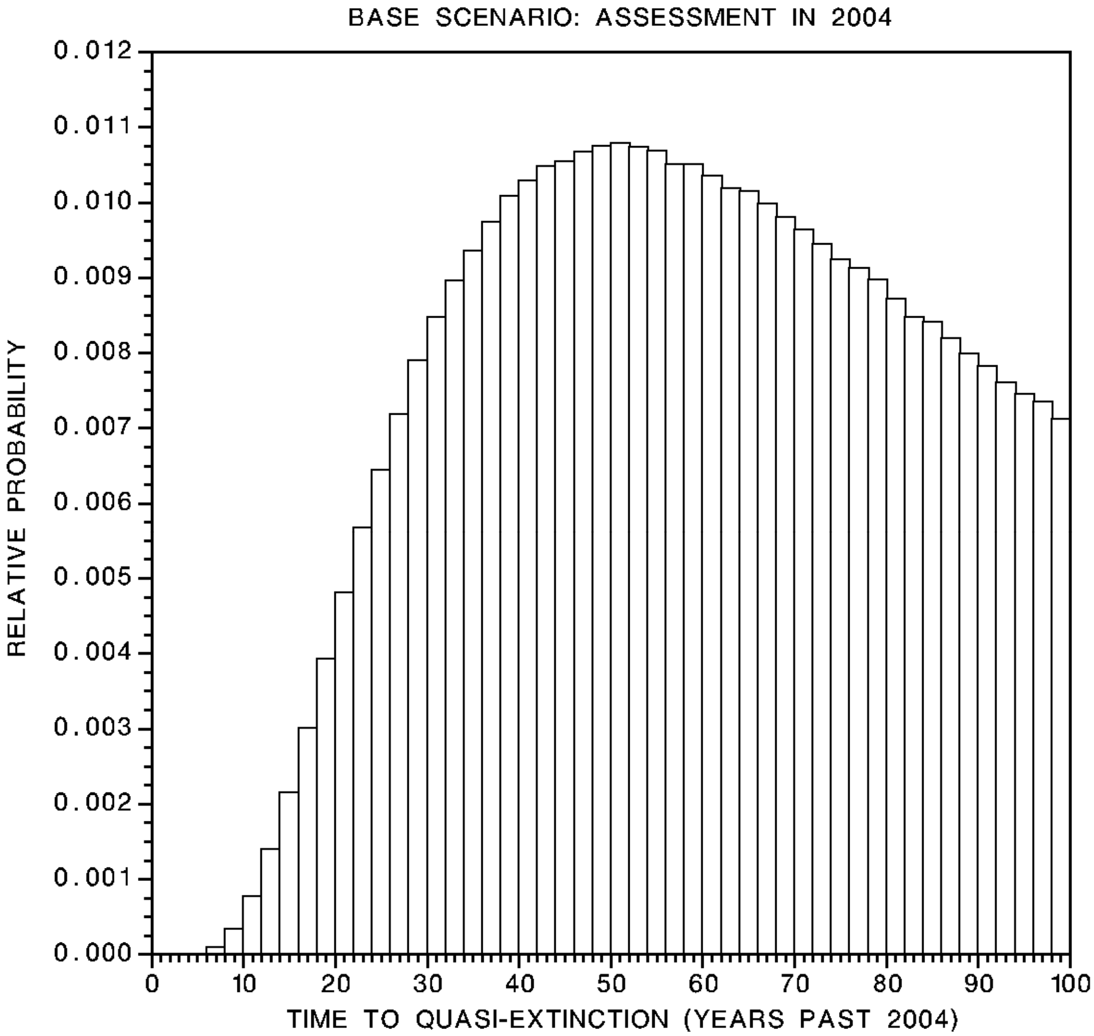

BASE SCENARIO: ASSESSMENT IN 2004
RELATIVE PROBABILITY
0.000
0.001
0.002
0.003
0.004
0.005
0.006
0.007
0.008
0.009
0.010
0.011
0.012
0
10
20
30
40
50
60
70
80
90
100
TIME TO QUASI-EXTINCTION (YEARS PAST 2004)

