Endangered Species Act — Section 7 Consultation
Biological Opinion

Activities Considered:	Authorization of groundfish fisheries under the Fishery Management Plan for Groundfish of the Bering Sea and Aleutian Islands Management Area Authorization of groundfish fisheries under the Fishery Management Plan for Groundfish of the Gulf of Alaska State of Alaska parallel groundfish fisheries
Agency:	National Marine Fisheries Service
Consultation by:	National Marine Fisheries Service Alaska Region
Date Issued:	11.24.10
Issued by:	James W. Balsiger, Ph.D. Administrator, Alaska Region

TABLE OF CONTENTS

APPENDICES

Appendix I	ADF&G Commercial Fisheries Emergency Order
Appendix II	Vessel Monitoring System Enabled Catch-In-Areas Database
Appendix III	Fisheries Catch Data Analysis for the Bering Sea, Aleutian Islands, and Gulf of Alaska
Appendix IV	Fisheries Catch Data in Critical Habitat- Figures and Tables
Appendix V	Differences in Expanded Observer and VMS-Catch-in Area catch estimates relative to NMFS' Official Catch Accounting estimates
Appendix VI	Russian Fisheries Catch Data
Appendix VII	The August 2, 2010 Draft Biological Opinion RPA and the Council's August 2010 Motion RPA

LIST OF FIGURES

LIST OF TABLES

ACRONYM LIST

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

AAC	Alaska Administrative Code
ABC	acceptable biological catch
ACIA	Arctic Climate Impact Assessment
ACL	annual catch limits
ACT	annual catch targets
ADEC	Alaska Department of Environmental Conservation
ADF&G	Alaska Department of Fish and Game
AFA	American Fisheries Act
AFSC	Alaska Fisheries Science Center
AKR	Alaska Regional Office
AM	accountability measure
ANCSA	Alaska Native Claims Settlement Act
ANHSC	Alaska Native Harbor Seal Commission
ANO	Alaska Native Organization
AO	Arctic Oscillation
AP	Advisory Panel
ASP	annual surplus production
BA	Biological Assessment
BP	Before Present
BC	British Columbia
BLM	Bureau of Land Management
B_{MSY}	biomass at maximum sustainable yield
BSAI	Bering Sea and Aleutian Islands
BUN	blood urea nitrogen
cADL	calculated aerobic dive limit
CAS	catch accounting system
CDQ	Community Development Quota
CFR	Code of Federal Regulations
CH/CVOA	Critical Habitat/Catcher Vessel Operational Area
C.I.	confidence interval
CIB	Cook Inlet Beluga
cm	centimeter
CO_2	carbon dioxide
Council	North Pacific Fishery Management Council
C/P	catcher/processor
CPUE	catch per unit effort
CSS	community size spectrum
CV	coefficient of variation
CVOA	catcher vessel operational area
dB	decibel
DFA	directed fishing allowance
DPS	distinct population segment
ECO	Ecosystem Office
ECOSIM	ecosystem dynamics model
EBS	Eastern Bering Sea

EEZ	Exclusive Economic Zone
EFH	essential fish habitat
EFP	exempted fishing permit
EIS	Environmental Impact Statement
EIT	echo integration-trawl
ENSO	El Nino Southern Oscillation
EOD	Expanded Observer Dataset
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act of 1973
F_{MSY}	fishing mortality rate associated with MSY
FO	frequency of occurrence
FFP	federal fisheries permit
FIT	fishery interaction team
FMP	Fishery Management Plan
FPP	federal processor permits
FR	Federal Register
FOCI	Fisheries Oceanography Coordinated Investigations
ft	feet
GHL	Guideline Harvest Level
GLOBEC	Global Ocean Ecosystem
GOA	Gulf of Alaska
GRS	groundfish retention standard
Guidelines	The National Standard Guidelines
Hg	mercury
HLA	harvest limit area
Hz	hertz
IAP	Interacademy Panel
ICA	intercooperative agreement
IFQ	individual fishing quota
IPA	incentive program agreement
IPCC	Intergovernmental Panel on Climate Change
IPHC	International Pacific Halibut Commission
IR/IU	improved retention/improved utilization
ITAC	initial total allowable catch
ITQ	Individual Transferable Quota
ITS	incidental take statement
IWC	International Whaling Commission
kg	kilogram
kHz	kiloHertz
kJ/g	kiloJoules per gram
km	kilometer
LAPP	limited access privilege program
LLP	License Limitation Program
LNG	liquefied natural gas
LOA	length overall
m	meter
MARPOL	International Convention for the Prevention of Pollution from Ships
MCA	Marine Conservation Alliance
MFMT	maximum fishing mortality threshold
mm	millimeter
MMPA	Marine Mammal Protection Act

MMS	Minerals Management Service
mmt	million metric ton
MRA	maximum retainable amounts
MSA	Magnuson-Stevens Fishery Conservation and Management Act
MSE	management strategy evaluation
MSFOR	multispecies model
MSRA	Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006
MSST	minimum stock size threshold
MSY	maximum sustainable yield
mt	metric tons
mtDNA	mitochondrial DNA
NAS	National Academy of Sciences
NGOS	North Gulf Oceanic Society
NEPA	National Environmental Policy Act
nm	nautical mile
NMFS	National Marine Fisheries Service
NMFS GOP	North Pacific Groundfish Observer Program Database
NMML	National Marine Mammal Laboratory
NOAA	National Oceanic and Atmospheric Administration
NPFMC	North Pacific Fishery Management Council
NPRB	North Pacific Research Board
NRC	National Research Council
NS1	National Standard 1
OC	organochlorine compound
OFL	overfishing level
OSP	optimum sustainable population
OY	optimum yield
PAH	polycyclic aromatic hydrocarbon
PBDE	polybrominated diphenyl ether
PBR	Potential Biological Removal
PCB	polychlorinated biphenyl
PDO	Pacific Decadal Oscillation
PDV	phocine distemper virus
pdf	probability density function
POP	Platform of Opportunity
ppmv	parts per million by volume
PRD	Protected Resources Division
PSC	prohibited species catch
PSEIS	Programmatic Supplemental Environmental Impact Statement
PSP	paralytic shellfish poisoning
PVA	Population Viability Analysis
RACE	Resource Assessment and Conservation Engineering
RCA	Rookery Cluster Area
RCRA	Resource Conservation and Recovery Act
REFM	Resource Ecology and Fisheries Management
RFRPA	revised final reasonable and prudent alternative
RIR	Regulatory Impact Review
RPA	Reasonable and Prudent Alternative
RPM	Reasonable and Prudent Measure
SAFE	Stock Assessment and Fishery Evaluation
SAR	Stock Assessment Report

SAT	surface air temperature
SCA	Steller Sea Lion Conservation Area
SCUBA	self-contained underwater breathing apparatus
SDR	satellite depth recorder
SEASWAP	Southeast Alaska Sperm Whale Avoidance Project
SEO	Southeast Outside
SER	spawning exploitation rate
SFD	Sustainable Fisheries Division
SL	standard length
SLP	sea level pressure
SLTDR	satellite-linked time-depth recorder
SPLASH	Structure of Populations, Levels of Abundance and Status of Humpbacks
SPR	spawning per recruit
SSB	spawning stock biomass
SSC	Scientific and Statistical Committee
SSLCA	Steller sea lion conservation area
SSLCZ	Steller sea lion foraging zone
SST	sea surface temperature
t	ton
TAC	Total allowable catch
TAD	time at depth
TAR	Third Assessment Report
TL	total length
TMDL	Total Maximum Daily Load
U.S.	United States
USC	United States Code
USCG	United States Coast Guard
USFWS	United States Fish and Wildlife Service
VMS	vessel monitoring system
VOE-CIA	VMS-Observer Enabled Catch-In-Areas
VRHS	voluntary rolling “hot spot”
WA/OR/CA	Washington/Oregon/California
WYK	West Yakutat
YOY	young-of-the-year

EXECUTIVE SUMMARY

This Biological Opinion, prepared in compliance with section 7 of the Federal Endangered Species Act (ESA), documents the consultation on the effects of the authorization of groundfish fisheries in the Bering Sea and Aleutian Islands region (BSAI) under the Fishery Management Plan (FMP) for Groundfish of the Bering Sea and Aleutian Islands Management Area, and on the authorization of groundfish fisheries in the Gulf of Alaska (GOA) under the FMP for Groundfish of the Gulf of Alaska, including the prosecution of parallel groundfish fisheries in Alaska state waters. This Biological Opinion is comprehensive in scope and considers the fisheries and the overall management framework established by the respective FMPs to determine whether that framework contains necessary measures to ensure the protection of listed species and critical habitat.

On August 2, 2010, NMFS released a draft of this Biological Opinion and accepted public comment on the draft analysis through September 3, 2010. The North Pacific Fishery Management Council (Council) and its Scientific and Statistical Committee and Advisory Panel convened a special meeting in August 2010 to review the draft Biological Opinion and the Reasonable Prudent Alternatives contained therein. During the comment period, NMFS received over 10,000 comments including many extensive scientific reviews of the document and the scientific underpinnings of its conclusions. NMFS also requested an internal Agency review of the scientific information contained in the Biological Opinion by NMFS scientists familiar with Steller sea lions, the North Pacific Ocean ecosystem, and the commercial fisheries prosecuted in the region. NMFS considered these comments and the review and made several revisions to the Biological Opinion as a result of comments and review received on the draft.

Background and Consultation History

In a letter to NMFS, on October 18, 2005, the Council recommended that NMFS commence the process to reinitiate a FMP-level formal Section 7 consultation on the effects of the Federal groundfish fisheries on ESA-listed species under U.S. Department of Commerce jurisdiction. On April 19, 2006, the Protected Resources Division of NMFS/Alaska Region (PRD) received a written request from the Alaska Region’s Sustainable Fisheries Division (SFD) to re-initiate section 7 consultation on the Alaska Groundfish FMPs to evaluate the effects of current federal fisheries management on listed species in light of information gained and management actions taken since previous consultations. That request was accompanied by a biological assessment that reviewed the likely effects of the proposed action on all twelve of the listed marine species found in waters off Alaska. The biological assessment concluded that the Steller sea lion (both the western and the eastern DPS), the North Pacific humpback whale, the North Pacific sperm whale, and the Northeast Pacific fin whale stock were likely to be adversely affected by the proposed action.

On June 21, 2006, PRD concurred with the SFD’s request and with the findings of the biological assessment, and formally initiated section 7 consultation on these three species and their designated critical habitat.

Action Area

The action area under consideration is the Exclusive Economic Zone (EEZ) of the BSAI and GOA, as well as state waters where the parallel fisheries occur. This action area covers all areas directly affected by fishing and indirectly affected by the removal of fish from nearby waters. The action area also encompasses all designated critical habitat for Steller sea lions in waters off Alaska.

Proposed Action

NMFS SFD, under the authority of the Magnuson-Stevens Fishery Conservation and Management Act (MSA), and the State of Alaska propose to: (1) authorize groundfish fisheries in the BSAI under the FMP for Groundfish of the BSAI; (2) authorize groundfish fisheries in the GOA under the FMP for Groundfish of the GOA; (3) authorize parallel fisheries within State-managed waters. Management of groundfish fisheries within the EEZ off Alaska and parallel fisheries in adjacent State waters is a continuing activity. The action being evaluated is an ongoing activity. Thus, NMFS has evaluated the potential effects of this action were it to continue with its current management policy into the foreseeable future.

NMFS first deconstructed this complicated action into its component parts. This deconstruction yields four main groups of interrelated activities which comprise the proposed action:

- fisheries management policy;
- exploitation strategy;
- annual fisheries assessment; and
- implementation of the fisheries.

This break-down in the description of the action permits examination of how parts of this action inter-relate. It facilitates an evaluation (in later chapters) of the effects of various aspects of this complicated action on listed species and designated critical habitat. For example, while policy on its own does not result in the removal of fish, it does set up limitations and expectations for removals. It sets in place a structure and process that affects how decisions about allowable levels and patterns of fishery removals are made in the face of differing sources and levels of information and uncertainty. Fishery management policy also affects fishery assessment practices and methodologies, as well as the treatment and reporting of components of assessments, once they are completed. Thus, while at first glance, the pattern and level of fishery removals occurring in different groundfish fisheries may seem independent, they are affected by, and highly inter-related with, policy choices, decisions about exploitation strategies, and stock and stock complex assessments that were made or accomplished earlier.

Recovery Plan for Steller Sea Lions

NMFS completed the first recovery plan for Steller sea lions in December 1992. The recovery plan became obsolete after the reclassification of Steller sea lions into two distinct population segments in 1997. Because of the continued decline in the western DPS, and nearly all of the recovery actions contained in the first plan had been completed, NMFS assembled a new Steller Sea Lion Recovery Team (Recovery Team) in 2001 to assist NMFS in revising the Recovery Plan to promote the conservation of the Steller sea lion. In March 2006, the Recovery Team submitted a draft of the Recovery Plan to NMFS, at which time it became an agency document. The Recovery Plan then went through peer and public review and NMFS completed the Revised Recovery Plan on February 29, 2008. The Revised Recovery Plan is a guidance document for the Agency as it continues to manage Steller sea lions and their habitat throughout their range. It contains recovery criteria which are described in several chapters of this Biological Opinion, and these are stated in performance measures by sub-region (sub-regions from west to east are: Russia/Asia; western, central, and eastern Aleutian Islands; western, central, and eastern Gulf of Alaska). The Recovery Team believed, and NMFS concurred, it was important to consider sub-population vital rates and demographic characteristics when considering the status of recovery of the western DPS. The Recovery Plan notes that 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 and would indicate a lack of recovery for the DPS as a whole. Thus, NMFS believes it was important to maintain viable sub-populations within the western DPS and not rely solely on the core of the range to provide for increasing population numbers over the short term.

For recovery, NMFS expects to see that both survival and pup production (natality) have increased to the point that the population is not only able to sustain itself (survival), but is able to grow at a modest rate (recovery) as required by the ESA. 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 in the recovery criteria in the Recovery Plan was influenced not only by the need to be confident that survival and natality had 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 Steller sea lion population can increase in all regimes, but it does lend further evidence that this population is robust enough to meet the requirements for downlisting or delisting.

Steller Sea Lion: Western Distinct Population Segment

The endangered western DPS of Steller sea lion has declined by almost 90% throughout its range, reaching its smallest size in 2000. Prior to the 1990s, the primary causes of the decline may have been commercial harvests of Steller sea lions, entanglement of juvenile Steller sea lions in commercial fishing gear, and intentional shooting by fishermen. However, since 1991 these effects have been nearly eliminated, yet the overall population in the western portion of the species' range is not increasing in abundance at a rate similar to that observed for the eastern portion of the species' range.

At present in the scientific community, there is no clear leading hypothesis to explain the decline of the western DPS of Steller sea lions in the 1990s or its apparent overall stability since 2000. Commercial fishing, nutritional stress, predation, and natural environmental changes are all considered to be factors in the decline and lack of recovery. The age groups originally thought most likely to be affected during the steep declines of the 1980s were juveniles and, to a lesser extent, adults; data collected in the last decade have put new emphasis on the potential risks to adult females.

There is general scientific agreement that the decline of the western DPS of Steller sea lion in the 1980s resulted primarily from declines in the survival of juvenile Steller sea lions. Evidence for this comes primarily from the central GOA, but trends in population counts along the range are consistent with this general conclusion. At the same time, however, smaller reductions in adult survival also occurred, and a three-decade long decline in natality in the central GOA began. There is less scientific agreement as to whether these changes had a nutritional cause, and whether fishery-induced changes in the forage base of Steller sea lions contributed to and continue to contribute to dietary or nutritional deficits. Groundfish fisheries managed under the two FMPs may adversely affect Steller sea lions by (a) competing for Steller sea lion prey and (b) affecting the structure of the fish community in ways that reduce the availability of alternative prey.

Under normal circumstances, the life history of Steller sea lions would protect them from short-term declines in the reproductive success of adult females or the survival of juvenile Steller sea lions. Steller sea lions are long-lived species with overlapping generations, a life-history strategy that protects them from short-term, environmental fluctuations. Their life history strategy would protect sea lion populations from variable survival and mortality rates caused by short-term climatic phenomena like El Niño Southern Oscillation. However, this life-history strategy may not protect Steller sea lions from changes in birth rates and juvenile survival that continue for two or three decades.

Recent data (2000-2008) indicate that the trend in population abundance for the western DPS as a whole has stabilized. Trends in population abundance are increasing in some sub-regions, however, trends in three sub-regions in the western DPS continue to decline. Improvements in juvenile survival rates are coincidental to management actions taken in the 1990s to reduce direct mortality factors and improvements in prey access resulting from fishery management measures implemented in the early 2000s. The data required to demonstrate cause and effect are not available.

Evidence provided in the Environmental Baseline indicates that current levels of mortality due to disease, contaminants, and direct human effects including direct fisheries effects (e.g., mortality due to catch in gear, shooting by fishermen, collisions with vessels), are unlikely to affect the ability of the western or eastern DPSs to survive or to recover, although their impact on survival and recovery cannot be ruled out.

Response of the Western DPS of Steller Sea Lion

Our review of the condition of the western DPS of Steller sea lion within the action area found that the overall western DPS decline was likely due to the cumulative effect of multiple factors, and that the marked change in the rate of the decline since 1990 suggests that the factors that contributed to the more rapid declines may not be the most significant stressors now operating. Direct sources of mortality likely were significant contributors to the Steller sea lion population declines observed prior to the 1990s, although this conclusion cannot be verified with available data. Since 1990, rates of mortality from these sources such as harvests, shooting, entanglement, and incidental catch have been substantially reduced and likely have contributed to a rebound in both juvenile and adult survival rates. Subsistence harvests of Steller sea lions continue but have declined substantially and are unlikely to be a substantial factor.

Killer whale predation also has the potential to be a significant top-down source of mortality. Available evidence from survival rates, isotope analyses, field observations of predation rates, and observations of prey types indicate that predation by killer whales is within the expected natural mortality level for Steller sea lions, although in some areas (e.g., central Aleutians), effects of killer whale predation could be amplified. However, the data to evaluate this hypothesis are unavailable.

Evidence that indirect or bottom-up factors may have contributed to the decline observed from the mid-1970s through the late 1990s include reductions in size at age, possible depressed late-term pregnancy rates, significantly reduced pregnancy rates for lactating females, and a decline in per capita natality of female sea lions at some rookeries. These responses by Steller sea lions are opposite to those predicted by direct, top-down, factors as body condition, growth rates, and natality should increase or remain the same when population abundance is reduced. These bottom-up factor(s) appeared to be affecting Steller sea lions as early as the 1960s and 1970s, at about the same time that large numbers of Steller sea lions were also killed directly (especially in the late 1970s and 1980s).

The carrying capacity of the North Pacific for Steller sea lions likely fluctuates in response to changes in the environment. Maschner *et al.* (2010) reported from the archeological record in the western GOA region that Steller sea lions likely have gone through at least three major declines in the last 1000 years and that these collapses are closely tied to reported regime shifts in the North Pacific ecosystem. Maschner *et al.* (2010) also report on the archeological record from historic Native village sites in the western Alaska Peninsula area and Unimak and Sanak Islands noting some Steller sea lion abundance response to Aleut harvest for skin kayaks during the period 1400-1700 AD, but that the paleoarchaelogical record from 250 to 4,500 years bp suggests sea lion abundance fluctuations are not necessarily correlated with human numbers but more likely climate and oceanographic conditions which are also reflected in these sites. Both regional and localized fisheries removals of prey could have exacerbated natural changes in carrying capacity, possibly in non-linear and unpredictable ways. Changes in the overall energy density of the prey field due to both climate shifts and long-term fisheries impacts (e.g., exploitation strategy), may have reduced the foraging efficiency of Steller sea lions and affected their ability to obtain adequate energy to maintain body condition and full reproductive potential.

In our review of climate and regime shifts, gadids were not necessarily affected across the range of Steller sea lions by the 1977 regime shift. Although it appears that EBS pollock did benefit from this change, GOA pollock and Atka mackerel likely were unaffected or affected in different ways that are still not clear. Also, the Steller sea lion population may have been increasing during the warm climate regime of the 1940s and 1950s, a period that was likely rich in gadids, but may have been affected by nutritional stress as early as the 1960s and 1970s, before the 1977 regime shift (Trites and Larkin 1992). Results by Hennen (2006) correlate sea lion declines with fisheries around rookeries in the 1980s, and find no correlation between fisheries and sea lion dynamics in the 1990s after conservation measures were enacted around rookeries and shooting was prohibited.

To summarize the Environmental Baseline impacts, climate and regime shifts, diseases, parasites, and predation have affected Steller sea lions throughout their 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 (e.g. regime shifts and natural variability) and predator abundance (e.g., killer whales and humans), as well as in response to the size of the Steller sea lion population itself. Steller sea lions persisted in the North Pacific despite the adverse impact of these stressors, and they did so without an apparent loss of genetic diversity, which would indicate that the population had gone through a “genetic bottleneck” (NMFS unpublished data). Therefore, for tens of thousands of years prior to the 1970s, Steller sea lions adapted to and accommodated fluctuations in their carrying capacity due to natural variability, disease and parasitism, killer whale predation, human-related kills, and apparently maintained, on average, a relatively large population size (i.e., above the point that would have resulted in an obvious genetic bottleneck). This is not to say that the population did not go through historical changes in population size or distribution as reported by Nelson (1887) and Maschner *et al.* (2010) or similar changes for seabirds (Causey *et al.* 2005).

The cumulative effects of future state, tribal, local, and private actions on Steller sea lions and their critical habitat include State fisheries, subsistence harvest, illegal shooting, entanglement in marine debris, disease, and disturbance from vessel traffic. The reported take levels associated with subsistence harvest in the eastern DPS are low and, as concluded in the Recovery Plan, should not impede recovery of the western DPS. Levels of Steller sea lion harvest in a few locations in the western DPS (e.g., Atka, where the total take estimate for 2007 was 54, with an upper range estimate of 87.2) could contribute substantially to the already downward trend in the local area and contribute to the overall downward trend in the subarea.

State-managed fisheries include salmon fisheries that occur in nearshore and offshore areas. Herring fisheries are also managed by the State. Both of these target high-energy species that may be important components (at least seasonally) of the diet of Steller sea lions. Additional cumulative effects include state groundfish fisheries in state waters and in the EEZ that are not managed under parallel regulations and are not included in a NMFS FMP. This includes lingcod, black, and blue rockfish, state waters sablefish, and the state waters Pacific cod fishery. The State of Alaska employs various management measures that indirectly provide some measure of protection to sea lions, and waters within 3 nm of important sea lion terrestrial sites in the western DPS are closed to vessel entry, including vessels fishing under state programs. These state fisheries as currently prosecuted would present similar issues and concerns as those associated with the proposed actions, albeit at much reduced levels. Generally these fisheries are small compared to the federal fishing associated with the proposed action. The amount to which state fisheries may add to the cumulative effects to the western DPS remains unknown, but could be significant in that they take place within near-shore areas and often target highly-concentrated, high value prey species for Steller sea lions.

Indirect or bottom-up factors may have contributed to the decline observed from the mid-1970s through the late 1990s. These bottom-up factor(s) appeared to be affecting Steller sea lions as early as the 1960s and 1970s, at about the same time that large numbers of Steller sea lions were also killed directly (especially in the late 1970s and 1980s). The combination of reduced population abundance and poor body condition indices is consistent with a substantial reduction in carrying capacity. Changes in vital rates may have been a function of acute direct mortality factors, as well as chronic nutritional stress resulting from a combination of reduced prey availability and quality. Two stressors were likely to have affected the prey field for Steller sea lions: (1) climate induced changes in the species composition, distribution, or nutritional quality of Steller sea lion prey and (2) fishery-induced changes in localized or overall prey abundance and quality. While the carrying capacity of the North Pacific for Steller sea lions likely fluctuates in response to changes in the environment, fisheries removals of prey likely have exacerbated natural changes in carrying capacity in some sub-regions.

Overall, as noted above, portions of the western DPS of Steller sea lion have had a statistically significant improvement in the rate of change from the 1990s to the 2000s. In this recent near-decade (2000-2008), four of the seven sub-regions have positive rates of population growth, two that are statistically significant ($\alpha = 0.1$)(Russia/Asia and eastern GOA). Three sub-regions are declining in abundance, one at a statistically significant rate (western Aleutian Islands) in this period. Over the last eight years, the numbers of sea lions in the western Aleutian Islands sub-region have declined by over 40%. If it were not for this one sub-region, it could be argued that the western DPS of Steller sea lions was moving toward recovery, as (1) overall the population is increasing and moving toward the number of animals required for downlisting, (2) no two juxtaposed sub-regions are in significant decline, and (3) no one sub-region has a decline in abundance of over 50%. However, because of the current decline in the western Aleutians (approximately 45%), as well as the slow decline observed in the central Aleutian sub-region, the recovery of this DPS is not meeting the Revised Recovery Plan criteria. If population growth trends in the western and central Aleutians continue at current rates, Steller sea lions may be extirpated from this portion of their range.

To facilitate a more in-depth examination of Steller sea lion pup and non-pup counts and trends, vital rates, prey resources and fishery harvests of prey, and other physical and biological data throughout the range of the western DPS, NMFS subdivided groups of rookeries that have similar demographic characteristics into Rookery Cluster Areas or RCAs. This analysis is used throughout this Biological Opinion and is described in detail in Chapter 5. The following is a summary of the status of Steller sea lions in each of these RCAs.

In the western DPS, Steller sea lions are in significant decline in the western Aleutian sub-region (RCA 1). In the western portion of the central Aleutian sub-region (RCAs 2 and 3), the sea lion population appears to be declining, but the rate of decline since 2000 is not significant.[1] In the eastern portion of the central Aleutian sub-region (RCAs 4 and 5), there are inconsistent trend estimates, but pup production is increasing in both areas. In the eastern Aleutians sub-region (RCA 6 and part of RCA 7), sea lion numbers are increasing at a robust rate (e.g., 3% or over per year, but non-significant). The western GOA sub-region is also increasing at a robust rate (over 3% but non-significant), although, in the eastern portion of this sub-region (part of RCA 8), sea lion numbers are have been stable since 2000. Finally, in the eastern GOA the rate of increase is both robust and statistically significant. However, in the western portion of the sub-region (RCA 9), the rate of recovery is not robust, while in the eastern portion of the sub-region, the rate of recovery is robust.

---

[1] In this biological opinion the term “significant” refers to statistical significance with a threshold defined as $P > 0.1$. Throughout the biological opinion the term “robust rate” refers to a discrete rate of population change $\geq 3\%$.

The western Aleutian Islands (RCAs 1-3) represent an ecosystem characterized by a narrow continental shelf. Relative to other subareas (Bering Sea or GOA), the Aleutian Islands region as a whole (RCAs 1-5) has the smallest total biomass of the important Steller sea lion prey. Throughout much of the Aleutian Islands, data indicate that Steller sea lions rely primarily on two key prey species: Atka mackerel and Pacific cod. Telemetry results indicated that juveniles foraged intensively inside of critical habitat and that they also ranged widely offshore, perhaps foraging within eddy systems to the west and north of the Aleutians. Since 1999, fisheries have removed a consistently high proportion of their total catch within critical habitat. During this same time Steller sea lions numbers continued to decline. While some regionally limited fishery changes have been implemented to reduce seasonally aggregated catch within critical habitat or to improve dispersal of catch (e.g., seasonal/spatial quotas, platoons), in general these measures have not been associated with any significant change in Steller sea lion trends. In the far west (RCAs 1-3), a combination of factors has possibly affected Steller sea lions, including: (a) removals of Atka mackerel and Pacific cod with a high proportion of catch inside critical habitat, (b) spatial heterogeneity of Steller sea lion foraging habitat (highest habitat patchiness and fewest prey options) and (c) killer whale predation (current surveys have yet to observe a transient killer whale in RCA 1 but high numbers have been surveyed in RCAs 2 and 3).

RCAs 4 and 5 are similar to RCAs 1-3 due to a narrow shelf, relatively small groundfish biomass, and a low diversity Steller sea lion diet. There has been no fishing for the primary groundfish prey species (Atka mackerel) within critical habitat in RCA 5 since 2002, nor any directed Atka mackerel fishery ever in RCA 4. In RCA 4 Steller sea lion non-pup numbers show a decline between 2000 and 2008, while pup production has increased during this time period. In RCA 5, sea lion numbers are increasing, but not robustly. A combination of a high proportion of Pacific cod catch within critical habitat in winter in an intermediate Steller sea lion foraging environment in RCAs 4-5 has possibly resulted in chronic long-term nutritional stress that has adversely affected survival and reproduction to the extent that the population is not recovering at a robust rate.

The eastern Bering Sea and GOA regions (RCAs 6-7) represent an ecosystem characterized by a broad continental shelf (larger in the EBS, less so in the GOA), the largest biomass of important Steller sea lion groundfish prey species (pollock and cod), and a relatively diverse Steller sea lion diet. Here, Steller sea lion numbers are increasing at a robust rate. Telemetry data from RCA 6 shows that juveniles forage primarily within critical habitat areas including the extensive Steller Sea Lion Conservation Area. In RCA 7, juveniles utilize critical habitat areas intensively, but they also move offshore into eddy systems. Since 1999, fishery measures have significantly reduced catch in critical habitat and have seasonally dispersed catch as well, particularly in RCA 6. Fishery management measures in this highly productive region appear to be sufficient to allow for recovery of these RCAs.

RCAs 8 and 9 are characterized by a continental shelf and groundfish prey biomass of intermediate magnitudes compared to Areas 1-5 (smaller) and Areas 6-7 (larger). The diet of Steller sea lions is relatively diverse in these areas, and the chief groundfish prey species are pollock, salmon, Pacific cod and arrowtooth flounder. A high proportion of the total catch of pollock and Pacific cod is caught in winter and within critical habitat. Steller sea lion numbers have stabilized over the last 20 years, but have shown only slight increases in the 2000s in these RCAs, suggesting that fishery measures may have provided for limited recovery. A combination of high catch amounts of pollock and Pacific cod within critical habitat in winter in RCAs 8-9 in an intermediate Steller sea lion foraging environment could possibly have resulted in chronic long-term nutritional stress that has adversely affected reproduction, but probably not survival, resulting in the current population stability but lack of recovery. To achieve a more robust recovery trajectory in these RCAs, additional measures to reduce the amount of Pacific cod caught in winter could be considered – although given the net increase in numbers over the last eight years and the robust recovery in the western and eastern GOA sub-regions, this recommendation would be considered a conservation measure, and not a mitigation measure to remove jeopardy or adverse modification of critical habitat.

RCA 10 is the area that contains the easternmost region of the western DPS. It contains Prince William Sound (PWS), the eastern GOA, and the Kenai coast. Steller sea lion numbers appear to be recovering in this region at a robust rate and are meeting the demographic criteria contained within the Revised Recovery Plan. The composition of Steller sea lion prey differs here from that in RCAs 1-9, shifting to a prey complex more similar to that found in southeast Alaska. Telemetry data indicates that most juveniles forage within critical habitat zones within PWS and the offshore islands. These areas are closed to trawling for pollock and Pacific cod. No changes in fishery management measures appear necessary within RCA 10.

In the Russian/Asian portion of the western DPS, 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 Steller sea lions at major rookeries and haul-outs in the Kuril Islands alone declined 74%. By 1990, the total Russian population had declined by approximately 50% to about 13,000 (the populations most proximate to the U.S. portion of the western DPS). Between the early 1990s and 2008, the Russia/Asian population (including pups) increased at a statistically significant and robust rate to about 25,000 overall.

Summary of Weight of Evidence for Risks to the Western DPS

The western Steller sea lion DPS declined significantly through the 1980s and 1990s. Since 2000, the decline has ceased and in most sub-regions the wSSL population is increasing. Sub-regions where the decline continues are the western and central Aleutian Islands (RCAs 1-5). Sea lion abundance is increasing at a statistically significant rate in four of remaining the five sub-regions.

The area inhabited by the western DPS is a fished ecosystem, from which large quantities of certain target species have been harvested since the 1960s, initially by foreign fisheries and by 1989 entirely domestic fisheries. The count of Steller sea lions in the western DPS in the Kenai to Kiska census area was over 100,000 animals (non-pups) by the end of the 1950s and about 90,000 around the end of the 1970s. Then a marked decline commenced with about 22,000 non-pups counted in this census area by 1990, and by 2000 the number of non-pups was at about 15,000. About 17,000 were counted as of 2008 in the Kenai to Kiska census area, the last survey date for non-pup animals.

Many have speculated on reasons for the decline in the 1980s and 1990s, including a climate regime change in the late 1970s that may have altered habitat conditions and prey abundance and diversity, increased predation, intentional and non-intentional human-caused mortalities, and fishery effects. It is generally agreed that the primary factor or factors responsible for the steep decline in the 1980s will never be identified with any assurance. Likely it is a combination of multiple factors (NRC 2003). In this last decade, the available information on birth and death rates indicates that adult and juvenile survival rates are similar to those pre-decline, but that natality with some exceptions has declined on the order of 30% relative to the pre-decline era. Our understanding about changes in these vital rates is limited as the number of sub-regions properly studied in the western DPS is limited to three at best (e.g.,Marmot Is. in RCA 9).

Pup to non-pup ratios are an indicator of reproductive rates (or natality) in sea lion populations. Chapter 3 describes some of the caveats about the interpretations of ratios of counts of pups to counts of non-pups. Values for the ratios of counts of pups to adult females on rookeries are provided in Table 3.6. Pup/adult female ratios in the western Aleutian Islands sub-region are the lowest of any of the western Steller sea lion DPS sub-regions (i.e., 0.29). All other sub-regions show pup/adult female ratios of 0.37-0.42, which are about 28% higher than in the western Aleutian Islands. In the central Aleutian Islands, a sub-region that has experienced a 36% decrease in natality over the past three decades, the pup/adult female ratio is 0.39. The pup/adult female ratio for the eastern DPS is in excess of 0.8, which is associated with a robust rate of increase in abundance for over 20 years (i.e., 3% per year).

Other measures of SSL natality have been published including models by Holmes *et al.* (2007) and Maniscalco *et al.* (2010). Brand/resight work, particularly in Russia, show that some SSLs move away from branding sites and other sea lions from other sites immigrate (Burkanov 2010), confounding calculation of pup/non-pup ratios as indicators of natality. Results of these various studies are equivocal in ascertaining a natality rate that would result in continued population increase. Given population increases in three of the six US sub-regions in the western DPS from 2000-2008, natality appears sufficient to ensure increases in most of the sub-regions. However, interpretation of a successful natality rate inferred from pup/adult female ratios is confounded as the ratio in the central Aleutian Islands (0.39) where the population is decreasing in numbers of non-pups and pups is greater than the ratio in the eastern Aleutian Islands and the western and eastern GOA where numbers of non-pups and pups are increasing (Table 5.3).

Pup to non-pup ratios based on data collected in 2009 suggest that natality rates of the western DPS are lower than those in southeast Alaska (DeMaster 2009). The pup to non-pup ratio from the two largest and oldest rookeries in southeast Alaska (Forrester Complex and Hazy Island), was 0.85. Rookery pup to non-pup ratios varied from 0.44 to 0.63 among sub-regions in the western DPS in 2009 and averaged 0.57. (DeMaster 2009). The most reasonable explanation for the pattern of natality in the western DPS relative to the eastern DPS is that portions of the western DPS may be nutritionally stressed as other hypotheses related to mechanisms associated with decreased natality (e.g., disease and contaminants) have for the most part been dismissed as not being significant, although killer whale predation remains a likely stressor in portions of the range of the western DPS (*cf* Horning and Mellish 2010a, b).

This Biological Opinion seeks to address the question of whether the groundfish fisheries in the GOA and BSAI areas are likely to cause jeopardy to the continued existence of the western DPS of Steller sea lions or likely to adversely modify its critical habitat. Given that in the seven sub-regions identified in the down- and de-listing criteria in the Revised Steller sea lion Recovery Plan trends in abundance are increasing in four sub-regions, our primary focus of concern in this Biological Opinion has been directed at the two sub-regions where substantial declines in abundance have been recorded since 2000. The continued decline in these sub-regions in light of continued fisheries, some of which target prey utilized by Steller sea lions, is of considerable concern to NMFS. We recognize existing fishery measures are likely mitigating impacts on Steller sea lions in other sub-regions, allowing increases in abundance to continue, and are not recommended for change; our focus here is on what additional measures, or changes in measures, may be required in the two sub-regions of primary concern.

It appears that the following factors may have acted or may continue to act individually or together to cause significant declines or otherwise limit the rate of recovery in one or more of the sub-regions that comprise the distribution of this DPS:

- Change in carrying capacity for SSLs; the North Pacific is likely a different ecosystem today than in the 1950s and 1960s. No major regime shift has occurred since the decline began in the late 1970s or early 1980s that might suggest another favorable foraging environment for Steller sea lions is on the horizon. Therefore, the habitat for the western DPS in the North Pacific may be close to its maximum capacity to sustain populations of SSLs, other large piscivorous marine mammals, and ichthyo-piscivorous competitors, and current fishing levels.

- Overlapping diets among SSL competitors, particularly arrowtooth flounder (Section 4.2.4; Boyd 2010), which is at a likely highest ever abundance in the North Pacific, may be a factor in placing competitive pressure on common prey items for Steller sea lions, and although data are unavailable, conceivably this competitive pressure could have a role in depressing the rate of recovery of the western DPS of Steller sea lions. This interaction may vary by sub-region.
- Killer whale predation can be locally high and could be suppressing SSL recovery in some parts of the Aleutian Islands (e.g.,Durban *et al.* [2010] reported very high numbers of transient killer whales in the central and eastern Aleutians, numbers sufficient to heavily prey on local Steller sea lions, although this ecotype of killer whales may not necessarily be present in high numbers at one time or place) and are thought by some to be the single greatest source of mortality for juvenile SSLs in the eastern GOA region (Horning and Mellish 2010a).
- The importance of nutritional stress in explaining the dynamics of the western DPS of Steller sea lions has been debated for decades. The current conclusion is that pollock in adequate availability can sustain healthy populations of SSLs, although it is equivocal how long sea lions can consume only pollock and continue to gain weight. For optimal foraging, SSLs require a diversity of prey species. SSLs have adapted to seasonal sources of high energy prey and to the mix of prey species present in their foraging areas (Rosen 2009, Trites *et al.* 2007, Womble *et al.* 2009, Winter *et al.* 2009, Sigler *et al.* 2009).
- Body condition of SSLs in the western DPS is relatively good (i.e., compared to body condition in animals from the eastern DPS), particularly for pups. Data recently collected on pups in the Aleutian Islands, Gulf of Alaska, Prince William Sound, and Southeast Alaska show that they are generally healthy and in good condition throughout their range in Alaska (Rea *et al.* 2010) indicating nutrition is sufficient to produce healthy young and sustain healthy juveniles and adults. Another indicator of SSL condition, skull size, hasn't changed as it likely would have if nutritional stress were the primary issue over the last 2-3 decades (Trites *et al.* 2008).
- Direct mortality of SSLs through a combination of historic commercial harvests, subsistence harvests, intentional kill, and incidental take in fisheries may explain a significant portion of the western Steller sea lion population decline that occurred through 1980 (Trites and Larkin 1992, Atkinson *et al.* 2008) but these sources have not likely affected the population in the past decade.
- Disease, parasites and contaminants could be a factor, but data are scarce and spotty and the preponderance of evidence does not support these potential stressors as being significant factors. Interestingly, PCB levels and mercury in SSL tissues are higher in the western portion of the range of the western DPS (Castellini *et al.* 2009). Preliminary data show higher stress protein (haptoglobin) levels in the eastern stock, where populations are more concentrated and higher prevalence of hookworm parasites have been found (Rea *et al.* 2010). Zenteno-Savin *et al.* (1997) found plasma haptoglobin levels are significantly higher in sea lions in the Aleutians and GOA than in SE Alaska.
- Correlations between western SSL trends in abundance to commercial groundfish fisheries are highly varied, some positive, some negative, and some spurious (e.g., Hennen 2006). Some have found no significant correlations between certain fishing techniques (such as longline fisheries) or target fisheries (such as Aleutian Islands Atka mackerel) and SSL trends (Calkins 2006, Trites 2010).

Past SSL telemetry data indicate SSLs in certain areas have tended to forage close to land, most within 20 nm. New spatial analyses indicate that SSLs indeed forage close to rookeries and haulouts, particularly in the 0 to 10 nm zone and also in the areas further offshore to 20 nm (Boor 2010, AFSC 2010b). In particular, recent telemetry information indicates that in RCAs 1, 2, and 3, a moderately large proportion of telemetered animals forage outside 20 nm (AFSC 2010b), although the number of telemetered animals is few and most are juvenile males. This may also be the case for parts of the central GOA sub-region. Therefore, it appears that foraging strategies of Steller sea lions vary by sub-region, and certainly also vary by gender of the animal, season of year, and age. While specific mechanisms related to competitive interactions between SSLs and commercial fisheries are difficult to verify, it appears that commercial fisheries, at least in the western and parts of the central Aleutian Islands, may remove fish that are prey for SSLs that forage there, or may draw down biomass levels in the general region, affecting prey availability in nearshore areas where SSLs prey most heavily.

Diet information indicates the dependence of Steller sea lions on certain prey species varies by sub-region. Steller sea lions in the western and central AI region heavily depend on Atka mackerel (96% Frequency of Occurrence [FO] in winter, 55 % FO in summer), which is the major target species harvested in commercial fisheries. Steller sea lions also require Pacific cod in the western and central AI sub-regions (6 % FO summer, 26 % FO winter).

The Atka mackerel annual quota in the Aleutian Islands is apportioned to each of the three NMFS Fishery Management Areas, with approximately one third to Area 541. Harvest in Area 541 has been largely outside critical habitat since most of Area 541 is closed to 20 nm, effectively closing most of Atka mackerel habitat to harvest leaving this area open to Steller sea lion foraging without fishery competition. Steller sea lion production in Areas 542 and 543, however, has been the lowest in the western DPS, and the Atka mackerel closures in these two areas have been the least restrictive with approximately two thirds of the harvest removed from Areas 542 and 543, most from inside critical habitat. While not conclusive, current fishery management measures that have limited Atka mackerel harvest from within critical habitat in the Aleutians east of 178ºW longitude (eastern portion of Area 542 and Area 541) could have contributed to improved Steller sea lion production in that area, while markedly less restrictive closures in Areas 542 and 543 may have contributed to the poor production of Steller sea lions in those areas.

The AI region also may be a more rigorous physical environment as evidenced by frequent stormy conditions, variable temperatures, and complex frontal features related to sea surface temperatures. The AI region may also be less hospitable for Steller sea lions (Lander *et al.* 2010) than other regions.

Conclusions

> ***After reviewing the current status of the endangered western population of Steller sea lions, the environmental baseline for the action area, the proposed actions, and the cumulative effects, it is NMFS' biological opinion that the action, as proposed, is likely to jeopardize the continued existence of the western DPS of Steller sea lion.***

Steller Sea Lion Critical Habitat: Western DPS

The response of critical habitat to the Environmental Baseline, Effects of the Action, and Cumulative Effects is presented in chapter 7. The essential features of this habitat most relative to this assessment are the principal species of Steller sea lion prey, which include walleye pollock, Atka mackerel, salmon, Pacific cod, cephalopods, arrowtooth flounder, and herring, among other species. These species may be highly dynamic, and heavily influenced by environmental variability (oceanographic conditions, regime shifts, climate change), ecological change, anthropogenic factors including commercial fisheries, and predation.

The potential response of Steller sea lions to reduced availability of prey in portions of critical habitat left open to fisheries would be chronic nutritional stress (see Section 3.1). Reduced prey availability can lead to physiological responses by Steller sea lions that directly (e.g., reduced natality) or indirectly (e.g., increased mortality from predators due to increased foraging) reduce their population growth. A sustained reduction of prey resources across a broad geographic region, or ecosystem, would thus reduce the carrying capacity of Steller sea lions. These impacts have generally been referred to as nutritional stress.

State-managed fisheries occur almost entirely within critical habitat. State-managed fisheries include salmon fisheries that occur in nearshore and offshore areas. Herring fisheries are also managed by the State. Both of these target high-energy forage species that may be important components (at least seasonally) of the diet of Steller sea lions. Additional cumulative effects include state groundfish fisheries in state waters and in the EEZ that are not managed under parallel regulations and are not included in a NMFS FMP. This includes lingcod, black, and blue rockfish, state waters sablefish, and the state waters Pacific cod fishery. The State of Alaska employs various management measures that indirectly provide some measure of protection to Steller sea lions, and all waters within 3 nm of shore within Steller sea lion critical habitat are closed to vessel entry, including vessels fishing under the State programs. These State fisheries would present similar issues and concerns as those associated with the proposed actions, albeit at much reduced levels. Generally these fisheries currently are small compared to the Federal fishing associated with the proposed action e.g., the 2000 State pollock harvest for the Gulf of Alaska was 1.7% of the Federal fishery, although the state cod harvest was 22.5% of the total Federal ABC. NMFS concludes based on available information that State managed fisheries for pollock, Pacific cod, herring, and salmon are likely to continue to compete for fish with foraging Steller sea lions. Given the importance of near shore habitats to Steller sea lions, this competition for fish may have consequential effects. More data on the foraging habits of Steller sea lions from research in key geographic areas could aid our understanding of where and when these effects might be most important.

Section 7.5.1 describes fisheries impacts to RCAs within critical habitat and the response of Steller sea lions and is incorporated here by reference.

Risk to Critical Habitat

The question in assessing the risk of the action to critical habitat is whether the proposed action will reduce the value of critical habitat for the conservation of the species. Thus, NMFS must determine whether affected designated critical habitat is likely to remain functional (or retain the ability to become functional) to serve the intended conservation role for the species in the near and long term under the effects of the action, environmental baseline and any cumulative effects. Much of the discussion in Section 7.5.2 is incorporated here as well, specifically impacts to critical habitat and Steller sea lion responses to habitat changes. Because the jeopardy analysis is primarily a habitat-based assessment, and the extent of critical habitat for the western DPS is so extensive, the two assessments are very similar. Therefore, rather than make the same arguments in both places, they will be incorporated here.

Prey resources are the most essential feature of marine critical habitat for Steller sea lions. The status of critical habitat is best described as the status and availability of the important prey resources contained within those areas, which include pollock, Atka mackerel, salmon, Pacific cod, arrowtooth flounder, Irish lord, rock sole, snailfish, herring, capelin, sand lance, other forage fish, squid, and octopus. Dominant prey items vary with region and season, but the most significant groundfish prey items for Steller sea lions in the western DPS are Atka mackerel, pollock, Pacific cod, and arrowtooth flounder, each of which has at least a 10% frequency of occurrence in the Steller sea lion diet.

A reduction in prey resources in critical habitat may result in a reduction in population growth rate, which would therefore be a reduction in the conservation value. Specifically, reduced prey availability can lead to physiological responses by Steller 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 Steller sea lions. These impacts have generally been referred to as nutritional stress and would be a reduction in the functionality of the conservation role of designated critical habitat.

Despite the many factors and changes to these prey species over the last several decades, the biomass for Pacific cod and walleye pollock are largely near levels assessed in the mid- to late-1970s. However, data on the important and relevant matter of how prey availability within critical habitat may have changed over time is often unavailable. Chapter 4 describes the great difficulty in assigning cause to changes in prey biomass or local abundance.

Removals of fish by these fisheries could have exacerbated natural changes in carrying capacity and may have contributed to declines in the western DPS. The implementation of conservation measures, in both the early 1990s and the late 1990s and early 2000s, is correlated with a reduction in the rate of decline of the western DPS (also see Section 5.1.2.2). However, the information necessary to determine if the conservation measures actually contributed to the reduced rate of decline is not currently available. However, it is possible that conservation measures implemented in the 2000s have had some positive effects on reducing the impacts of fishery removals of prey species that are important to Steller lions, but these conservation measures may be insufficient to provide for the survival and recovery of the western DPS.

We have discussed how Steller sea lions foraging near rookery sites in critical habitat depend on the availability of food supplies in the vicinity to meet the energetic needs involved in reproduction (adult females and males, and pups). The issue of prey availability and nutritional adequacy is highly complex and very likely specific to the eco-region or habitat site in question.

Differences in western Steller sea lion population response since 2000 is likely reflective of three main factors, (a) overall ecosystem productivity (i.e., Aleutian Islands is less than GOA which is less than the EBS), (b) fishery intensity within critical habitat on important Steller sea lion prey species, and (c) predation pressure (e.g., central Aleutian sea lions may be more susceptible than sea lions in sub-regions to the east). For example, Steller sea lion populations in RCAs located in the Aleutian Islands and central GOA (low to medium productivity) experienced higher fishery intensities inside critical habitat (RCAs 1-3, 8-9). Steller sea lions in these RCAs performed poorer than those located in the EBS or western GOA (medium to high productivity), which experienced lower fishery intensities inside critical habitat (RCAs 6-7). Critical habitat areas in the Aleutian Islands west of 178°W are open to Atka mackerel directed fishing up to 60% of their annual catch, and open to the directed Pacific cod fixed gear fisheries with few restrictions on catch (other than BSAI TAC). Steller sea lion populations in these areas (RCAs 1-3) continue to decline and have shown no recovery since conservation measures were implemented in the early 1990s (e.g., no shooting). Critical habitat areas in the central GOA (RCAs 8-9) continue to provide a high proportion of the catches of pollock and Pacific cod, particularly in winter. Steller sea lion populations in these areas continued to decline through the 1990s and have been approximately stable since 2000. By contrast, Steller sea lion populations in regions with lower proportions of catch in critical habitat have stabilized or increased in the 2000s and may depend on the overall productivity of the region. Those areas in the EBS or western GOA (RCAs 6-7) have performed better than those in the Aleutian Islands (RCAs 1 - 4).

Conclusion

From these data and observations, NMFS concludes that the relative intensity of groundfish fisheries as currently prosecuted in the western and central Aleutian Islands sub-regions, particularly within critical habitat, is negatively associated with Steller sea lion population trends since 2000 and that these adverse effects on the availability of important Steller sea lion prey within critical habitat are exacerbated in areas of low ecosystem productivity and habitat spatial heterogeneity. Based on this analysis of the effects of the action, and considering the ongoing nature of this action, it is unlikely that designated critical habitat within the western DPS of Steller sea lion will remain functional (or retain the ability to become functional) to serve the intended conservation role for the western DPS in the near and long-term.

After reviewing the current status of critical habitat that has been designated for the western population of Steller sea lions, the environmental baseline for the action area, the proposed action for Alaska Groundfish in the Bering Sea and Aleutian Islands and Gulf of Alaska, and the cumulative effects, it is NMFS' biological opinion that the action, as proposed, is likely to adversely modify the designated critical habitat for the western DPS of Steller sea lion.

Reasonable and Prudent Alternative (RPA)

Steller Sea Lion (Western DPS) and Critical Habitat

This Biological Opinion includes one RPA, which has multiple management measures or elements that are essential to avoid the likelihood of the groundfish fisheries jeopardizing the continued existence of the endangered western DPS of Steller sea lion or adversely modifying its designated critical habitat. Together these measures are designed to ameliorate adverse effects of removing prey biomass and avoid competition in the short- and long-term.

Prior to finalizing this Biological Opinion, NMFS provided to the public a draft Biological Opinion with a draft Reasonable and Prudent Alternative. The public review process involved a special meeting of the North Pacific Fishery Management Council in August 2010 and its Scientific and Statistical Committee and Advisory Panel. Public comments were sought and over ten thousand were provided to NMFS. The Council submitted a recommendation for an alternative RPA, initially crafted by its Advisory Panel. The Council's Scientific and Statistical Committee also reviewed the draft Biological Opinion and RPA and drafted comments on the scientific analyses and the logic of the underpinning science supporting NMFS' recommended draft RPA. All comments and the Council's suggested alternative RPA were evaluated by NMFS and analyzed for possible inclusion in a revised RPA. NMFS ultimately accepted eight modifications to the RPA in the draft Biological Opinion; the analyses of these revisions indicated that their inclusion did not reduce the RPA's ability to meet the jeopardy and adverse modification standard. The revised draft RPA was presented to the Council and the public in October 2010, and then included in the RPA described below.

This RPA must be implemented quickly in order to halt the immediate effects of the fisheries on the acute population decline in the western portion of the range of the western DPS of Steller sea lion. Its measures are intended to support the recovery of the population as a whole. The risk factors identified in this Biological Opinion in combination with present and ongoing threats dictate the elements of the RPA that must be implemented by NMFS.

Existing fishery management measures as established in previous Biological Opinions must be maintained. The measures required below are in addition to the existing measures contained within regulation (see Table 2.16) as described below.

Because this Biological Opinion has found that jeopardy and adverse modification of critical habitat is likely, the action agency (SFD) is required to notify PRD (NMFS) of the date of the implementation of the reasonable and prudent alternative contained herein.

This RPA requires changes in groundfish fishery management in Management Sub-areas 543, 542, and 541 in the Aleutian Islands Management Area.

<u>In Area 543</u>:

- Prohibit retention by all federally permitted vessels of Atka mackerel and Pacific cod.
- Establish a TAC for Atka mackerel sufficient to support the incidental discarded catch that may occur in other targeted groundfish fisheries (e.g., Pacific ocean perch).
- Eliminate the Atka mackerel platoon management system in the HLA.

<u>In Area 542</u>:

Groundfish

- Close waters from 0–3 nm around Kanaga Island/Ship Rock to directed fishing for groundfish by federally permitted vessels.

Pacific cod

- Close 0–6 nm zone of critical habitat to directed fishing for Pacific cod by federally permitted vessels using nontrawl gear year round. For vessels 60 ft or greater, close critical habitat from 6 nm–20 nm to directed fishing for Pacific cod using nontrawl gear by federally permitted vessels from January 1 to March 1 12:00 noon.
- Close critical habitat to directed fishing for Pacific cod by federally permitted vessels 60 ft or greater using hook and line gear from January 1 to August 15, 12:00 noon.
- Between 177 E to 178 W longitude, close critical habitat from 0-20 nm to directed fishing for Pacific cod by federally permitted vessels using trawl gear year round.
- Between 178 W to 177 W longitude, close critical habitat from 0-10 nm to directed fishing by federally permitted vessels using trawl gear year round. Between 178 W to177 W longitude, close critical habitat 10-20 nm to directed fishing for Pacific cod using trawl gear by federally permitted vessels from June 10 12:00 noon to November 1, 12:00 noon.
- Prohibit directed fishing for Pacific cod by all federally permitted vessels from November 1, 12:00 noon to January 1. (This extends the trawl gear restriction to nontrawl gear.)
- Reinitiate ESA consultation if the nontrawl harvest of Pacific cod exceeds 1.5 percent of the BSAI Pacific cod acceptable biological catch (ABC) (equivalent to the Area 542 maximum annual harvest amount from 2007 through 2009). Similarly, reinitiate ESA consultation if the trawl harvest of Pacific cod exceeds 2 percent of the BSAI Pacific cod ABC (equivalent to the Area 542 maximum annual harvest amount from 2007 through 2009).

Atka mackerel

- Set TAC for Area 542 to no more than 47 percent of the acceptable biological catch (ABC).
- Between 177 E to 179 W longitude and 178 W to 177 W longitude, close critical habitat from 0–20 nm to directed fishing for Atka mackerel by federally permitted vessels year round.

- Between 179 W to 178 W longitude, close critical habitat from 0-10 nm to directed fishing for Atka mackerel by federally permitted vessels year round. Between 179 W and 178 W longitude, close critical habitat from 10-20 nm to directed fishing for Atka mackerel by federally permitted vessels not participating in a harvest cooperative or fishing a CDQ allocation.
- Add a 50:50 seasonal apportionment to the CDQ allocation to mirror seasonal apportionments for Atka mackerel harvest cooperatives.
- Limit the amount of Atka mackerel harvest allowed inside critical habitat to no more than 10 percent of the annual allocation for each harvest cooperative or CDQ group. Evenly divide the annual critical habitat harvest limit between the A and B seasons.
- Change the Atka mackerel seasons to January 20, 12:00 noon to June 10, 12:00 noon for the A season and June 10, 12:00 noon to November 1, 12:00 noon for the B season.
- Eliminate the Atka mackerel platoon management system in the HLA.

In Area 541:

Pacific cod

- Close 0–10 nm of critical habitat to directed fishing for Pacific cod by all federally permitted vessels year round.
- Limit the amount of catch that can be taken in the 10 nm–20 nm area of critical habitat based on gear type used:
    - Close critical habitat 10 nm–20 nm to directed fishing for Pacific cod using nontrawl gear by federally permitted vessels January 1 to March 1, 12:00 noon.
    - Close critical habitat 10 nm–20 nm to directed fishing by for Pacific cod using trawl gear by federally permitted vessels June 10 12:00 noon to November 1, 12:00 noon.
- Prohibit directed fishing for Pacific cod by federally permitted vessels November 1, 12:00 noon to January 1. (This extends this trawl gear restriction to nontrawl gear).
- Reinitiate ESA consultation if the nontrawl harvest of Pacific cod exceeds 1.5 percent of the BSAI Pacific cod ABC (equivalent to the Area 541 maximum annual harvest amount from 2007 through 2009). Similarly, reinitiate ESA consultation if the trawl harvest of Pacific cod exceeds 11.5 percent of the BSAI Pacific cod ABC (equivalent to the Area 541 maximum annual harvest amount from 2007 through 2009).

Atka mackerel

- Change the Bering Sea Area 541 Atka mackerel seasons to January 20, 12:00 noon to June 10, 12:00 noon for the A season and June 10,12:00 noon to November 1, 12:00 noon for the B season.
- Close the Bering Sea subarea year round to directed fishing for Atka mackerel.

Intended Effect of the RPA

Implementation of this RPA is expected to:

- Greatly reduce or eliminate local competition between Steller sea lions and the Atka mackerel and Pacific cod fisheries in the western Aleutian Islands sub-region. This action is expected to improve foraging success and prey availability for juvenile and adult Steller sea lions, which in turn is expected to lead to higher survival and natality rates.
- Significantly reduce the competitive overlap between Steller sea lions and fisheries for Atka mackerel and Pacific cod in the central Aleutian Islands sub-region. This is expected to improve foraging success and prey availability for Steller sea lions, particularly adult females with dependent young in winter, which in turn is expected to lead to higher natality rates.
- Implement an adaptive management strategy for exploited groundfish forage species that explicitly accounts for the prey requirements of listed species and allows them to reach the recovery goals in the western and central Aleutian Islands sub-regions. This is expected to allow NMFS to better evaluate the impact of the conservation measures on the recovery of Steller sea lions in this region.

NMFS designed the RPA described above to ensure that the groundfish fisheries would not likely jeopardize the continued existence of the western DPS of Steller sea lions or adversely modify critical habitat. This RPA was structured to mitigate effects of the fishery in sub-regions where Steller sea lion abundance continues to decline (western and central Aleutian Islands) and where available information indicates that reproduction may be reduced to a level that cannot support positive population growth (the western Aleutian Islands). The western and central Aleutian Islands were the two sub-regions where population growth was negative from 2000-2008 and of most concern. Currently, the western DPS of Steller sea lion is growing at a rate of 1.4% per year. However, as explained in Chapter 7, the western DPS is not meeting the criteria of a recovering population as determined by the Revised Recovery Plan and is in jeopardy of going extinct in the western portion of its range in U.S. waters.

This RPA was structured to address observed declines in Steller sea lions where there was a reasonable basis to support that commercial fisheries, by removing sufficient quantities of prey species important to Steller sea lions for basic nutrition, are likely to appreciably reduce the reproduction and thus the numbers of Steller sea lions in the western and central Aleutians Islands. The RPA also was designed to address impacts to habitat where there is a reasonable basis to support that commercial fisheries, by affecting the prey field, are likely to adversely modify the value of designated critical habitat for the survival and recovery of the western DPS of Steller sea lions. It is recognized that competition with fisheries for prey is likely one component of an intricate suite of natural and anthropogenic factors affecting Steller sea lion numbers and reproduction. While natural factors may be contributing, NMFS must ensure that actions authorized by NMFS are not likely to appreciably reduce the likelihood of survival and recovery of the western DPS of Steller sea lions.

While effects of the RPA on the response of the Steller sea lion population cannot be projected with any amount of certainty with the available information, NMFS has determined that by conserving important prey species in the areas and seasons of utmost importance to foraging Steller sea lions commensurate with the rate of decline observed in each fishery management area will be adequate to reduce the effect of the fisheries to the point that they would not be likely to appreciably reduce the survival and recovery of the species. This determination is premised on the expectation that fishery harvests displaced from important times and areas via this RPA are not subsequently concentrated in a manner that would result in effects that are not currently anticipated. Should modifications to the fisheries result in increases in spatial and temporal concentrations of fishing effort outside the realm of recent fishing patterns (e.g., 2000-2008), that would constitute “new information” requiring NMFS to reinitiate consultation.

It is NMFS’ opinion, based on the best available information, that the RPA described above will remove the likelihood that the fisheries will jeopardize the continued existence of the western DPS of Steller sea lions or adversely modify their designated critical habitat. Indisputably, measures that are more conservative than the RPA described above would also ensure that the fisheries are not likely to jeopardize Steller sea lions or adversely modify their critical habitat. However, NMFS must design RPAs to be consistent with the intended purpose of the action. Given the available information, the prescribed RPA is deemed to be necessary and sufficient to allow NMFS to conclude that the groundfish fisheries are not likely to jeopardize the continued existence of Steller sea lions or adversely modify their designated critical habitat. However, NMFS is committed to continue to monitor Steller sea lion population trends, especially in the western and central Aleutians sub-regions, to evaluate whether fishery conservation measures are adequately insuring the conservation of Steller sea lions and their critical habitat. To that end, NMFS recommends that the groundfish fisheries subject to this Biological Opinion be managed in an adaptive approach as further described in Chapter 8.

Steller Sea Lion: Eastern Distinct Population Segment

The eastern DPS range extends east from Cape Suckling at 144° W longitude along Alaska's southern coast and south to California. In 2008, NMFS estimated the population of sea lions in the eastern DPS at 63,000 animals. In contrast to the decline and lack of recovery documented in the western DPS, the eastern DPS increased at over three percent per year between the late 1970s and 2002, more than doubling in Southeast Alaska, British Columbia, and Oregon. Currently, this population is estimated to be at its highest level in recent history.

Since 1996, pup production in southeast Alaska has increased at a rate of 5.0 percent per year, or 1.3 percent faster than the increase in numbers of non-pups on rookeries. Surveys of pups and non-pups conducted in British Columbia in 2006 and in southeast Alaska, California, and Oregon in 2009 indicate that the overall eastern DPS trend continued through 2009 and is particularly strong in the northern portion of the eastern DPS in southeast Alaska and British Columbia.

NMFS recognizes that a small number of western DPS animals may be present in southeast Alaska, but this constitutes a small percentage of the overall population. Most of the increase in southeast Alaska comes from population increases in the eastern DPS. No fisheries for the principal groundfish prey species in critical habitat are found in southeast Alaska, and only very small fisheries for those species in the range of the eastern DPS. Given migration of some animals from the western DPS to the eastern DPS and evidence of pupping of those females, it is likely that habitat conditions in the eastern DPS provide for adequate survival and the ability to recover based on long-term demographics. Few cumulative effects impact the eastern DPS.

Based on the analysis conducted in this Biological Opinion, NMFS concludes that the action, as proposed, is not likely to jeopardize the continued existence of the eastern DPS of Steller sea lion. Nor is the proposed action likely to destroy or adversely modify critical habitat of the eastern DPS of Steller sea lion. Based on the recovery criteria established in the revised Steller Sea Lion Recovery Plan (NMFS 2008a), the eastern DPS is not in danger of extinction nor likely to become endangered and has, most likely, recovered from its previous threatened status.

Humpback Whales

Based on the analysis conducted in this document, NMFS finds that the BSAI and GOA groundfish fisheries and the parallel fisheries, as authorized under the FMPs, are not likely to jeopardize the continued existence of the North Pacific humpback whales or to destroy or adversely modify their critical habitat.

Humpback whales are found throughout the North Pacific and most Alaskan waters. They feed mainly on small schooling fishes, euphausiids, and other large zooplankton. Fish prey species in the North Pacific include Pacific herring, capelin, juvenile walleye pollock, and sand lance. Should the animals not get enough food during the time spent in Alaska, compensation will not occur in other locations or at other times of the year, as humpback whales fast while on their breeding grounds.

During the summer, humpback whales migrate short distances through their summer feeding range, and then migrate a long distance to spend the winter on breeding grounds in warmer waters. Data from the Structure of Populations, Levels of Abundance, and Status of Humpbacks (SPLASH) project conducted between 2004 to 2006 indicate that population structure and migration patterns are much more complex than previously understood, but also that humpback whales exhibit a high degree of site fidelity to specific feeding areas, with little interchange among them.

Humpback whales were listed as endangered under the ESA in 1973 due to commercial exploitation that severely depleted their populations. Prior to 1905, an estimated 15,000 humpback whales lived in the entire North Pacific; by 1966, commercial harvest had reduced that population to an estimate of between 1,000 and 1,200 animals. In 1965, the International Whaling Commission (IWC) banned commercial whaling in the Pacific Ocean, but Soviet whalers continued to harvest humpback whales until 1980. Currently, some illegal whaling continues, but we do not know how many whales are actually taken.

Since the listing of these whales as endangered and IWC's prohibiting commercial harvest of humpbacks, the Central and Western North Pacific humpback whale populations have increased substantially. We currently estimate that between 18,000 and 20,000 individuals live in the North Pacific, a significant increase from a previous estimate of 6,010 animals. Almost certainly, the overall increase results in part from the protective legislation enacted by both the United States and Canada during the early 1970s, which restrictions successfully reduced mortality at a time when the population was below carrying capacity.

Effects of contaminants, toxins, and disease on humpback whale populations are largely unknown. Killer whales prey upon humpback whales, although such attacks are observed relatively infrequently. Younger animals may be more vulnerable to this type of predation during migration when group size is smaller than in summering or wintering areas.

Western North Pacific Population

The Aleutian Islands and Bering Sea are important foraging habitats for this population. Until recently, the Western North Pacific humpback whale population was estimated at about 400 animals; currently, results from the SPLASH project estimate a population of 6,000 to 14,000 humpbacks in the Bering Sea and Aleutians, and 100 to 700 in Russian waters.

Data from the SPLASH project indicate that some humpbacks that forage in the Aleutians and Bering Sea spend their winter in areas researchers do not know about; consequently, these whales were not well-represented in any of the winter sampling areas. Although researchers believe it reasonable to assume that this breeding ground would be a region in the eastern central North Pacific, the location is uncertain given the complexities around migratory pathways.

Central North Pacific Population

Most humpback whales in the Central North Pacific population spend the winter months in Hawaii where they breed and give birth to and nurse their calves. Some animals, however, remain on the feeding grounds year-round.

Humpback whales in this population typically show fidelity to either the southeast Alaska or the Prince William Sound feeding areas. The current humpback whale abundance estimate in the combined feeding areas of Southeast Alaska and Northern British Columbia is approximately 3,000-5,000 animals. The current best adult survival rate in the Central North Pacific population is estimated at 0.963, and the rate of increase for this population is currently acknowledged as 7 percent per year.

Although measures of abundance continue to indicate an increasing trend, abundance estimates alone cannot be relied upon as accurate measures of population recovery without a long-term understanding of demographic parameters and variability in the population and the effects of natural and anthropogenic stressors on the status of the population. In addition, the species may be vulnerable to catastrophic or random events that could result in significant declines and increase the species' risk of extinction. However, on the basis of total abundance, current distribution, and regulatory measures that are currently in place, it is unlikely that this species is in danger of extinction in the foreseeable future, although some stocks may be at risk. NMFS initiated a status review under the ESA on August, 12 2009 (74 FR 40568) to determine whether humpback whales should be delisted or reclassified from endangered to threatened. NMFS is also considering whether to recognize Distinct Population Segments (DPSs) of humpback whales according to the 1996 DPS policy (61 FR 4722). DPSs of vertebrate species, as well as subspecies of all listed species, may be proposed for separate reclassification or for removal from the list. Critical habitat has not been designated for humpback whales anywhere throughout their range.

Humpback whale prey species are not targeted or taken in significant amounts by the fishery actions evaluated in this Biological Opinion. Gear entanglements are not uncommon for these whales and have been associated with the fisheries under the proposed actions. However, it is unclear to what extent entanglements reported to the stranding network in Alaska involve groundfish fishing gear. Overall, the number of entanglements that might result from interactions with groundfish fisheries appears to be low in contrast with other gear types. Also, the incidents of entanglement are not expected to reach such an extent that they will have negative population-level consequences for humpback whales in the North Pacific. The incidence of ship strikes leading to death or serious injury from vessels involved in the groundfish fisheries also appears to be negligible and unlikely to have population-level consequences for these whales.

Humpback whales may be disturbed by noise from fishing vessel engines. The whales typically react to the presence of a vessel with visible changes in behavior, such as leaving the area if sufficiently disturbed. However, the displacement of individual humpback whales by such disturbances is not likely to reach a level that would compromise the recovery or survival of the species.

Cumulative effects on humpback whales include direct and indirect impacts from state-managed fisheries. Over 100 entanglement incidents have been reported to the NMFS Alaska stranding program over the last 30 years, many involving pot gear or gill net gear from fisheries in the inside waters of southeast Alaska and in areas around Kodiak, Homer, and Seward. For many of these incidents, when disentanglement is not possible or the animal is not re-sighted, the ultimate fate of the animal remains unknown. State-managed fisheries represent an additional source of anthropogenic impact beyond those posed by the proposed action through entanglements of whales in the Central North Pacific and Western North Pacific populations.

After reviewing the current status of the Central Pacific population and Western Pacific population of humpback whales, the environmental baseline for the action area, the effects of the Federal and parallel groundfish fisheries off Alaska, and the cumulative effects, it is NMFS' biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Central North Pacific

population or the Western North Pacific population of humpback whales. No critical habitat has been designated for this species; therefore, none will be affected.

Sperm Whales

Based on the analysis conducted in this Biological Opinion, NMFS finds that the BSAI and GOA groundfish fisheries and the parallel fisheries, as authorized by the FMPs, are not likely to jeopardize the continued existence of the North Pacific sperm whales or to destroy or adversely modify their critical habitat.

Sperm whales are found throughout the North Pacific and throughout most Alaskan waters, usually in waters greater than 300 m in depth. Mature female and immature sperm whales of both sexes are found in temperate and tropical waters; sexually mature males join these groups throughout the winter. During the summer, mature male sperm whales are thought to move north into the Aleutian Islands, GOA, and the Bering Sea.

Reliable estimates for population abundance, status, and trends for the Alaska population of sperm whales are not available. However, the number of sperm whales in the eastern North Pacific has been estimated to be 39,200 animals. Alternatively, sperm whale density extrapolations have resulted in a practical working estimate of sperm whale abundance for the entire North Pacific ranging from 100,000-200,000 animals.

Sperm whales feed primarily on mesopelagic squid, but also consume octopus, other invertebrates, and fish. Of fish in the North Pacific, sperm whales are known to feed on salmon, lantern fishes, lancetfish, Pacific cod, pollock, saffron cod, rockfishes, Atka mackerel, sculpins, lumpsuckers, lamprey, skates, rattails, and sablefish. Male sperm whales have been known to take sablefish directly from longline gear in the GOA.

Sperm whales were listed as endangered under the ESA in 1973. Between 1947 and 1987, commercial whalers harvested more than 250,000 sperm whales in the North Pacific and severely depleted the Bering Sea population. In addition to the ESA listing, sperm whales were protected from commercial harvest by the International Whaling Commission in 1981. Nevertheless, Japanese whalers continued to harvest sperm whales in the North Pacific until 1988.

The estimated annual rate of human-caused mortality and serious injury appears to be minimal for this population. On the basis of total abundance, current distribution, and regulatory measures that are currently in place, this species may not be in danger of extinction in the foreseeable future, however a recent ESA status review concluded that due to the lack of sufficient and reliable information on the severity of multiple potential threats to the recovery of sperm whale populations, as well as population structure, species abundance and population trends, the status of the sperm whale should remain as "endangered" (NMFS 2009c). Critical habitat has not been designated for sperm whales anywhere throughout their range.

While the potential exists for competition between sperm whales foraging for prey species and groundfish fisheries in the GOA, scientists have no evidence that the groundfish fisheries in Alaska compromise sperm whale diet. The whales are known to take fish off of longline fishing gear in Alaskan waters, mainly the GOA, and this depredation has been increasing throughout the last decade, increasing the potential for ship strikes and entanglement in fishing lines. But the incidence of sperm whale entanglement in Alaska appears to be low, and would not be expected to reach a level that would have population-level consequences. And the potential for ship strikes, too, seems minimal and unlikely to result in an adverse population level effect for sperm whales in Alaska.

Any impact to sperm whales due to disturbance by vessels is uncertain. Given that many individual sperm whales are attracted to the sound of groundfish vessel engines and gear hauling catch, it would appear that they often do not interpret such noise as disturbance. Additionally, as depredation behavior in Alaska is only known to involve male sperm whales, it is unlikely vessel disturbance would present a concern for the species.

After reviewing the current status of the North Pacific sperm whale population, the environmental baseline for the action area, the effects of the Federal and parallel groundfish fisheries off Alaska, and the cumulative effects, it is NMFS' biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the North Pacific sperm whale. No critical habitat has been designated for this species; therefore, none will be affected.

Fin Whales

Based on the analysis conducted in this document, NMFS finds that the BSAI and GOA groundfish fisheries and the parallel fisheries, as authorized under the FMPs, are not likely to jeopardize the continued existence of the Northeast Pacific fin whales.

The fin whale has been listed as endangered under the ESA since its passage in 1973. Although most populations were depleted by modern whaling in the mid-twentieth century, there are still tens of thousands of fin whales worldwide. Commercial whaling for this species ended in the North Pacific in 1976.

For management purposes, three stocks of fin whales are currently recognized in U.S. waters in the Pacific: 1) Alaska (Northeast Pacific [NEP]), 2) California/Washington/Oregon, and 3) Hawaii. New information from Mizroch *et al.* (2009) suggests that this structure should be reviewed and updated, if appropriate, to reflect current data. The NEP stock of fin whales is likely to be affected by the Federal and parallel groundfish fisheries off Alaska.

Information on abundance of fin whales in Alaskan waters has improved considerably in the past few years. Although the full range of the NEP stock of fin whales in Alaskan waters has not been surveyed, a rough estimate of the size of the population west of the Kenai Peninsula could include the sums of the estimates from Moore *et al.* (2002) and Zerbini *et al.* (2006). Using this approach, the provisional estimate of the fin whale population west of the Kenai Peninsula would be 5,700. The most recent estimate of the annual fin whale population growth rate in Alaska was 4.8% (95% CI: 4.1-5.4%) for the period 1987-2003 (Zerbini *et al.* 2006).

Fin whales feed primarily on zooplankton and fish and may depend on some of the same species harvested in the groundfish fisheries (e.g., pollock). This results in the potential for competition between fin whales foraging for prey species and groundfish fisheries in the GOA and BSAI; however, the extent of this impact is currently not well understood. The potential does exist for a reduction in prey biomass to occur in the future, which could threaten fin whales' ability to successfully forage. Currently, however, there is no evidence that fin whale diet is compromised by the groundfish fisheries in Alaska.

Prior to 1999, there were no observed or reported mortalities of fin whales incidental to commercial fishing operations within the range of the Northeastern Pacific fin whale stock. However, in 1999, one fin whale was killed incidental to the Gulf of Alaska pollock trawl fishery. This take occurred in federal waters of statistical area 620, southwest of Kodiak Island. Between 2002 and 2006, there was one observed incidental mortality of a fin whale in the Bering Sea/Aleutian Island pollock trawl fishery, resulting in a mean annual mortality/serious injury of 0.23 fin whale/yr incidental to commercial fisheries in Alaska for this 5 yr period.

Based on the one mortality reported and investigated during 2002-2006, the minimum mean annual mortality/serious injury from ship strikes is 0.20 fin whales per year in Alaska. The estimated minimum annual total human-caused mortality and serious injury rate for the NEP stock of fin whales in the US EEZ for 2002-2006 is 0.43 whales per year. Accordingly, total human-caused mortality and serious injury is below 10 percent of PBR (1.14) for this stock. Because all total human-related serious injuries and mortalities are less than 0.1 PBR, NMFS determined that mortality and serious injury incidental to commercial fisheries will have a negligible impact on the NEP fin whale stock (75 FR 68767).

After reviewing the current status of the Northeast Pacific fin whale population, the environmental baseline for the action area, the effects of the Federal and parallel groundfish fisheries off Alaska, and the cumulative effects, it is NMFS' biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Northeast Pacific fin whale population. No critical habitat has been designated for this species; therefore, none will be affected.

1 BACKGROUND AND CONSULTATION HISTORY

1.1 Purpose

The Endangered Species Act of 1973 (ESA) (16 USC 1531-1544), amended in 1988, establishes a national statute for protecting and conserving threatened and endangered species of fish, wildlife, and plants and the habitat on which they depend. Section 7(a)(2) of the ESA, requires each federal agency to ensure that any action authorized, funded, or carried out by such agency be not likely to jeopardize the continued existence of any endangered species or threatened species, or to destroy or adversely modify critical habitat of such species. When the action of a federal agency may adversely affect a protected species, that agency (i.e., the "action" agency) is required to consult with the National Marine Fisheries Service (NMFS) and/or the United States Fish and Wildlife Service (USFWS), depending upon the protected species that may be affected. For the proposed action evaluated in this consultation – the management of the federal fisheries off Alaska - the action agency is NMFS Alaska Region, Sustainable Fisheries Division (SFD) and the consulting agency is NMFS Alaska Region, Protected Resources Division (PRD).