Figure 6

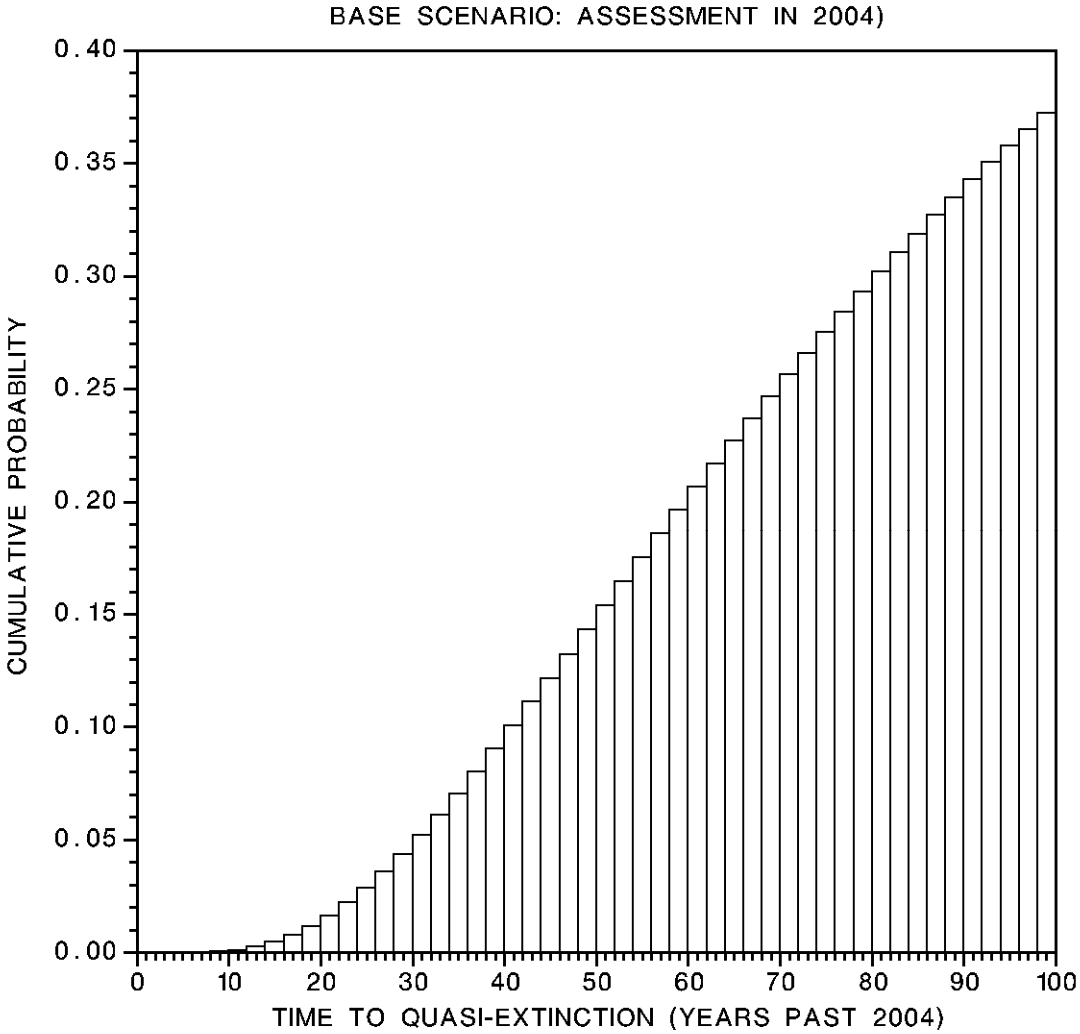

BASE SCENARIO: ASSESSMENT IN 2004)
CUMULATIVE PROBABILITY
0.40
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
0
10
20
30
40
50
60
70
80
90
100
TIME TO QUASI-EXTINCTION (YEARS PAST 2004)

Figure 7

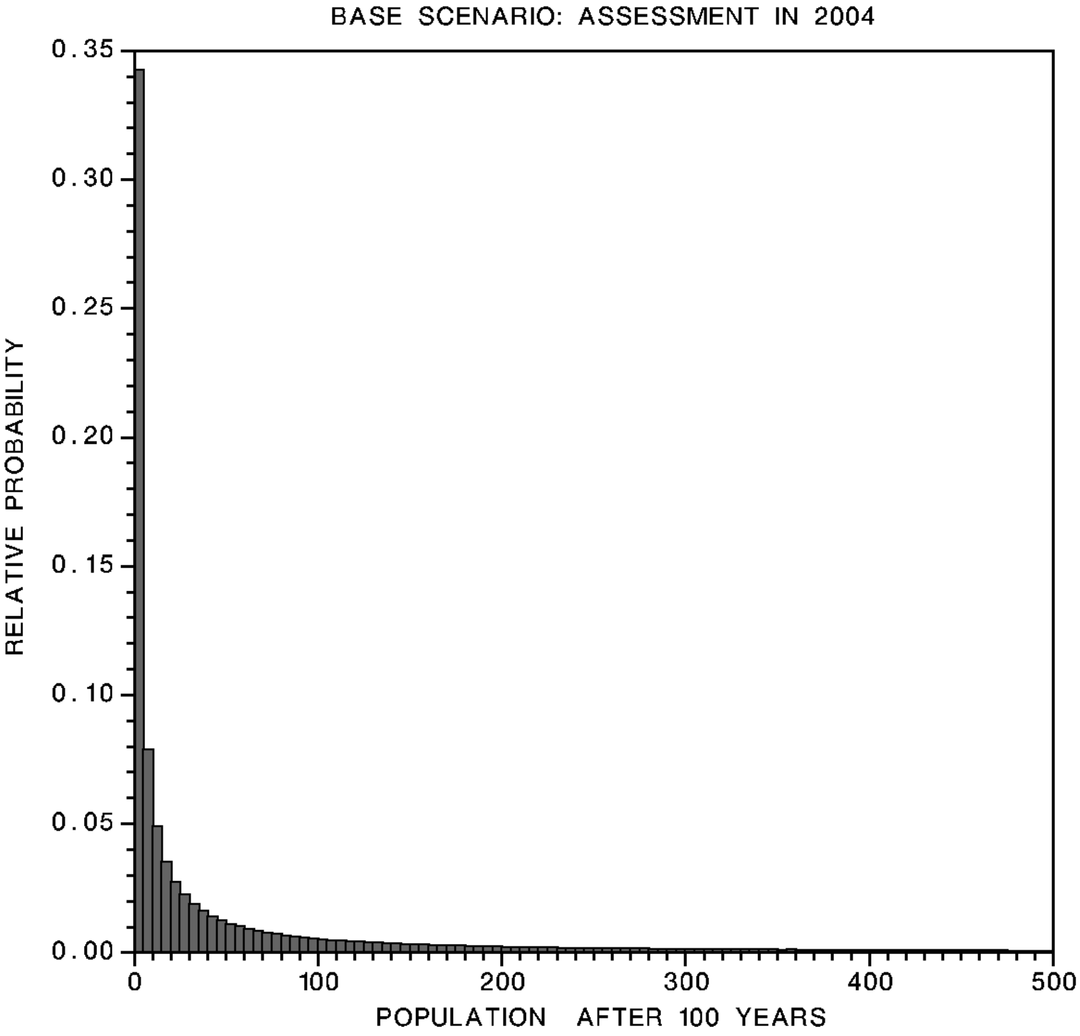

BASE SCENARIO: ASSESSMENT IN 2004
RELATIVE PROBABILITY
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
0
100
200
300
400
500
POPULATION AFTER 100 YEARS