In a letter to NMFS, on October 18, 2005, the North Pacific Fishery Management Council (Council) recommended that NMFS commence the process to reinitiate a FMP-level formal Section 7 consultation on the effects of the Federal groundfish fisheries on ESA-listed species under U.S. Department of Commerce jurisdiction. On April 19, 2006, PRD received a written request from SFD to re-initiate formal ESA section 7 consultation on the Alaska groundfish Fishery Management Plans (FMPs), in order to evaluate the effects of current federal fisheries management on listed species in light of any new information gained since the previous consultations in 2000 and 2001. The request for re-initiation was accompanied by a Biological Assessment (BA). PRD concurred with this request and formally re-initiated consultation on June 21, 2006.

In the BA, SFD reviewed the status of the species and their critical habitat, the likely effects of the proposed actions, and the potential impacts to the species. For each species, SFD determined whether the actions were likely or not likely to adversely affect ESA-protected species and designated critical habitat. NMFS has determined that the action is not likely to adversely affect the following ESA-listed species: the blue whale, the bowhead whale, the North Pacific right whale and its critical habitat, the sei whale, the fin whale, the Olive Ridley sea turtle, the loggerhead sea turtle, the green sea turtle, and the leatherback sea turtle (Table ES.1 of the BA and subsequent re-initiation letters between SFD and PRD). Therefore, formal consultation on these species is not required.

Additionally, in 2009, SFD consulted with PRD on Amendment 91 to the Bering Sea and Aleutian Islands (BSAI) groundfish FMP for Cook Inlet Beluga (CIB) whales. NMFS determined that due to the behavior of CIB, the location and harvest amounts of potential prey species in the groundfish fisheries, and the minimizing of Chinook salmon bycatch under Amendment 91, Alaska groundfish fisheries may affect, but are not likely to adversely affect, CIB either directly through vessel interactions or indirectly through prey competition.

The NMFS Alaska Region SFD informally consulted with the NMFS SW Region on the southern DPS of green Sturgeon. Of the entire Alaska groundfish fishery, sturgeon are rarely taken incidentally in trawl fisheries in the Bering Sea. Since 2000 only 4 sturgeon were recorded taken in the Alaska groundfish fisheries. Of these four, one was a green sturgeon taken in the Bering Sea Pacific cod nonpelagic trawl fishery. The remaining sturgeons were taken in the nonpelagic trawl flatfish fisheries in the Bering Sea and Gulf of Alaska (Catch Accounting System September 28, 2010). Of the four sturgeon taken, two were identified as green sturgeon. These fish may have come from either North America or Asia. When possible, the NMFS Alaska Region collects tissue samples from incidentally caught sturgeon for genetic analysis to determine origin. Two acoustically tagged southern DPS green sturgeon have been detected by an acoustic array outside of Glacier Bay in Southeast Alaska. Trawling is prohibited in the Southeast Outside District where the southern DPS of green sturgeons were detected. Green sturgeon occurs primarily in nearshore waters less than 110 m depth. The coastline north of Yakutat is deeper than 110 m, which likely prevents the movement of the Southern DPS of green sturgeon further north. Because the incidental catch of any sturgeon is rare in the entire Alaska groundfish fisheries, and the detection of the southern DPS green sturgeon is limited to a location where trawling is prohibited, the Alaska groundfish fisheries are unlikely to adversely affect the southern DPS of green sturgeon.

NMFS has determined that the following species, however, are likely to be adversely affected by the action, and thus require formal section 7 consultation under the ESA:

(i) Western Distinct Population Segment (DPS) of Steller Sea Lion (*Eumetopias jubatus*); listed as threatened on November 26, 1990 (55 FR 40204); listed as endangered on May 5, 1997 (62 FR 30772); critical habitat designated on August 27, 1993 (58 FR 45269)
(ii) Eastern Distinct Population Segment of Steller Sea Lion (*Eumetopias jubatus*); listed as threatened on November 26, 1990 (55 FR 40204); critical habitat designated on August 27, 1993 (58 FR 45269)
(iii) North Pacific Humpback Whale (*Megaptera novaeangliae*) listed as endangered upon passage of the ESA of 1973 (16 U.S.C. 1531 et seq.)
(iv) North Pacific Sperm Whale (*Physeter macrocephalus*) listed as endangered upon passage of the ESA of 1973 (16 U.S.C. 1531 et seq.)

This Biological Opinion and incidental take statement (ITS) were prepared by NMFS in accordance with Section 7(b) of the ESA, and implementing regulations at 50 Code of Federal Regulations (CFR) Part 402. The Biological Opinion is based on an evaluation of both the direct and indirect effects of the action on listed species and their critical habitat, together with the effects of other activities that are interrelated or interdependent with that action. The opinion presents NMFS' review of the status of the listed species considered in this consultation, the condition of the critical habitat, the environmental baseline for the action area, all the effects of the action as proposed, and cumulative effects (50 CFR 402.14 (g)). For the jeopardy analysis, NMFS analyzed those combined factors to determine whether the proposed action is likely to appreciably reduce the likelihood of survival and recovery of the affected listed species. With respect to critical habitat, the analysis relies only on the statutory provisions of the ESA, and not on the regulatory definition of "destruction or adverse modification" at 50 CFR Part 402.02.

This Biological Opinion includes (1) the opinion of the agency as to whether or not the Federal action is likely to jeopardize the continued existence of a listed species, or result in destruction or adverse modification of designated critical habitat; (2) a summary of the information on which that opinion is based; and (3) a detailed discussion of the effects of the action on listed species and designated critical habitat. In this Biological Opinion, NMFS PRD has evaluated the effects of three actions:

- Authorization of groundfish fisheries under the Fishery Management Plan for Groundfish of the Bering Sea and Aleutian Islands Management Area;
- Authorization of groundfish fisheries under the Fishery Management Plan for Groundfish of the Gulf of Alaska; and
- State of Alaska parallel groundfish fisheries [consultation requested by the State of Alaska on March 31, 2006; see letter from McKie Campbell, Commissioner, to Robert D. Mecum, Acting Administrator, NMFS, Alaska Region and further clarified in Memorandum from Sue Salveson Sustainable Fisheries Division to Kaja Brix Protected Resources Division on January 25, 2010.]

The objective of this formal consultation is to determine if the BSAI groundfish fisheries, GOA groundfish fisheries, and State of Alaska parallel groundfish fisheries, as implemented under their respective FMPs and State management regulations, are likely to jeopardize the continued existence of listed species and/or are likely to destroy or adversely modify designated critical habitat. Based on the directives of the ESA and implementing regulations, as well as court findings with respect to previous opinions, the scope of this consultation and resulting opinion are intended to be comprehensive. Through this consultation, NMFS has considered not only the effects of the fisheries themselves, but also the overall management framework as established under the respective FMPs. It is NMFS' intent to determine if that management framework includes sufficient conservation and management measures to ensure the protection of listed species and their critical habitat.

A growing body of literature and scientific studies has explored the various factors that may have contributed to the decline of Steller sea lions in recent years. Many of these factors (such as predation, environmental variability, disease, entanglement, etc.) are described in the Revised Steller Sea Lion Recovery Plan (Recovery Plan) dated March 2008 (cited as NMFS 2008a in this Biological Opinion). The purpose of this Biological Opinion is to evaluate the Effects of the Action (Chapter 5; federal and parallel groundfish fisheries) on Steller sea lions and other affected species and their critical habitat, including direct and indirect effects of fishing. Other past impacts are considered as part of the Environmental Baseline (Chapter 4), and future effects are discussed in Cumulative Effects (Chapter 6).

If NMFS determines that the action under consideration is likely to jeopardize the continued existence of an ESA-listed species or destroy or adversely modify designated critical habitat, reasonable and prudent alternatives must be identified for the action that will enable the action agency to avoid jeopardy and destruction or adverse modification of critical habitat and meet other regulatory requirements (50 CFR Part 402.02).

NMFS developed this Biological Opinion after reviewing information provided in the BA, the June 2004 Final Programmatic Supplemental Environmental Impact Statement (PSEIS) for the Alaska groundfish fisheries, previous Biological Opinions and National Environmental Policy Act (NEPA) documents for SFD and the North Pacific Fishery Management Council (NPFMC) actions, the 2008 final revised Steller Sea Lion Recovery Plan, and the best available data, such as published and unpublished information on the biology and ecology of listed species in the action area, the history of fisheries in the action area, published and unpublished information on fishing efforts and fisheries management, published and unpublished information on the ecosystems in which the action may occur, and published and unpublished information on human activities in the action area, relevant to the environmental baseline and potential cumulative effects. A complete administrative record of this consultation is on file at NMFS Alaska Regional Office (AKR).

1.2 Consultation History

A history of recent, relevant consultations and actions leading up to this Biological Opinion is presented below.

January 26, 1996 Biological Opinions on the FMPs for the BSAI Groundfish Fishery and the GOA Groundfish Fishery, the proposed 1996 TAC Specifications and their effects on Steller Sea Lions. These opinions concluded that the BSAI and GOA FMPs, fisheries, and harvests under the proposed 1996 TAC specifications were not likely to jeopardize the continued existence of Steller sea lions or to result in the destruction or adverse modification of their critical habitat. With respect to these opinions, the agency also concluded that the reasons for the decline of Steller sea lion populations and the possible role of the fisheries in the decline remain poorly understood.

December 3, 1998 Biological Opinion on authorization of the BSAI Atka mackerel fishery, BSAI pollock fishery, and GOA pollock fishery under their respective FMPs for the period from 1999 to 2002. The opinion concluded that the Atka mackerel fishery was not likely to jeopardize the western population of Steller sea lion or adversely modify its critical habitat, but that the pollock fisheries were likely to cause jeopardy and adverse modification. These conclusions and the reasonable and prudent alternatives (RPAs) developed for the pollock fisheries were challenged in court; the conclusions were upheld, but the RPAs were found arbitrary and capricious for lack of sufficient information. The court ordered preparation of revised final reasonable and prudent alternatives (RFRPAs), which were issued by NMFS on October 15, 1999 and were implemented for the 2000 fisheries.

December 22, 1998 Biological Opinion on authorization of the BSAI and GOA groundfish fisheries based on TAC specifications recommended by the Council for 1999. The opinion concluded that based on the 1999 TAC specifications, the groundfish fisheries were not likely to cause jeopardy or adverse modification for listed species or their critical habitat. The opinion was challenged in court and subsequently found to be arbitrary and capricious for failing to include a sufficiently comprehensive analysis of the groundfish fisheries and their individual, combined, and cumulative effects. Based on this finding, the court determined that NMFS was out of compliance with the ESA (Green Peace v. National Marine Fisheries Service, 80 F. Supp. 2d 1137 [WD. Wash. 2000]).

December 23, 1999 Biological Opinion on authorization of the BSAI and GOA groundfish fisheries based on TAC specifications recommended by the Council for 2000, and on authorization of the fisheries based on statutes, regulations, and management measures to implement the American Fisheries Act of 1998 (AFA). The opinion concluded that based on the 2000 TAC specifications and implementation of the AFA, the groundfish fisheries would not cause jeopardy or adverse modification for listed species or their critical habitat. The opinion was not challenged in court.

November 30, 2000 Biological Opinion (FMP Biological Opinion) on authorization of groundfish fisheries in the BSAI under the FMP for the BSAI Groundfish Fisheries, and the authorization of groundfish fisheries in the GOA under the FMP for Groundfish Fisheries of the GOA. The opinion was comprehensive in scope and considered the fisheries and the overall management framework established by the respective FMPs to determine whether that framework contained necessary measures to ensure the protection of listed species and their critical habitat. The Biological Opinion determined that the BSAI or GOA groundfish fisheries, as implemented under the respective FMPs, jeopardize the continued existence of the western and eastern populations of Steller sea lions and adversely modified their critical habitat. The Biological Opinion provided an RPA which was partially implemented in 2001. Full implementation of the RPA was scheduled for 2002; however, the action considered in the 2001 Biological Opinion described below took the place of that RPA.

In January 2001, an RPA committee, comprised of members of the fishing community, the conservation community, NMFS, State agencies and the Council's Scientific and Statistical Committee (SSC), was formed to develop an alternative RPA.

October 19, 2001 Biological Opinion on Authorization of the BSAI and GOA groundfish fisheries under their respective FMPs, specifically the Pacific cod, pollock, and Atka mackerel fisheries and the parallel fisheries for Pacific cod, pollock, and Atka mackerel as authorized by the State of Alaska within 3 nm of shore. In July of 2001, the action agency (SFD) proposed an alternative RPA developed by the RPA committee to replace the components of the original FMP action that had resulted in the jeopardy and adverse modification finding in the 2000 FMP-level consultation. In 2001, NMFS prepared a project level Biological Opinion (NMFS 2001) which reviewed the revised action and determined that it was not likely to jeopardize or adversely modify critical habitat. This 2001 Biological

Opinion evaluated the direct and indirect effects of that proposed action on Steller sea lions and their critical habitat, together with the effects of other activities that are interrelated or interdependent with that action. These effects were considered in the context of an Environmental Baseline and Cumulative Effects. State waters, so-called "parallel fisheries", were also included in this Biological Opinion in part because of their intricate connection with the federal fisheries being considered, and also due to the State of Alaska's request to formally include this fishery in the consultation. This was re-iterated by the State in a comment dated September 12, 2001 (from Frank Rue, Commissioner, Alaska Department of Fish and Game [ADF&G]). This opinion determined that the action was not likely to jeopardize or adversely modify critical habitat.

In the 2001 Biological Opinion (2001:8) NMFS specified that:

> "…the FMP level biological opinion will remain in effect as NMFS' coverage at the plan level, and this opinion (the 2001 opinion) will address the project level effects on listed species that would be likely to occur if the Council's preferred action were implemented."

The court reviewed the 2001 Biological Opinion and found that it was arbitrary and capricious and remanded the opinion back to NMFS for revision.

June 19, 2003 Supplement to the 2001 Biological Opinion on Authorization of the BSAI and GOA groundfish fisheries under their respective FMPs, specifically the Pacific cod, pollock, and Atka mackerel fisheries and the parallel fisheries for Pacific cod, pollock, and Atka mackerel as authorized by the State of Alaska within 3 nm of shore. In response to the court order remanding the 2001 Biological Opinion back to NMFS, NMFS prepared a supplement (NMFS 2003) to the 2001 Biological Opinion (NMFS 2001), which affirmed NMFS' conclusions that the revised FMP actions were not likely to jeopardize ESA-listed species or adversely modify their critical habitat. NMFS presented background information on the decision making process in the 2001 Biological Opinion as a requirement of the court order.

March 9, 2006 Biological Opinion on the issuance of an exempted fishing permit (EFP) to support a feasibility study using commercial fishing vessels for acoustic surveys of pollock in the Aleutian Islands subarea. Formal consultation was initiated on January 17, 2006. The opinion evaluated the potential effects of a permit that authorized the harvest of pollock inside designated critical habitat. NMFS determined that the action would not jeopardize listed species or adversely modify their critical habitat.

On October 18, 2005, the NPFMC (also referred to as the Council) requested that NMFS SFD reinitiate consultation on the BSAI and GOA FMPs. The Council's request was based on the recognition that a substantial amount of new research on Steller sea lions had been published since NMFS completed the 2000 Biological Opinion, such that an evaluation of the FMPs in light of that new information would be prudent. NMFS agreed and on April 19, 2006 SFD requested re-initiation of formal ESA section 7 consultation on the Alaska Groundfish FMPs and the State of Alaska parallel groundfish fisheries.

On March 7, 2006, the NMFS notified the ADF&G that the NPFMC had recommended that NMFS reinitiate formal consultation on the effects of the FMPs for groundfish under Section 7 of the ESA. NMFS requested that the State respond to the letter regarding its intention to participate in the consultation to allow the State an opportunity that ensure that any decisions in the opinion that may affect state fisheries are based on the most recent information regarding the action and potential effects.

On March 31, 2006, The State of Alaska (ADF&G) notified the NMFS of its desire to participate in the consultation and to have the State parallel groundfish fisheries included in the consultation.

On April 19, 2006, SFD sent PRD a written request to re-initiate formal ESA section 7 consultation on the Alaska Groundfish FMPs. The request specified that the purpose of this reinitiation was to evaluate the effects of current fisheries actions and management measures on listed species, in light of any new information gained since completion of the previous consultations in 2000 and 2001. The request for re-initiation was accompanied by a biological assessment (BA). PRD concurred with this request and re-initiated formal consultation on the effects of the fisheries on the eastern DPS of Steller sea lions and its designated critical habitat, the western DPS of Steller sea lions and its designated critical habitat, humpback whales and sperm whales on June 21, 2006.[2]

Because of the complexity of the analysis, the consulting agency and the action agency agreed to an extended timeline beyond the statutory deadline of 135 days for completion of the consultation.

On March 18, 2008, SFD provided PRD with a listing and description of all FMP amendments and regulatory amendments for the Alaska groundfish fisheries since April 2006.

On May 1, 2008, NMFS informed the NPFMC that the agency required additional time to complete the analyses for this opinion, and thus that its release would be delayed beyond the anticipated May 7, 2008 release date.

On June 11, 2008, the NPFMC requested that NMFS provide a new schedule and timeline for completion of the Biological Opinion, NEPA documents, and milestones for NMFS' interactions with the Council and its Steller Sea Lion Mitigation Committee.

On September 18, 2008, NMFS provided new schedules and timelines for completion of the Biological Opinion, for the NEPA process, and for completion of NEPA documents to the NPFMC. NMFS provided two timelines related to these processes: one in the case of no-jeopardy, no destruction/adverse modification conclusion in the opinion and one in the case of a jeopardy and/or a destruction/adverse modification of critical habitat conclusion. In the letter to the Council, NMFS also noted that, in the event that the "…draft opinion concluded jeopardy or adverse modification of critical habitat, NMFS has built into the schedule…anticipated collaboration with the Council on specific mitigation measures for the Reasonable and Prudent Alternatives…"

On April 9, 2009, following receipt of a letter from the Marine Conservation Alliance (MCA), public comment, and its concurrence with issues raised in the MCA letter, the Council requested that NMFS incorporate specific new information, including information which had yet to be collected, into the draft Biological Opinion before its release. The new information which the Council recommended be included in the opinion was: 1) information from Steller sea lion pup surveys that were scheduled to be conducted in the summer of 2009; 2) information from Steller sea lion non-pup surveys that were to be conducted in the summer of 2009; and 3) new information on reproductive rates, with reference to a recently released report from researchers at the Alaska SeaLife Center. The Council wrote that it believed "…strongly that it is critical to include this new information in the Biological Opinion as it has the potential to significantly affect the findings within that BiOp…The Council recognizes that incorporation of this

---

[2] Subsequent to reinitiating section 7 consultation on the Alaska groundfish fisheries, a fin whale was killed incidental to the Bering Sea/Aleutian Islands pollock trawl fishery. Due to this incidental mortality, NMFS determined that the groundfish fisheries may adversely affect the Northeast Pacific stock of fin whales and thus, included this stock to this Biological Opinion to determine whether or not effects from the fisheries were likely to jeopardize the continued existence of this stock.

information could potentially delay release of the draft BiOp…" and it requested that NMFS provide the Council with its best indication of how the Biological Opinion schedule might be affected.

On May 6, 2009, NMFS (2009b) notified the Council that NMFS had concluded that the new surveys could potentially reduce uncertainty and controversy about important elements in the analysis in the opinion, and thus, that NMFS was extending the schedule for release of the opinion to allow for incorporation of this information. NMFS informed the Council that it planned to have the opinion available for Council consideration in March of 2010. However, because of the timeframe needed to finalize a formal review from NOAA leadership, the release was delayed until August 2010, at which time a draft Biological Opinion was released.

In August 2010, the Council convened a special meeting to review the draft Biological Opinion and to provide opportunity for public comment. Concurrent with Council review, NMFS offered the public an opportunity to submit written comments. In September and October 2010, NMFS reviewed public comments and the Council's suggestion for an alternative Reasonable and Prudent Alternative, and continued the consultation process while evaluating these comments. As a consequence of this public and Council review process, additional analyses of comments were completed and the results of these analyses as well as new information were incorporated into the Biological Opinion. This final Biological Opinion is the culmination of the above-described consultation process.

1.3 Presentation of the Analysis in this Opinion

Biological Opinions are constructed in sections that meet specific requirements imposed by the ESA and implementing regulations. These sections contain different portions of the overall analytical approach. This section is intended as a basic guide to the reader of the other sections of this Biological Opinion by describing the analyses that can be found in each section.

Description of the Proposed Action – This section contains a basic summary of the proposed Federal action and any interrelated and interdependent actions. This description forms the basis of the first step in the analysis where we consider the various elements of the action and determine the stressors expected to result from those elements. The nature, timing, duration, and location of those stressors define the action area and provide the basis for our exposure analyses.

Status of the Species – This section provides the reference condition for the species and critical habitat at the listing and designation scale. These reference conditions form the basis for the determinations of whether or not the proposed action is likely to jeopardize the species or result in the destruction or adverse modification of critical habitat. Other key analyses presented in this section include critical information on the biological and ecological requirements of the species and critical habitat and the impacts to species and critical habitat from existing stressors.

Environmental Baseline – This section provides the reference condition for the species and critical habitat within the action area. By regulation, the baseline includes the impacts of past and present future actions (except the effects of the proposed action) on the species and critical habitat. In this Biological Opinion, some of this analysis is contained within the *Effects of the Proposed Action* section because the proposed action is a continuation of the on-going action (i.e., the baseline) of prosecuting Federal and parallel fisheries in Alaska. This section also contains summaries of the impacts from stressors that will be ongoing in the same areas and times as the effects of the proposed action (future baseline). This information forms part of the foundation of our exposure, response, and risk analyses.

Effects of the Proposed Action – This section details the results of the exposure, response, and risk analyses NMFS conducted for listed species and elements, functions, and areas of critical habitat.

Cumulative Effects – This section summarizes the impacts of future State, tribal, local, or private actions, not involving Federal activities, that are reasonably certain to occur in the action area considered in this Biological Opinion. Similar to the rest of the analysis, if cumulative effects are expected, NMFS determines the exposure, response, and risk posed to individuals of the species and features of critical habitat.

Integration and Synthesis of Effects – In this section of the Biological Opinion, NMFS presents the summary of the effects identified in the preceding sections and then details the consequences of the risks posed to individuals and features of critical habitat to the species or DPS at issue. Finally, this section concludes whether the proposed action may result in jeopardy to the continued existence of a species or the destruction or adverse modification of designated critical habitat.

2 DESCRIPTION OF THE PROPOSED ACTION

NMFS SFD, under the authority of the Magnuson-Stevens Fishery Conservation and Management Act (MSA), and the State of Alaska propose to: (1) authorize groundfish fisheries in the BSAI under the FMP for Groundfish of the BSAI management area; (2) authorize groundfish fisheries in the GOA under the FMP for Groundfish of the GOA; (3) authorize parallel fisheries within State-managed waters. Management of groundfish fisheries within the Exclusive Economic Zone (EEZ) off Alaska and parallel fisheries in adjacent State waters is a continuing activity. As noted in Section 1, the action evaluated in this opinion includes not only the fisheries covered under the FMPs but also the overall management framework under which the fisheries are promulgated. Thus, in the current section, we describe both levels of this action.

The purpose of this section is to provide a description of the MSA, the two FMPs for Alaska groundfish fisheries, and management of State parallel groundfish fisheries sufficient to: a) permit NMFS to fully evaluate the potential effects of the action on listed species and designated critical habitats and b) summarize enough information for the reader to understand and evaluate the action under consideration in this Biological Opinion. In order to do so, we first deconstruct this complicated action into its component parts (Figure 2.1). This deconstruction yields four main groups of interrelated activities which comprise the proposed action:

- fisheries management policy;
- exploitation strategy;
- annual fisheries assessment; and
- implementation of the fisheries.

This break-down in the description of the action permits examination of how parts of this action inter-relate. It facilitates an evaluation (in later chapters) of the effects of various aspects of this complicated action on listed species and designated critical habitat. For example, while policy on its own does not result in the removal of fish, it does set up limitations and expectations for removals. It sets in place a structure and process that affects how decisions about allowable levels and patterns of fishery removals are made in the face of differing sources and levels of information and uncertainty. Fishery management policy also affects fishery assessment practices and methodologies, as well as the treatment and reporting of components of assessments, once they are completed. Thus, while at first glance, the pattern and level of fishery removals occurring in different groundfish fisheries may seem independent, they are affected by, and highly inter-related with, policy choices, decisions about exploitation strategies, and stock and stock complex assessments that were made or accomplished earlier.

The entire action being evaluated (as described above) is an ongoing action that is undergoing a reinitiated consultation under Section 7 of the ESA due to the availability of new information. Thus, as consultation is occurring, the action is proceeding. We are evaluating the potential effects of this action were it to continue, with its current management policy, exploitation strategy, fisheries assessment, and fisheries prosecution into the foreseeable future.

Fishery management policy, assessment, and implementation are dynamic processes. Ideally, they are responsive to new information. Changes have occurred to the proposed action since the initiation of consultation. Due to the need to complete this Biological Opinion in a timely manner, we have included all information about the action provided to us by the action agency through January 2010.

The current (December 2009) BSAI and GOA groundfish FMPs are available on the NPFMC website http://www.fakr.noaa.gov/npfmc/default.htm. In order to understand the proposed action we have reviewed these FMPs, the MSA itself, the Goodman *et al.* (2002) report (see description below) and associated Alaska Fisheries Science Center (AFSC) response documents, the PSEIS for the Alaska groundfish fisheries (NMFS 2004), relevant Stock Assessments and Fishery Evaluation (SAFE) reports from 2002-2009 (available at http://www.fakr.noaa.gov/npfmc/SAFE/SAFE.htm), State of Alaska groundfish fisheries management documents, as well as other papers, reports, and regulatory documents cited throughout this opinion.

Large portions of some of these documents (e.g., the FMPs) are excerpted in this chapter. The Goodman *et al.* (2002) independent scientific review was prepared, at the request of the NPFMC, in part to "…develop an educational primer on the Council's current procedure" (Goodman *et al.* 2002:11). In their review of the Goodman *et al.* (2002) report, the AFSC (2002:2) summarized that "While there are a number of specific points to which some objection could be made, for the most part, the AFSC agrees with the Panel's depiction of our current harvest system…(M)uch of the material is presented in the form of an introductory course…". Thus, it is easy to understand. However, the AFSC (2002:2-3) cautioned that "…one disadvantage is that…the authors occasionally overstate things or otherwise make conclusions sound more general than they really are. Readers should be cautioned that some of the material in the report is best viewed as an introduction to the subject, not as the final word…" on it.

In addition to review of written documents, we have also gained insight into this action through discussions with fishery managers and assessment specialists, as well as through discussions during recent Groundfish Plan Team Meetings and NPFMC meetings.

2.1 Purpose

As described by SFD, there are two fundamental and interrelated purposes underlying the management of the Alaska groundfish fisheries: to maximize the social and economic benefits of the groundfish resources to the people of the United States (U.S.) and to conserve the resource to ensure its sustained availability to current and future generations. The use and conservation of the fisheries need to be managed so that one objective—whether related to biological conservation or to socioeconomic well-being—does not take priority over the other, except when the resource itself is at risk of being depleted. NMFS fisheries management strives to balance these two fundamental objectives to prevent such depletion of the resource.

2.2 Fisheries Management Policy

When considering the impacts of commercial groundfish fisheries, one must investigate which species of fish are harvested, how much, when, and in what manner harvest occurred. Yet, in order for this ultimate effect (fishing) to occur, a long series of guiding documents has been prepared, interpreted, and their policies implemented. In this section, we focus on the first major area: fisheries management policy.
As summarized by Goodman *et al.* (2002:16):

> "Fisheries are complex dynamic systems, involving physical, biological and human dimensions. Within those dimensions, innumerable elements inter-relate and change through time. Observing those elements and understanding the relationships between them is difficult, and being able to predict the fate of all these elements accurately is impossible.

> And yet, despite this complexity and limited predictability, the goal of fisheries management is, as far as possible, to make sense of the various dimensions and elements, and to make decisions on alternative policies in the face of sometimes high levels of uncertainty but based on the best available information."

2.2.1 The MSA and Other Applicable Law

The MSA (16 USC § 1851) is the primary domestic legislation governing management of marine fishing activities in federal waters (all marine waters from 3 nautical miles offshore the coast of Alaska or its baseline to 200 nautical miles offshore, to the 1990 United States/Russia maritime boundary line and to the United States/Canada maritime boundary). This area became known as the EEZ in 1983. First passed in 1976, the MSA was reauthorized in 1996 by the U.S. Congress to include, among other things, a new emphasis on the precautionary approach in U.S. fishery management policy. The MSA contains 10 National Standards, with which all FMPs must conform and which guide fishery management. The Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006 (MSRA) amended the MSA to include requirements for annual catch limits (ACLs), accountability measures (AMs), and other provisions related to both ending and preventing overfishing as well as rebuilding fisheries. To incorporate these new requirements into current National Standard 1 (NS1) guidance, NMFS initiated a revision of the NS1 guidelines in 50 CFR 600.310. In January 2009 (74 FR 3178), NMFS amended the guidelines for the NS1 of the MSA to: 1) provide guidance on how to comply with the new annual catch limit (ACL) and accountability measure (AM) requirements for ending overfishing of fisheries managed by FMPs; and 2) clarify the relationship between ACLs, acceptable biological catches (ABCs), maximum sustainable yield (MSY), optimum yield (OY), and other applicable reference points. In the final rule, NMFS stated that this action was necessary to facilitate compliance with MSA requirements to rebuild overfished stocks, achieve OY, and to end and prevent overfishing.

Besides the MSA, U.S. fisheries management must be consistent with the requirements of other regulations including the Marine Mammal Protection Act (MMPA), the ESA, the Migratory Bird Treaty Act, and other Federal laws.

The MSA created eight regional fishery management councils that are primarily charged with preparing fishery management plans and plan amendments. The Councils are authorized to prepare and submit to the Secretary of Commerce (Secretary) for approval, disapproval or partial approval, FMPs and any necessary amendments, for each fishery under their authority that require conservation and management. The Councils conduct public meetings so as to allow all interested persons an opportunity to be heard in the development of FMPs and amendments, and review and revise, as appropriate, the assessments and specifications with respect to the optimum yield (OY) from each fishery (16 USC 1852(h)).

To date, the NPFMC has prepared, and NMFS has approved and implemented, 6 FMPs, most now with numerous amendments. These FMPs not only must comply with the MSA, but with the requirements of other federal laws, such as the ESA.

The MSA contains provisions for taking into account the requirements of other laws, as well as provisions related to the protection of marine ecosystems and the environment, some of which are contained in the definitions of OY and "conservation and management":

In the MSA, the term "optimum", with respect to the yield from a fishery, means the amount of fish which–

> (A) will provide the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, taking into account the protection of marine ecosystems;

> (B) is prescribed as such on the basis of the MSY from the fishery, as reduced by any relevant economic, social, or ecological factor; and
> (C) in the case of an overfished fishery, provides for rebuilding to a level consistent with producing the MSY of such fishery" (16 USC § 1802(3)(28)).

The term "conservation and management" refers to all of the rules, regulations, conditions, methods, and other measures: (A) which are required to rebuild, restore, or maintain, and which are useful in rebuilding, restoring, or maintaining, any fishery resources and the marine environment; and (B) which are designed to assure that–

> (i) a supply of food and other products may be taken, and that recreational benefits may be obtained, on a continuing basis;
> (ii) irreversible or long-term adverse effects on fishery resources and the marine environment are avoided; and
> (iii) there will be a multiplicity of options available with respect to future uses of these resources" (16 USC § 1802(3)(5)) (emphasis added).

Fishery Management Councils have considerable autonomy but most prepare FMPs, create regulations, and generally make decisions that are consistent with the provisions of the MSA (Goodman *et al.* 2002). Section 301(a) of the MSA sets forth national standards for conservation and management with which FMPs and regulations must be consistent. In addition, NMFS established 10 National Standard Guidelines to assist in the development and review of FMPs, amendments, and regulations prepared by the Fishery Management Councils and the Secretary (50 CFR 600 Subpart D). The National Standards are as follows:

1. Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the OY from each fishery for the U.S. fishing industry.
2. Conservation and management measures shall be based upon the best scientific information available.
3. To the extent practicable, an individual stock of fish shall be managed as a unit throughout its range, and interrelated stocks of fish shall be managed as a unit or in close coordination.
4. Conservation and management measures shall not discriminate between residents of different states. If it becomes necessary to allocate or assign fishing privileges among various U.S. fishermen, such allocation shall be A) fair and equitable to all such fishermen; B) reasonably calculated to promote conservation; and C) carried out in such manner that no particular individual, corporation, or other entity acquires an excessive share of such privileges.
5. Conservation and management measures shall, where practicable, consider efficiency in the utilization of fishery resources; except that no such measure shall have economic allocation as its sole purpose.
6. Conservation and management measures shall take into account and allow for variations among, and contingencies in, fisheries, fishery resources, and catches.
7. Conservation and management measures shall, where practicable, minimize costs and avoid unnecessary duplication.
8. Conservation and management measures shall, consistent with the conservation requirements of this Act (including the prevention of overfishing and rebuilding of overfished stocks), take into account the importance of fishery resources to fishing communities in order to A) provide for the sustained participation of such communities, and B) to the extent practicable, minimize adverse economic impacts on such communities.
9. Conservation and management measures shall, to the extent practicable, A) minimize bycatch and B) to the extent bycatch cannot be avoided, minimize the mortality of such bycatch.
10. Conservation and management measures shall, to the extent practicable, promote the safety of human life at sea.

National Standard 1 is undoubtedly the most influential in decisions made by the Council on fisheries management. National Standard 2 ensures that science plays a key role in determining how fisheries are prosecuted (Goodman *et al.* 2002). As noted above, to facilitate compliance with MSA requirements to rebuild overfished stocks, achieve OY, and to end and prevent overfishing, NMFS (2009a) amended the guidelines for the National Standard 1 in January 2009 (74 FR 3178). The main features of these guidelines including the setting of harvest limits and ensuring management measures are implemented to prevent overfishing are modeled after the groundfish fishery management practices used by the NPFMC.

These guidelines: 1) provide guidance on how to comply with the ACL and AM requirements for ending overfishing of fisheries managed by FMPs; and 2) clarify the relationship between ACLs, ABCs, MSY, OY, and other applicable reference points. According to NMFS (2009a), some of the major items covered in the proposed NS1 guidelines were:

1) "A description of the relationship between MSY, OY, overfishing limits (OFL), ABC, ACLs, and annual catch targets (ACT);
2) guidance on how to combine the use of ACLs and AMs for a stock to prevent overfishing when possible, and adjust ACLs and AMs, if an ACL is exceeded;
3) statutory exceptions to requirements for ACLs and AMs and flexibility in application of NS1 guidelines;
4) ‘‘stocks in the fishery’’ and ‘‘ecosystem component species’’ classifications;
5) replacement of MSY control rules with ABC control rules and replacement of OY control rules with ACT control rules;
6) new requirements for scientific and statistical committees (SSC);
7) explanation of the timeline to prepare new rebuilding plans;
8) revised guidance on how to establish rebuilding time targets;
9) advice on action to take at the end of a rebuilding period if a stock is not yet rebuilt; and
10) exceptions to the requirements to prevent overfishing".

The main substantive change made to the proposed action in the final rule pertained to ACTs. The final action retains the concept of an ACT and an ACT control rule, but does not require them to be included in FMPs. After taking public comment NMFS decided that ACTs are better addressed as AMs. Several components of these changes, including but not limited to guidance on how it classify "stocks in the fishery" and "ecosystem component species", may have important impacts on Steller sea lions and their critical habitat. However, as noted above, the timing of notification from NMFS SFD to NMFS PRD of notice and evaluation of the effects of these changes was too late to allow us to adequately evaluate or incorporate these guidance changes into our analysis.

The NPFMC is scheduled in February 2010, for initial review of the analysis for Amendments 96 and 87 to the BSAI and GOA groundfish FMPs, respectively, which would incorporate provisions of the National Standard 1 guidelines into the FMPs. Implementation of the amendments is scheduled for January 1, 2011, and any ESA consultation on the amendments would be completed before this date.

2.2.2 The FMPs and Implementing Regulations

The FMPs govern groundfish fisheries of the Gulf of Alaska and Bering Sea and Aleutian Islands Management Areas (NPFMC 2009a and 2009b). Coverage of species and locations of fisheries under these plans are detailed in Section 3.1 of the BSAI and GOA FMPs.

The BSAI groundfish FMP was approved by the Secretary on October 27, 1979, and implemented by regulations published on December 31, 1981 (46 FR 63295, corrected January 28, 1982, 47 FR 4083;

NPFMC 2009a). As of November 2009, it has been amended over 90 times, and its focus has changed from the regulation of mainly foreign fisheries to the management of fully domestic groundfish fisheries. The geographical extent of the BSAI groundfish FMP management unit is the U.S. EEZ of the Bering Sea, including Bristol Bay and Norton Sound, and that portion of the North Pacific Ocean adjacent to the Aleutian Islands which is between 170° W. longitude and the U.S.-Russian Convention Line of 1867 (Figure 2.2). The BSAI groundfish FMP covers fisheries for all stocks of finfish and marine invertebrates except salmonids, shrimps, scallops, snails, king crab, Tanner crab, Dungeness crab, corals, surf clams, horsehair crab, lyre crab, Pacific halibut, and Pacific herring. The BSAI FMP was revised in January 2005 to incorporate previous amendments and to better organize the document. The most recent (December 2009) version of the BSAI FMP is available from the Council's website at http://alaskafisheries.noaa.gov/npfmc/fmp/bsai/BSAI.pdf.

The GOA groundfish FMP was approved by the Secretary on February 24, 1978, and implemented by regulations published on November 14, 1978 (44 FR 52709; NPFMC 2009c). Since that time, it has been amended over 80 times, and its focus has changed from the regulation of mainly foreign fisheries to the management of fully domestic groundfish fisheries. The geographical extent of the GOA FMP management unit is the U.S. EEZ of the North Pacific Ocean, exclusive of the Bering Sea, between the eastern Aleutian Islands at 170° W. longitude and Dixon Entrance at 132°40' W. longitude (Figure 2.2). The GOA groundfish FMP covers fisheries for all stocks of finfish except salmon, steelhead, Pacific halibut, Pacific herring, and tuna. In terms of both the fishery and the groundfish resource, the GOA groundfish fishery forms a distinct management unit. The GOA groundfish FMP was revised in January 2005 to incorporate previous amendments and to better organize the document. The December 2009 version of the GOA FMP is available from the Council's website at http://alaskafisheries.noaa.gov/npfmc/fmp/goa/GOA.pdf.

2.2.2.1 Objectives of the FMPs

The history of fishery development, target species and species composition of the commercial catch, bathymetry, and oceanography differ between the GOA and the adjacent BSAI management area. Although many species occur over a broader range than the BSAI or GOA management areas, with only a few exceptions (e.g., sablefish), stocks of common species in each management area are believed to be different from those in the other management area. Each FMP contains management policies and measures for the groundfish fisheries occurring in the management area. These policies and measures are explained in detail in the BA (Section 2.2.2 and 2.2.3). They are subject to annual review by the NPFMC (NPFMC 2009:4) to enable adaptive management. The management objectives given in Section 2.2.1 of the 2009 BSAI FMP are listed below.