Figure 8

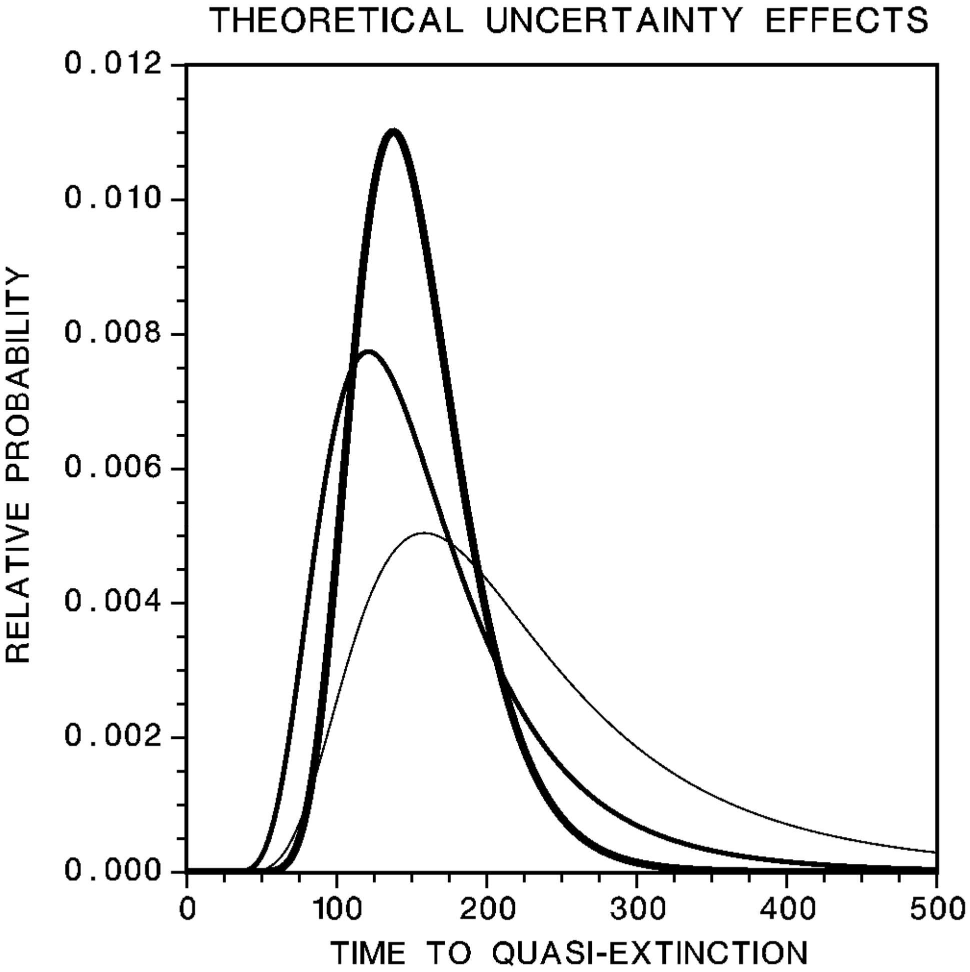

THEORETICAL UNCERTAINTY EFFECTS
RELATIVE PROBABILITY
0.012
0.010
0.008
0.006
0.004
0.002
0.000
0
100
200
300
400
500
TIME TO QUASI-EXTINCTION