Prevent Overfishing:

1. Adopt conservative harvest levels for multi-species and single species fisheries and specify optimum yield.
2. Continue to use the 2 million mt OY cap for the BSAI groundfish fisheries.
3. Provide for adaptive management by continuing to specify optimum yield as a range.
4. Provide for periodic reviews of the adequacy of F_{40} and adopt improvements, as appropriate.
5. Continue to improve the management of species through species categories.

Promote Sustainable Fisheries and Communities:

6. Promote conservation while providing for OY in terms of the greatest overall benefit to the Nation with particular reference to food production, and sustainable opportunities for recreational, subsistence, and commercial fishing participants and fishing communities.
7. Promote management measures that, while meeting conservation objectives, are also designed to avoid significant disruption of existing social and economic structures.

8. Promote fair and equitable allocation of identified available resources in a manner such that no particular sector, group or entity acquires an excessive share of the privileges.
9. Promote increased safety at sea.

Preserve Food Web:

10. Develop indices of ecosystem health as targets for management.
11. Improve the procedure to adjust acceptable biological catch levels as necessary to account for uncertainty and ecosystem factors.
12. Continue to protect the integrity of the food web through limits on harvest of forage species.
13. Incorporate ecosystem-based considerations into fishery management decisions, as appropriate.

Manage Incidental Catch and Reduce Bycatch and Waste:

14. Continue and improve current incidental catch and bycatch management program.
15. Develop incentive programs for bycatch reduction including the development of mechanisms to facilitate the formation of bycatch pools, vessel bycatch allowances, or other bycatch incentive systems.
16. Encourage research programs to evaluate current population estimates for non-target species with a view to setting appropriate bycatch limits, as information becomes available.
17. Continue program to reduce discards by developing management measures that encourage the use of gear and fishing techniques that reduce bycatch which includes economic discards.
18. Continue to manage incidental catch and bycatch through seasonal distribution of total allowable catch and geographical gear restrictions.
19. Continue to account for bycatch mortality in total allowable catch accounting and improve the accuracy of mortality assessments for target, prohibited species catch, and noncommercial species.
20. Control the bycatch of prohibited species through prohibited species catch limits or other appropriate measures.
21. Reduce waste to biologically and socially acceptable levels.
22. Continue to improve the retention of groundfish where practicable, through establishment of minimum groundfish retention standards.

Avoid Impacts to Seabirds and Marine Mammals:

23. Continue to cooperate with USFWS to protect ESA-listed species, and if appropriate and practicable, other seabird species.
24. Maintain or adjust current protection measures as appropriate to avoid jeopardy of extinction or adverse modification to critical habitat for ESA-listed Steller sea lions.
25. Encourage programs to review status of endangered or threatened marine mammal stocks and fishing interactions and develop fishery management measures as appropriate.
26. Continue to cooperate with NMFS and USFWS to protect ESA-listed marine mammal species, and if appropriate and practicable, other marine mammal species.

Reduce and Avoid Impacts to Habitat:

27. Review and evaluate efficacy of existing habitat protection measures for managed species.
28. Identify and designate essential fish habitat and habitat areas of particular concern pursuant to MSA rules, and mitigate fishery impacts as necessary and practicable to continue the sustainability of managed species.
29. Develop a Marine Protected Area policy in coordination with national and state policies.
30. Encourage development of a research program to identify regional baseline habitat information and mapping, subject to funding and staff availability.

31. Develop goals, objectives and criteria to evaluate the efficacy and suitable design of Marine Protected Areas and no-take marine reserves as tools to maintain abundance, diversity, and productivity. Implement marine protected areas if and where appropriate.

Promote Equitable and Efficient Use of Fishery Resources:

32. Provide economic and community stability to harvesting and processing sectors through fair allocation of fishery resources.
33. Maintain the license limitation program, modified as necessary, and further decrease excess fishing capacity and overcapitalization by eliminating latent licenses and extending programs such as community or rights-based management to some or all groundfish fisheries.
34. Provide for adaptive management by periodically evaluating the effectiveness of rationalization programs and the allocation of access rights based on performance.
35. Develop management measures that, when practicable, consider the efficient use of fishery resources taking into account the interest of harvesters, processors, and communities.

Increase Alaska Native Consultation:

36. Continue to incorporate local and traditional knowledge in fishery management.
37. Consider ways to enhance collection of local and traditional knowledge from communities, and incorporate such knowledge in fishery management where appropriate.
38. Increase Alaska Native participation and consultation in fishery management.

Improve Data Quality, Monitoring and Enforcement:

39. Increase the utility of groundfish fishery observer data for the conservation and management of living marine resources.
40. Develop funding mechanisms that achieve equitable costs to the industry for implementation of the North Pacific Groundfish Observer Program.
41. Improve community and regional economic impact costs and benefits through increased data reporting requirements.
42. Increase the quality of monitoring and enforcement data through improved technology.
43. Encourage a coordinated, long-term ecosystem monitoring program to collect baseline information and compile existing information from a variety of ongoing research initiatives, subject to funding and staff availability.
44. Cooperate with research institutions such as the North Pacific Research Board (NPRB) in identifying research needs to address pressing fishery issues.
45. Promote enhanced enforceability.
46. Continue to cooperate and coordinate management and enforcement programs with the Alaska Board of Fish, ADF&G, and Alaska Fish and Wildlife Protection, the U.S. Coast Guard (USCG), NMFS Enforcement, International Pacific Halibut Commission, Federal agencies, and other organizations to meet conservation requirements; promote economically healthy and sustainable fisheries and fishing communities; and maximize efficiencies in management and enforcement programs through continued consultation, coordination, and cooperation

2.2.2.2 Stocks in the GOA

Stocks governed by the GOA groundfish FMP include all finfish, except salmon, steelhead, halibut, herring, and tuna, which are distributed or are exploited in the area described above. Harvest allocations and management are based on the calendar year.

Five categories of species or species groups are likely to be taken in the groundfish fishery. Species may be split or combined within the “target species” category according to procedures set forth in the FMP without amendments to this FMP, notwithstanding the designation listed in the FMP. The optimum yield concept is applied to all except the "prohibited species" category. These categories are described as follows (and in the table below):

1. Prohibited Species – are those species and species groups the catch of which must be avoided while fishing for groundfish, and which must be immediately returned to sea with a minimum of injury except when their retention is authorized by other applicable law. Groundfish species and species groups under the FMP for which the quotas have been achieved shall be treated in the same manner as prohibited species.
2. Target species – are those species that support a single species or species complex (consisting of multiple species) target fishery, are commercially important, and for which a sufficient data base exists that allows each to be managed on its own biological merits. Accordingly, a specific TAC is established annually for each target species or species complex. Catch of each species must be recorded and reported. This category includes walleye pollock, Pacific cod, sablefish, shallow and deep-water flatfish, rex sole, flathead sole, arrowtooth flounder, Pacific ocean perch, shortraker rockfish, rougheye/blackspotted rockfish, northern rockfish, "other slope" rockfish, pelagic shelf rockfish, demersal shelf rockfish, thornyhead rockfish, Atka mackerel, and skates.
3. Other Species – are those species or species groups that currently are of slight economic value and not generally targeted. This category, however, contains species with economic potential or which are important ecosystem components, but insufficient data exist to allow separate management. The "other species" category is managed with a single gulf wide OFL, ABC, and TAC recommended by the NPFMC annually. Catch of this category as a whole must be recorded and reported. The category includes squid, sculpins, sharks, and octopus.
4. Forage fish species – are those species which are a critical food source for many marine mammal, seabird and fish species. The forage fish species category is established to allow for the management of these species in a manner that prevents the development of a commercial directed fishery for forage fish. Management measures for this species category have been specified in regulations and include such measures as a prohibition on directed fishing, limitations on allowable bycatch retention amounts, limitations on the sale, barter, trade or any other commercial exchange, as well as the processing of forage fish in a federally regulated commercial processing facility.
5. Nonspecified species – are those species and species groups of no current economic value taken by the groundfish fishery only as an incidental catch in the target fisheries. Virtually no data exist which would allow population assessments. No record of catch is necessary. The allowable catch for this category is the amount that is taken incidentally while fishing for target and other species, whether retained or discarded.

Groundfish stocks in the GOA (NPFMC 2009b)

Management Group	Species
Prohibited Species[1]	Pacific halibut Pacific herring Pacific salmon Steelhead trout King crab Tanner crab
Target Species[2]	Walleye pollock Pacific cod Sablefish Flatfish (shallow-water flatfish, deep-water flatfish, rex sole, flathead sole, arrowtooth flounder) Rockfish (Pacific ocean perch, northern rockfish, shortraker rockfish, rougheye rockfish, blackspotted rockfish,other slope rockfish, pelagic shelf rockfish, demersal shelf rockfish[3], thornyhead rockfish) Atka mackerel Skates (big and longnose skates, other skates)
Other Species[4]	Squid Sculpins Sharks Octopus
Forage Fish Species[5]	Osmeridae family (eulachon, capelin, and other smelts) Myctophidae family (lanternfishes) Bathylagidae family (deep-sea smelts) Ammodytidae family (Pacific sand lance) Trichodontidae family (Pacific sand fish) Pholidae family (gunnels) Stichaeidae family (pricklebacks, warbonnets, eelblennys, cockscombs, and shannys) Gonostomatidae family (bristlemouths, lightfishes, and anglemouths) Order Euphausiacea (krill)

[1] Must be immediately returned to the sea
[2] TAC for each listing
[3] Management delegated to the State of Alaska
[4] Aggregate TAC for group
[5] Management measures for forage fish are established in regulations implementing the FMP

2.2.2.3 Stocks in the BSAI

Stocks governed by the FMP include all stocks of finfish and marine invertebrates except salmonids, shrimps, scallops, snails, king crab, Tanner crab, Dungeness crab, corals, surf clams, horsehair crab, lyre crab, Pacific halibut, and Pacific herring.

Five categories of species or species groups are likely to be taken in the groundfish fishery. The optimum yield concept is applied to all except the "prohibited species" category. These categories are described as follows (and in the table below):

1. Prohibited Species – are those species and species groups the catch of which must be avoided while fishing for groundfish, and which must be returned to sea with a minimum of injury except when their retention is authorized by other applicable law. Groundfish species and species groups under the FMP for which the quotas have been achieved shall be treated in the same manner as prohibited species.
2. Target species – are those species that support either a single species or species complex (consisting of multiple species) target fishery, are commercially important, and for which a sufficient data base exists that allows each to be managed on its own biological merits. Accordingly, a specific TAC is established annually for each target species. Catch of each species must be recorded and reported. This category includes pollock, Pacific cod, sablefish, yellowfin sole, Greenland turbot, arrowtooth flounder, rock sole, flathead sole, Alaska plaice, "other flatfish", Pacific ocean perch, northern rockfish, shortraker rockfish, rougheye rockfish, "other rockfish", Atka mackerel, and squid.
3. Other Species – are those species or species groups that currently are of slight economic value and not generally targeted. This category, however, contains species with economic potential or which are important ecosystem components, but insufficient data exist to allow separate management. Accordingly, a single TAC applies to this category as a whole. Catch of this category as a whole must be recorded and reported. The category includes sculpins, sharks, skates, and octopus.
4. Forage fish species – are those species which are a critical food source for many marine mammal, seabird and fish species. The forage fish species category is established to allow for the management of these species in a manner that prevents the development of a commercial directed fishery for forage fish. Management measures for this species category have been specified in regulations and include such measures as a prohibition on directed fishing, limitations on allowable bycatch retention amounts, or limitations on the sale, barter, trade or any other commercial exchange, as well as the processing of forage fish in a federally regulated commercial processing facility.
5. Nonspecified species – are those species and species groups of no current economic value taken by the groundfish fishery only as an incidental catch in the target fisheries. Virtually no data exist which would allow population assessments. No record of catch is necessary. The allowable catch for this category is the amount which is taken incidentally while fishing for target and other species, whether retained or discarded.

Groundfish stocks in the BSAI (NPFMC 2009a)

Management Group	Species	
	Finfish	**Marine Invertebrates**
Prohibited Species[1]	Pacific halibut Pacific herring Pacific salmon Steelhead	King crab Tanner crab
Target Species[2]	Walleye pollock Pacific cod Sablefish Yellowfin sole Greenland turbot Arrowtooth flounder Rock sole Flathead sole Alaska plaice Other flatfish Pacific ocean perch Northern rockfish Shortraker rockfish Rougheye rockfish Blackspotted rockfish Other rockfish Atka mackerel	Squid
Other Species[3]	Sculpins Sharks Skates	Octopus
Forage Fish Species[4]	Osmeridae family (eulachon, capelin, and other smelts) Myctophidae family (lanternfishes) Bathylagidae family (deep-sea smelts) Ammodytidae family (Pacific sand lance) Trichodontidae family (Pacific sand fish) Pholidae family (gunnels) Stichaeidae family (pricklebacks, warbonnets, eelblennys, cockscombs, and shannys) Gonostomatidae family (bristlemouths, lightfishes, and anglemouths)	Order Euphausiacea (krill)

[1] Must be returned to the sea
[2] TAC for each listing
[3] Aggregate TAC for group
[4] Management measures for forage fish are established in regulations implementing the FMP

2.2.2.4 Fishery Management Measures

Descriptions of management measures contained in the FMPs are provided in Tables 2.1a and 2.1b. In some cases, management measures are specific (e.g., the Pribilof Islands Habitat Conservation Area) while some measures are implemented in more detail in regulation. The specific management measures, implemented in regulations, which pertain to the conservation of Steller sea lions, are described in Section 2.5.2.

General regulations governing U.S. fisheries appear at 50 CFR Part 600, and regulations specifically governing the groundfish fisheries in the EEZ off Alaska appear at 50 CFR Part 679. The regulations therein prescribe the existing regulatory framework for the federally managed groundfish fisheries off Alaska. Groundfish management areas are provided in Figure 2.2.

2.2.3 The Decision Making Process – Implementing Policy

There are two major decision making areas initiated by the Council: the implementation of FMP and regulatory amendments; and the setting of the annual TAC specifications. The following description of the management process is intended to be generic, illustrating the process by which FMP amendments and regulatory amendments are developed. The setting of TACs will be described below in the section on the annual fisheries assessment and specifications.

2.2.3.1 Involved Entities

The following entities inter alia are integral in the decision making process for the implementation of the FMPs and the harvest of groundfish fisheries in Alaska:

NMFS

The Alaska groundfish fisheries are managed under the authority of the Secretary of Commerce, who delegates that authority through the Under Secretary and Administrator of the National Oceanic and Atmospheric Administration (NOAA) to the Assistant Administrator for Fisheries (NMFS) and to the NMFS Regional Administrator, Alaska Region. The Secretary may rescind this delegation at any time or for any management decision. NMFS is responsible for the day-to-day management of the fisheries. The agency cooperates with the Council to develop fishery policies, conducts rulemaking to implement FMP or regulatory amendments, conducts analyses on the effects of the fisheries on the human environment, monitors the fisheries, and enforces the rules and regulations implemented under the MSA and other applicable law.

NMFS also conducts research programs required to support the fisheries. For the Alaska groundfish fisheries, research activities are conducted primarily by the AFSC. Groundfish stocks in the BSAI and GOA are surveyed by the Resource Assessment and Conservation Engineering (RACE) Division, stock assessment is conducted by the Resource Ecology and Fisheries Management (REFM) Division, and research on marine mammals (including listed large cetaceans and Steller sea lions) is conducted by the National Marine Mammal Laboratory (NMML), also a division of the AFSC.

NMFS is also the principal management agency responsible for the recovery of a number of listed or protected species in the BSAI and GOA regions. Species described in Chapter 3 of this document includes both the western and eastern DPS of Steller sea lion, North Pacific sperm whale, and Eastern North Pacific and Central North Pacific humpback whale.

U.S. Coast Guard

The USCG provides services essential to the implementation of the fisheries, including monitoring for safety and compliance with regulations, enforcement of such regulations, and field assistance with research. The USCG designates a non-voting representative to the Council to act as an enforcement advisor, ensuring that conservation and management measures reflect the practical realities of enforcement in the region. That member also advises Council members and NMFS of the safety impacts of proposed conservation and management measures.

The USCG enforces compliance with fishery regulations and supports NOAA management objectives. Using airborne and at-sea assets, the USCG:

- Prevents encroachment by foreign fishing vessels on the EEZ;
- Ensures compliance by U.S. fishermen with domestic living marine resource laws and regulations within the EEZ;
- Enforces regulations implemented under laws such as the MMPA and ESA and protects threatened marine resources, and;
- Ensures compliance with international agreements for the management of living marine resources on the high seas.

The Coast Guard also provides enforcement policy guidance to domestic lawmakers and regulators, and to U.S. representatives in the international arena, ensuring national and international policy objectives are achievable and enforceable.

U.S. Fish and Wildlife Service

The U.S. Fish and Wildlife Service (USFWS) Alaska Regional Director or his/her designee sits on the Council as a non-voting member. The USFWS partners with NMFS and the Council to provide information relevant to potential fishery effects on seabirds, certain marine mammals (sea otters, walrus, polar bear), or other issues or resources administered by the USFWS in Alaska.

U.S. Department of State

A representative from the U.S. State Department sits on the Council as a non-voting member. The State Department partners with NMFS and the Council to assure fishery management decisions take into account U.S. relations with its international neighboring countries, including treaty obligations, as well as international cooperative research and management issues, or other areas of U.S national and international interest.

Pacific States Marine Fisheries Commission

The Pacific States Marine Fisheries Commission representative on the Council (also non-voting) provides liaison between NMFS and the Council in Alaska and other Pacific coastal states of the U.S. The PSMFC also provides to the NMFS and Council process data gathering and data management services.

State of Alaska

Since the MSA was passed in 1976, fisheries off Alaska have been managed by a combination of state and federal agencies. Article VIII of the Alaska State constitution directs the Alaska legislature and executive branch to manage state fisheries in such a way as to achieve maximum benefit to its people and management of renewable resources on a sustained yield basis. The ADF&G monitors state fisheries, conducts fisheries research, assesses stock condition, and determines appropriate harvest levels for salmon, herring, crabs and other invertebrates, lingcod, rockfish, and Pacific cod. The ADF&G also has in-season emergency authority to open and close fisheries. The Commercial Fisheries Entry Commission is a second state agency that has authority to establish moratoria or limited-entry systems for state-managed fisheries. The Alaska State Legislature created the Alaska Board of Fisheries to provide public access to the fishery management process and to give policy direction to ADF&G. The Board of Fisheries is responsible for developing state fishery management plans, making allocative decisions, and promulgating regulations. The Department of Public Safety enforces State fishery regulations. State fisheries will be considered below in the chapters on the Environmental Baseline (Chapter 4) and Cumulative Effects (Chapter 6).

State of Alaska Parallel Fisheries

Annually at the beginning of each calendar year, ADF&G issues an emergency order (EO) to define the parallel groundfish fisheries in state waters; the EO for 2010 was issued on December 31, 2009 (emergency order 4-GF-01-10, Appendix 2-1). Fishing in state waters, i.e., within three nm of the coastline, is managed by ADF&G, which coordinates parallel groundfish fisheries management in state waters with that of adjacent federal fisheries in the EEZ. This coordination generally includes opening dates, in-season adjustments, gear requirements, Steller sea lion closure areas, vessel monitoring system (VMS) requirements for Pacific cod, pollock, and Atka mackerel (except jig gear vessels), as well as seabird avoidance measures and essential fish habitat closures. Fish harvested in the parallel fisheries are counted against the federal TAC to ensure that the total harvest level is within the constraints on the federal fishery. Harvests from these parallel fisheries are small in comparison to harvest from the federal fisheries, ranging up to 3% of the TAC in the BSAI and up to 9% of the TAC in the GOA (except Demersal Shelf Rockfish which comprised 16% of TAC in 2009). See Table 2.27 for harvest amounts. These parallel fisheries include harvest of Pacific cod, pollock, Alaska plaice, arrowtooth flounder, flathead sole, Greenland turbot, northern rockfish, Pacific ocean perch, rockfish species, rock sole, sablefish, squid, and yellowfin sole in the BSAI; and in the GOA, parallel fisheries harvest arrowtooth flounder, skates, rockfish species, flathead sole, Pacific ocean perch, rex sole, sablefish, and deep and shallow water flatfish. All of these fisheries are considered part of the federal action in this Biological Opinion.

Additional ADF&G groundfish fisheries in state waters and in the EEZ that are not managed under parallel regulations and are not included in a NMFS FMP include lingcod, black and blue rockfish, state waters sablefish, and the state waters Pacific cod fishery, which is managed under a Guideline Harvest Limit (GHL). The Pacific cod GHL is a portion of the ABC for the federal Pacific cod fishery; the federal harvest amounts are adjusted downward to provide for the GHL State fishery. These fisheries are described in more detail in the Cumulative Effects chapter of this document, but are not considered part of the Federal Action in this Biological Opinion.

North Pacific Fishery Management Council

The Council, which is composed of 11 voting members, serves six main functions (16 U.S.C. 1852 § 302(h)(1-6)):

1. prepares and submits FMPs for each fishery that requires conservation and management, as well as amendments to each plan;
2. prepares comments on certain applications for foreign fishing and on FMPs or amendments prepared by the Secretary;
3. conducts public meetings to allow public participation in the management process;
4. submits to the Secretary reports that it deems necessary or that were requested by the Secretary;
5. for each fishery, reviews on a continuing basis the assessments and specifications necessary to achieve OY from, the capacity and extent to which United States fish processors will process U.S. harvested fish from, and the total allowable level of foreign fishing in, each fishery; and
6. conducts any other activities required by the MSA or necessary and appropriate to the foregoing functions.

In addition to the main Council body, the Council maintains four main committees and panels related to groundfish fishery management. The Advisory Panel consists primarily of representatives of the fishing industry and is intended to advise the Council on any matters pertaining to how the FMPs and amendments may impact the fishing industry. The SSC consists of appointed scientists and is intended to assist in the development, collection, and evaluation of statistical, biological, economic, social, and other scientific information necessary for development and amendment of FMPs. The two remaining committees are Plan Teams for the BSAI and GOA groundfish fisheries. These teams review stock assessment methods and results, and make recommendations on harvest levels to the Council based on the status and trends of each stock and its tolerance for fishery removal. The Council appoints other committees as needed to advise the Council on other issues relating to groundfish fishery management (e.g., the Steller Sea Lion Mitigation Committee has been utilized to inform the Council on Steller sea lion related fishery management issues).

2.2.3.2 Council and NMFS Fishery Management Policy

The Council has developed a management policy and objectives to guide its development of management recommendations to the Secretary. The Council's policy is to apply judicious and responsible fisheries management practices, based on sound scientific research and analysis, proactively rather than reactively, to ensure the sustainability of fishery resources and associated ecosystems for the benefit of future, as well as current, generations. The productivity of the North Pacific ecosystem is acknowledged to be among the highest in the world. For the past 30 years, the Council management approach has incorporated forward looking conservation measures that address differing levels of uncertainty. This management approach has in recent years been labeled the precautionary approach. Recognizing that potential changes in productivity may be caused by fluctuations in natural oceanographic conditions, fisheries, and other non-fishing activities, the Council intends to continue to take appropriate measures to ensure the continued sustainability of the managed species. It will carry out this objective by considering reasonable, adaptive management measures, as described in the MSA and in conformance with the National Standards, the ESA, NEPA, and other applicable laws. This management approach takes into account the National Academy of Science's recommendations on Sustainable Fisheries Policy.

As part of its policy, the Council intends to consider and adopt, as appropriate, measures that accelerate the Council's precautionary, adaptive management approach through community-based or rights-based management, ecosystem-based management principles that protect managed species from overfishing, and where appropriate and practicable, increase habitat protection and bycatch constraints. All management measures will be based on the best scientific information available. Given this intent, the fishery management goal is to provide sound conservation of the living marine resources, provide socially and economically viable fisheries for the well-being of fishing communities, minimize human-caused threats to protected species, maintain a healthy marine resource habitat, and incorporate ecosystem-based considerations into management decisions.

This management approach recognizes the need to balance many competing uses of marine resources and different social and economic goals for sustainable fishery management, including protection of the long-term health of the resource and the optimization of yield. This policy will use and improve upon the Council's existing open and transparent process of public involvement in decision-making.

2.2.3.3 Implementing the FMPs

FMPs and amendments to FMPs are developed by the Council, submitted to the Secretary for review, and may be approved, disapproved, or partially approved (Section 304(a)(3) of the MSA). Amendments also may require implementing regulations which are recommended and deemed necessary by the Council and generally developed by NMFS. Regulations without an FMP amendment also may be recommended and deemed necessary by the Council and developed by NMFS. Once the regulations are effective, NMFS has responsibility for day-to-day management of the fisheries. Enforcement of the regulations is carried out jointly by NMFS and the USCG. Disapproved and partially approved FMPs and FMP amendments are returned by NMFS to the Council with an explanation of the reasons for disapproval. The Council may then decide whether to revise and resubmit the FMP/amendment. If the Council fails to develop a necessary FMP/amendment, or fails to revise an FMP/amendment following Secretarial disapproval or partial approval within a reasonable period of time, the Secretary may develop a Secretarial FMP/amendment.

Amendments to FMPs may be necessitated by a variety of events including new or triggered statutory requirements, operational need, or changes in the fisheries. Occasionally, the Council will solicit FMP and regulatory amendment proposals from the public. These proposals are then reviewed, and qualitatively ranked in terms of analytical difficulty and priority for consideration. If a proposal is selected for consideration, then the next step is the preparation of an initial analysis of the proposal. These analyses serve at least three functions. First, they fulfill requirements under certain statutes and executive orders. Second, they provide opportunity for interested or affected members of the public to bring information to the Council's attention regarding the proposed and alternative actions. And third, they help the Council to contrast and compare the potential effects of alternative actions to their stated policy goals and objectives, and make a well-reasoned decision on which amendment proposal to recommend to the Secretary.

Additional analytical requirements may include environmental assessments or environmental impact statements (EIS) as required by NEPA; a Regulatory Impact Review (RIR) under Executive Order 12866; a regulatory flexibility analysis under the Regulatory Flexibility Act (RFA); an assessment of potential impacts on marine mammals under the MMPA; a review of effects on essential fish habitat under the MSA; a review of effects on the state's coastal zone management program (under the Coastal Zone Management Act); an assessment under the Paperwork Reduction Act; a predissemination review under the Information Quality Act; consultation under the ESA; and possibly a federalism impact statement under Executive Order 13132.

The next step for the Council is to review a draft of the initial analysis to determine whether it should be released for public review and comment. In making this decision, the Council relies on the advice it receives from its Advisory Panel and SSC and the public. The Council decision at this point may be to release the initial draft analysis for formal public review as it is, instruct staff to make certain minor revisions to it before releasing it, or request major revisions to it and another Council review before releasing it. Or the Council may decide to suspend further action on the analysis, which would stop further development of the proposal, at least temporarily. If the Council decides to release the initial draft analysis for public review, the public review period normally is the time period before the next Council meeting, usually at least four weeks.

After a period of public review, the next action by the Council on a management proposal is to decide on its preferred alternative. The Council's choice of a preferred alternative (other than the "no action" alternative) frequently is referred to as the final action of the Council to adopt an FMP or FMP/regulatory amendment for recommendation to the Secretary.

Once the Council has determined its final recommendation, the recommendation is transmitted to the Secretary. The principal documents that are submitted include (a) the proposed FMP text or text changes in the case of an FMP amendment, (b) the draft analysis of potential environmental and socioeconomic impacts of the preferred alternative and other alternatives considered by the Council, and (c) any proposed regulations that would implement the action, if the amendment is approved. A notice of availability of a proposed FMP amendment or FMP is published in the Federal Register for a 60 day comment period. The proposed implementing regulations are published in the Federal Register as a notice of proposed rule-making with a 45 day comment period. These comment periods generally overlap. Comments received on either the FMP amendment or on the proposed regulations are considered in the Secretarial approval of the FMP amendment and regulations.

After receipt of the official FMP/amendment review package, the Secretary must immediately commence review of the package to determine whether the proposed FMP or FMP amendment is consistent with MSA, including the National Standards, and other applicable law and must immediately publish a notice of availability in the Federal Register to start the 60 day period of public review. Within 30 days after the end of the public comment period, the Secretary must approve, disapprove or partially approve the FMP amendment by written notice to the Council. If Secretarial action is not taken within the required time period, then the FMP amendment takes effect as if it were fully approved.

Thus, the MSA vests the Councils with the primary role of developing management measures. The role of the Secretary (normally NMFS, on behalf of the Secretary) is usually limited to approval, disapproval, or partial approval of a Council's FMP recommendation and determination that the proposed rule is consistent with the FMP, FMP amendment, and other applicable laws, including ESA. Section 304(a)(3) states that if an FMP or FMP amendment is disapproved or partially approved, the written notice to the Council must specify the applicable law with which the FMP/amendment is inconsistent, the nature of the inconsistency, and recommendations for correcting the inconsistency.

When the Council recommends regulations to implement an FMP or amendment, the Secretary reviews them to determine their consistency with the underlying FMP. If NMFS determines that the proposed regulatory amendment is consistent, then it is published in the Federal Register, but if the determination is negative, NMFS must notify the Council in writing specifying the inconsistencies and providing recommendations for revision that would make the proposed regulation consistent. An approved FMP, FMP amendment or regulatory amendment is implemented by publication of a notice of approval (for FMP amendments without regulations) or a final rule in the Federal Register. The rule normally is not effective for an additional 30 days after it is published, as required under the Administrative Procedure Act.

2.3 Exploitation Strategy

In 2002, the Council convened a panel to provide an independent scientific review of the current harvest strategy embodied in the FMPs (Goodman *et al.* 2002). The focus of the review was on describing the harvest policy, the role of F40% as a reference point, and to determine whether changes should be made to account for particular species or ecosystem needs in accordance with the MSA. This opinion relies heavily on the Goodman *et al.* (2002) report and the SSC and AFSC responses to concerns identified in the report; they provide an excellent review of the harvest strategy and the potential consequences to non-target species and the ecosystem as a whole. See the Goodman *et al.* (2002) report for further background on fisheries management and exploitation strategy. This section of the opinion focuses on describing the exploitation strategy as it relates to harvests of prey species important to ESA-listed species and incorporates much of the descriptive text from Goodman *et al.* (2002) and SSC and AFSC clarifications where necessary.

Harvests in the BSAI and GOA fisheries are governed by the BSAI and GOA groundfish FMPs. The FMPs allow for a wide range of possible harvests for any given stock in any given year, such that the plans are consistent with a large number of harvest strategies. However, any harvest allowed by the FMPs is required to be consistent with the National Standards described in the MSA. Of particular relevance in this regard is National Standard 1, which states, "Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry" (Title III, Section 301(a)(1)).

The FMPs contain the following definitions of terms that are used in this opinion and are important for understanding the action:

> Maximum sustainable yield (MSY) is the largest long-term average catch or yield that can be taken from a stock or stock complex under prevailing ecological and environmental conditions.
>
> Optimum yield (OY) is the amount of fish which:
> a) will provide the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, and taking into account the protection of marine ecosystems;
> b) is prescribed as such on the basis of the MSY from the fishery, as reduced by any relevant economic, social, or ecological factor; and
> c) in the case of an overfished fishery, provides for rebuilding to a level consistent with producing the MSY in such fishery.
>
> Overfishing level (OFL) is a limit reference point set annually for a stock or stock complex during the assessment process. Overfishing occurs whenever a stock or stock complex is subjected to a rate or level of fishing mortality that jeopardizes the capacity of a stock or stock complex to produce MSY on a continuing basis. Operationally, overfishing occurs when the harvest exceeds the OFL.
>
> Acceptable biological catch (ABC) is an annual sustainable target harvest (or range of harvests) for a stock or stock complex, determined by the Plan Team and the SSC during the assessment process. It is derived from the status and dynamics of the stock, environmental conditions, and other ecological factors, given the prevailing technological characteristics of the fishery. The National Standard Guidelines (Guidelines) distinguish between limit reference points (which management seeks to avoid) and target reference points (which management seeks to achieve). The target reference point is set below the limit reference point for overfishing.
>
> Total allowable catch (TAC) is the annual harvest limit for a stock or stock complex, derived from the ABC by considering social and economic factors.

As noted in the FMPs, ABCs and TACs are specified annually for each stock or stock complex within the "target species" and "other species" categories. The OY range is constant and is specified for the groundfish fishery (comprising target species and other species categories) as a whole.

According to both current FMPs (NPFMC 2009 a, b), the sum of the stock-specific ABCs may fall within or outside of the OY range. However, if the sum of annual TACs falls outside of the OY range, either the TACs must be adjusted or the FMP must be amended. In the case of the BSAI, Consolidated Appropriations Act of 2004 sets the upper limit of the BSAI OY at 2 million mt. Thus, for a change to the BSAI OY, congressional action would be necessary for a statutory amendment.

2.3.1 Background: Principles of Sustainable Fisheries and Surplus Production

The following discussion is based on material provided to PRD by the SFD. A tenet of current fisheries management is that the average biomass (weight of the stock) at which a stock persists depends on the relationship between the spawning (breeding) stock biomass and the average production of new fish, modified after reproduction by how well the new fish (sometimes called "recruits") survive, grow, reproduce, and distribute themselves in space after they enter the fishery. The actual and assumed relationships between stock biomass and production are of major importance in fisheries management. From a single target stock point of view, and ignoring potential ecosystem interactions or consequences, it is theoretically possible to have sustainable fishing of that stock at almost any level of stock biomass, so long as the catch that is taken balances the production. In principle, therefore, sustainable fishing could take place anywhere between very low or very high stock sizes. However, the ability to manage with confidence a stock to any given stock size depends on how well basic population parameters can be estimated and predicted over time, how well catches can be controlled to match production, the dynamics of the stock's response to deviations from the intended level of catch, and a variety of other difficult and uncertain factors. We discuss these issues in some detail in the effects chapter.

By definition, the level of stock size that produces the maximum possible production is the biomass at maximum sustainable yield (BMSY). From a single target stock point of view, and because of economic and social objectives, as well as uncertainty, fishery managers in the fisheries of issue here have concluded that there are good reasons for trying to manage fish stocks near to, but somewhat above, the stock size that confers MSY. The MSY is the highest theoretical production (yield, or catch) that can be continuously taken from a stock under constant environmental conditions. The MSY is estimated from models based on surplus production, stock-recruitment relationships, and other methods. In practice, MSY, and the level of fishing effort needed to take it, are difficult to assess. Nevertheless, MSY is a benchmark in fisheries theory, international agreements and national legislation; as such, it is the basis for important reference points used in fishery management.

The assessment of a fish stock (and the potential for harvest) is generally based on the assumption that the fished populations are closed (no immigration or emigration). Under this assumption, a population can increase in number only through recruitment and can decrease in number only through mortality, i.e., it is replenished only by the annual addition of a new cohort or year-class. In terms of biomass, the population may change by additions due to recruitment and physical growth, and by losses due to natural and fishing mortality.

The number of fish constituting the fished part of a population is determined, then, by the combination of the survival rates of all cohorts and annual recruitment of a new cohort. Mortality may result from natural causes (i.e., natural mortality), direct or indirect effects of fishing (i.e., fishing mortality), or other human-related actions. Recruitment is determined by a number of factors which may vary in importance over time, by stock, by area, and due to other factors. Which factors have the greatest impacts on recruitment in different species is a matter of considerable debate and research. For example, the Fisheries-Oceanography Coordinated Investigations (FOCI) program was initiated by the NOAA in 1984 to investigate the factors determining recruitment of pollock in the GOA.

From the single-species standpoint, for an unfished stock of a particular size, recruitment may occur at levels greater than necessary to maintain the stock at its present size. Such "excess" however, is essential for population growth. From the standpoint of deterministic single-species fishery management of a non-depleted stock, this excess is considered a surplus that can be removed by fishing without harm to the stock. The concept of surplus recruitment is illustrated by the Ricker (1954) stock-recruitment relation in Figure 2.3. The Ricker curve indicates a density-dependent relation between stock and recruitment where recruitment varies as a function of some measure of stock size (e.g., number or biomass). The Ricker curve, which has a dome-shaped pattern, suggests that recruitment reaches a peak at an intermediate stock level and then declines with increasing stock size. The Beverton-Holt stock-recruit relationship has the feature that recruitment increases as a function of spawning biomass to an asymptotic level. The excess or surplus recruitment in this case is represented by the vertical difference between the stock-recruitment line and the replacement line. The biological mechanism(s) that are thought to explain this peak for some species include cannibalism of adults on small fish and crowding effects due to overescapement. It assumes density-dependent population behavior. In the simplest single-species case, without random variability and where the fishable stock consists of a single age group, and without the population response of other harvesters (e.g., natural predators), this excess represents sustainable yield. At some stock size, the "excess" reaches a maximum, which is the MSY.

While a decline in a stock could indicate changes in both reproduction of the stock and mortality of pre-recruits, Ricker (1954) attributed it to compensatory mortality of pre-recruits through mechanisms such as predation and, in particular, cannibalism. Thus, the number of young produced probably continues to increase with increasing stock size, but fewer young survive to recruitment. The remainder is "lost" to various forms of mortality, possibly including predation.

2.3.2 Overview of the Harvest Strategy

The following description is largely excerpted from Goodman *et al.* (2002), with minor editing and modification for brevity and clarification:

> The current harvest strategy is essentially a maximum sustainable yield (MSY) single-species approach, modified by some formal safeguards incorporated to ward against overfishing as defined from the single-species standpoint, and with opportunities of a less-structured nature for reducing harvest rates further in response to perceived social, economic and ecological concerns. No quantitative standards or specific decision rules are stated for each of these latter considerations, except as they are imposed from outside the MSA, by the ESA or the MMPA, and only for particular populations. An overall quantitative limit is applied to the harvests amount by the optimal yield specified in each FMP and is considered by the Council as the scientifically recommended harvest level is further reduced for social, economic, and ecological concerns. The fishing mortality rate used to estimate the overfishing level (OFL) for each stock is an estimate either of the fishing mortality rate associated with MSY (F_{MSY}) or an estimate of a surrogate for F_{MSY}. The OFL is treated in the management system as a limit that should not be exceeded except with a very low probability. The acceptable biological catch (ABC) set for each stock is an estimate of a target rate, which is intended to establish some margin between it and the OFL. The hope is that managing so as to achieve this target on average will accomplish the desired compliance with exceeding the limit (OFL) only rarely. If it occurs, the ad hoc downward adjustments of harvest in response to other social, economic, and ecological considerations takes place in the deliberations where the total allowable catch (TAC) is set subject to the constraints that each individual TAC be less than or equal to the corresponding ABC and the sum of the TACs falls within the OY range.
>
> The formulaic component of the reduction of harvest rate from the theoretical MSY harvest rate (from OFL to ABC) is by an amount that is often modest, when expressed as a fraction of the harvest rate; but in terms of the total tonnage involved, or its dollar value, the amount is considerable. The margin may also be small relative to natural variation. It may be small relative to the underlying uncertainty about key population parameters, the overall status of many target stocks, stock complexes, trophic guilds, and about ecosystem effects of various harvest levels. The rate of reduction of the TAC from the ABC has for some stocks and some years been large relative to the reduction from OFL to ABC, but there is no explicit and general formula for this reduction. The FMPs specify that the setting of TAC must be less than or equal to ABC (section 3.2.5.1), and the setting of ABC must be no more than OFL and must be less than OFL for stocks with less information, as further explained below.
>
> The formal and standardized quantitative portions of the process of determining OFL and ABC begin with the assignment of each stock to one of six "Tiers" based on the availability of

information about that stock. There should be the most (relative to stocks in other tiers) information about stocks in Tier 1 and the least information about stocks placed in Tier 6. The $F_{40\%}$ fishing mortality rate plays a prominent role in some of the Tiers (2, 3, and 4) but not the others. Notably, in Tier 3 (which is where many of the major BSAI/GOA stocks are assigned) and Tier 4, the estimate of $F_{40\%}$ is used as a surrogate for a fishing mortality rate that is somewhat below FMSY.

$F_{40\%}$ is the calculated fishing mortality rate at which the equilibrium spawning biomass per recruit is reduced to 40% of its theoretical value in the equivalent unfished stock. This is a useful measure of the amount by which the associated fishing rate reduces the stock size, in the long run. The useful features of this particular measure are two-fold. First, its calculation is less sensitive to the details of the stock-recruitment relationship than is the calculation of F_{MSY}, so it is practical to estimate $F_{40\%}$ for stocks that are not well enough studied for estimation of F_{MSY}. The second is that, for a range of dynamics encompassing many, but not all, of the BSAI and GOA target groundfish stocks, modeling studies have shown that harvesting at $F_{35\%}$ accomplishes about the same thing as harvesting at F_{MSY}, so harvesting at the slightly lower rate, $F_{40\%}$, establishes a modest margin of safety.

2.3.3 MSY and OY

Estimates of the groundfish complex have been computed by summing MSY estimates for individual species and species groups. However, the NPFMC (2009b:14; citing an in press paper by Walters *et al.*, which would be updated with the implementation of Amendment 87 to Walters *et al.* 2005) summarized that: "…current multi-species models suggest that the sum of single-species MSYs provides a poor estimate of MSY for the groundfish complex as a whole...because biological reference points for single stocks, such as FMSY, may change substantially when multi-species interactions are taken into account." Under the MSA, OY is prescribed on the basis of the MSY from each fishery, as reduced by any relevant economic, social, or ecological factor (16 U.S.C. 1802 § 3(28)(B)). In both the BSAI and GOA groundfish FMPs the concept of optimum yield has been applied to the total groundfish catch in these regions. The OY of the BSAI groundfish complex (which consists of stocks listed in the "target species" and "other species" categories, as listed in Table 3-1 of the BSAI groundfish FMP) is set at 85% of the historical estimate of MSY, or 1.4 to 2.0 million mt, plus the incidental harvest of non-specified species. (NPFMC 2009a). Thus, the endpoints of the OY range were determined by subtracting 15% from the endpoints of the range of MSY estimates available at the time the current OY specification was adopted. The BSAI groundfish FMP justified the 15% reduction by stating that it 1) reduces the risk associated with relying upon incomplete data and questionable assumptions in assessment models used to determine the condition of stocks, and 2) is probably a conservatively safe level for the groundfish complex.

The range of OY specified in the GOA groundfish FMP is 116,000-800,000 mt of groundfish for the target species and the "other species" categories, to the extent this can be harvested consistently with the management measures specified in this FMP. This range was established in 1987 based on the examination of historical and recent catches, recent determinations of ABC, and recent and past estimates of MSY for each major groundfish species. This derivation from historical estimates of MSY and fishery performance reflects the combined influence of biological, ecological, and socioeconomic factors. The end points of the range were derived as described below.

For the minimum value, 116,000 mt was approximately equal to the lowest historical groundfish catch during the 21-year period 1965-1985 (116,053 mt in 1971, NPFMC 1986). In that year catches of pollock, Pacific cod, and Atka mackerel were all at very low levels. Given the status of the groundfish resources and the present management regime, it was considered extremely unlikely that future total harvest would fall below this level. Thus, the TACs must be established so as to result in a sum of at least 116,000 mt.

The upper end of the GOA groundfish FMP OY range, 800,000 mt, was derived from MSY information. The MSY for all species of groundfish (excluding the other species category) between 1983 and 1987 ranged from 804,950 mt in 1983 to 1,137,750 mt for the 1987 fishing year. The average MSY over the five-year period was 873,070 mt. Therefore, the upper end of the range has been approximately equal to 92% of the mean MSY for the five-year period. However, we clarify that while this is the fact of the relationship between OY and mean MSY over this period, there is no rule that sets the upper end of the OY range equal to this percentage of the mean five-year MSY. The ABC summed for all species ranged from 457,082 mt in 1985 to 814,752 mt in 1987. Most of the variation in the ABC and catch over the five-year interval resulted from changes in the status of two species: pollock and flounder. Pollock ABC ranged from 112,000 mt in 1987 to 516,600 mt in 1984; while flounder ABC ranged from 33,500 mt in 1985 to 537,000 mt in 1987. Therefore, the 800,000 mt upper end of the OY range was selected in consideration of the volatility in pollock and flounder ABC, and the potential for harvesting at MSY.

2.3.4 Harvest Control Rules

The National Standard Guidelines (Guidelines) distinguish between limit reference points (which management seeks to avoid) and target reference points (which management seeks to achieve). However, the use of the term “target” should not be interpreted as indicating this point can actually be achieved with some “surgical” like precision. As pointed out during review of this document, if errors are distributed symmetrically about the target, targets will be exceeded 50% of the time, regardless of what the target is. In the case of target harvest levels or rates, the Guidelines encourage a precautionary approach as follows (50 CFR § 600.310(f)(5)).

(1) Target reference points should be set safely below limit reference points.
(2) A stock that is below its MSY level should be harvested at a lower rate than if the stock were above its MSY level.
(3) Criteria used to set target catch levels should be explicitly risk averse, so that greater uncertainty regarding the status or productive capacity of a stock corresponds to greater caution in setting target catch levels.

The Guidelines envision that limit and target fishing mortality rates will often be cast in the form of “harvest control rules,” which are functions that determine fishing mortality based on stock size (50 CFR § 600.310(c)(2), § 600.310(f)(4)(ii)). In particular, the Guidelines presume that MSY will be estimated using an “MSY control rule” which describes how the Council would set harvest rates if maximization of long-term average yield were its primary goal. An MSY control rule would be an example of a limit reference point. A wide variety of functional forms can be used to define harvest control rules (Restrepo *et al.* 1998).

The BSAI and GOA groundfish FMPs define two sets of harvest control rules which follow the precautionary approach outlined above to a considerable extent. One set of control rules defines the limit harvest rate that is used to determine the overfishing level (OFL), and the other defines the upper boundary for the target harvest rate that is used to determine the ABC.

The two sets of harvest control rules in the BSAI and GOA groundfish FMPs are prescribed through a set of six tiers which are listed below and which are numbered to correspond to descending order of information availability about the stock (e.g., there should be better information about key parameters of stocks in Tier 1 than in Tier 2; see below).

As indicated in the 2009 BSAI groundfish FMP (NPFMC 2009a):

> "Overfishing is defined as any amount of fishing in excess of a prescribed maximum allowable rate. This maximum allowable rate is prescribed through a set of six tiers which are listed below in descending order of preference, corresponding to descending order of information availability. The Council's Scientific and Statistical Committee (SSC) will have final authority for determining whether a given item of information is "reliable" for the purpose of this definition, and may use either objective or subjective criteria in making such determinations."

For Tiers 1-3, the coefficient "a" is set at a default value of 0.05, with the understanding that a different value for a specific stock or stock complex may be used if supported by the best available scientific information. For Tiers 2-4, a designation of the form "F" refers to the estimate of fishing mortality (F) associated with an equilibrium level of spawning per recruit (SPR) equal to X% of the estimated equilibrium level of spawning per recruit in the absence of any fishing. For Tier 3, the term B40% refers to the estimate of the long-term average biomass that would be expected under average recruitment and F=F40%. Tiers under which fished stocks are currently managed are listed in Table 2.8.

- Tier 1) Information available: Reliable point estimates of B and B_{MSY} and reliable probability density function (pdf) of F_{MSY}.
  - 1a) Stock status: $B/B_{MSY} > 1$
    - $F_{OFL} = m_A$, the arithmetic mean of the pdf
    - $F_{ABC} \leq m_H$, the harmonic mean of the pdf
  - 1b) Stock status: $a < B/B_{MSY} \leq 1$
    - $F_{OFL} = m_A \times (B/B_{MSY} - a)/(1 - a)$
    - $F_{ABC} \leq m_H \times (B/B_{MSY} - a)/(1 - a)$
  - 1c) Stock status: $B/B_{MSY} \leq a$
    - $F_{OFL} = 0$
    - $F_{ABC} = 0$
- Tier 2) Information available: Reliable point estimates of B, BMSY , FMSY , F35% , and F40%.
  - 2a) Stock status: $B/B_{MSY} > 1$
    - $F_{OFL} = F_{MSY}$
    - $F_{ABC} \leq F_{MSY} \times (F_{40\%} / F_{35\%})$
  - 2b) Stock status: $a < B/B_{MSY} \leq 1$
    - $F_{OFL} = F_{MSY} \times (B/B_{MSY} - a)/(1 - a)$
    - $F_{ABC} \leq F_{MSY} \times (F_{40\%} / F_{35\%}) \times (B/B_{MSY} - a)/(1 - a)$
  - 2c) Stock status: $B/B_{MSY} \leq a$
    - $F_{OFL} = 0$
    - $F_{ABC} = 0$
- Tier 3) Information available: Reliable point estimates of B, B40% , F35%, and F40%.
  - 3a) Stock status: $B/B_{40\%} > 1$
    - $F_{OFL} = F_{35\%}$
    - $F_{ABC} \leq F_{40\%}$
  - 3b) Stock status: $a < B/B_{40\%}$ # 1
    - $F_{OFL} = F_{35\%} \times (B/B_{40\%} - a)/(1 - a)$
    - $F_{ABC} \leq F_{40\%} \times (B/B_{40\%} - a)/(1 - a)$
  - 3c) Stock status: $B/B_{40\%} \leq a$
    - $F_{OFL} = 0$
    - $F_{ABC} = 0$
- Tier 4) Information available: Reliable point estimates of B, F35%, and F40%.
    - $F_{OFL} = F_{35\%}$
    - $F_{ABC} \leq F_{40\%}$
- Tier 5) Information available: Reliable point estimates of B and natural mortality rate M.
    - $F_{OFL} = M$

$F_{ABC} \leq 0.75 \times M$

Tier 6) Information available: Reliable catch history from 1978 through 1995.

OFL = the average catch from 1978 through 1995, unless an alternative value is established by the SSC on the basis of the best available scientific information

ABC ≤ 0.75 × OFL

The following is a description of the tier system excerpted from Goodman *et al.* (2002), but edited for brevity and clarity and updated:

> The dynamics of only a few stocks covered by the FMP, BS pollock, yellowfin sole, and rock sole are considered well-enough quantified to qualify for Tier 1 (NPFMC 2009 BSAI SAFE Report). In Tier 1 the limiting F_{OFL} is the equivalent of the point estimate of F_{MSY} (that is to say, roughly, the "best" estimate without adjusting for uncertainty), and the target F_{ABC} is the harmonic mean of the distribution of the estimate for F_{MSY}. The harmonic mean has the mathematical property that it is less than the simple average (roughly, the point estimate) by an amount that increases with the spread of the distribution, so this establishes a margin that increases with the uncertainty in the estimate.
> Tier 2 differs from Tier 1 in that only point estimates of the key population parameters are available, so the distribution of the estimate for F_{MSY} is not known. In this Tier, the limiting F_{OFL} is the point estimate of F_{MSY}, much as in Tier 1, but a different formula (based on the adjustment used in Tier 3) is used for adjusting the F_{ABC} downward from F_{OFL}. The mathematics of the different formulas used for adjusting the F_{ABC} downward from F_{OFL} in Tier 1 and Tier 2 does not guarantee that the margin so established in Tier 2 will be wider than the margin in Tier 1.
>
> Tier 3 differs from Tier 2 in that information is insufficient for any estimation of MSY. In this Tier, the limiting F_{OFL} is the point estimate of $F_{35\%}$ and the target F_{ABC} is the point estimate of $F_{40\%}$. The width of the margin between F_{ABC} and F_{OFL}, in this Tier, therefore, will be essentially the same as in Tier 2, and the relation to the width of the margin in Tier 1 is variable. Most of the major target stocks in the BSAI/GOA are in Tier 3.
>
> Tier 4 differs from Tier 3 in that information is insufficient for estimation of target biomass levels. In this Tier, the limiting F_{OFL} is the point estimate of $F_{35\%}$, and the target F_{ABC} is the point estimate of $F_{40\%}$, both as in Tier 3. The width of the margin between F_{ABC} and F_{OFL}, in this Tier, therefore, will be identical to that in Tier 3, and essentially the same as in Tier 2, and the relation to the width of the margin in Tier 1 is variable.
>
> Tier 5 differs from Tier 4 in that information is insufficient for estimating $F_{40\%}$ or $F_{35\%}$, so the limits and targets use different surrogates to attempt to approximate management for MSY. In this Tier, the limiting F_{OFL} is the point estimate of the natural mortality rate of the stock, and the target F_{ABC} is three fourths of that value. The limiting F_{OFL} in this Tier may be either conservative or aggressive relative to the limiting F_{OFL} of $F_{35\%}$ in the three Tiers above. Theoretical work [Deriso 1982 among others and Thompson] has shown that M is often higher than F_{MSY}, so it would be a better as a limit than a target. The margin between F_{ABC} and F_{OFL} in this Tier, corresponding to a 25% reduction of fishing mortality rate, is wider than the margin in Tiers 2 through 4. Most of the minor target stocks in the BSAI/GOA are in Tier 5.
>
> Tier 6 differs from Tier 5 in that information is insufficient for estimating any of the stock parameters, and all that is known is the catch history. In this Tier, the limiting F_{OFL} is the average historic catch, and the target F_{ABC} is three fourths of that value. In practice, without estimates of stock size, the control is exerted simply through a limit on amount of catch. The margin between F_{ABC} and F_{OFL}, in this Tier, considered as a fractional reduction, is the same as in Tier 5.

In Tiers 1 through 3 there are provisions for rapid rebuilding of stocks from an overfished condition, by reductions in the target fishing mortality rate triggered whenever the estimate of stock biomass is below the target biomass. There is no such provision in Tiers 4 through 6. In Tiers 1 through 5, the information on the stock is sufficient to give clear indications if the stock status is departing substantially from the management goals. In Tier 6, this is not the case.

2.3.4.1 Stock Status: Overfished and Overfishing

The MSA requires the Secretary to "report annually to the Congress and the Councils on the status of fisheries within each Council's geographical area of authority and identify those fisheries that are overfished or are approaching a condition of being overfished" (16 USC § 304(e)(1)). The Guidelines define two "status determination criteria" to be used in making this identification. The first of these, the "maximum fishing mortality threshold" (MFMT), is used to determine whether a stock is being subjected to a rate of fishing mortality that is too high. The second, the "minimum stock size threshold" (MSST), is used to determine whether the stock has fallen to a level of biomass that is too low. Exceeding the MFMT results in a determination that the stock is being subjected to overfishing. Falling below the MSST results in a determination that the stock is overfished.

More specifically, the Guidelines require that the MFMT be at least as conservative as the MSY control rule (50 CFR 600.310(d)(2)(i)), and they define the MSST as whichever of the following is greater: one-half the MSY stock size, or the minimum stock size at which rebuilding to the MSY level would be expected to occur within 10 years if the stock were exploited at the MFMT (50 CFR 600.310(d((2)(ii)).

When expressed in units of catch, the MFMT is equivalent to OFL in the BSAI and GOA groundfish FMPs, and when expressed in units of fishing mortality, the MFMT is equivalent to FOFL. Thus, prevention of overfishing is accomplished simply by insuring that catch does not exceed OFL in any given year.

For each BSAI and GOA groundfish stock managed under Tiers 1-3, the following algorithm is used to determine stock status with respect to MSST (Figure 2.4):

- If the stock is below ½ B_{MSY}, it is below MSST.
- If the stock is above B_{MSY}, it is above MSST.
- If the stock is between ½ B_{MSY} and B_{MSY}, then 1000 simulations are conducted in which the population is projected forward 10 years with randomly varying recruitment and with fishing mortality set equal to F_{OFL} in all years. Recruitment is drawn from a probability distribution based on recruitment estimates from 1978 to 1998.
- If the average ending stock size in these simulations is above B_{MSY}, the stock is above its MSST.
- If the average ending stock size in these simulations is below B_{MSY}, the stock is below its MSST.

MSSTs cannot be estimated for certain stocks because the necessary reference stock levels cannot be estimated reliably. These stocks are (by definition) managed under harvest Tiers 4-6.

The stock is considered to be approaching an overfished condition if NMFS (for the Secretary) estimates that the stock will become overfished within two years (16 USC 1854 § 304(e)(1)). For each BSAI and GOA groundfish stock managed under Tiers 1-3, the determination as to whether the stock is approaching an overfished condition is made on the basis of 1000 simulations in which the population is projected forward 12 years with randomly varying recruitment and with fishing mortality set equal to the maximum permissible value of FABC for the first two years and equal to FOFL thereafter:

- If the mean spawning biomass for the third year is below ½ BMSY, the stock is approaching an overfished condition.
- If spawning biomass for the third year is above BMSY, the stock is not approaching an overfished condition.
- If spawning biomass for the third year is between ½ BMSY and BMSY, the determination depends on the mean spawning biomass at the end of 12 years.
- If the average ending stock size in these simulations is below BMSY, the stock is approaching an overfished condition.
- If the average ending stock size in these simulations is above BMSY, the stock is not approaching an overfished condition.

2.3.4.2 Stock and Recruitment

Recruitment is the only source of replenishment for the numbers of individuals in the fished portion of a population. Biomass may be increased by somatic growth, but the biomass of a cohort is also a function of the number of individuals in that cohort. Thus, recruitment can be viewed as one process by which fished populations are maintained and their future status assured. The factors and processes that determine recruitment have been a source of extensive discussion and debate in fisheries biology. The debate has focused largely on three questions: (1) is the process of recruitment density-independent or density-dependent, (2) if density-dependent, what is the nature of the relation between recruitment and stock size, and (3) the role of environmental factors in determining recruitment.

Current harvest policies under the FMP are based on a single-species approach to fisheries management designed to be precautionary. However, if the distribution of recruitments that would be produced from biomasses in the $B_{20\%}$-$B_{40\%}$ range is significantly different from the distribution of recruitments observed since the 1976-1977 regime shift, and if environmental conditions are such that an anomalous string of poor recruitments pushes the stock well below B40% despite harvest rates being set conservatively, then it is possible that a stock could be below the MSST without management knowing it.

When stock-recruitment relationships are uncertain, the FABC and FOFL are based on estimates of current stock status and considerations of spawning per recruit (SPR). A designation of the form “$F_{X\%}$” refers to the F associated with an equilibrium level of spawning per recruit (SPR) equal to X% of the equilibrium level of spawning per recruit in the absence of any fishing. The use of SPR analyses to derive biological reference points for fisheries management has undergone broad scientific review and is used to form the basis of harvest control rules in several systems throughout the world (Clark 1991, Clark 1993, Thompson 1993). The use of F35% as a proxy for FMSY stems in part from the work of Clark (1991) who showed that a large fraction of the potential yield from a typical groundfish stock could be obtained at a rate of F35% across a discrete set of plausible stock-recruitment relationships, including both Ricker and Beverton-Holt forms. Subsequent analyses showed that F40% would reduce the probability of low biomass if recruitment was highly variable or autocorrelated (Clark 1993). Research continues to refine estimates of biological reference points. For example, analyses have focused on considerations of reproductive rates at low stock sizes (Myers *et al.* 1996) and applications of Clark’s general approach to species that possess similar life history characteristics (Dorn 2002).

2.3.4.3 Natural Mortality

Natural mortality (M) refers to the rate of decline of a stock as a consequence of natural processes. These include predation by other fishes, marine mammals, and seabirds, as well as some level of mortality due to disease, injury, starvation, etc. The relation between M and fishing mortality (F) is an important consideration in the fishery management strategy. Ironically, natural mortality is one of the most difficult parameters of a population to estimate.

Natural mortality (M) is a fixed parameter and not estimated in the pollock, Pacific cod and Atka mackerel stock assessments. For Eastern Bering Sea (EBS) pollock, the reference model has assumed fixed natural mortality-at-age values of M=0.9, 0.45, and 0.3 for ages 1, 2, and 3+ respectively since at least the 2005 assessment (Ianelli *et al.* 2005, 2008). The EBS mortality-at-age values were originally estimated in a cohort analysis by Wespestad and Terry (1984). For GOA pollock the stock assessment assumed a fixed natural mortality rate of 0.3 for all ages (Dorn *et al.* 2005). The GOA value of 0.3 is based on an analysis by Hollowed and Megrey (1990) which estimated natural mortality using a variety of methods. For EBS and GOA Pacific cod, the stock assessment presented three models; Models 1 and 2 assumed a value of M fixed at the traditional value of 0.37 and Model 3 estimated M internally (Thompson and Dorn 2005). The 2006 Pacific cod ABC is based on Model 2 which assumed a fixed value of 0.37 for all ages. The value of 0.37 was originally estimated in the 1993 BSAI Pacific cod assessment (Thompson and Methot 1993), and all subsequent assessments of BSAI and GOA Pacific cod assessments used this value, with one exception in the GOA. However, based on further analysis, in the 2007 assessments, the value of M was set at 0.34 for BSAI Pacific cod and at 0.38 for GOA Pacific cod (Thompson *et al.* 2008) and these values were also used in the 2008 and 2009 assessments. The BSAI Atka mackerel assessment assumed a fixed value of 0.3 for all ages based on the regression model of Hoenig (1983, Lowe *et al.* 2005), which is based on the longevity of the species. Lowe and Fritz (1997) explored several alternative methods to estimate natural mortality for Atka mackerel; the current assumed value of 0.3 is consistent with values derived from a variety of methods (Lowe *et al.* 2005).

In the single species stock assessments model there is no explicit accounting for other consumers. Hollowed *et al.* (2000) developed a model for GOA pollock that explicitly includes predation (e.g., consumption by arrowtooth flounder, Pacific halibut, Steller sea lions, and Pacific cod). The total natural mortality (predation plus residual M) was higher than the assumed M used in the pollock assessment. The role of pollock as prey in the GOA ecosystem cannot be fully evaluated using a single species assessment model (Hollowed *et al.* 2000). The current pollock assessment includes both a single species model, and an ecosystem considerations section that includes results from ecosystem models (Dorn *et al.* 2005). The 2008 BSAI Pacific cod assessment includes a similar appendix.

Harvest policies in the North Pacific take into consideration a variety of factors including fishing mortality rates that reduce the level of spawning per recruit to some percentage of the unfished level ($F_{X\%}$). For many stocks, 40% of the unfished spawning biomass per recruit is used, i.e., the harvest policy is based on an F40% fishing mortality rate. There is a positive relationship between M and $F_{40\%}$. The $F_{X\%}$ harvest policies are designed to maintain sufficient spawning biomass to ensure adequate recruitment to the stock. If M is higher, an average recruit would not live as long and thus it would produce less spawning biomass over its lifetime. Consequently, a higher fishing mortality rate is needed to reduce spawning biomass to 40% of the unfished level. Clark (1999) found that specifying a conservative (lower) natural mortality rate is typically more precautionary (from a single species perspective) when natural mortality rates are uncertain.

Stock assessment models are used to project these stocks based on the assumption of constant natural mortality rates. TACs are set each year at values consistent with the harvest control rules and other provisions of the FMPs (e.g., the OY caps). For some stocks in some years, this amounts to fishing at the maximum permissible ABC. In such instances, the recommended fishing mortality rate typically varies directly with M. For example, if the intent is to fish at a rate of F40% and M happens to be over-estimated while all other parameters are estimated without error, the recommended fishing mortality rate will exceed the true value of $F_{40\%}$. However, over-estimation of M leads not only to errors in the estimate of F40% but to errors in the estimate of stock size as well. Errors in estimated stock size resulting from over-estimation of M can be either positive or negative (Thompson 1994). The combined effects of these two errors can result in a recommended short-term catch that is either higher or lower (Thompson 1994) than the short-term catch corresponding to the intended harvest strategy. In the long term, however, catch tends not to be sensitive to error in M except when gross under-estimates occur, in which case catches tend to be lower than those corresponding to the intended harvest strategy.

The effect of reductions in prey biomass on other consumers in the environment has received little treatment in traditional fisheries management but there is literature on this issue which we review and discuss in the effects section of this opinion.

2.3.5 Ecosystem Considerations

The MSA and implementing regulations require that relevant social, economic, and ecological factors be considered in the setting of OY for a fishery. The regulations (50 CFR § 600.310 (f)(3)(iii)) provide the following examples of ecological factors:

> "stock size and age composition, the vulnerability of incidental or unregulated stocks in a mixed-stock fishery, predator-prey or competitive interactions, and dependence of marine mammals and birds or endangered species on a stock of fish. Also important are ecological or environmental conditions that stress marine organisms such as natural or manmade changes in wetlands or nursery grounds, and effects of pollutants on habitat and stocks."

The FMP process considers the species managed under it as parts of functioning ecosystems. However, ecosystem management is extremely complex. In setting the harvest rate, managers also attempt to be sufficiently protective of the larger ecosystem in which the harvesting occurs. An Ecosystem Considerations chapter has been added to the SAFE documents since 1995 and is evolving to be more operational, and other multi-species studies have been undertaken. Ecosystem indicators are being evaluated, and multi-species models have been constructed. The multi-species models have allowed consideration of ecosystem impacts in a way that single-species models cannot. They are not viewed as a replacement of the single-species approach, which remains the primary determinant of ABC specifications, but rather they add insight into potential ecosystem effects. To date, the multi-species modeling studies reported in the SAFE and Supplemental EIS documents have suggested that fishery impacts on fish species in the BSAI and GOA systems seem to be about the same order of magnitude as what is shown in single-species models. Nevertheless, it is known from theoretical models of harvest dynamics in a predator-prey-competition system that harvesting at single-species MSY levels will not achieve MSY for the aggregate because of species interactions. This knowledge is one of the reasons that the BSAI OY cap was set at 85% of the single-species MSYs.

2.4 Annual Fisheries Assessment

The annual fisheries management cycle consists of activities that can be grouped into three main functions: (1) resource surveys, (2) stock assessment and setting the ABC, and (3) setting the TAC levels. Target species were discussed in Section 2.2.2 and are provided in Table 2.2.

The purpose of stock assessment is to describe those stocks that are targeted by the fisheries and the nature and magnitude of fishery effects on those stocks (i.e., the stocks' tolerance for fishing). Consistent with the fundamental approach to fishery management, two of the primary objectives of stock assessments are to estimate biomass and the size-age structure of target stocks. The following sections provide a basic description of the stock assessment process.

2.4.1 Resource Surveys and Biomass Assessment

Stock assessment consists of two main functions: (1) determining the status (a measure of population size and trend) of the stock, and (2) evaluating its tolerance to fishing. Resource surveys, along with the fishery observer program and catch statistics, are essential for assessment of the stocks fished under the BSAI and GOA groundfish FMPs. In general, these surveys involve deployment of standardized sampling gear according to consistent protocols to catch or measure fish abundance or biomass at a particular location. Estimates of overall fish abundance or biomass are then based on average catch rates per sampled location multiplied by the size of the total area. The results can be expressed as an index or estimate of abundance or biomass. Results from single surveys may be used separately to generate such indices/estimates, or results from multiple surveys may be combined.

2.4.1.1 Groundfish Surveys

The purpose of the groundfish surveys is to estimate the distribution and abundance and age structure of groundfish species. This information is essential to the annual stock assessments used in the determination of the annual ABCs and TACs. Current surveys are designed to provide information to manage groundfish harvests on a single species basis. Management of groundfish harvests on a multi-species level requires additional information and understanding that are not currently available.
Three types of surveys are currently conducted, including bottom trawl for shellfish and bottom fishes, hydroacoustic or echo integration-trawl (EIT) for pollock, and longline for bottom fishes (e.g., sablefish) of the deeper waters of the continental shelf and slope. Summer bottom trawl surveys of the EBS Bering Sea shelf have been conducted annually since 1972, with the current standardized time series beginning in 1979. These surveys follow a systematic grid of sampling stations. A triennial bottom trawl survey of the Bering Sea continental slope was conducted triennially from 1979 to 1991 and then resumed on a biennial schedule since 2000 (the planned 2006 survey was canceled due to lack of funding). Triennial summer bottom trawl surveys for the Aleutian Islands and the GOA shelf and upper continental slope began in 1980 and 1984, respectively. In 1999 the GOA was changed from a triennial to a biennial bottom trawl survey and extended to cover the continental slope out to 1,000 m depth. The Aleutian Islands survey moved to a biennial schedule in 2000. The GOA, Aleutian Islands, and EBS continental slope surveys are based on area and depth-stratified random sampling among a set of predetermined stations.

EIT surveys in the Bering Sea and GOA have been conducted on a series of winter and summer annual and biennial surveys. Annual winter EIT surveys were initiated in 1981 to study abundance of spawning pollock in Shelikof Strait (except in 1982 and 1999), and in 1988 to study pollock abundance in the vicinity of Bogoslof Island (except in 1990 and 2004). Winter surveys have also been conducted in the Gulf of Alaska in the Shumagin Islands/Sanak Trough area in 1994-96 and 2001-2006 and on the shelf break east of Kodiak Island in 2002-2006. Summer EIT surveys of pollock on the Bering Sea shelf were surveyed triennially from 1979 to 1994 and in 1996, 1997, and 1999 and then changed to a biennial schedule in 2000.

Summer longline surveys were initiated by Japanese scientists in 1979 to assess sablefish abundance over the upper continental slope in the GOA. These surveys are now conducted by U.S. scientists, and have been extended to the Aleutian Islands and the EBS slope, where they are conducted in alternate years. Current surveys are as follows:

1. Annual summer bottom trawl surveys of the EBS shelf,
2. Biennial summer bottom trawl surveys in the Aleutian Islands and GOA (shelf and continental slope),
3. Annual summer longline surveys for estimation of sablefish abundance,

4. Annual winter EIT surveys in the Bogoslof, Shelikof, Shumagin Islands/Sanak Trough, and shelf break east of Kodiak areas,
5. Biennial summer EIT survey of EBS shelf pollock, and
6. Biennial summer EIT survey of GOA pollock.

Ianelli *et al.* (2008) note that scientific research catches are reported to fulfill requirements of the MSA. Different kinds of surveys are undertaken to provide estimates that aid in the assessment of the distribution, abundance or stock biomass of groundfish stocks. In addition, they also provide important information on age and sex composition, recruitment of young fish to the fished stock, length and weight at age, reproductive status or condition, food habits, and other pertinent biological characteristics. Assessment of each of these parameters may be affected by sampling variability, measurement error, or systematic bias. Considerable effort is directed at minimizing measurement error and bias, but sampling variability may still occur and be evaluated and reported to provide an indication of the confidence with which final parameter estimates may be used. Table 2.3 provides an indication of the sampling variability observed for each assessed stock. The error is expressed as the coefficient of variation (CV) which is equal to ([standard error/estimate]*100).

Survey information is used to spatially allocate TACs to management areas. Surveys in the GOA and Aleutian Islands are used to allocate TACs in proportion to biomass. The use of commercial fishing vessels for acoustic surveys of pollock in the Aleutian Islands was investigated by the AFSC to determine if the data from such surveys could be used for near real time management of pollock fishing in limited areas (NMFS 2006)(http://alaskafisheries.noaa.gov/analyses/efp/aecpollock1206ea.pdf). This work was conducted under an exempted fishing permit and with support from the NRPB. The NPRB-funded study determined that information from commercial fishing vessels was reliable and comparable to information collected by a NOAA research vessel, the Oscar Dyson. No more field work on this issue is planned and the final report is under review by the NPRB as of January 2010. The results show that it is feasible to use a commercial fishing vessel to do pollock surveys in a localized area to determine biomass that may be used to set harvest levels and manage fisheries in a local area. This method is unlikely to be further pursued due to lack of pollock fishing in the Aleutian Islands (S. Barbeaux, AFSC, personal communication December 2009).

2.4.1.2 Stock Structure

Research on stock structure for groundfish species is ongoing (e.g., Hinckley 1987, Bailey *et al.* 1999, Kotenev and Glubokov 2007, Cunningham 2007, Cunningham *et al.* 2009). Currently, the best available information is based on limited tagging data for sablefish, Atka mackerel, and Pacific cod and morphometrics or genetic studies for pollock, Atka mackerel, Pacific ocean perch, and a few other rockfish.

Pollock will be used in this section as an example to describe some of the patterns in stock structure that have been observed in the past. Pollock in the BSAI are managed as three units: EBS, Aleutian Islands, and the Aleutian Basin/Bogoslof Island (Basin). Recruitment to the Basin stock is thought to occur primarily as density-dependent migration of pollock from the EBS shelf stock. Large cohorts of shelf pollock appear to be the source of most of the pollock in the Basin, which suggests that the Basin stock itself is not self-sustaining. Fishing on the Basin stock was terminated in 1992 by international agreement, but it has since failed to recover. Given the reduced recruitment in the 1990s compared to the large year classes in the late 1970s and 1980s, the Basin stock would have been expected to decline in size even in the absence of fishing. The extent to which spawning in the Bogoslof region contributes to recruitment of the shelf stock is unknown. For example, overfishing in the basin may have exacerbated the decline of the Basin stock, and it may have adversely affected recruitment in the shelf stock.

Pollock stocks in the Aleutian Islands region have also declined since the mid-1980s, from a high of 496,000 mt in 1983 to 105,000 mt in 1997 and 130,000 mt in 2004 (Barbeaux *et al.* 2005). Since the decline of pollock in the Aleutian Islands parallels that of the Basin, the two stocks may be closely related. Several explanations for the lack of population recovery in the Aleutians might be explained primarily as a series of years with poor recruitment. Barbeaux *et al.* (2005) describe the pattern of pollock fishing in the Aleutians in the 1990s, where the fishery moved increasingly westward apparently because spawning aggregations in the eastern portion had disappeared (i.e., around Kanaga Island and in Amukta Pass). It is not known whether spawning from these basin aggregations contributed to the Aleutian stock. The degree to which Aleutian Islands pollock abundance depends on movement from the EBS is also unknown. It is possible that the EBS fishery causes some interception of potential Aleutian Islands recruitment.

2.4.1.3 Stock Complexes

Under the FMPs, many stocks have been placed in complexes (i.e., groupings of stocks). Uncertainty is a concern for species managed in complexes because they often are placed into complexes if the available information is insufficient to manage a species as a single target stock. The risk of fishery effects on a single species may be greater when the species is fished as part of a complex.

Numerous species are incorporated into management complexes: GOA deep-water flatfish (3 species), GOA shallow-water flatfish (8+ species), GOA other slope rockfish (12+ species), GOA pelagic shelf rockfish (3 species), GOA demersal shelf rockfish (7 species.), GOA other skates (6 species), GOA other species (sculpin, sharks, squid and octopus), BSAI other flatfish (15 species), other rockfish (7 species), and BSAI other species (sculpin, sharks, skates, and octopus)(see Chapter 3 in the BSAI FMP Table 3-1) (Table 2.2).

Some of the large complexes listed above (e.g., BSAI and GOA other species) are composed of a very diverse assemblage of species, some of which are prey for listed species (e.g., squid, octopus, and sculpins). While the magnitude of fishing effects on any single species in the other species assemblage is not thought to be large given the group catch amounts, the limited or non-existent information on the status or catch of any single species makes this determination uncertain. One example of precautionary management that addresses this is the establishment of retention thresholds and prohibition on directed fishing for forage fish (e.g., osmerids and myctophids) to prohibit the establishment of new commercial fisheries. In general, the ecological consequences of fishing on groundfish complexes cannot be evaluated very precisely due to the lack of data on the stock structure of individual species.

2.4.1.4 Species and Stock Distribution

As noted in the above description of stock surveys, information on the distribution of affected (fished and unfished) stocks is vital to assessment of stock status, fishery and environmental effects on stocks, and ways in which fisheries can affect predators of those fish stocks. The distribution of a species is an important determinant of the ecological role it plays in local marine communities, including availability to predators.

2.4.1.5 Stock Biomass

Biomass is used to describe or estimate stock status and trend, tolerance for fishing, and reproductive capacity. A fishing mortality rate for a species is set on the basis of its effect on target stock biomass and its reproductive capacity. That is, the fishing mortality rate is intended to maintain the species at BMSY or a proxy for it (B35%). Further, the stock-recruitment relation fundamental to the MSY concept is currently typically based on recruitment as a function of spawning biomass. Thus, stock biomass is clearly an important measure of the stock and a basis for evaluating potential fishery yields. Female spawning biomass from 1980 to 2005 is provided for pollock, Pacific cod, and Atka mackerel in Table 2.4; also shown is the current B100% level, and the relationship to that benchmark in each year. Accurate estimates of stock biomass depend both on information from surveys and from the fishery (total removals and catch age composition). Estimates of stock biomass for the early years of the pollock fishery are uncertain because of limited and potentially biased information from both sources. In the Bering Sea, the trawl survey began in the late 1960s, but the survey was initially designed to survey crab populations and did not encompass the range of the pollock stock (Bakkala *et al.* 1985, Megrey and Wespestad 1990). In 1975, the survey was expanded to cover most of the EBS shelf, and has been conducted annually since 1979. Catch information from the foreign fishery during the 1970s was submitted by the fishing nations at bilateral meetings or under provisions of the International North Pacific Fisheries Commission. Since this was prior to the development of fisheries observer programs, there was no way to verify the accuracy of the catch information, and there were often questions about the credibility of some the reported fisheries data (Megrey and Wespestad 1990).

Based on a 1999 pollock assessment (Ianelli *et al.* 1999), pollock age 3+ biomass in the 1970s ranged from 5.2 mmt (million metric tons) in 1971 to 2.0 mmt in 1974 (Figure 2.5). In contrast, Megrey and Wespestad (1990) reported that pollock in the EBS ranged from about 8 mmt to 12 mmt for the same time period. The precision of the Ianelli *et al.* (1999) estimates is depicted by the 95% confidence intervals in Figure 2.5, which suggest that biomass in 1970s may have been as high 7.1 mmt (in 1971) or as low as 1.1 mmt (in 1974). These estimates of uncertainty are only approximate and also rely on assumptions of known natural mortality, relatively precise and unbiased total catch estimates and correct model specification. Therefore, the actual variance is likely to be larger than that indicated in Figure 2.5 (National Research Council [NRC] 1996). Furthermore, fishery selectivity estimates from Ianelli *et al.* (1999) were allowed to vary over time to reflect the fact that the fleet composition has changed over time from foreign vessels to joint venture operations to the current domestic fleet. This increases the overall variance of the model. Another effect of time-varying fishery selectivity can change the interpretation of “available” biomass and simple exploitation rates comparing total catch compared to age 3+ biomass. For example, in 1974 about 23% of the “available” biomass was aged 1 and 2. This was quite high and compares to an average of 3% for the entire period 1964-1999. This is due to the fact that the 1972 year class was quite strong and that the gear selectivity at that time was more concentrated on young pollock. Harvest rates during the mid-1970s is depicted by Ianelli *et al.* (2005) to be about 40% with the 95% confidence interval near 50% (Figure 2.5).