Figure 9

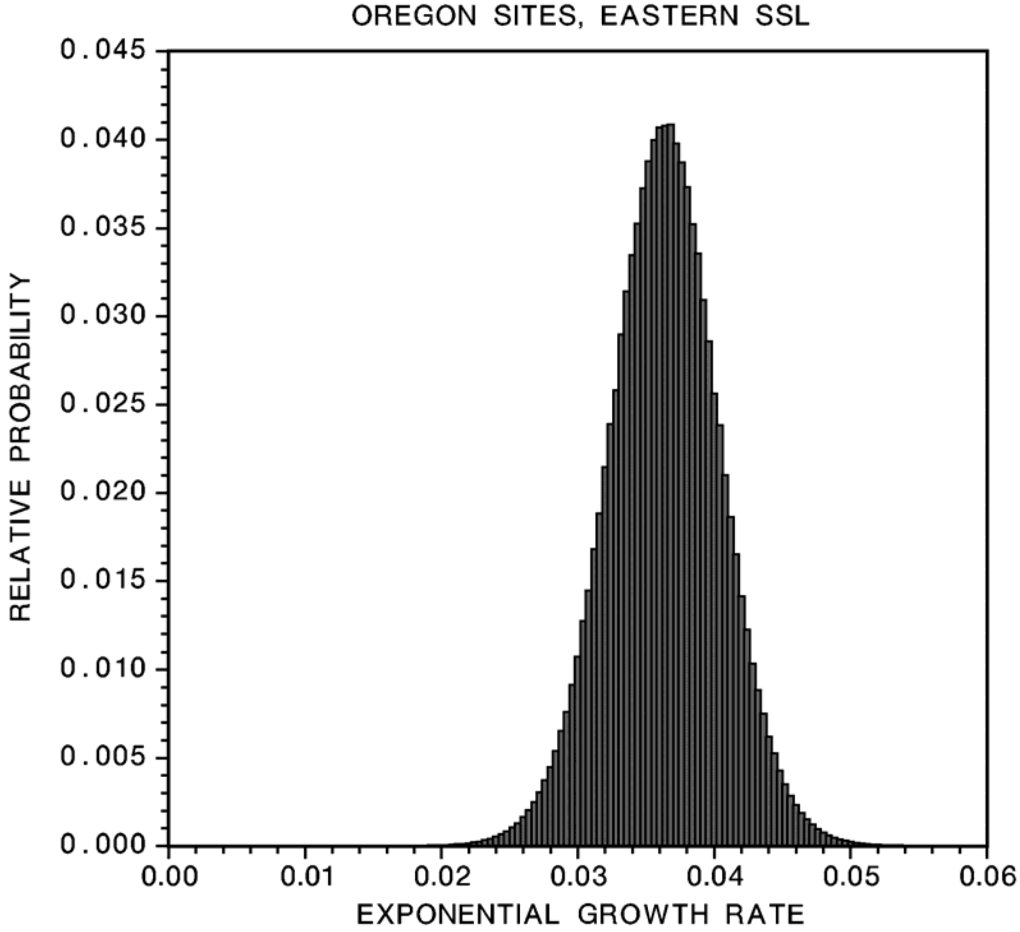

OREGON SITES, EASTERN SSL
RELATIVE PROBABILITY
0.000
0.005
0.010
0.015
0.020
0.025
0.030
0.035
0.040
0.045
0.00
0.01
0.02
0.03
0.04
0.05
0.06
EXPONENTIAL GROWTH RATE

Figure A1

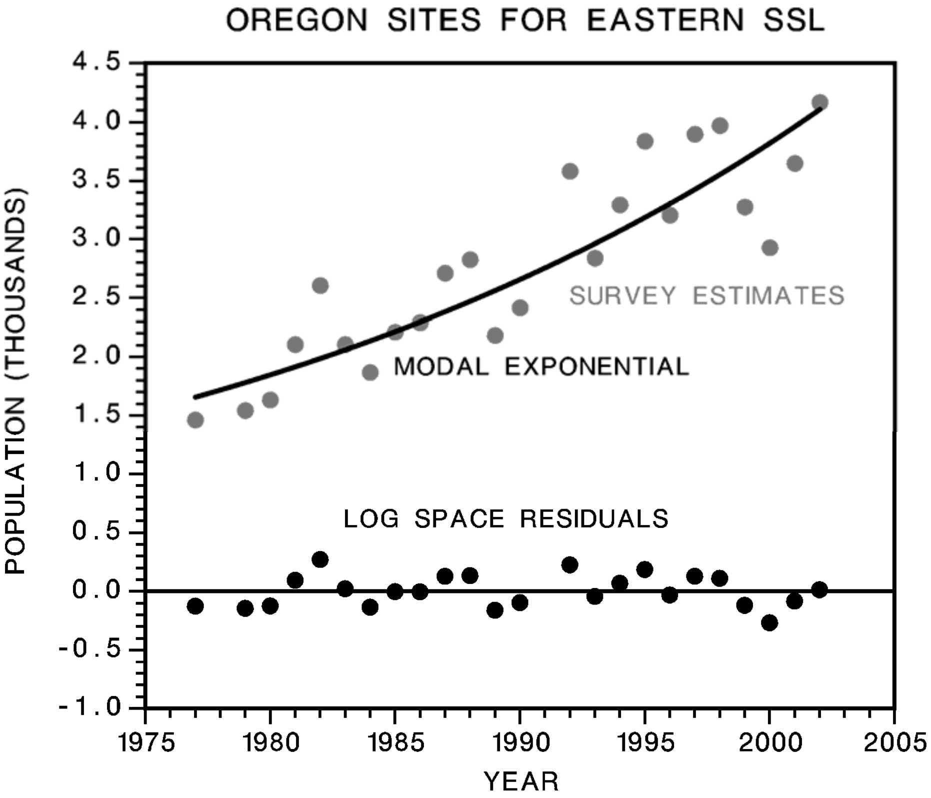

OREGON SITES FOR EASTERN SSL
POPULATION (THOUSANDS)
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
-0.5
-1.0
SURVEY ESTIMATES
MODAL EXPONENTIAL
LOG SPACE RESIDUALS
1975
1980
1985
1990
1995
2000
2005
YEAR