At present, biomass estimates or indices are available for 37 of the 45 species or species groups listed in Table 2.3. For approximately 21 out of 45 of these stocks, biomass by age is not available. However, no groundfish stock in the BSAI or GOA is currently being subjected to overfishing (a fishing mortality rate higher than the maximum allowable rate) and regardless of the level of information on each species, given an absence of a history of overfishing, it is unlikely that any stock would be in an overfished condition defined using the single species criteria (biomass has fallen so low that a special rebuilding plan is needed). Again, to address the question of whether harvests based on imperfect biomass information for groundfish stocks affects listed species (for example biomass estimates are not available for 8 of the 45 species in Table 2.3), it is informative to go back to the ecosystem concept and relate it back to foraging behavior of the listed species. In general, the stocks for which the least information is available are lightly fished, of relatively low abundance, and among the least commercially desirable stocks.

2.4.2 Stock Assessment

Each year, scientists from NMFS and ADF&G collect data, and compile and update databases on catch, age and size composition, and survey biomass. Stock assessment scientists from these agencies analyze the data and calculate estimates of key population parameters. In most cases, contemporary stock assessment models are constructed to integrate the scientific information, except when information is not sufficient for model construction. The techniques of stock assessment are summarized in the texts by Hilborn and Walters (1992), Quinn and Deriso (1999), and Haddon (2001). An overview of issues related to stock assessment points out the difficulties and challenges (NRC 1998). The processes of stock assessment and harvest strategy development are interrelated. Stock assessment parameters are used in development of the harvest strategy, and the current biomass-based harvest strategy utilizes the most recent biomass estimates in determining ABC, OFL, TAC, and whether overfishing is occurring. A goal of harvest strategy development is to provide a stable, quantitative set of control rules for operating the fisheries, and a goal of stock assessment is to use the best available scientific information to determine the status of the population in reference to the quantities that are inputs to the rules.

2.4.2.1 Modeling

The second major process in stock assessment is modeling of each stock to further describe its status and investigate its tolerance to fishing. The information required for modeling comes from the stock surveys, from the fisheries themselves, and from other studies. For a given target stock, the objective of modeling is to (1) estimate the state of the population by creating a simulated population that is most consistent with the data on the wild population, and (2) estimate the tolerance of the wild population to fishing based on the characteristics of the simulated population.

Models can take a large variety of forms, but in essence they all serve the same purpose: they allow thoughts, theories, and data (observations of the world) to be organized and simplified such that complicated issues can be cut through and clear logic applied. Theoretical models may be used to follow through to logical conclusions. Statistical models may be used to “fit” data and estimate parameter values to be used elsewhere. Simulation models may be used to combine theory, knowledge and data to consider what might be and to ask “what if?” questions. Models as used in ecology and fisheries are often highly complex, using state-of-the-art mathematics, statistics and computing approaches, but they always represent major simplifications of real systems.

In a deterministic model all processes are treated as completely predictable in principle. Therefore, if all parameters are known and fixed, a deterministic model run repeatedly from the same starting point will repeatedly result in the same sequence of outputs. In fact, this is only partially true–some deterministic models can behave chaotically (apparently randomly within bounds) for certain parameter inputs and can actually be used as “random number” generators. In a stochastic model, there is random variability in some of the parameters or processes. Running a stochastic model many times will, therefore, result in different outcomes. Stochastic models are in principle closer to reality, but only if the variability can be properly incorporated; this is very difficult and makes stochastic models difficult to set up and apply. A stochastic model may be fitted to data from the history of a population, but it will not predict a unique future for that population.

Stock assessment models used in fisheries are standard tools of fisheries science. Single species stock assessment models are used to consider the data collected from fisheries or research on fish stocks. Those data contain information on how fish age, grow and mature, how fish die and how fisheries select fish of different sizes or ages. The data, though, are never perfect and there are always many things that assessment scientists have to make assumptions about, often based on experience elsewhere. What the assessment models do, given data, assumptions and prior knowledge, is allow inferences to be made about the past and present state of stocks. This allows scientists to advise managers as to the status of stocks: whether or not stocks have been, or are currently, overfished, and whether or not overfishing is taking place. In addition to assessment models to determine stock status, it is common also to forecast the future state of stocks under different catch levels or rates. Forecasting involves updating the estimated current status using assumptions or models to determine how many new fish (recruits) there will be in the future.

Assessment models and forecasts may be deterministic or stochastic and they take many different forms. Production models represent the state of a stock by a single variable (stock biomass) and estimate production (yield, or catch) from its relationship with biomass. Age (or size) structured models represent the state of a stock by the number of fish in each age (or size) class. They differ from production models in that whilst a stock may have had the same biomass at different points in history, the yields produced would have been different because the stock would have been made up of differently aged (or sized) fish. A deterministic model is in equilibrium when all of the variables stay the same from year to year. This kind of constancy does not occur in the real world; it is an attribute of a model. Although equilibrium results are hypothetical, they are nevertheless widely used to obtain reference points for fishery management. A production model would be in equilibrium once the catch equals the yield, because this will maintain the biomass at a constant value. An age (or size) structured model, however, would only be in equilibrium when the numbers of fish in each age (or size) group in both the stock and the catch is the same each year. The equivalent to equilibrium for a stochastic model is a stationary distribution, where the relevant variables exhibit a kind of consistent range of variation over time, though they are not constant. Analysis of stochastic models is more involved than analysis of deterministic models, and even the definition of appropriate indicators of good performance requires much more thought with stochastic models. Nevertheless, variability is a feature of the world, so there is merit to examining reference points from the perspective of stochastic models.

Different models or modeling approaches have been, and are currently used, for the stocks fished under the BSAI and GOA groundfish FMPs (Table 2.3). There is an ongoing iterative process to improve models based on both the best information about the stock and analytical methodologies. In general, these models include a range of elements from simple numerical or accounting procedures to complex mathematical functions. The nature and blend of these elements depends, in part, on the information that is available and the preferences of the scientist(s) modeling the stock. Nonetheless, all have the same general purpose of describing the wild stock and evaluating its tolerance to fishing.

The stock synthesis approach has been an important modeling tool in past assessments and is still used currently for a few species. The approach was developed by Methot (1990) to conduct an age- or length-structured analysis using life history, catch, survey, and other information, as well as the level of uncertainty in such information. Given a set of values for the model parameters (e.g., annual fishing mortality rates and recruitment), a simulated stock is created and subjected to simulated fisheries and surveys for comparison with the real catch and survey data. The degree of similarity between the simulated data and the real data is referred to as the “goodness of fit,” which is expressed in terms of a “likelihood”. The likelihood is then assessed as the probability of the data given the model parameters. The best simulated population (i.e., the one in most agreement with the data) is found by adjusting the model parameters of the simulated population until the likelihood expression is maximized (accomplished using a computer “optimization” routine). The stock assessment authors then complete their assessment by weighing and considering the best simulated population, along with other reasonable or possible model outcomes.

AD Model Builder is essentially a set of pre-programmed computer subroutines that enable faster and more reliable estimation of various parameters used in stock assessment modeling and which also enable efficient calculation of the probabilities of alternative parameter values. The equations representing population dynamics and statistical likelihood in models developed under AD Model Builder can take exactly the same form as those in the stock synthesis approach or they can take different forms, thereby enabling exploration of alternative modeling assumptions. In effect, AD Model Builder expands the capabilities of the stock assessment modeling efforts.

"Survey index modeling" encompasses a variety of assessment approaches that are used to describe the wild population and its tolerance for fishing when the available data are too limited to conduct a full age- or length-based assessment. They are frequently based on indices of the population derived from survey estimates alone.

Where the data allow, the general modeling approach is to create a simulated population of a particular size (number) and age/sex composition. That is, the model is based on year-classes or cohorts. A new cohort enters the model population in each year of the simulation. The numerical abundance of a cohort at the age where it first enters the model population is usually treated as a parameter to be estimated by the model. This is sometimes referred to as "recruitment" to the model population, which may occur at a different age than recruitment to the surveyed population or recruitment to the fished population. For example, for a particular stock the model population might begin at age 1, even though fish in that stock are seldom detected by the survey before age 2 or caught in the fishery before age 3. After the age of recruitment to the model, each cohort decays over time due to natural mortality and fishing mortality (when appropriate). As a cohort ages over time in the model, the average length, weight, maturity, and selectivity of fish in the cohort are assumed to vary in predictable fashion. In the wild, these functions may vary unpredictably under a number of influences, including density-independent factors (e.g., environmental conditions) or density-dependent factors (e.g., stock size). In modeling, however, these functions are often treated as fixed or constant parameters, although some models allow length at age to be described by a statistical distribution and allow selectivity to vary over time. The processes of growth, maturation, reproduction, natural mortality, fishing mortality, and recruitment are described in further detail below.

Growth

Individuals in a cohort grow over time. Information on physical size and growth is important because the simulated and wild populations consist of numbers of individuals, but harvests are measured in terms of biomass. Thus, growth information is necessary to convert numbers available to biomass available. Growth is assessed using samples taken during surveys and from the fisheries catch. The estimated relations may include length as a function of age, weight as a function of age, or weight as a function of length. Age is usually estimated using the ear bones (otoliths), which exhibit annual growth layers or rings. Weight at age and numbers at age are necessary to determine overall biomass. Weight also appears to be an important determinant of fecundity (number of viable eggs produced by a female).

Maturation

Maturation is an expression of the reproductive capacity of an individual. While individuals are generally described as "immature" or "mature" (i.e., fully one or the other), maturation may involve physiological and behavioral changes that are not abrupt but transition over a period of time. For example, young females in the process of maturing may be able to produce eggs, but those eggs may not be as viable as the eggs of an older female. Maturation is expressed most often as a function of age but, weight may also be an important determinant of the maturation process. Maturity is assessed using samples taken during surveys and from the fisheries catch. Maturation of all individuals in a cohort may occur over a single year or over a period of several years.

Reproduction

As females mature they begin to produce eggs. The number and viability of a female's eggs determine the contribution of that female to the new cohort. However, the size of the cohort at recruitment age is also a function of environmental (e.g., currents, temperature) and ecological (e.g., predators, prey) factors that determine growth and survival from fertilization to recruitment. Depending on the method used for modeling recruitment, reproductive functions may or may not be essential or important for the modeling effort. For example, if recruitment is modeled as a density-independent random variable based on estimates of past recruitment, then reproduction by adult females need not be included explicitly in the model.

Natural Mortality

Natural mortality refers to the instantaneous rate of decline of a population or cohort due to natural causes such as disease or predation. The rate of decline may vary as a function of age, but for most fish populations harvested in the BSAI and GOA groundfish fisheries, natural mortality is generally treated as constant for cohorts at or above the age of recruitment to the fishery. In most age- or length-structured stock assessments the natural mortality rate is assumed to be known from previous studies, although occasionally it is estimated within the stock assessment model itself. For fish populations, natural mortality is most often expressed as M in the function

$$N_1 = N_0 * e^{-(\mathrm{M} + \mathrm{F})},$$

where N_0 and N_1 represent numbers at time 0 and time 1.

Fishing Mortality

F in the above equation, is the instantaneous rate of decline of a population or cohort due to fishing. Age- or length-structured stock assessment models estimate annual fishing mortality rates for each year or season in a time series as parameters of the model.

Recruitment

Recruitment is the process by which fish enter some portion of the population, such as the portion available to the fishery. The process may be defined in terms of the age or size of the fish, which are usually closely related. The numbers or biomass of fish recruited to the fishery in a given year is determined by the quantity and quality of reproductive output by mature fish, plus factors that affect the growth and survival of individuals from fertilized egg up to recruitment. Defining the age of recruitment to the model population is largely a matter of convenience and may be governed by such considerations as the youngest age observed in the survey or the youngest age above which natural mortality can reasonably be viewed as constant. Above the age of recruitment to the model population, most stock assessment models treat fishery selectivity as a continuous function of age or size, making designation of "the" age of recruitment to the fishery a somewhat tenuous exercise.

The modeling of recruitment is a crucial component of population models used for fishery evaluation and projection. The population models used for these fished stocks are "closed" in the sense that they do not include immigration or emigration in or out of the population (except for the possibility that recruitment to the model population could potentially include an immigration component). Therefore, as cohorts are stepped through time (years) they can only diminish in numbers due to natural or fishing mortality. In terms of numbers, the stock or population is replenished only through the addition (recruitment) a new cohort each year.

Recruitment can be incorporated into fisheries models in a variety of ways, two of which will be described here. First, recruitment can be modeled as a function of the reproductive stock (based on either numbers, biomass, or other measure of reproductive capacity) (Figure 2.3). The shape of an assumed or demonstrated stock-recruitment function is a crucial consideration in modeling recruitment.

The second approach to modeling recruitment is to assume that it is independent of stock size (i.e., density independent). For BSAI and GOA groundfish, the assumption is that while spawning biomass (used as a proxy for number of eggs produced) may be an important determinant of subsequent year class strength when stock size is low, spawning biomass in not an important determinant of subsequent year class strength at stock sizes typically observed. Because stock-recruitment functions have not been identified for the majority of stocks fished under the BSAI and GOA groundfish FMPs, recruitment is modeled as a density-independent random variable based on past recruitment levels.

In summary, the significance of these processes in the model depends on the sensitivity of model results to each function and the extent to which the real processes are appropriately and accurately represented in the modeling process. Again, all of the above processes except recruitment are incorporated into the models as fixed rates or schedules, some estimated within the model and others estimated from separate studies. Recruitment is the only process that is treated stochastically in the current set of BSAI and GOA age-structured models. Uncertainty is incorporated into the model for input data collected in the field (e.g., catch at age, age-length relation, survey biomass).

2.4.2.2 Target Harvest Rates

The TAC-setting framework establishes B40% as a reference point in defining the maximum permissible value of ABC. Stocks above that level may be reduced through harvesting. Stocks below that level may still be harvested, but at reduced rates to allow the stock to recover over time to B40%. If the stock is falling well below B40%, the Plan Team and SSC may recommend reduced fishing pressure on the stock, including up to no directed fishing on pollock, Pacific cod, and Atka mackerel stocks that are approaching B20%. For other groundfish stocks, the Plan Team and SSC may recommend reduced fishing on the stock, including up to no directed fishing on the stocks that are approaching B 2%. Regulations currently allow bycatch to be taken for most stocks unless a stock reaches its ABC, in which case the directed fishing on that stock would be prohibited, commonly referred to as putting the stock on “prohib status.” If the stock is approaching its OFL, NMFS also may consider prohibiting directed fishing on other groundfish species where incidental catch of the prohibited stock may occur. For Steller sea lions, a modified control rule was implemented in 2001 which halts all directed fishing for pollock, Pacific cod, and Atka mackerel if spawning biomass is estimated to be below 20% of pristine levels (NMFS 2000). Figure 2.6 provides an example for GOA pollock which further includes additional conservation measures implemented by the assessment author (Barbeaux *et al.* 2005). This ensures that as biomass decreases to relatively low levels, there will no longer be any directed fishing effect on these important prey species for Steller sea lions (NMFS 2000). This does not ensure that biomass will not drop further below B20%; poor recruitment, bycatch, or increased natural mortality could drive the stock even lower, but the modified control rule does remove the effect of a directed fishery.

No stocks in Tiers 1 to 3 have approached the 2% level in the history of the FMPs. In 2003, GOA pollock biomass reached a low of 26% of theoretical pristine levels (Dorn *et al.* 2005). In 2010, EBS pollock biomass is estimated to be at about 27% of theoretical unfished levels (BSAI 2009 SAFE report). Figure 2.5 (particularly the lower figure from the BSAI Pollock SAFE) shows that spawning exploitation rate (SER-annual percent removal of spawning females due to the fishery) was below 20% from 1980 – 2006 (except 1990), went well above 20% from 2006-2008, and then back below 20% in 2009 due to reductions in TAC from the ABC control rules.

Stocks in Tiers 1 to 3 can be evaluated with respect to the reference points in Figure 2.7 (B_{MSY}, or the proxy $B_{35\%}$, ½ B_{MSY}, and $0.05B_{MSY}$). None of these values can be estimated for stocks in tier 4. Thus, the status of stocks in Tier 4 cannot be determined relative to an unfished level, nor can they be determined relative to their MSST.

Stocks in Tier 5 cannot be assessed with respect to their unfished level or their MSST. These stocks can be harvested at an FABC of 0.75*M. To evaluate the potential effect of this strategy on a Tier 5 stock, an example was developed using an M value of 0.3, age of recruitment of three, and a growth schedule consistent with pollock (Ianelli *et al.* 1999). Harvesting at F = M*0.75 would reduce the spawning stock biomass to about 50% of its unfished level under this scenario.

Stocks in Tier 6 also cannot be assessed with respect to their unfished level or their MSST. BSAI squid, sharks, and octopus, as well as GOA Atka mackerel and deep-water flatfish other than Dover sole, fall into Tier 6. The Tier 6 guidelines suggest that the OFL should be set at the mean catch from 1978 to 1995, unless an alternative (unspecified) level is set by the Council's SSC. The ABC level is then set at 0.75*OFL. While these guidelines would not necessarily ensure the protection of a stock in Tier 6, catches of squid in the BSAI and GOA (less than 700 mt in 2009 per NMFS inseason reports accessed on 12/2/09) are relatively low compared to squid biomass estimates based on predation models in the EBS (Sobelevsky 1996). The harvest control rules are based on the assumption that a stock that has tolerated a certain mean level of catch for a long time can continue to tolerate that level (or that level times 0.75) indefinitely. The overfished status of the stocks in Tier 6 cannot be determined.

2.4.2.3 Uncertainty

Uncertainty is inherent throughout the process by which TACs are set and is documented in the associated NEPA documents. Biomass and projected yield estimates are uncertain. For EBS pollock, the model specified for providing these estimates has been extensively evaluated for sensitivity to assumptions and was selected to reflect uncertainties due to alternative models (Ianelli *et al.* 2005). Other sources of uncertainty from this model include observation errors associated with survey and fishery data, and the synthesis of a wide variety of information.

Uncertainty in current and future biomass can be expressed as a cumulative probability plot (Figure 2.8). This figure shows that the 2005 level has about 7% probability of being below the (uncertain) B40% level. The level of uncertainty increases into the future (under alternative constant-catch scenarios listed as an example) due to the impact of variable and uncertain future recruitment. In practice, these uncertainties decrease as information on the recruitment strengths is obtained.

Uncertainty in projected yield specifications is due to a number of sources. These include uncertainty in biomass (above), uncertainty in future selectivity-at-age patterns, and uncertainty in natural mortality estimates (if specified). Since the Council's SSC has considered this stock to be managed under Tier 1 of Amendment 56, the uncertainty in this calculation is adjusted to be formally risk-averse and the harmonic mean yield (which is smaller as uncertainty increases) results in an ABC value of 813,000 mt for EBS pollock in 2010 (BSAI 2009 SAFE report).

2.4.2.4 Draft Stock Assessment and Fishery Evaluation Report

ABC and OFL are first recommended by the stock assessment authors, who evaluate the biological state of the fished stock and its tolerance for fishing. Their recommendations are summarized in SAFE reports. SAFE reports provide the Council with "a summary of information concerning the most recent biological condition of stocks and the marine ecosystems in the fishery management unit and the social and economic condition of the recreational and commercial fishing interests, fishing communities, and the fish processing industries. They summarize, on an annual basis, the best available scientific information concerning the past, present, and possible future condition of the stocks, marine ecosystems, and fisheries being managed under Federal regulation" (50 CFR 600.315(e)(1)). Each SAFE report must be scientifically based and should contain (50 CFR 600.315(e)(2-3)):

(1) Information on which to base harvest specifications,
(2) A description of the maximum fishing mortality threshold and the minimum stock size threshold for each stock or stock complex, along with information by which the Council may determine (a) whether overfishing is occurring or any stock is overfished, and whether overfishing or overfished conditions are being approached, and (b) any measures necessary to rebuild an overfished stock.

Each report may also contain "additional economic, social, community, EFH, and ecological information pertinent to the success of management or the achievement of objectives of each FMP" (50 CFR 600.315(e)(4)).

The BSAI and GOA FMPs require the following minimum contents of the SAFE reports:
(1) Current status of BSAI or GOA major species or species group.
(2) Estimates of MSY and ABC.
(3) Estimates of groundfish species mortality from non-groundfish fisheries, subsistence fisheries, and recreational fisheries, and differences between groundfish mortality and catch, if possible.
(4) Fishery statistics (landings and value) for the current year.
(5) The projected responses of stocks and fisheries to alternative levels of fishing mortality.
(6) Any relevant information relating to changes in groundfish markets.
(7) Information to be used by the Council in establishing prohibited species catch (PSC) limits for prohibited species and fully utilized species with supporting justification and rationale.
(8) Any other biological, social, or economic information which may be useful to the Council.

2.4.3 Setting the Catch Specifications

Descriptions of procedures for setting harvest quotas are provided in the GOA and BSAI groundfish FMPs and are incorporated herein by reference (NPFMC 2009a, 2009b). This includes a description of MSY and OY as they pertain to conservation of target stock biomass and sustainability of the groundfish complex in the BSAI and GOA. The FMPs also include a definition of Overfishing Level (OFL) which is a limit reference point set annually for a stock or stock complex during the assessment process. Overfishing occurs whenever a stock or stock complex is subjected to a rate or level of fishing mortality that jeopardizes the capacity of a stock or stock complex to produce MSY on a continuing basis. Operationally, overfishing occurs when the harvest exceeds the OFL. The Council has developed a procedure to set annual harvest levels by specifying a TAC for each species and species group managed by the FMP.

Scientists from the AFSC, ADF&G, and other agencies and universities prepare SAFE documents annually. These documents are first reviewed by the Groundfish Plan Team, and then by the Council's SSC and Advisory Panel (AP), and the Council. Reference point recommendations are made at each level of assessment. The SSC recommends values for ABC and OFL, and the AP recommends values for TAC. The Council has final authority to approve only the TAC and must use the SSC's recommendations for ABC and OFLs. The Council focuses on setting TACs so that OY is achieved and OFLs are not exceeded.

The procedure for setting TAC consists of the following steps:

1. Determine the ABC for each managed species or species group. ABCs are recommended by the Council's SSC based on information presented by the Plan Team.
2. Determine a TAC based on biological and socioeconomic information. The TAC must be lower than or equal to the ABC. The TAC may be lower if bycatch considerations or socioeconomic considerations cause the Council to establish a lower harvest.

3. TACs are specified so that the sum of TACs for “target species” and “other species” falls within the OY range specified for the groundfish complex in the FMP. If the sum falls outside this range the TACs must be adjusted or the FMP amended. The upper limit of the BSAI OY range is set in statute and can not be changed without congressional action.

2.4.3.1 Plan Team Review of Stock Assessments

In Section 2.3 we discussed the development of the draft SAFE report by the stock assessment author. After the draft SAFE has been prepared, the stock assessments and recommendations are then reviewed by the BSAI and GOA groundfish plan teams, which consist of members from the AFSC, NMFS SFD, ADF&G, the Washington Department of Fisheries, the USFWS, the International Pacific Halibut Commission, and the University of Alaska at Fairbanks. The plan teams then prepare their recommendations to the Council’s AP and SSC, and the main body of the Council. The Council’s SSC has final authority for determining whether a given item of information is "reliable" for the purpose of determining ABCs and OFLs, and may use either objective or subjective criteria in making such determinations. The SSC’s recommendations for ABCs and OFLs must be used by the AP and the Council for the setting of TACs.

2.4.3.2 Council Process and the Development of Multi-Year TACs

Based on the reviews and recommendations of the stock assessment authors, the plan teams, the SSC, and the AP, the Council, at its October meeting, then considers the proposed ABC and OFL levels for each stock, and pertinent social, economic, and ecological information to determine a TAC for each stock or stock complex under the BSAI and GOA FMPs for the following two years. The proposed harvest specifications recommended in October are based on the previous year’s SAFE report and any additional information that may be available during the September Plan Team meetings. The survey information collected during the summer is not available for the Council’s consideration in October. These proposed harvest specifications are modified based on the November SAFE when the Council makes its final harvest specifications recommendations in December, as further explained below.

The TAC for a specific stock or stock complex may be sub-divided for biological and socio-economic reasons according to percentage formulas established in FMP amendments. For particular target fisheries, TAC specifications are further allocated within management areas (eastern, central, western Aleutian Islands; Bering Sea; eastern, central, western GOA; Figure 2.2), among management programs (limited access, open access, or community development quota program), processing components (inshore or offshore), specific gear types (trawl, hook-and-line, pot, jig), and seasons according to regulations.
The Council will provide proposed recommendations for harvest specifications to the Secretary after its October meeting, including detailed information on the development of each proposed specification and any future information that is expected to affect the final specifications. As soon as practicable after the October meeting, the Secretary will publish in the Federal Register proposed harvest specifications for the following two years based on the Council’s October recommendations and make available for public review and comment, all information regarding the development of the specifications, identifying specifications that are likely to change, and possible reasons for changes, if known, from the proposed to final specifications. The prior public review and comment period on the published proposed specifications will be a minimum of 15 days, but usually for 30 days.

At its December meeting, the Council reviews the final SAFE reports developed by its Plan Teams in November, recommendations from the Groundfish Plan Teams, SSC, AP, and public comments. The Council will recommend final harvest specifications for the following two years to the Secretary for review, approval, and publication. When new final annual specifications are made by the Secretary (typically in February or March), they supersede the existing annual specifications and are effective upon publication in the Federal Register.

Harvest specifications provided in regulations implementing the FMPs (CFR § 679.20) include two years of TACs and apportionments thereof, and reserves for each stock or stock complex in the "target species" and "other species" categories. Notwithstanding designated stocks or stock complexes listed by category, the Council may recommend splitting or combining stocks or stock complexes in the "target species" category for purposes of establishing a new TAC if such action is desirable based on commercial importance of a stock or stock complex and whether sufficient biological information is available to manage a stock or stock complex on its own merits.

2.4.3.3 Harvest Specifications

The amount of harvest during a year for each groundfish species is controlled by the harvest specifications. In 2008, the Council made recommendations for TAC for the Alaska groundfish fisheries for 2009 and 2010 (combined areas for pollock, Pacific cod, and Atka mackerel through 2010). Similarly, in 2009, the Council made two-year recommendations for TAC in these fisheries for 2010 and 2011. These are shown in Tables 2.5 (GOA), 2.6 and 2.7 (BSAI). Incidental harvest of groundfish species also is limited by the maximum retainable amounts (MRAs) specified in the regulations. These MRAs are in Tables 10 and 11 to 50 CFR Part 679 and are included in the BA as Tables 2.6.6.3 (GOA) and 2.6.6.4 (BSAI). Note that for Table 2.6.6.3, the other species MRA for the arrowtooth flounder fishery was changed by Amendment 69 to 20%, effective April 12, 2006 (71 FR 12626, March 13, 2006) and increased up to 20% for other target groundfish species in April 2009 (74 FR 13348, March 27, 2009).

NMFS SFD reviews the final harvest specifications recommended by the Council in December and may adjust the harvest amounts available at the beginning of the year by an inseason action. This type of action is used to ensure that the fishery is started on the best available information and prevents the potential for allowing more harvest than recommended by the Council and allowed for under seasonal apportionment for the fishing year. This ensure the entire fishing year is managed under the Steller sea lion protection measures and based on the best available science for Steller sea lion principal prey species.

2.5 Commercial Fisheries Prosecution

The prosecution of the fisheries can be grouped into two main functions: (1) implementation of the fisheries, and (2) monitoring the catch and fisheries effects.

2.5.1 Implementation of the Fisheries

2.5.1.1 Fishery Status

The fishery for a target species may be categorized as open to directed fishing, closed to directed fishing, or prohibited retention. A different set of management regulations apply to each open, closed, or prohibited retention status.

Management of a Species Open to Directed Fishing

When a species fishery is open to directed fishing, vessels are allowed to target and retain within the allocations and apportionments specific to the fishery that are described in the annual harvest specifications. If the catch is expected to reach the TAC and some amount of TAC must be held in reserve for incidental catch in other fisheries, then a portion of the TAC may be established as a "directed fishing allowance," meaning that directed fishing is allowed only on that portion of the TAC. For example, for the BSAI pollock fishery, after subtracting 10% for the community development quota allocation, 4% of the TAC is established as an "incidental catch allowance" and the directed fishery is based on the remaining 86% of the TAC. For fisheries other than BSAI pollock, Atka mackerel, flathead sole, rock sole, yellowfin sole, Aleutian Islands Pacific ocean perch, and fixed gear Pacific cod, the amount for a "directed fishing allowance" is determined by NMFS as the season progresses, and is established by an in-season regulatory action.

Management of Species, when Directed Fishing is Prohibited

When a groundfish species is closed to directed fishing in the GOA or BSAI, vessels are allowed to retain up to the MRA of that species at any time during the fishing trip, but are prohibited from exceeding the MRA during the fishing trip. These MRAs are calculated as a percentage of retained catch for the species that is closed to directed fishing for a given amount of the basis species that is open to directed fishing. Each basis species and the corresponding MRA for a species that is closed to directed fishing is listed in Table 10 to 50 CFR Part 679 for the BSAI, and in Table 11 to 50 CFR Part 679 for the GOA. All catch, whether retained or discarded is counted against the TAC.

Fishery managers may prohibit a species from directed fishing for a number of reasons. Once the directed fishing allowance for a species is taken, the fishery is closed to directed fishing to assure that annual catches will stay within an established TAC. Other directed fishing closures may be intentionally set at intervals throughout a fishing year to slow the intensity and rate of fishing during a fishing year. This provision does allow targeting for the species on a haul-by-haul basis, as long as the MRA for the trip is not exceeded. Closing a species to directed fishing has been used effectively by managers throughout the BSAI and GOA to contain groundfish catches within an annual TAC. Selective closures to directed fishing have also been applied to certain Steller sea lion protection areas to reduce the rate and amount of a prey species, such as Atka mackerel, that are removed.

While the provision for MRA accounting supports the suite of Steller sea lion protection measures for the BSAI and GOA, the effectiveness of this provision has a potential loophole. In a single haul, it is possible for a vessel operator to haul through a Steller sea lion protection area, and account for the MRA of a species, by using basis species from outside the Steller sea lion protection area. This loophole could allow for targeting of Atka mackerel or Pacific cod on a haul-by-haul basis, as long as the MRA for the trip is not exceeded. The Council had recommended a provision to relax MRA accounting that would close that loophole, as discussed in "Pending Actions." During the proposed rule stage, public comment indicated that this action would not meet the goal of relaxing MRA accounting and this proposed rule was withdrawn (74 FR 65503, December 10, 2009).

Management of a Species Prohibited from Retention

Retention is prohibited if the total TAC is caught before the end of the year. Prohibiting retention removes any incentive to increase incidental catch as a portion of other fisheries. If the ABC is taken and the trajectory of the catch indicates that the OFL may be approached, additional closures may be imposed. To prevent overfishing, specific fisheries identified by gear and area that incur the greatest incidental catch may be closed. Closures expand to other fisheries if the rate of take is not sufficiently slowed. Overfishing closures are rare.

2.5.1.2 Access and Permits

Both the GOA and BSAI FMPs prescribe requirements that restrict the participation of individuals and vessels in the groundfish fisheries. These programs include requirements for Federal groundfish fishing licenses, species and/or gear endorsements requirements for these licenses, and harvesting sideboards.

These fisheries also may be managed under certain gear or time and area restrictions. Limited access provisions related to pollock harvest based on the AFA and the 2004 Consolidated Appropriations Act are incorporated herein by reference. The permits can be used to identify those vessels that must comply with certain fisheries management requirements. For instance, all federally permitted vessels that are endorsed for Atka mackerel, Pacific cod, or pollock fishing on their Federal Fisheries Permit must use a vessel monitoring system. Additional information regarding permits are in the regulations at 50 CFR 679.4.

License Limitation Program

A Federal groundfish license is required for catcher vessels (including catcher/processors) participating in all BSAI groundfish fisheries, other than fixed gear sablefish. However, the following vessel categories are exempt from the license program requirements:

a. vessels fishing in State of Alaska waters (0-3 miles offshore);
b. vessels less than 32 feet (ft) LOA in the BSAI (26 ft LOA in the GOA); or
c. jig gear vessels less than 60 ft LOA using a maximum of 5 jig machines, one line per machine, and a maximum of 15 hooks per line.

Any vessel that meets the License Limitation Program (LLP) qualification requirements will be issued a license, regardless of whether they are exempt from the program or not. The specifics of the LLP program and the species and gear endorsements described in Section 3.3.1 of the BSAI and GOA groundfish FMPs are incorporated here by reference.

Limited Access Privilege Programs

Amendment 80 to the FMP for Groundfish of the BSAI primarily allocates several BSAI non-pollock trawl groundfish fisheries among fishing sectors, and facilitates the formation of harvesting cooperatives in the non-AFA trawl catcher/processor sector. The amendment establishes a limited access privilege program (LAPP) for the non-AFA trawl catcher/processor sector. This action increases resource conservation and improves economic efficiency for harvesters who participate in the BSAI groundfish fisheries. The number of vessels fishing may decrease as forming cooperatives allows participants to increase efficiency.

Exempted Fishing Permits

The Regional Administrator, after consulting with the Director of the AFSC and with the Council, may authorize for limited experimental purposes fishing activities that would otherwise be prohibited. This could include the targeted or incidental harvest of groundfish and prohibited species and fishing in areas that are closed to directed fishing, for continued fishing with gear otherwise prohibited, or for continued fishing for species for which the quota has been reached. EFPs are issued by means of procedures contained in regulations (50 CFR 679.6 and 600.745).

As well as other information required by regulations, each application for an EFP must provide the following information: 1) experimental design (e.g., staffing and sampling procedures, the data and samples to be collected, and analysis of the data and samples), 2) provision for public release of all obtained information, and 3) submission of interim and final reports.

As specified in the FMPs, the Regional Administrator may deny an EFP for reasons contained in regulations, including a finding that:

a. according to the best scientific information available, the harvest to be conducted under the permit would detrimentally affect living marine resources, including marine mammals and birds, and their habitat in a significant way;
b. issuance of the EFP would inequitably allocate fishing privileges among domestic fishermen or would have economic allocation as its sole purpose;
c. activities to be conducted under the EFP would be inconsistent with the intent of the management objectives of the FMP;
d. the applicant has failed to demonstrate a valid justification for the permit;
e. the activity proposed under the EFP could create a significant enforcement problem; or
f. the applicant failed to make available to the public information that had been obtained under a previously issued EFP.

2.5.1.3 Sector and Gear Allocations

Gear types authorized by the FMPs are trawl, hook-and-line, pot, jig, and other gear as defined in regulations at 50 CFR Part 679 (authorized gear types and fisheries are displayed in Table 2.8). Gear types and sector allocations for specific BSAI fisheries are described in detail in the annual harvest specifications (Appendix 1). The complexity of the allocation scheme has grown since 2000 and is only described in general terms here based on the final 2009 harvest specifications (74 FR 7365, February 17, 2009).

Fifteen percent of each target species or species group, except for pollock, the hook-and-line and pot gear allocation of sablefish, and the Amendment 80 species, are placed in a non-specified reserve category in the BSAI. In the EBS, pollock is allocated among four sectors, with 10% of the TAC allocated to the Community Development Quota (CDQ) Program, 4% held in reserve for incidental catch (ICA), and the remainder split among the inshore, catcher/processor, and mothership sectors in the ratio of 50:40:10, respectively. The Aleutian Islands pollock fishery is allocated 10% to CDQ, a portion determined by NMFS inseason management to cover the ICA (9% in 2009), and the remainder to the Aleut Corporation. For BSAI arrowtooth flounder and Bering Sea Greenland turbot, 10.7% of the non-specified reserve is held as reserve for CDQ. For the Amendment 80 species Atka mackerel, Aleutian Islands Pacific ocean perch, yellowfin sole, rock sole, flathead sole and Pacific cod, 10.7% of the TAC is held as reserve for CDQ. After removal of CDQ reserve for Pacific cod, the remainder is allocated to jig (1.4%), hook-and-line and pot (60.8%) and trawl (37.8%), with the trawl portion split among catcher vessels, AFA catcher/processors and Amendment 80 vessels. The hook-and-line and pot gear allocation is further allocated as follows: 48.7% hook-and-line catcher/processors, 0.2% hook-and-line catcher vessels greater than 60 ft LOA, 1.5% pot catcher/processors, 8.4% pot catcher vessels greater than 60 feet LOA, and 2.0% to catcher vessels under 60 ft length overall using hook-and-line or pot gear. For sablefish in the Bering Sea, hook-and-line and pot together are allocated 50%, and trawl is allocated 50%. For sablefish in the Aleutian Islands, hook-and-line and pot receive 75% and trawl 25% (20% of hook-and-line/pot allocation is held as CDQ reserve, as is 7.5% of the trawl allocation). For Atka mackerel, 0.5% of the Bering Sea and eastern Aleutian Islands allocation goes to jig gear.

In the GOA, 20% of pollock, Pacific cod, flatfish and “other species” is held for initial reserve, and the remainder of the pollock allocation goes to the inshore sector, as reduced by the amount needed to support incidental catch in the offshore sector. For Pacific cod, the allocation is split 90% to the inshore sector and 10% to the offshore sector. Sector allocations are made to trawl and longline gear for the central GOA Pacific ocean perch, northern rockfish, and pelagic shelf rockfish fisheries. Sector allocations are not made for flatfish or other species in the GOA. The purpose of the reserves is to give management the flexibility needed to prevent the catch from exceeding the TAC. In 2009, NMFS reapportioned all the reserves in the final 2009 harvest specifications.

2.5.1.4 Spatial and Temporal Restrictions

In addition to temporal and spatial allocation of TACs, certain areas are closed seasonally, year-round, or under special circumstances as established in regulations. Prohibitions specific to the protection of Steller sea lions or their habitat are described separately in Section 2.5.2. General time/area closures are as follows (see FMPs Section 3.5 for detailed descriptions and maps). State parallel groundfish fisheries generally mirror adjacent federal closures in state waters, and are discussed below.

Figures depicting these time and area closures are available at 50 CFR 679 and on NOAA Fisheries Alaska Region Website at: http://fakr.noaa.gov/rr/figures.htm.