Figure A2

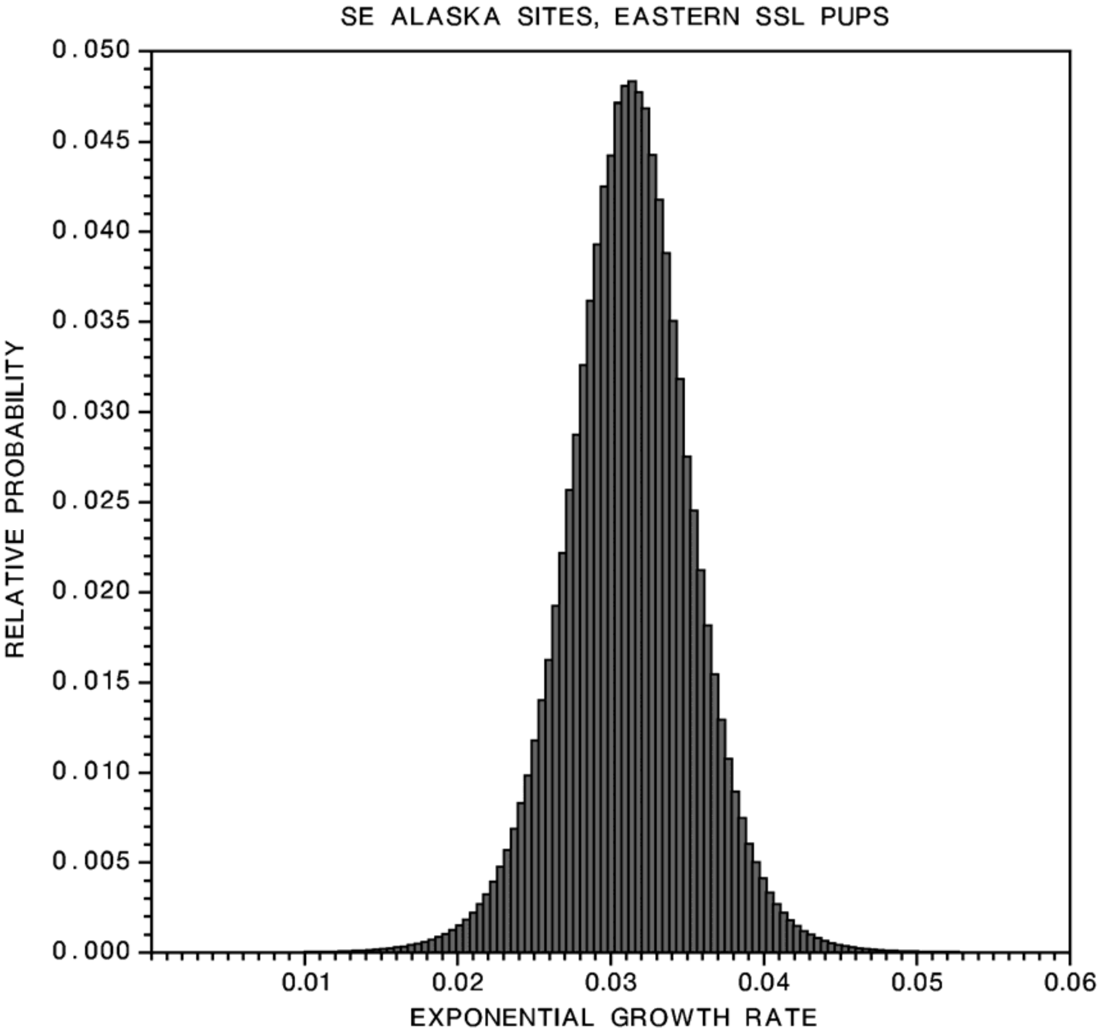

SE ALASKA SITES, EASTERN SSL PUPS
RELATIVE PROBABILITY
0.050
0.045
0.040
0.035
0.030
0.025
0.020
0.015
0.010
0.005
0.000
0.01
0.02
0.03
0.04
0.05
0.06
EXPONENTIAL GROWTH RATE

Figure A3

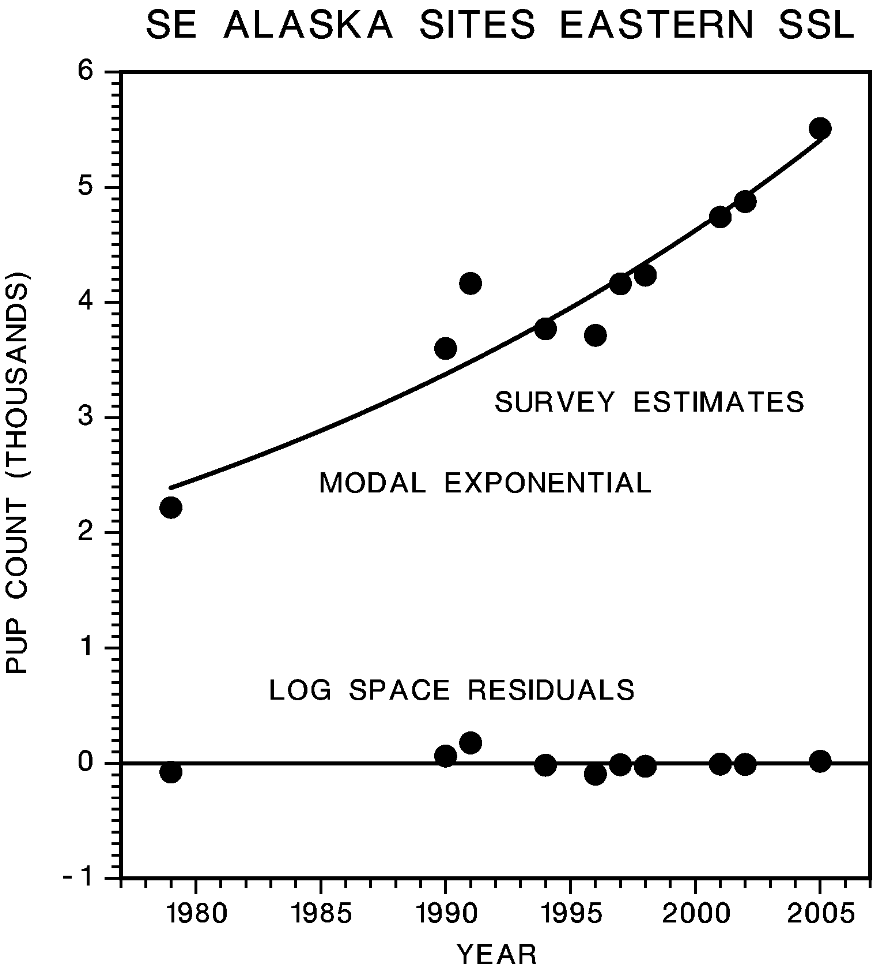

SE ALASKA SITES EASTERN SSL
PUP COUNT (THOUSANDS)
6
5
4
3
2
1
0
-1
SURVEY ESTIMATES
MODAL EXPONENTIAL
LOG SPACE RESIDUALS
1980
1985
1990
1995
2000
2005
YEAR