GOA Area Restrictions

- Sitka Pinnacles Marine Reserve – All vessels
  The Sitka Pinnacles Marine Reserve encompasses an area totaling 2.5 square nm off Cape Edgecumbe. Vessels holding a Federal fisheries permit are prohibited at all times from fishing for groundfish or anchoring in the Sitka Pinnacles Marine Reserves. This closure is outside state waters.
- King Crab Closure Areas around Kodiak Island – Trawl gear only
  A time/area closure has been developed to protect and rebuild the King Crab stock around Kodiak. Three area types have been designated as follows. In Type I areas, bottom trawling is closed year round. In Type II areas, bottom trawling is prohibited during the soft-shell season (February 15 to June 15). Type III areas are those that may be converted to Type I or Type II if a recruitment event occurs. A Type III area is open to bottom trawling until the number of females assessed for the area meets or exceeds the number required to hold a crab fishery. If a crab fishery is initiated, then no closure is in effect. If no crab fishery is initiated, then the Regional Administrator may designate the Type III area as a Type I or II area based on the information available. Adjacent state waters follow the same closure guidelines.
- Cook Inlet non-Pelagic Trawl Closure Area
  The use of non-pelagic trawl gear is prohibited in Cook Inlet north of a line extending between Cape Douglas and Point Adam. This prohibition is intended to reduce crab bycatch and assist in the rebuilding of crab stocks. Adjacent state waters follow the same closure guidelines.
- Southeast Outside Trawl Closure
  Use of any gear other than non-trawl gear is prohibited at all times in the Southeast Outside district.
- EFH closures (pub. 6/28/06, effective July 28, 2006)
  Closure areas are described in Tables 22, 26 and 27 to 50 CFR Part 679. Adjacent state waters follow the same closure guidelines.

  Alaska Seamount Habitat Protection Areas
  No bottom tending gear or anchoring allowed.

  Gulf of Alaska Slope Habitat conservation Areas
  No bottom trawling allowed.

  GOA Coral Habitat Protection Areas
  No bottom tending gear or anchoring allowed.

BSAI Area Restrictions

The following time and area restrictions apply to some or all trawl vessels. Other time and area restrictions that may apply to trawl vessels are triggered by the attainment of a bycatch limit.

- Crab and Halibut Protection Zone
  The crab and halibut protection zone is closed to all trawling from January 1 to December 31. For the period March 15 to June 15, the western border of the zone extends westward. See Figure 3-2 (NPFMC 2009a). This area was superseded by the Nearshore Bristol Bay Trawl Closure and is proposed to be removed from the FMP by Amendment 94.
- Pribilof Island Area Habitat Conservation Zone
  The Pribilof Islands Area Habitat Conservation Zone is closed to all trawling from January 1 to December 31.
- Chum Salmon Savings Area
  The Chum Salmon Savings Area is closed to trawling from August 1 through August 31. Trawling is also prohibited in this area upon the attainment of an "other salmon" bycatch limit.
- Chinook Salmon Savings Areas
  Closed to pollock trawling when 29,000 Chinook salmon limit is attained until April 15 or from September 1 through December 31 or both time periods (679.21(e)(7)(viii), depending on when limit is reached.
  Vessels participating in an intercooperative agreement (ICA) to reduce salmon bycatch under Amendment 84 are exempt from both the Chum Salmon Savings Area and Chinook Salmon Savings Areas closures. Nearly all pollock vessels are members of the ICA with only one vessel not participating in the ICA in 2008. Member of the ICA are required to stay out of salmon hot spots as identified by the ICA manager based on near real time rates of salmon bycatch in the fishery. This amendment is further explained in section 2.5.1.7. Amendment 84 may be replaced by proposed Amendment 91 which is under NMFS review at this time. Amendment 91 would remove the Chinook Salmon Savings Area in the Bering Sea and would establish hard caps on Chinook salmon bycatch in the Bering Sea pollock fishery, along with an incentive program for the industry to minimize salmon bycatch.
- Red King Crab Savings Area
  The Red King Crab Savings Area is closed to non-pelagic trawling year round, except that when the Regional Administrator of NMFS, in consultation with the Council, determines that a guideline harvest level for Bristol Bay red king crab has been established, he or she may open a subarea of the Red King Crab Savings Area to non-pelagic trawling.
- Nearshore Bristol Bay Trawl Closure
  The Nearshore Bristol Bay area is closed to all trawling on a year round basis, except a subarea that remains open to trawling during the period April 1 to June 15 each year.
- Catcher Vessel Operational Area
  Catcher/processors identified in the AFA are prohibited from engaging in directed fishing for pollock in the catcher vessel operational area (CVOA) during the non-roe ("B") season, unless they are participating in a community development quota fishery.
- EFH Closures.

  Alaska Seamount Habitat Protection Areas (Table 22 to 50 CFR Part 679)
  No bottom contact gear or anchoring allowed.

  Aleutian Islands Coral Habitat Protection Areas (Table 23 to 50 CFR Part 679)
  No bottom contact gear or anchoring allowed.

  Aleutian Islands Habitat Conservation Area (Table 24 to 50 CFR Part 679)
  No bottom trawling allowed.

Bowers Ridge Habitat Conservation Zone (Table 25 to 50 CFR part 679)
No mobile bottom contact gear allowed.

- Bering Sea Habitat Conservation Measures
  Nunivak Island, Etolin Strait, Kuskokwim Bay Habitat Conservation Area
  No bottom trawling allowed.

  Saint Lawrence Island Habitat Conservation Area
  No bottom trawling allowed.

  Saint Matthew Island Habitat Conservation Area
  No bottom trawling allowed.

  Bering Sea Habitat Conservation Area
  No bottom trawling allowed.

  Northern Bering Sea Research Area
  No bottom trawling allowed. Bottom trawling may be done under an exempted fishing permit that comports with a research plan to study the impacts of bottom trawling on bottom habitat.

Closures in State Waters

Additionally, most state waters are closed to non-pelagic trawling (Figure 2.13). Most areas are closed year-round, and some areas are closed seasonally (Shelikof Strait and Sanak Islands). Additionally, a portion of eastern Prince William Sound is closed to pelagic trawl gear during the pollock fishery (5 Alaska Administrative Code [AAC] 28.263) and most of eastern Prince William Sound is closed to all (non-pelagic and pelagic) trawling year-round (5 AAC 39.165), see Figure 2.14. These trawl closures were established by the Alaska Board of Fisheries to protect seafloor habitats, shellfish species (including depressed crab populations), and non-target demersal fishes. In March 2007 the board also adopted specific gear and area closures for EFH in both the Aleutian Islands and the GOA waters to match federal regulations.

2.5.1.5 Harvest of TAC

We provide updated tables of catch of pollock, Pacific cod, Atka mackerel, and total groundfish in Table 2.9 including the percentage of total groundfish for each of the three species.

2.5.1.6 Incidental Catch

While fishery participants may target a certain species, they are not 100% effective in limiting their catch to that specific target. Other fishes and marine life are also caught to varying degrees depending on target species, gear type and fishing method, area fished and habitat type, season, depth, and other physical and biological factors. These other fishes and marine life are referred to as "incidental catch" or "bycatch."3 Whether a species or stock is caught as a target by a fishing vessel, or incidentally by a vessel targeting another species, the catch is included against the overall total allowed for a species or stock. That is, TACs are intended to represent the sum of all catch including targeted catch and incidental catch.

---

[3] The terms "incidental catch" and "bycatch" are often used to mean catch of species or marine life not targeted. In regulations, the terms are given specific meanings. "Incidental catch" is defined as fish that is caught and retained while targeting another species but does not include discarded fish that were returned to the sea. "Bycatch" is defined as fish that is caught and released when targeting another species or targeting the same species.

2.5.1.7 Bycatch of Prohibited Species

When a target fishery, as specified in regulations implementing the FMP, attains a PSC limit apportionment or seasonal allocation as described in the FMPs and specified in regulations implementing the FMPs, the bycatch zone(s) or management area(s) to which the PSC limit apportionment or seasonal allocation applies will be closed to that target fishery (or components thereof) for the remainder of the year or season, whichever is applicable. Some fisheries are also closed due to no PSC available to support a directed fishery such as closure of the Pacific cod fishery due to no halibut PSC available to support the bycatch that would occur in the Pacific cod fishery. The procedure for apportioning PSC limits is detailed in section 3.6.2.3 of the FMPs (NPFMC 2009a,b).

Prohibited species include Alaska king crab, Tanner and snow crab, Pacific halibut, Pacific salmon species, steelhead trout, and Pacific herring. With some exceptions, retention is prohibited in the BSAI and GOA groundfish fisheries to eliminate any incentive to target these species. A description of the individual PSC limits can be found in Section 3.6.2.1 of the FMPs.

A variety of management measures have been used to control the bycatch of prohibited species, including 1) PSC limits by fishery for selected prohibited species (red king crab, Tanner and snow crab, Pacific halibut, Pacific salmon, and Pacific herring in the BSAI and Pacific halibut in the GOA); 2) time and area closures; 3) seasonal apportionments of groundfish TACs; 4) gear restrictions; 5) groundfish TAC allocations by gear type; 6) reductions in groundfish TACs; 7) at-sea and on-shore observer programs to monitor bycatch; 8) required retention of Pacific salmon bycatch until counted by an observer; 9) Individual Transferable Quota (ITQ) management for the fixed-gear Pacific halibut and sablefish fisheries; 10) careful release regulations; and 11) public reporting of individual bycatch rates.

Groundfish fisheries or fisheries under the FMPs for which the TAC has been reached shall be treated in the same manner as prohibited species. Species identified as prohibited must be avoided while fishing groundfish and must be immediately returned to the sea with a minimum of injury when caught and brought aboard, except when their retention is authorized by other applicable law.

In the mid-1990s, the Council and NMFS implemented regulations to control the bycatch of chum salmon and Chinook salmon taken in the BSAI trawl fisheries. These regulations established closure areas in areas and at times when salmon bycatch had been highest based on historical observer data. Information from the fishing fleet indicates that bycatch may have been exacerbated by the current regulatory closure regulations, as much higher salmon bycatch rates were reportedly encountered outside the closure areas. Some of these bycaught salmon include Chinook and chum stocks of concern in western Alaska. Further, the closure areas impose increased costs on the pollock fleet and processors. To address this immediate problem, the Council recommended other means to control salmon bycatch. Amendment 84 (available from the NMFS Alaska Region website at http://alaskafisheries.noaa.gov) was implemented November 28, 2007. Under Amendment 84, certain trawl vessels are exempt from the Chum Salmon Savings Area and Chinook Salmon Savings Areas closures. The exemption is in effect so long as the pollock cooperatives and CDQ groups belong to an intercooperative agreement (ICA) for salmon bycatch reduction. This ICA includes a salmon bycatch voluntary rolling “hot spot” (VRHS) closure system to avoid salmon bycatch. Nearly all pollock vessels participate in the ICA and reports from the first year indicate that salmon bycatch rates were reduced by using the VRHS closure system (Halinger, Gruver and Christenson 2008). Fishing patterns may change under this amendment, as the pollock fishery is no longer mandatorily forbidden to fish in the established savings areas. This action was included in an ESA Section 7 consultation on ESA-listed salmon consultation with NMFS NW Region.

The Council has recommended Amendment 91 to further minimize Chinook salmon bycatch in the Bering Sea pollock fishery. This amendment is under review by NMFS and is intended to replace the salmon bycatch management established under Amendment 84. The amendment would provide hard caps on Chinook salmon bycatch in the Bering Sea pollock fishery and includes an incentive program for the industry to minimize their bycatch. The industry will continue to use hot spot avoidance methods to reduce salmon bycatch which is likely to result in spatial distribution similar to the distribution under Amendment 84. This action was subject of an ESA Section 7 consultation on ESA-listed salmon with the NMFS NW Region completed in December 2009.

2.5.1.8 Retention and Utilization Requirements

Retention and utilization requirements in the BSAI and GOA consist of various types of retention requirements for groundfish, utilization of round catch, and utilization of pollock roe, specified by area and fishing sector.

All vessels participating in the BSAI groundfish fisheries are required to comply with improved retention/improved utilization (IR/IU) requirements (50 CFR § 679.27).

In the BSAI, IR/IU species are pollock and Pacific cod. When directed fishing for an IR/IU species is open, a catcher vessel must retain catch of all IR/IU species brought onboard the vessel, and all catcher/processors and motherships must process a primary product from each IR/IU species brought onboard, regardless of gear type employed and target fishery. When directed fishing for an IR/IU species is prohibited, a catcher vessel must retain all catch of that species up to the maximum retainable amount in effect for that species. These retention requirements are superseded if retention of an IR/IU species is prohibited for retention by other regulations. No discarding of whole fish of these species is allowed, either prior to or subsequent to that species being brought on board the vessel. At-sea discarding of any processed product from any IR/IU species is also prohibited, unless required by other regulations.
IR/IU requirements in the BSAI also include a groundfish retention standard (GRS), implemented in the Amendment 80 program for non-AFA trawl catcher/processor vessels. All non-AFA trawl catcher/processors (Amendment 80 vessels), regardless of size, are required to meet GRS requirements in the BSAI. The GRS requires that a vessel, or designated group of vessels, must retain an annually increasing percentage of total groundfish caught, that starts at 65% in 2008 and increases to 85% in 2011, and all years thereafter. For Amendment 80 vessels harvesting in the BSAI under the authority of an Amendment 80 cooperative, GRS requirements apply collectively to all vessels harvesting under the authority of the cooperative rather than on a vessel-specific basis. An Amendment 80 cooperative is required to meet the GRS on an aggregate basis for all vessels in the Amendment 80 cooperative. The GRS also apply to all non-AFA trawl catcher/processor vessels that have not entered a cooperative, on an individual vessel basis.

Roe-stripping of pollock is prohibited, and the Regional Administrator is authorized to issue regulations to limit this practice to the maximum extent practicable. It is the Council's policy that the pollock harvest shall be utilized to the maximum extent possible for human consumption.

IR/IU in the GOA

In the GOA IR/IU species are pollock, Pacific cod and shallow-water flatfish. All IR/IU species caught in the GOA must be either (1) processed at sea subject to minimum product recovery rates and/or other requirements established by regulations, or (2) delivered in their entirety to onshore processing plants for which similar processing requirements are implemented by state regulations. When directed fishing for an IR/IU species is open, a catcher vessel must retain catch of all IR/IU species brought onboard the vessel, and all catcher/processors and motherships must process a primary product from each IR/IU species brought onboard, regardless of gear type employed and target fishery. When directed fishing for an IR/IU species is prohibited, a catcher vessel must retain all catch of that species up to the maximum retainable amount in effect for that species. These retention requirements are superseded if retention of an IR/IU species is prohibited for retention by other regulations. No discarding of whole fish of these species is allowed, either prior to or subsequent to that species being brought on board the vessel. At-sea discarding of any processed product from any IR/IU species is also prohibited, unless required by other regulations.

To monitor discards of shallow-water flatfish in the GOA, the Secretary has approved a GOA groundfish FMP amendment (Amendment 72) to revise the FMP to state that the Council will annually review information on the discard of shallow-water flatfish in GOA groundfish fisheries. After review of this annual information, the Council may recommend revisions to retention and utilization requirements if the discard rate for shallow-water flatfish falls above or below a specified threshold. This action supports the Council's initiatives to monitor and reduce bycatch in the GOA groundfish fisheries. The intended effect of this action is to conserve and manage the groundfish resource in the GOA.

2.5.2 Steller Sea Lion Protection Measures

The proposed action, authorization of the BSAI and GOA groundfish FMPs (NPFMC 2009a, 2009b), includes mitigation measures for the pollock, Atka mackerel, and Pacific cod fisheries which were included in the action to avoid jeopardy and adverse modification of Steller sea lions and their critical habitat (NMFS 2000, 2001, and 2003). The mitigation measures were implemented in 2001 and took the place of a RPA required as part of the jeopardy and adverse modification finding in the 2000 FMP Biological Opinion. Following the 2000 FMP Biological Opinion and the partial implementation of its RPA, the Council recommended an alternative suite of management measures intending to be substituted for the measures contained within the RPA of the 2000 Biological Opinion. Using the best available scientific and commercial data available at the time, NMFS (2001) determined that these alternative protection measures avoided jeopardy and adverse modification to critical habitat for both the western and eastern distinct population segments of Steller sea lion. Therefore, the new measures recommended by the Council and adopted by NMFS, although not specifically required by an RPA, were considered necessary to avoid jeopardy and destruction or adverse modification of critical habitat as they replaced the specific measures in the RPA from the 2000 Biological Opinion.

Greenpeace, American Oceans Campaign, and the Sierra Club challenged the 2001 Biological Opinion on December 18, 2002. In U.S. District Court for the Western District of Washington, Judge Zilly granted motion for summary judgment on Greenpeace, American Oceans Campaign, and Sierra Club v. NMFS *et al.*, No. C98-492Z). The opinion was remanded to NMFS for further consideration of issues as required by the court. On June 19, 2003 NMFS prepared a supplement to the 2001 Biological Opinion (NMFS 2003) which further evaluated the fisheries and their interactions with Steller sea lions and affirmed the determination that the pollock, Pacific cod, and Atka mackerel fisheries did not jeopardize the species or adversely modify their critical habitat. The supplement evaluated fishery catch data from both before and after implementation of the conservation measures, which provided a unique perspective for a consultation. The supplement showed that some conservation components to the action worked quite well while others did not perform up to expectations (NMFS 2003, Table IV-1). Because fisheries are dynamic, biomass amounts change, fish move, and the fleet is constantly adapting to changes in both the physical and economic environment, some of this is expected. Overall, the action was conservative enough to avoid jeopardy, while some elements certainly could be improved upon as described in the supplement. However, changes were not required to avoid jeopardy or adverse modification in 2003.

Under the Steller sea lion protection measures implemented in 2002, a complex suite of open and closed areas was used based upon the individual fishery. For that reason, it is impossible to easily sum these various closures and determine how much of the area is closed to fishing. The protection measures, which represent, more of a mosaic, are best described (for closure areas) by looking at each individual fishery and area to determine what is open or closed inside Steller sea lion critical habitat. A summary table of the measures required in the 2000 RPA and the action currently implemented is provided in Table 2.16. We acknowledge that this action has built-in measures, designed in 2001 using the best available scientific and commercial data available at the time, to avoid jeopardy to Steller sea lions and adverse modification of Steller sea lion critical habitat. We consider those measures as part of the existing and proposed future action in light of the new information that is available to us and that prompted this re-initiation of consultation.

Note that in the past several years, additional regulations were implemented in the BSAI and GOA groundfish fisheries to facilitate research on the interaction between groundfish fishing activities and Steller sea lions. These measures included temporary and season-specific closures of some areas to allow research to continue – e.g.,near Unimak Island for Pacific cod research and Chiniak Gully for pollock studies. The Chiniak Gully closure is effective August 1 up to September 20 for the years 2006-2010 (71 FR 31105, June 1, 2006). The study was not conducted in 2007 through 2009 due to lack of funding, and the closures were rescinded. Additional information can be found in regulations at 50 CFR 679.22.

2.5.2.1 No Transit Zones

Neither State nor federally permitted fishing vessels, nor any other type of vessel, are allowed within 3 nm of Steller sea lion rookeries listed in 50 CFR Part 223.202(a)(3) (Table 2.17). The State of Alaska enforces similar regulations which are important since marine waters 0-3 nm from shore are state waters.

2.5.2.2 Global Control Rule

The setting of TAC for the pollock, Pacific cod, and Atka mackerel fisheries is based on a global control rule which is modified from the one detailed in the FMP Biological Opinion. The ABC for pollock, Pacific cod, and Atka mackerel in the BSAI and GOA would be reduced when the spawning biomass is estimated to be less than 40% of the projected unfished (pristine) biomass. The reduction would continue at the present rate established under the tiers described in the groundfish FMPs, but when the spawning biomass is estimated to be less than 20% of the projected unfished biomass, directed fishing for a species would be prohibited (see example at Figure 2.6).

2.5.2.3 Area Specific Measures

The Steller sea lion protection measures include a complicated set of area closures depending on the location, target species, gear type and time of year (Figures 2.9-2.11). These are intended to provide spatial dispersion of fishing effort to reduce the potential effect on prey fields. The measures also include seasonal apportionments depending on the species, gear type, and locations. Seasonal apportionments provide for temporal dispersion of fishing activities to reduce the potential for competition for prey.
A brief history of these management measures is presented below. Then, the closures are described in more detail in following sections. Maps are included as Figures 2.9 – 2.21.

Table 2.10. Chronology of Steller sea lion protection measures in Alaska groundfish fisheries.

1990	Under the ESA, NMFS lists Steller sea lion as endangered. • Shooting banned to reduce Steller sea lion mortality. • Established 3 nm No Transit/No Fishing Areas around Steller sea lion rookeries. State of Alaska also adopted these closures. (Figure 2.15)
1992	No Trawl zones around Steller sea lion rookeries established. (Figure 2.16) • 10 nm around rookeries • 20 nm around some rookeries in the A Season. • Donut Hole closed by international convention

1993	Steller Sea Lion Critical Habitat Established under 50 CFR Part 226. (Figure 2.17) • 20 nm around Rookeries and Haulouts, and the Bogoslof, Shelikof, and Seguam Pass Foraging Areas.
1995	Biological Opinion issued stating that fisheries of the Gulf of Alaska and Bering Sea, Aleutian Islands "not likely to jeopardize the continued existence of Steller Sea Lions".
1997	Steller sea lions west of 144 degrees West longitude are listed as endangered.
1998	• Atka Mackerel and Pollock Biological Opinion 1 issued. It concludes that these fisheries jeopardize the recovery of Steller sea lions. • Groundfish Fishery Biological Opinion 2 issued. • Ban on forage fishing as important to marine mammals.
1999	No trawl zones- Figure 2.18 • NMFS closes the Aleutian Islands Subarea (541, 542, 543) to directed fishing for Pollock • Critical Habitat/Catcher Vessel Operational Area (CH/CVOA) in the Bering Sea Created inside the Bogoslof Foraging Area • Atka Mackerel TAC for inside and outside of Steller sea lion Critical Habitat • Additional RFRPA Pollock No Trawl Areas – A Season and/or B Season. • In 2000 Additional Pollock RFRPA's added.
2000	• July - Judge Zilly issues Trawl Injunction for all Steller sea lion Critical Habitat, as per 50 CFR 226. • November. Biological Opinion 3 issues with 13 Closed and Restricted Areas. (Figure 2.19).
2001	• Emergency Rule issued with Management from January 2000. • Emergency Rule Expired June 2001. Management reverted to November 30 Biological Opinion (Figure 2.19). • July 17, 2001 Final Rule issued with Current Management based on recommendations by the Steller Sea Lion Reasonable and Prudent Alternative Committee. Steller sea lion management broken out by Critical Habitat limits, target fisheries of pollock, Pacific cod, and Atka mackerel and gear type (Figure 2.20).
2004	• Small changes to GOA Steller sea lion protection measures based on informal consultation (Figure 2.21).

Closure Areas around Rookeries and Haulouts: Groundfish, Atka Mackerel, Pollock, and Pacific Cod Fisheries

Fishery closures are located in 50 CFR Part 679.22 in Tables 4 through 6 and Table 12 (provided here as Tables 2.17 through 2.20). Individual haulouts and rookeries and their associated closures are identified for each fishery in the tables. Detailed maps of the closures are provided in Figures 2.22 to 2.24.

After the 2001 Biological Opinion, regulations were changed in 2003 to allow pot fishing within 0-3 nm at Cape Barnabas and Caton Island, and further changed in 2004 implementing several additional changes in the GOA at Castle Rock, Kak Island, Puale Bay, and Cape Douglas (Figure 2.34). Those measures are incorporated into the tables below. Thus, the following tables and text reflect the current status of groundfish fishery regulations that relate to Steller sea lion protection measures in the GOA and BSAI.

Gulf of Alaska Fisheries

Steller sea lion protection measures for the GOA include area closures to pollock and Pacific cod fishing as shown in Tables 4 and 5 to 50 CFR part 679. Table 12 of 50 CFR part 679 contains groundfish fishing closures within 3 nm of rookeries. Vessels using jig gear to target Pacific cod are exempt from all GOA area closures, except the 0-3 nm no transit closures around rookeries under 50 CFR 223.202 and 0-3 nm no groundfish fishing zones around rookeries. Directed pollock fishing and directed fishing for Pacific cod using trawl gear in general are prohibited within 20 nm of most rookeries and within 10 nm of most haulouts. Pacific cod fishing with hook-and-line gear and pot gear is less restrictive in the GOA with many haulout areas open to the shore and only the haulouts near Chignik closed to 20 nm. Directed fishing for Atka mackerel is prohibited (§ 679.22(b)(2)).

Bering Sea and Aleutian Islands Fisheries

Steller sea lion protection measures for the BSAI include area closures to Atka mackerel, pollock, and Pacific cod fishing as shown in Tables 4-6 to 50 CFR Part 679. Table 12 of 50 CFR Part 679 contains groundfish fishing closures within 3 nm of rookeries. Vessels using jig gear to target Pacific cod are exempt from all BSAI area closures, except the 0-3 nm no transit closures around rookeries under 50 CFR 223.202, the 0-3 nm no groundfish fishing zones around rookeries, and the Seguam and Bogoslof foraging area closures. Directed pollock fishing is prohibited within 20 nm of all haulouts and rookeries in the Aleutian Islands and all rookeries in the Bering Sea. Pollock fishing is also prohibited within either 10 nm or 20 nm of haulouts in the Bering Sea. Directed fishing for Pacific cod using trawl gear in general is prohibited within 10 to 20 nm of most rookeries and within 3 to 20 nm of most haulouts. Directed fishing for Pacific cod using hook-and-line or pot gear is prohibited within 3 to 20 nm around BSAI rookeries and closures around haulouts vary from no closures to closures up to 20 nm.

Seasonal Apportionments and Fishery Allocations

Pollock (GOA)

In the GOA, pollock is apportioned by season and area, and is further allocated for processing by inshore and offshore components. Pursuant to CFR § 679.20(a)(5)(iv)(B), the annual pollock TAC specified for the Western and Central Regulatory Areas of the GOA is apportioned into four equal seasonal allowances of 25%. As established by CFR § 679.23(d)(2)(i) through (iv), the A, B, C, and D season allowances are available from January 20 through March 10, March 10 through May 31, August 25 through October 1, and October 1 through November 1, respectively.

Pollock TACs in the Western and Central Regulatory Areas of the GOA in the A and B seasons are apportioned among Statistical Areas 610, 620, and 630 in proportion to the distribution of pollock biomass based on a composite of NMFS winter surveys and in the C and D seasons in proportion to the distribution of pollock biomass based on the four most recent NMFS summer surveys. Since 2003, the Council has recommended averaging the winter and summer distribution of pollock in the Central Regulatory Area for the A season to better reflect the distribution of pollock and the performance of the fishery in the area during the A season. Within any fishing year, the underage or overage of a seasonal allowance may be added to, or subtracted from, subsequent seasonal allowances in a manner to be determined by the Regional Administrator. The rollover amount of unharvested pollock is limited to 20% of the seasonal apportionment for the statistical area. Any unharvested pollock above the 20 percent limit could be further distributed to the other statistical areas, in proportion to the estimated biomass in the subsequent season in those statistical areas (CFR § 679.20(a)(5)(iv)(B)). The West Yakutat (WYK) and Southeast Outside (SEO) Districts of GOA pollock TACs are not allocated by season.

CFR § 679.20(a)(6)(i) requires the allocation of 100% of the pollock TAC in all regulatory areas and all seasonal allowances to vessels catching pollock for processing by the inshore component after subtraction of amounts that are projected by the Regional Administrator to be caught by, or delivered to, the offshore component incidental to directed fishing for other groundfish species. The amount of pollock available for harvest by vessels harvesting pollock for processing by the offshore component is that amount actually taken as incidental catch during directed fishing for groundfish species other than pollock, up to the maximum retainable amounts allowed by § 679.20(e) and (f). These incidental catch amounts are determined during the fishing year.

At 50 CFR § 679.7 a prohibition for trawl vessels is included to limit retention of pollock to 136 mt at anytime during a fishing trip. This trip limit regulation was intended to temporally disperse pollock catches in the GOA as one of several protection measures implemented to protect Steller sea lions under the ESA. Since 2005, pollock delivery practices of some trawl catcher vessels in the GOA have caused seasonal pollock quotas to be exceeded, and if allowed to continue could conflict with the intent of these Steller sea lion protection measures. Current regulations for the pollock directed fishery prohibit catcher vessels from landing in excess of 136 mt of pollock in a fishing trip. Since trip limits were implemented in 1999, these regulations have become less effective, as multiple trips during a day and partial offloads of pollock product during a trip have allowed for increasing amounts of pollock to be caught in some areas of the GOA. This problem was solved with a regulatory amendment that placed a daily limit of 136 mt on GOA pollock catcher vessels (section 679.7(b)(2)). This regulation also prohibits landing a cumulative amount of unprocessed pollock harvested from any GOA reporting area from a catcher vessel issued a federal fisheries permit (FFP) during a directed fishery that exceeds the daily limit multiplied by the number of calendar days that occur during the time period the directed fishery is open in that reporting area. The objective of this prohibition is to prevent catcher vessels from circumventing the intent of trip limit regulations when making deliveries of pollock. The effect of this regulation is to continue to effectively disperse catches of Steller sea lion prey species temporally and spatially in Steller sea lion protection areas of the GOA. Trip limits were revised in a similar manner by the State of Alaska for their groundfish fisheries in 2007, in anticipation of this change to the Federal regulations.

Pacific Cod (GOA)

Pacific cod fishing is divided into two seasons in the Western and Central Regulatory Areas of the GOA. For hook-and-line, pot, and jig gear, the A season begins on January 1 and ends on June 10, and the B season begins on September 1 and ends on December 31. For trawl gear, the A season begins on January 20 and ends on June 10, and the B season begins on September 1 and ends on November 1 (CFR § 679.23(d)(3)). After subtraction of incidental catch needs by the inshore and offshore components in other directed fisheries through the A season ending June 10, 60% of the annual TAC will be available as a directed fishing allowance during the A season for the inshore and offshore components. The remaining 40% of the annual TAC will be available for harvest during the B season and will be apportioned between the inshore and offshore components (§ 679.20(a)(6)(ii)). Any amount of the A season apportionment of Pacific cod TAC under or over harvested will be added to or subtracted from the B season apportionment of Pacific cod TAC (§ 679.20(a)(11)(ii)). The dates for the A season and the B season for the Pacific cod fishery differ from those of the A, B, C, and D seasons for the pollock fisheries.

Section 679.20(a)(6)(ii) requires the allocation of the Pacific cod TAC apportionment in all regulatory areas between vessels catching Pacific cod for processing by the inshore and offshore components. Ninety percent of the Pacific cod TAC in each regulatory area is allocated to vessels catching Pacific cod for processing by the inshore component. The remaining 10% of the TAC is allocated to vessels catching Pacific cod for processing by the offshore component. These seasonal apportionments and allocations of the Pacific cod TACs are shown below:

Table 2.11. Pacific cod allocations by season and area in the GOA

Area	**Gear**	**Season**	**TAC Apportionment**	**Inshore**	**Offshore**
W and C Regulatory Areas	H&L Pot Jig	Jan 1 – June 10	60	90	10
		Sept 1 – Dec 31	40	90	10
W and C Regulatory Areas	Trawl	Jan 20 – June 10	60	90	10
		Sept 1 – Nov 1	40	90	10
E Regulatory Area	All	Jan 1 – Dec 31 (Jan 20 for trawl)	100	90	10

Fishery Restrictions in the BSAI

Protection measures in the BSAI are more complicated than in the GOA because of additional types of areas that require protection beyond those listed in Tables 4 through 6 and 12 to 50 CFR Part 679. All closures in the BSAI are in 50 CFR 679.22. Rookery and haulout designations are based on Tables 4-6 and 12 of 50 CFR Part 679 and 50 CFR 223.202(a)(2)(i). The table below gives a general overview of closures:

Table 2.12. General Steller sea lion protection area closures in BSAI

Area	Restriction	Season	Exceptions
Rookeries BSAI	No groundfish fishing and no vessel transit 0-3 n mi	All year	
Rookeries Aleutian Islands	• No directed fishing for pollock 0-20 nm. • No directed fishing for Atka mackerel with trawl gear 0-10 W of 178º W and 0-20 nm E of 178ºW. • No directed fishing for P. cod by trawl 0-20 during harvest limit area (HLA) fishery, 0-10 after HLA fishery, and 0-10 nm E of 178ºW. • No directed fishing for P. cod by pot or hook-and-line 0-3 in W and C AI and W portion of EAI, and 0-20 in the E portion of EAI.	All year	Buldir I. closed to Atka mackerel trawling 0-15 nm, and no P. cod pot and hook-and-line directed fishing 0-10. Agligadak I. closed to P. cod trawl 0-20 nm.
Haulouts AI	• No directed pollock fishing 0-20 nm • No P. cod trawl 0-3 nm. • No P. cod trawl 0-20 during HLA Atka mackerel fishery. • No P. cod pot or hook and line fishing 0-20 nm in E portion of EAI. • No Atka mackerel trawl fishing 0-3 nm W of 178º W and 0-20 nm E of 178ºW.	All year	
Haulouts BS	• No directed pollock fishing varies from 0-3 nm to 0-20 nm. • No directed fishing for P. cod by trawl varies 0-3 nm to 0-20 nm. • No directed fishing for P. cod with pot or hook-and-line gear varies 0-3 nm to 0-20 nm. • No directed fishing for Atka mackerel 0-20 nm.	All year	
Area	**Restriction**	**Season**	**Exceptions**
Rookeries BS	• No directed trawl fishing for P. cod or pollock 0-10 nm. • No directed fishing for P. cod with hook and line or pot gear 0-3 nm. • No directed fishing for Atka mackerel with trawl 0 to 20 nm.	All year	Sea Lion Rocks (Amak) no pot or hook-and-line fishing for P. cod within 0-7 nm. Bogoslof I. and Adugak I. are in the Bogoslof Foraging Area, closed to directed fishing for pollock, P. cod and Atka mackerel.
Pribilof Is. haulouts	No directed trawl fishing for P. cod or pollock 0-3 nm	All year	
East of 178˚ W, trawl gear	Rookeries closed 0-10 n mi; haulouts closed 0-3 nm	All year	Agligadak closed 0-20 nm
West of 178˚ W, trawl gear	Rookeries & haulouts closed 0-20 nm until Atka mackerel fishery inside SSL CH is closed (applies to A & B seasons), then P cod trawling closed 0-3 nm of haulouts and 0-10 n mi of rookeries	All year	
Pot, H&L gear in	Closed in SSL CH east of 173˚ W to 170˚ W;	All year	

Aleutian Islands	Buldir rookery closed 0-10 nm; Agligadak rookery closed 0-20 nm		
Seguam foraging area	Closed to pollock, P. cod and Atka mackerel	All year	
Bogoslof foraging area	Closed to Atka mackerel, P. cod, and pollock directed fishing	All year	Bogoslof Pacific Cod Exemption Area H&L and jig vessels < 60' targeting P. cod allowed S of line extending from a point 3 nm N of Bishop Point to Cape Tanak
St. Lawrence & Hall Is., Cape Newenham, Round Is. haulouts	Closed 0-20 nm to pollock, P. cod and Atka mackerel	All year	
Unalaska/Bishop Point & Akutan I./Reef-Lava haulouts	No directed H&L fishing for P. cod 0-10 nm	All year	Vessels <60' E of 167ºW are exempt. Jig vessels prohibited in 10 nm Bishop Pt. Closure W of 167º W
Steller Sea Lion Conservation Area (SCA)	No directed fishing for pollock	A season	
Catcher Vessel Operating Area (CVOA)	No directed trawl C/P fishing for pollock	B season	

Pacific Cod

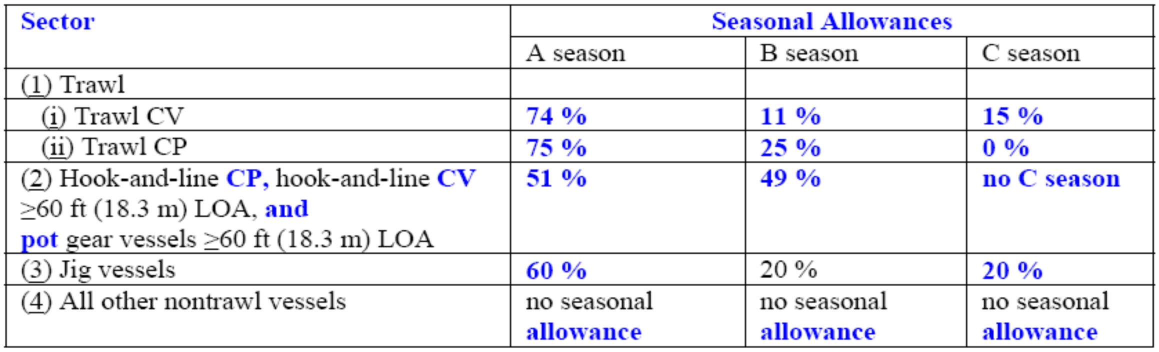
Amendment 85 was implemented January 1, 2008. This amendment to the BSAI FMP changes the amount of the BSAI Pacific cod TAC allocated to various sectors. The allocation is based on historic retained catch by each fishing sector, with some exceptions for the <60 ft fixed gear and jig gear sectors, yet remains within the overall intent of Steller sea lion protection measures as they pertain to seasonal and sector allocation requirements. This action modified the current allocations of BSAI Pacific cod TAC among various harvest sectors and seasonal apportionments thereof, establishes a hierarchy for reallocating projected unharvested amounts of Pacific cod from certain sectors to other sectors, revises catcher/processors (C/P) sector definitions, modifies the management of Pacific cod incidental catch that occurs in other groundfish fisheries, eliminates the Pacific cod nonspecified reserve, subdivides the annual PSC limits currently apportioned to the Pacific cod hook-and-line gear fisheries between the catcher vessel and C/P sectors, and modifies the sideboard restrictions for AFA C/P vessels. In addition, this action increased the percentage of the BSAI Pacific cod TAC apportioned to the CDQ Program.

In general, the amendment revises the initial annual allocations to each sector to reflect historic catch and mirrors the current temporal distribution of catch by overall gear sector, to account for TAC that is annually reallocated among gear sectors late in the year. Thus, while each sector's initial allocation is modified, the amount of the TAC allocated to each overall gear sector (trawl, fixed, and jig gear) in the first half of the year does not differ from status quo. The final Council motion is provided at: http://alaskafisheries.noaa.gov/npfmc/analyses/AM85motion406.pdf.

Pursuant to § 679.20(a)(7)(ii)(A), 1.4% of the Pacific cod initial total allowable catch (ITAC) is allocated to vessels using jig gear, 60.8% to vessels using hook-and-line or pot gear, 10.7% to CDQ, and 37.8% to vessels using trawl gear, including Amendment 80 sector vessels. Section 679.20(a)(7)(ii)(A) further allocates the portion of the Pacific cod ITAC allocated to trawl gear as 22.1% to catcher vessels, 13.4% to Amendment 80 vessels, and 2.3 percent to AFA C/P. CFR § 679.20(a)(7)(ii)(B) sets aside a portion of the Pacific cod ITAC allocated to hook-and-line or pot gear as an ICA of Pacific cod in directed fisheries for groundfish using these gear types. Based on anticipated incidental catch in these fisheries, the Regional Administrator currently specifies an ICA of 500 mt. The remainder of Pacific cod ITAC is further allocated to vessels using hook-and-line or pot gear as the following directed fishing allowance (DFAs): 48.7% to hook-and-line catcher/ processors, 0.2% to hook-and-line catcher vessels greater than 60 feet length overall (LOA), 1.5% to pot catcher/processors, 8.4% to pot catcher vessels greater than 60 ft LOA, and 2.0% to catcher vessels under 60 ft (18.3 m) LOA using hook-and-line or pot gear.