Figure A4

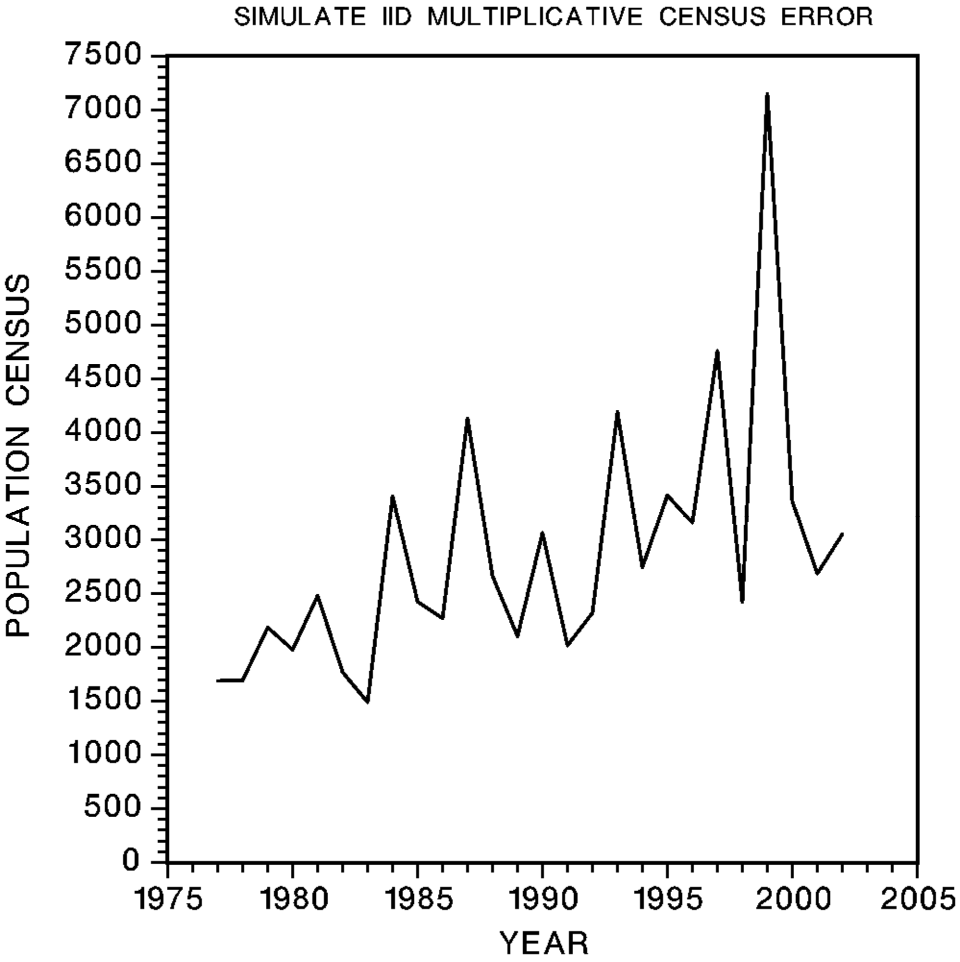

SIMULATE IID MULTIPLICATIVE CENSUS ERROR
POPULATION CENSUS
7500
7000
6500
6000
5500
5000
4500
4000
3500
3000
2500
2000
1500
1000
500
0
1975
1980
1985
1990
1995
2000
2005
YEAR

Figure A5

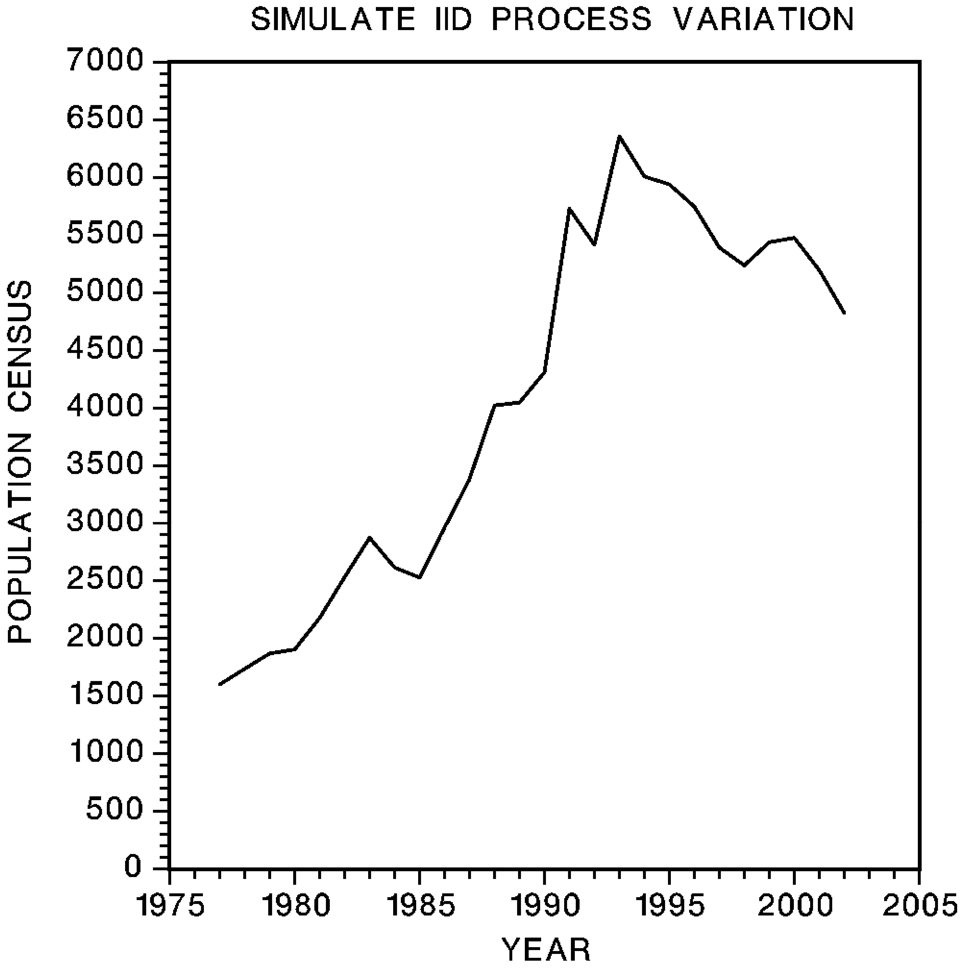

SIMULATE IID PROCESS VARIATION
POPULATION CENSUS
7000
6500
6000
5500
5000
4500
4000
3500
3000
2500
2000
1500
1000
500
0
1975
1980
1985
1990
1995
2000
2005
YEAR

Figure A6

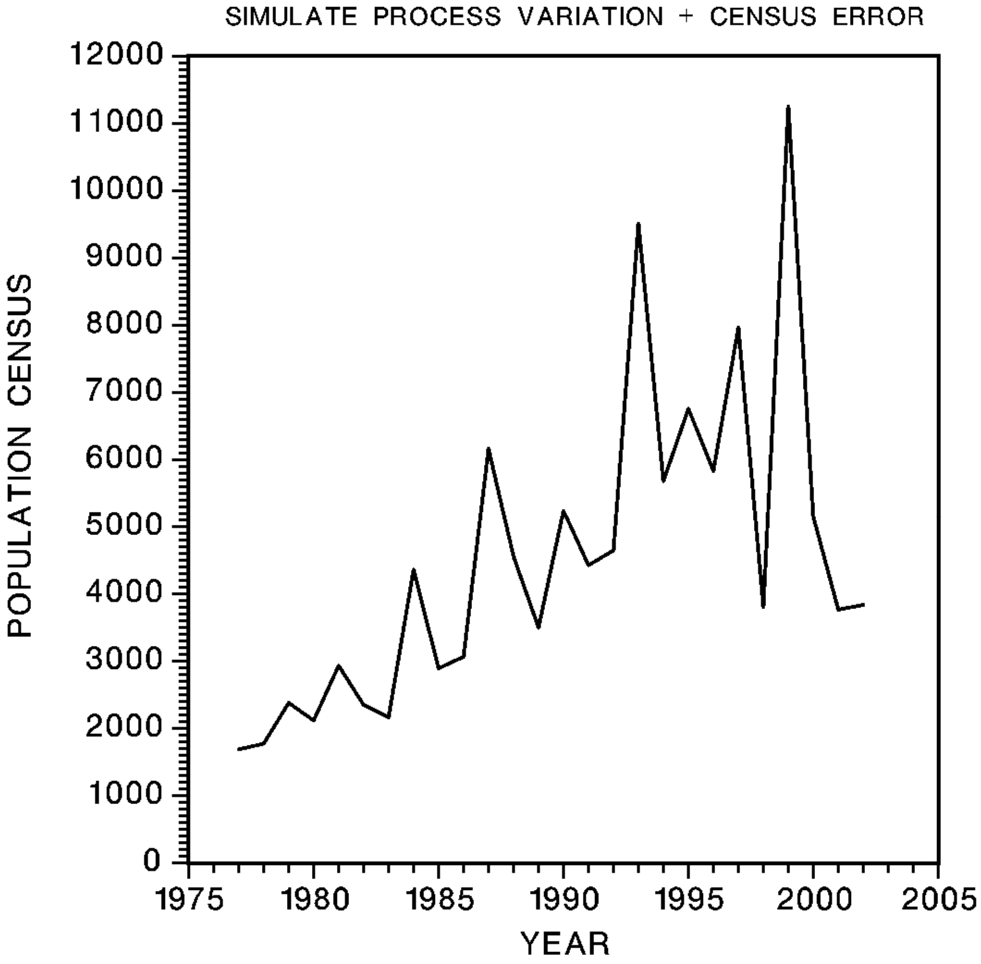

SIMULATE PROCESS VARIATION + CENSUS ERROR
POPULATION CENSUS
0
1000
2000
3000
4000
5000
6000
7000
8000
9000
10000
11000
12000
1975
1980
1985
1990
1995
2000
2005
YEAR

Figure A7

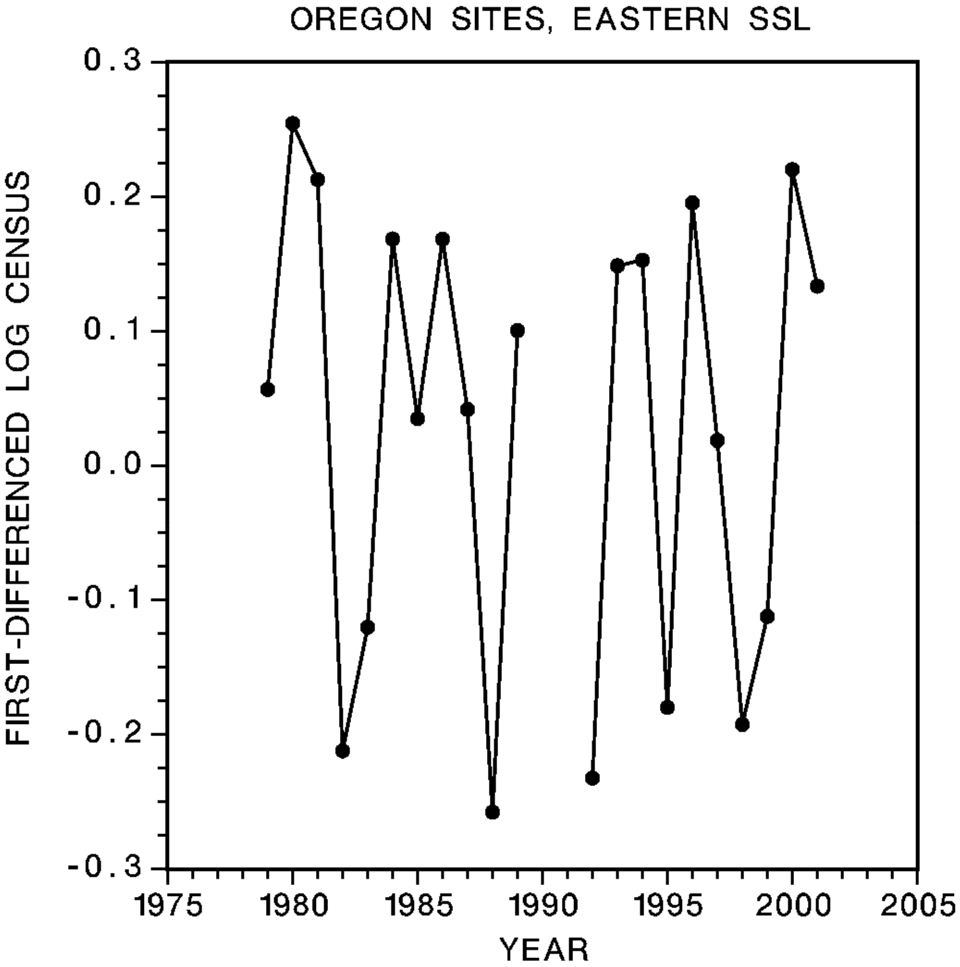
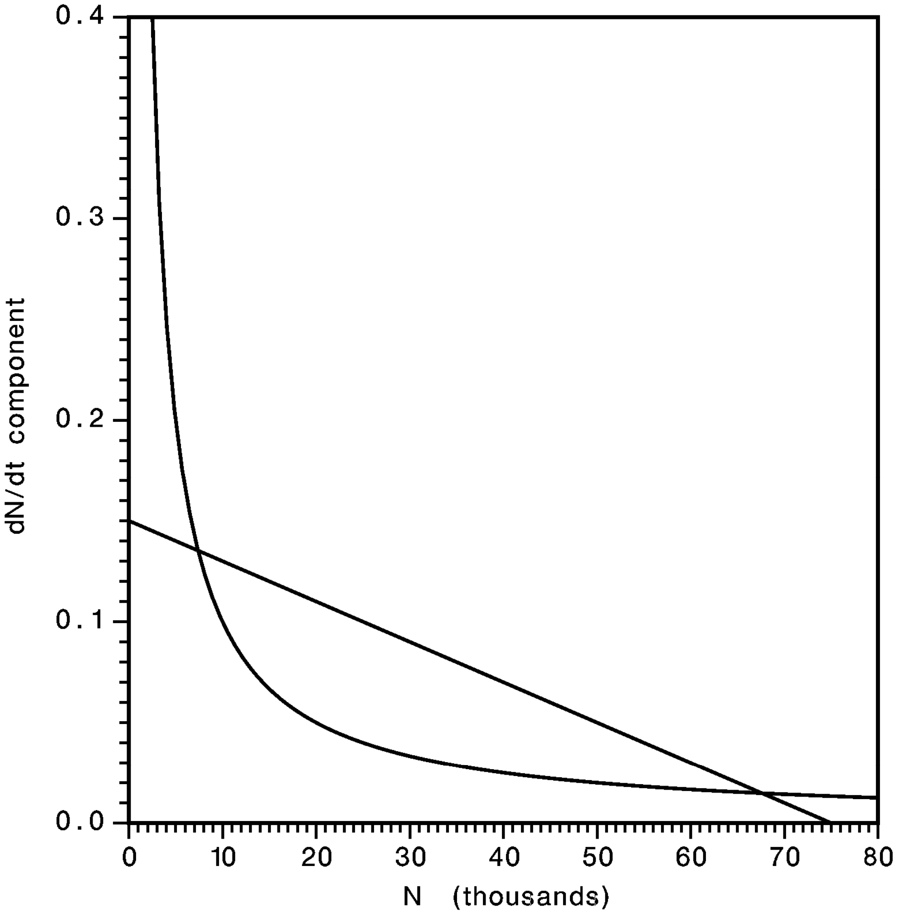

OREGON SITES, EASTERN SSL
FIRST-DIFFERENCED LOG CENSUS
0.3
0.2
0.1
0.0
-0.1
-0.2
-0.3
1975
1980
1985
1990
1995
2000
2005
YEAR

Figure A8



0.4
0.3
0.2
0.1
0.0
dN/dt component
0
10
20
30
40
50
60
70
80
N (thousands)

Figure 1B