Due to concerns about the potential impact of the Pacific cod fishery on Steller sea lions and their critical habitat, the apportionment of the ITAC disperses the Pacific cod pot and hook-and-line fisheries into two seasonal allowances (see CFR § 679.20(a)(7)(iv)(A) and 679.23(e)(5)). For pot and hook-and-line gear vessels greater than 60 ft LOA and for all hook-and-line C/Ps, the first seasonal allowance of 51% of the ITAC is made available for directed fishing from January 1 to June 10, and the second seasonal allowance of 49% of the ITAC is made available from June 10 (September 1 for pot gear) to December 31. No seasonal harvest constraints are imposed for the Pacific cod fishery by catcher vessels less than 60 ft (18.3 m) LOA using hook-and-line or pot gear. For trawl gear catcher vessels, the first season is January 20 to April 1 and is allocated 74% of the ITAC. The second season, April 1 to June 10, and is allocated 11%. The third season, June 10 to November 1, is allocated 15%. The trawl catcher processors Pacific cod ITAC is allocated as 75% in the first season and 25% in the second season. For jig gear, the first season allocation is 60 percent and the second and third seasons are each allocated 20% of the ITAC. The table below lists the allocations and seasonal apportionments of the Pacific cod ITAC as specified in CFR § 679.20(a)(7)(iv)(A). In accordance with CFR § 679.20(a)(7)(iv)(B), any unused portion of a seasonal Pacific cod allowance will become available at the beginning of the next seasonal allowance, unless the unused allowance is in the jig fishery, which is distributed to pot and hook-and-line vessels less than 60 ft LOA.

Table 2.13. Pacific cod allocations by season in the BSAI (non-CDQ fisheries)



Sector	**Seasonal Allowances**		
	A season	B season	C season
(1) Trawl			
(i) Trawl CV	**74 %**	**11 %**	**15 %**
(ii) Trawl CP	**75 %**	**25 %**	**0 %**
(2) Hook-and-line **CP,** hook-and-line **CV** ≥60 ft (18.3 m) LOA, **and** **pot** gear vessels ≥60 ft (18.3 m) LOA	**51 %**	**49 %**	**no C season**
(3) Jig vessels	**60 %**	20 %	**20 %**
(4) All other nontrawl vessels	no seasonal **allowance**	no seasonal **allowance**	no seasonal **allowance**

Pollock

CFR § 679.20(a)(5)(i)(A) requires that the pollock TAC apportioned to the Bering Sea subarea, after subtraction of the 10 percent for the CDQ program and the 4% for the ICA, will be allocated as a DFA as follows: 50% to the inshore component, 40% to the catcher/processor component, and 10% to the mothership component. In the Bering Sea subarea, the A season (January 20–June 10) is allocated 40% of the DFA and the B season (June 10–November 1) is allocated 60% of the DFA. The AI directed pollock fishery allocation to the Aleut Corporation is the amount of pollock remaining in the Aleutian Islands subarea after subtracting 1,900 mt for the CDQ DFA (10%) and 1,600 mt for the ICA. When the Aleutian Islands pollock ABC is less than 19,000 mt, the annual TAC will be no greater than the ABC. When the AI pollock ABC equals or exceeds 19,000 mt, the annual TAC will be equal to 19,000 mt. In the Aleutian Islands subarea, 40% of the ABC is allocated to the A season and the remainder of the directed pollock fishery is allocated to the B season.

CFR § 679.20(a)(5)(i)(A)(4) also includes several specific requirements regarding pollock and pollock allocations. First, 8.5% of the pollock allocated to the C/P sector will be available for harvest by AFA catcher vessels with C/P sector endorsements, unless the Regional Administrator receives a cooperative contract that provides for the distribution of harvest among AFA catcher/processors and AFA catcher vessels in a manner agreed to by all members. Second, AFA catcher/processors not listed in the AFA are limited to harvesting not more than 0.5% of the pollock allocated to the catcher/processor sector.

The table below lists seasonal apportionments of pollock and harvest limits within the Steller Sea Lion Conservation Area (SCA). The harvest within the SCA, as defined at CFR § 679.22(a)(7)(vii), is limited to 28% of the annual DFA until April 1. The remaining 12% of the 40% of the annual DFA allocated to the A season may be taken outside the SCA before April 1 or inside the SCA after April 1. If the 28% of the annual DFA is not taken inside the SCA before April 1, the remainder is available to be taken inside the SCA after April 1. The A season pollock SCA harvest limit will be apportioned to each sector in proportion to each sector's allocated percentage of the DFA.

Table 2.14. Pollock allocations by season and area in the BSAI

Area	DFA	Season	DFA Allocation	Restriction
Bering Sea	Inshore 50% C/P 40% Mothership 10%	Jan 20 – June 10	40%	No more than 28% from the SCA before Apr 1
		June 10 – Nov 1	60%	
Aleutian Islands	Aleut Corp 100%	Jan 20 – June 10	40%	
		June 10 – Nov 1	60%	
Bogoslof		Closed		

Atka Mackerel

CFR § 679.20(a)(8)(ii) allocates the Atka mackerel TACs, after subtraction of the CDQ reserves, jig gear allocation, and ICAs, to the Amendment 80 and BSAI trawl limited access sectors. The allocation of the ITAC for Atka mackerel to the Amendment 80 and BSAI trawl limited access sectors is established in Table 33 to Part 679 and § 679.91. The CDQ reserve is 10.7% of the TAC for use by CDQ participants (CFR § 679.20(b)(1)(ii)(C) and 679.31).

Pursuant to § 679.20(a)(8)(i), up to 2% of the eastern Aleutian District and the Bering Sea subarea Atka mackerel ITAC may be allocated to jig gear. The amount of this allocation is determined annually by the Council based on several criteria, including the anticipated harvest capacity of the jig gear fleet. Currently there is a 0.5% allocation of the Atka mackerel ITAC in the Eastern Aleutian District and the Bering Sea subarea to the jig gear.

CFR § 679.20(a)(8)(ii)(A) apportions the Atka mackerel ITAC into two equal seasonal allowances. After subtraction of the jig gear allocation, the first seasonal allowance is made available for directed fishing from January 1 (January 20 for trawl gear) to April 15 (A season), and the second seasonal allowance is made available from September 1 to November 1 (B season) (see table below).

Pursuant to CFR § 679.20(a)(8)(ii)(C)(1), the Regional Administrator will establish a harvest limit area (HLA) limit of no more than 60% of the seasonal TAC for the western and central Aleutian Districts. A lottery system is used for the HLA Atka mackerel directed fisheries to reduce the amount of daily catch in the HLA by about half and to disperse the fishery over two districts (CFR § 679.20(a)(8)(iii)).

Table 2.15. Atka mackerel allocations by season and area in the BSAI

Gear	ITAC Gear Split	Area	Spatial ITAC Split	Season	Seasonal Allocation	Restrictions
Jig	~2%			Jan 1 – Dec 31		
Trawl	~98%	W & C Regulatory Areas	~68%	Jan 20 – Apr 15	50%	Each season's harvest limited to 60 percent of seasonal apportionment in W & C HLAs (see regulations)
				Sept 1 – Nov 1	50%	
		E Reg Area & Bering Sea	~32%	Jan 20 – Apr 15	50%	
				Sept 1 – Nov 1	50%	

2.5.2.4 Area Closed

Table 2.21 displays the amount of area closed and area composed of each critical habitat zone and for each fishery and area. Table 2.22 presents this information as a percentage of each zone which is closed within critical habitat. The amount of area that would have been closed under the 2000 FMP Biological Opinion is displayed in Table 2.23. Gear types are not listed separately because these closure areas are identical for all gear types. Overall, 63% of critical habitat was closed, with 65% of the 0-10 nm area was closed. One important difference in the closure areas was that under the 2000 FMP Biological Opinion, all closed areas were closed to all three species which would ensure no competition for any of the three, whereas under the 2001 conservation measures this is not the case. An area closed to pollock fishing may be open to Pacific cod or Atka mackerel fishing.

2.5.3 Monitoring and Evaluation of Fisheries Catch

Catch data used to manage the groundfish fisheries under the BSAI and GOA FMPs are collected from vessels, processors, and fishery observers trained by NMFS. This section discusses recordkeeping and reporting requirements, catch estimation, and the in-season fishery management programs. Monitoring of the fisheries is necessary to ensure that they are prosecuted in compliance with management regulations and do not threaten the health and status of the target stocks or the ecosystem, including listed species and critical habitat. The catch is monitored by a catch accounting system (CAS) which is designed to support in-season management of complex allocation schemes and harvest limits and ensures that fisheries do not exceed TACs or violate other fishery restrictions, such as time and area closures. NMFS also may use data from vessel monitoring systems to track effort in certain locations for estimating potential catch and timing of the fishery and to track compliance with harvest limitations in relation to Steller sea lion protection areas.

2.5.3.1 Recordkeeping and Reporting Requirements

Fishery participants issued FFPs and federal processor permits (FPPs) are required to comply with record keeping and reporting requirements to report groundfish harvest, discard, and production (50 CFR § 679.5). Reporting requirements include: logbooks, production reports, and landing reports.

Hard-copy (paper) logbooks are required to be completed and submitted for all vessels harvesting groundfish in the BSAI or GOA (except vessels less than 60). Logbooks contain information on the fishing gear and haul specific data on area fished. Logbook information is mainly used by NMFS for enforcement purposes.

Daily production reports are required for C/Ps, motherships, shoreside processors, and stationary floating processors that are issued an FFP or an FPP under 50 CFR §679.4. Production reports are submitted electronically to NMFS on a daily basis. Production reports include gear type, area fished, and a breakdown of the weight of each species and product. At-sea production reports also include the weight or number of each species that was discarded at sea.

Shoreside processors, stationary floating processors, and motherships that receive groundfish from vessels with an FFP are required to electronically submit landing reports to NMFS. Landing reports are trip-based for shoreside processors and weekly for motherships. Landing reports include the fishing start date, and the delivery date, fishing gear, area fished, and a breakdown of the weight and condition of each species delivered. Delivering vessels also provide information on at-sea discards which is included on landing reports.

2.5.3.2 Observer Requirements

The observer program for federally managed groundfish fisheries was authorized in 1990 and implemented by NMFS, effective February 7, 1990 (55 CFR 4839, February 12, 1990). Under this program, NMFS provides operational oversight, certification training, definition of observer sampling duties and methods, debriefing of observers, and management of the data. Vessel and processing plant owners pay the cost of the observers, and the costs associated with managing the program are paid for by the Federal Government. The design and focus of this observer program is on estimating fish and prohibited species catch and bycatch; protected species (marine mammal and seabird) bycatch information is also recorded.

Observer coverage is generally determined based on the size of the vessel prosecuting the fishery and, in several fisheries, the fishery in which the vessel is participating. Coverage levels for vessels harvesting groundfish under the FMPs is divided into three general categories: (1) vessels under 60 ft LOA are not required to carry observers; (2) vessels at least 60 ft and shorter than 125 ft are required to carry observers on 30% of their fishing days; (3) vessels 125 ft and longer are required to carry observers on 100% of their fishing days. Shoreside processors that process between 500 mt and 1000 mt of groundfish in a calendar month are required to have observers present 30 percent of the days that they receive or process groundfish. Shoreside processors that process 1,000 mt or more of groundfish in a calendar month are required to have observers present 100% of the days that they receive or process groundfish.

Observer coverage levels have been increased to implement certain limited access programs with increased monitoring needs, such as the western Alaska CDQ Program, AFA pollock fishery and, more recently, in the Rockfish Pilot Program and BSAI groundfish FMP Amendment 80 program.

Detailed information about the observer sampling protocols and extrapolations can be found in the North Pacific Groundfish Observer Sampling Manual (http://www.afsc.noaa.gov/fma/document.htm#Manuals1). In general, observers follow a standard sampling protocol (e.g., simple random sampling or systematic sampling). The observer information is shared with NMFS Alaska Regional Office (AKR), which uses the data provided from observed hauls, together with industry reported data to calculate catch and discard levels on unobserved hauls and trips.

2.5.3.3 Estimation of Groundfish Catch

NMFS' Catch Accounting System (CAS) was developed to receive catch reports from multiple sources, evaluate data for duplicate or errors, estimate the total catch by species or species category, and determine the appropriate "bin" or account to attribute the catch. These accounts are established to mirror the myriad combinations of gear, area, sector, and season that are established in the annual groundfish harvest specifications. The procedures for estimating catch and bycatch accommodate two important management components: first, the estimation procedures are designed to provide a quick turn-around of the data so that in-season management has useable rates as quickly as possible after receiving the landing reports and the observer data. The system makes maximum use of small amounts of observer data quickly (at coarser aggregation levels) which are updated and refined as more data become available. Secondly, although complex, the system was designed so that changes to the management structure could be reflected in the catch accounting structure to allow in-season management to stay current with fisheries regulations and specifications.

Groundfish Landings and Discard Estimates

Different accounting methods are used to estimate groundfish retained catch and discards for vessels that catch and process fish (C/Ps), vessels that do not process fish (catcher vessels), vessels that only receive and process fish (motherships), and shoreside processing plants. For large C/Ps and motherships that typically have 100% or more observer coverage, observer information is used for retained catch and discard accounting. For C/Ps and catcher vessels with 30% coverage, production reports are used to estimate retained catch. At-sea groundfish discards are estimated from observer data collected from the same component of the fleet (e.g.,catcher vessels using the same gear and fishing in the same area and target fishery). Discard rates on observed vessels are then applied to the reported groundfish catch on unobserved vessels. Any catch that is delivered to a shoreside plant and then discarded is accounted for using discards reported by the plant.

Prohibited Species Bycatch Estimates

Bycatch management measures for groundfish fisheries in the BSAI and the GOA have specific means to limit or reduce the incidental catch species traditionally harvested by other fisheries. These species include salmon, Pacific halibut, Pacific herring, red king crab, Tanner crab, and Snow crab. Collectively, these species are called PSC. Regulations require that in the groundfish fisheries, PSC are returned to the sea with no additional injury.

PSC bycatch estimates for unobserved catcher vessels using algorithms implemented in the CAS. Haul-specific observer information is used by the CAS to create PSC bycatch rates from observed vessels that are applied to total groundfish catch in each delivery by an unobserved vessel. The CAS is programmed to extrapolate information from observed vessels to unobserved vessels by matching the type of information available from observed vessels with that of an unobserved vessel. Surrogate bycatch rates are applied using the most closely available data from an observed catcher vessel by:

- processing sector (catcher vessel, C/P, or mothership)
- week ending date,
- target fishery,
- fishing gear,
- special area, and
- federal reporting area.

The PSC rate is calculated using the observed PSC bycatch divided by the groundfish weight, which results in a measure of PSC per metric ton of groundfish caught. As landings data are received from unobserved vessels, PSC estimates are created by finding the best possible matching rate and multiplying the groundfish catch by the PSC bycatch rate. PSC bycatch rates are calculated separately for each PSC species and are calculated in numbers of individuals for crab and salmon, and total weight of halibut and herring. All rates use information within a calendar year (i.e., 2006 information is not used for 2007).

2.5.3.4 In-season Management of TAC Apportionments

The sub-allocation of TACs among areas, sectors, and seasons results in a set of quotas monitored by NMFS. The CDQ program receives a percentage of the TAC for target groundfish species or species group fished in the BSAI, and a percentage of allowed limits for PSC. The overall CDQ suballocation is further divided into six quotas for each of the six CDQ groups. These quotas are monitored based on observer data, shoreside processor reports, or reports of IFQ landings. The sablefish IFQ fishery is monitored based on records from a real-time transaction processing system. The AFA pollock fishery TAC is divided among the CDQ groups, C/P sector, mothership sector, and inshore sector with seven inshore cooperatives and an open-access allocation for inshore vessels not participating in a cooperative. All pollock caught by vessels using pelagic trawl gear is attributed to directed fishing, and pollock caught with bottom trawl gear or non-trawl gear is considered incidental catch. The CDQ groups and inshore pollock cooperatives actively monitor their harvest and cease fishing activity before exceeding their quota. NMFS monitors the pollock harvest for the C/P and mothership sectors and can close a cooperative fishery if needed.

Separate pollock quotas have been established for the SCA in the Bering Sea. NMFS monitors pollock catch to ensure that the pollock quota inside the SCA is not exceeded. For catcher processors and motherships, the haul retrieval location as recorded by the observer is used to establish the location of catch. For catcher vessels, the State statistical area reported on the fish ticket is used to establish if the catch is inside the SCA. All vessels directed fishing for pollock are required to have a VMS unit that provides detailed information on the vessel location and speed. NMFS uses the VMS data to verify the State statistical areas reported on the fish tickets.

In 2008, NMFS implemented Amendment 80 for six groundfish species and halibut and crab PSC limits in the BSAI among trawl fishery sectors. Amendment 80 also facilitates the formation of harvesting cooperatives in the non-American Fisheries Act (non-AFA) trawl C/P sector. Amendment 80 species include: Aleutian Islands Pacific ocean perch, BSAI Atka mackerel, BSAI flathead sole, BSAI Pacific cod, BSAI rock sole, and BSAI yellowfin sole.

For the general groundfish fishery, which is all groundfish fishing that is not under the CDQ, individual fishing quota (IFQ), and cooperative programs for AFA and Amendment 80, NMFS monitors catch and issues regulatory notices to open and close specific fisheries. In some cases catch is monitored from daily or weekly reports and the closure date is projected by extrapolating catch rates. In cases where fishing effort is high relative to the available quota, NMFS will estimate the length of the fishery using historic effort and catch rates, and open the fishery for a specific length of time, ranging from as little as six hours up to several days.

A running total of PSC is maintained from a combination of observer reports from vessels and processors, extrapolated when necessary to unobserved vessels and processors.

2.6 Pending Actions

2.6.1 Salmon Bycatch Reduction, Amendment 91

Amendment 91 was developed by the Council and NMFS and evaluated in an EIS (NMFS 2009). This amendment would add salmon bycatch reduction measures to the current system of salmon bycatch reduction through an incentive program agreement (IPA) among pollock vessels. The action includes a hard cap on salmon bycatch and an industry IPA that would encourage minimizing the potential bycatch of Chinook salmon for the Bering Sea pollock fishery. This action is likely to result in constraints on the amount and location of pollock fishing in the Bering Sea. Effects on Steller sea lions and other ESA-listed species are likely dependent on the effects on the pollock and salmon stocks and the locations of fishing due to the industry's efforts to avoid salmon bycatch. No changes to the current Steller sea lion protection areas are in this amendment. The rulemaking for Amendment 91 is under development and if approved is scheduled for implementation in 2011. Effects on ESA-listed species are described in the EIS for this action, and a Section 7 consultation was completed on the potential effects on ESA-listed salmon with the NMFS NW region in December 2009.

2.6.2 National Standard 1 Revisions to the FMPs

The MSA requires FMPs to include a mechanism for specifying annual catch limits in the plan (including a multiyear plan), implementing regulations, or annual specifications, and setting harvests levels such that overfishing does not occur in the fishery and measures to ensure accountability. To assist in meeting this requirement, the National Standard 1 guidelines were revised in January 2009 to provide guidance on how to develop annual catch limits and the accountability measures to prevent overfishing. Many of the provisions of the guidelines were modeled after the current practices used to manage the Alaska groundfish fisheries. The FMPs will be amended to clarify language in the FMPs to reflect current practices for addressing uncertainty in the setting of harvest levels and the accountability measures used to ensure overfishing would not occur, all consistent with the NS1 guidelines.

Implementation is scheduled for January 2011 for the groundfish fisheries. One of the alternatives under consideration for this action would eliminate the other species assemblage and would set OFLs, ABCs, and TACs for each group of sharks, squids (in the GOA), octopi and sculpins. Minimal to no effects on Steller sea lions are expected from this action as it is not expected to change the management of the groundfish fisheries; it would only provide better descriptions in the FMPs of current practices which are currently described in the SAFE reports and implemented by the NMFS SFD. The splitting of the other species assemblage into their component groups may benefit Steller sea lions as the management of each of these groups would ensure the harvest limits are set based on a stock assessment of each group, rather than the current practice at the assemblage level. Management of the other species assemblage is done by summing the specifications for each component group harvest limit, which may allow for an individual group to be harvested beyond a level that is appropriate. Steller sea lions diet studies have shown predation on octopus and sculpin, so improved management of these species groups may reduce the potential for impacting these prey species, as is possible under current management.

2.7 Interrelated and Interdependent Actions

By regulation, the effects of an action include the direct and indirect effects of an action on listed species or designated critical habitat, together with the effects of other activities that are interrelated or interdependent with that action. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration (50 CFR 402.02).

Although not directly contained within the FMPs themselves, regulations at 50 CFR Part 600 implement the MSA provisions. Specifically, the issuance of scientific research permits may be provided by the Regional Administrator by § 600.745 (Scientific research activity, exempted fishing, and exempted educational activity). Except for the exempted fishing permits, these are authorized under the MSA only and provide the support for the resource assessment program and represent an interrelated action. This research may be conducted by either fishery research vessels or fishing vessels chartered by NMFS.

2.8 Action Area

The action area means “all areas to be affected directly or indirectly by the Federal action, and not merely the immediate area involved in the action” (50 CFR §402.02(d)). The action area is determined by the effects of the BSAI and GOA groundfish fisheries which remove fish biomass from the North Pacific Ocean and Bering Sea. The action area for this Biological Opinion includes the EEZ and state waters within the areas of the BSAI and GOA groundfish FMPs (Figure 2.2). These regions encompass those areas directly affected by fishing, and those that are likely affected indirectly by the removal of fish at nearby sites. The action area also includes state waters as they are areas that will be affected by the State parallel groundfish fisheries.

Of those fisheries identified in the FMPs within the action area, fisheries previously found to adversely affect Steller sea lions are the Atka mackerel, pollock, and the Pacific cod fisheries. In addition, the Alaska groundfish fisheries were determined to be likely to adversely affect sperm and humpback whales.

2.9 Critical habitat within the action area

Critical habitat designated for Steller sea lions occurs within the action area. Steller sea lion critical habitat is described in Section 3, and listed in 50 CFR part 226.202 (provided here as Tables 2.25 and 2.26 and Figures 3.2 and 3.3). Only those areas designated within Alaska are within the action area. Thus, critical habitat areas in Oregon and California are outside of the action area and are unlikely to be affected by the proposed action.

3 STATUS OF SPECIES AND CRITICAL HABITAT

Section 3(15) of the ESA, as amended states: "(T)he term "species" includes any subspecies of fish or wildlife or plants, and any distinct population segment of any vertebrate fish or wildlife which interbreeds when mature" (16 USC § 1532). Thus, under the ESA, DPS and subspecies are included in the definition of species and such entities are sometimes listed separately from other subspecies and/or DPSs of the same biological species.

Based on the best available scientific and commercial information, NMFS has determined that the action being considered in the Biological Opinion may adversely affect the following species and critical habitat listed under the ESA:

Steller sea lion (*Eumetopias jubatus*)

- Endangered Western Distinct Population Segment (western DPS)
- Threatened Eastern Distinct Population Segment (eastern DPS)
- Designated Critical Habitat for Steller sea lions

Humpback Whale (*Megaptera novaeangliae*) Endangered

Sperm Whale (*Physeter macrocephalus*) Endangered

Fin whale (*Balaenoptera physalus)* Endangered

The following summaries of information about the status of listed species and designated critical habitat are based on information found in numerous sources that include, but are not limited to: the Alaska Marine Mammal Stock Assessments (Angliss and Outlaw 2005, 2006; Angliss and Allen 2009) (available at http://www.nmfs.noaa.gov/pr/pdfs/sars/); the Biological Assessment for the current action provided by the SFD at the initiation of formal consultation (NMFS 2006); the FMP Biological Opinion (NMFS 2000); the Biological Opinion on Authorization of the BSAI and GOA groundfish fisheries (NMFS 2001) and Supplement (NMFS 2003); the Final Revised Steller Sea Lion Recovery Plan (NMFS 2008a); Environmental Impact Statements and Assessments related to research permitting (NMFS 2002, 2007) and fishery management (NMFS 2007); the NRC (2003) volume "Decline of the Steller Sea Lion in Alaskan waters: untangling food webs and fishing nets"; chapters within the volume "Sea Lions of the World" (Trites *et al.* 2006); the "Summary Document Steller Sea Lion and Northern Fur Seal Research Final Programmatic *Environmental Impact Statement January 2009*" (NMFS 2009a); the FPEIS (NMFS 2007); the Policy and Guidance document; the 2009 Biological Opinion on Full Implementation of the Preferred Alternative of the Programmatic Environmental Impact Statement (PEIS) for Research on Steller Sea Lions and Northern Fur Seals (NMFS 2009c); Stock Assessment and Fishery Evaluation Reports for the Gulf of Alaska and the Bering Sea (NPFMC 2009a, b); the Structure of Populations, Levels of Abundance and Status of Humpbacks (SPLASH) Final Report (Calambokidis *et al.* 2008); the Humpback Whale Recovery Plan (*Megaptera novaeangliae*) (NMFS 1991); the Sperm Whale Draft Recovery Plan (*Physeter macrocephalus*) (NMFS 2006c); the Fin Whale Recovery Plan (NMFS 2010); peer-reviewed scientific literature; white papers, unpublished reports, and research summaries from government agencies, academic institutions, and private industry; and communication with species experts and observers as identified in the literature cited. Our summary of the status of listed species and their critical habitat is based on the best scientific and commercial data available, and information related to the status, distribution, abundance, habitat use, and ecology of these species.

We provide a concise summary of key findings from the status section in the synthesis portion of the Conclusions Section. Below, we provide more detailed summaries, updates, and some discussion of information on aspects of the status of species and critical habitat that are helpful for understanding the effects of the proposed action on the listed species and designated critical habitat.

3.1 Steller Sea Lions: Western and Eastern Distinct Population Segments

3.1.1 Species Description

The species Steller sea lion (*Eumetopias jubatus*) is classified within the Order Carnivora, Suborder Pinnipedia, Family Otariidae, and Subfamily Otariinae. The genus *Eumetopias* contains one species, the Steller (also called northern) sea lion, *E. jubatus*. As the Steller sea lion is the only extant representative of its genus, the extinction of the Steller sea lion would result not only in the loss of a species, but in the loss of a genus.

Steller sea lions are the largest otariid and show marked sexual dimorphism with males 2-3 times larger than females. The average standard length is 282 centimeters (cm) for adult males and 228 cm for adult females; weight of males averages 566 kg and females 263 kilograms (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 six months (Daniel 2003). A more detailed physical description is given in Loughlin *et al*. (1987) and Hoover (1988).

The marked sexual dimorphism and large size of adults, but especially of adult males, are both features of Steller sea lion morphology that are important to understanding effects of human activities on this species and to understanding why it is difficult to obtain certain kinds of information about adults. For example, smaller animals are vulnerable to injury or even death if trampled by adults, especially males. The large size of Steller sea lion adults also makes their capture, handling, and even salvage (of dead animals) more challenging than many other pinnipeds, thus making it difficult to obtain certain kinds of information (e.g., body condition) from live adults or even dead large adult males.

3.1.1.1 Distribution

The range of the Steller sea lion 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 (Figure 3.1) (Kenyon and Rice 1961, Loughlin *et al*. 1984, 1992). Seal Rocks, at the entrance to Prince William Sound, Alaska, is the northernmost rookery (60°09'N). Currently, Año Nuevo Island off central California is the southernmost rookery (37°06'N), although in the past (until 1981) some pups were born farther south at San Miguel Island (34°05'N).

Prior to the Steller sea lion decline in the west, most large rookeries were in the GOA and Aleutian Islands (Kenyon and Rice 1961, Calkins and Pitcher 1982, Loughlin *et al*. 1984, 1992, Merrick *et al*. 1987). Historically, these areas were supportive of very large numbers of Steller sea lions. However, as the decline continued, rookeries in the west became progressively smaller. Consequently, the largest rookeries are now in Southeast Alaska and British Columbia.

Most adult Steller sea lions occupy rookeries[4] 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 haulouts. Adult males, in particular, may disperse widely after the breeding season. During fall and winter many Steller 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 pups 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. Subsequent observations documented pup movements with mothers of over 800 km. Juvenile animals were seen at much greater distances from their rookery of birth (up to 1,785 km). Sightings of adults were generally less than 500 kilometers (km) away from the natal rookery, although adult males have since been seen over 1000 km from the rookery where they held a territory.

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.

3.1.1.2 Population Structure

The Steller sea lion was originally listed as one population under the ESA but was reclassified into two distinct population segments (DPS) based on genetic studies and phylogeographical analyses which indicated genetic differentiation among the eastern and western portion of its range (62 FR 24345). For a population to be listed as a DPS under the ESA it must be genetically or geopolitically[5] discrete and significant in terms of maintaining continuity in a species range, or evolutionarily significant as evidenced by marked differences in behavior or biology (including genetics). The supporting information for evaluating Steller sea lion DPS designations included a review of genetics, distribution, population response, and phenotypic characteristics (Loughlin 1997).

Genetic differentiation between Steller sea lions in the eastern and western DPSs was found primarily through studies using mitochondrial DNA (mtDNA) to evaluate patterns of genetic variability within and among Steller sea lions from various locations throughout their range. This type of genetic material exhibits maternal inheritance, meaning that offspring inherit their mother’s sequence (barring mutation). Several studies in the late 1990s analyzing mtDNA in Steller sea lions found a high level of genetic diversity with a large number of haplotypes occurring at a relatively low frequency (Bickham *et al.* 1996, 1998a, and 1998b) from samples in the Commander Islands, Kuril Islands, Gulf of Alaska, British

---

[4] Throughout this document a rookery refers to a site where pups are born (usually a count of 50 or more pups), 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.

[5] Under the ESA, a species may be considered a discrete population if its range crosses international boundaries which would preclude the authority of the ESA.

Columbia, Oregon, and California. 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.

The resulting regulatory division between DPSs occurs at Cape Suckling (144º W longitude) in the northeast Gulf of Alaska. The eastern DPS includes Steller 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) throughout their range including Russia/Asia.

Since the early mtDNA studies, substantial additional genetic research has been conducted with larger sample sizes from throughout the Steller sea lion range, including most rookeries in Asia. The results of these studies generally confirm the strong eastern/western DPS delineation, but differ in their description of further structure within the western DPS (e.g., Trujillo *et al.* 2004, Baker *et al.* 2005, Harlin-Cognato *et al*. 2006, O'Corry-Crowe *et al*. 2006, Hoffman *et al.* 2006, 2009, NMFS unpublished data). Research conducted by O'Corry-Crowe *et al.* 2006 confirmed genetic differentiation between Steller sea lions in western versus southeastern Alaska (again supporting current DPS designations within Alaska). This study also found a clear separation in mtDNA between oceanic and shelf rookeries within the western DPS as well as significant differentiation among rookeries in the western DPS with a disjunction at Samalga Pass. This split coincides with differences in Steller sea lion population trends and foraging ecology, as well as oceanographic divergence between ocean basin and continental shelf waters.

Further genetic studies have been conducted examining the possibility of a third population, termed the Asian population, just west of the Commander Islands in Russia (Baker *et al.* 2005, Hoffman *et al.* 2006). Using mtDNA analysis, Baker *et al.* (2005) recommended that the western DPS be partitioned from Asian Steller sea lions west of the Commander Islands, yielding a western DPS that ranges from Prince William Sound west to and including the Commander Islands, and an Asian DPS that includes rookeries from the Kamchatka Peninsula, Kuril Islands, and Sea of Okhotsk. In another study, an examination of nuclear DNA found little evidence to support the separation of an Asian DPS due to potentially extensive male gene flow (Hoffman *et al.* 2006), but investigators did detect a clear phylogenetic break between animals in U.S. and Asian parts of the western DPS versus those in the eastern DPS, concluding that population structuring reflects a genuine discontinuity within the range. A morphometric study of Steller sea lion skulls by Phillips *et al.* (2009) indicated that two subspecies of *Eumetopias jubatus* be recognized based on observed morphological differences, one including the Asian and western DPSs, and the other the eastern DPS.

It should be noted that some public comments challenged the merits of managing the species as two distinct population segments (e.g., Boyd 2010). It is the Agency position that such concerns are not appropriately debated in a Section 7 consultation. Rather, the merits of a given DPS designation are best addressed during periodic status reviews or the review of the current Recovery Plan.

3.1.1.3 Movement between Populations

Steller sea lions may disperse from their rookeries of birth and breed at other rookeries within their parent populations. This breeding dispersal has the potential to affect local population dynamics and patterns of underlying genetic variation. Some authors (e.g., O'Corry-Crowe *et al*. 2006, Hoffmann *et al.* 2009) have concluded that the population structure within the western DPS is consistent with one or more metapopulations (*sensu* Hanski and Simberloff 1997). Occasional dispersal of animals from their natal rookeries may also have important consequences for expansion of the eastern population and possible recovery of the western DPS, as it provides a mechanism for occupying new territory or re-occupying areas that were once rookeries or haulouts but which are now vacant (Raum-Suryan *et al*. 2002). New rookeries were established in Southeast Alaska as population size increased. These rookeries have been colonized in part from 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 in part by breeding females from the western DPS (NMFS unpublished, Pitcher *et al.* 2007). Brand/resight studies in the Russia/Asia portion of the western DPS range indicate fairly extensive movements (Burkanov 2010), in some cases many hundreds of kilometers from the site where the animal was marked.

Recent studies have confirmed movement of animals across the eastern DPS/western DPS boundary (Raum-Suryan *et al.* 2002, Gelatt *et al*., 2006, Pitcher *et al.*, 2007). Animals branded as pups in one DPS have occasionally been resighted at haulouts and rookeries within the other DPS. Recently observed movement of animals has also indicated that the geographic boundary between the western and eastern populations as it existed at the time of the listing of two DPSs may be changing or blurring at the edges (Gelatt *et al*. 2006, Pitcher *et al.* 2007, NMFS unpublished). Of the two most recently established rookeries in the eastern DPS, Graves Rock and White Sisters, about 70% and 45%, respectively, of the pups born in 2002 (a maximum of 159 pups total from both rookeries used in the analysis) had mtDNA haplotypes that had previously been found only in western DPS females (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). If dispersal from one DPS to another for breeding and pupping became more common, the pattern of genetic variation throughout portions of the range, especially at the boundary between the two DPSs, would likely change over time.

3.1.2 Listing Status

In the 1950s, the worldwide abundance of Steller sea lions 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. In the 1980s, annual rates of decline in the range of what is now recognized as the western population were as high as 15% per year. By 1990, the U.S. portion of the population had declined by about 80%. Numbers on certain rookeries had declined by 63% since 1985 and by 82% since 1960. On April 5, 1990, NMFS (55 FR 12645) issued an emergency interim rule to list the Steller sea lion as threatened and request public comment. On November 26, 1990, NMFS issued the final rule (55 FR 49204) to list Steller sea lions as a threatened species under the ESA. After listing, the rate of decline decreased to about 5% per year.

NMFS reclassified Steller sea lions as two distinct population segments under the ESA in 1997. At this time, the western DPS, extending from Japan around the Pacific Rim to Cape Suckling in Alaska (144°W), was up-listed to endangered status due to its continuous decline and lack of recovery (Figure 3.1 depicts the 144°W boundary). This endangered status listing was supported by Population Viability Analysis (PVA) which indicated that a continued decline at the 1985-1994 rate would result in extinction of the western DPS in 100 years or a 65% chance of extinction if the 1989-1994 trend continued for 100 years (62 FR 24354).

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, the larger decline overall in the U.S. population, human interactions, and the lack of recovery in California (62 FR 24354). However, at present, the eastern DPS appears to have met the recovery criteria laid out in the Final Revised Steller Sea Lion Recovery Plan (NMFS 2008a). NMFS is currently evaluating the status of this stock for possible de-listing.

Critical habitat for Steller sea lions 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 items (Figures 3.2 and 3.3).

In the final rule, NMFS stated that essential habitat for Steller sea lions includes terrestrial, air, and aquatic areas, and that physical and biological features within this habitat that support reproduction, foraging, rest and refuge are essential to the conservation of the species. With respect to the terrestrial habitat, NMFS concluded that the suitability of a particular area for Steller sea lions is influenced by substrate, exposure to wind and waves, the extent and type of human activities and disturbance in the region, and proximity to prey. For the aquatic habitat areas, the essential at-sea activity is presumed to be feeding and access to adequate food resources. An in-depth description of critical habitat areas for Steller sea lions is provided in section 3.2.

3.1.3 Population Status and Trends

NMFS monitors the status of Steller sea lion populations by conducting counts of animals during the breeding season at a consistently surveyed set of terrestrial rookeries and haulouts called trend sites (since they are used to assess population trend; NMFS 2008a). Trend sites include the majority of animals observed in each survey, and consist of two groups: those consistently monitored since the mid 1970s (70s trend sites) and 1991 (90s trend sites). Count data used to estimate the trend of Steller sea lion population abundance and pup production are of two types: counts of pups about 1 month of age on rookeries and counts of animals over 1 year of age (i.e., non-pups) on rookeries and haulouts.

Until recently, pups were counted by observers on rookeries, herding the non-pups into the water, and walking through the rookery (Calkins and Pitcher 1982, Sease *et al*. 2001). Beginning in 2005, vertical film or digital photography has been used to count pups in the Alaska portion of the western stock (Westlake *et al*. 1997, Snyder *et al*. 2001, Fritz *et al*., 2008). There may be a higher chance of inclusion of dead pups in the counts based on aerial survey data than those from on-site counts. Kaplan *et al.* (2008) examined SSL pup mortality from birth to 3 weeks at a rookery near Forrester Island in the eastern DPS. Mean survival rate at multiple sites was 0.68, with highest mortality in day 1. They reported large numbers of dead pups unseen by standard observation techniques, including pups wedged between rocks, or pups swept to sea and lying dead on the nearshore seafloor.

Prior to 2004, non-pups in Alaska were counted from 35 millimeter (mm) color slides taken from the side windows of aircraft during the breeding season (Calkins and Pitcher 1982, Merrick *et al.* 1987, Sease *et al.* 2001). Since 2004, vertical high resolution photography has been used. Counts from oblique 35 mm and vertical photographs were highly correlated but, on average, were 3.6% higher than vertical high resolution photographs, requiring an adjustment when counts from both methods are used in the same analysis (Fritz and Stinchcomb 2005).

Counts of pups on rookeries are conducted near the end of the birthing season in late June through mid-July. Based on an estimated ratio of pups to non-pups in the population, these counts can be utilized to estimate approximate total population size (Calkins and Pitcher 1982, Trites and Larkin 1996). Using estimates of birth rate and sex and age structure of a stable Steller 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. Estimates of total population size obtained using this methodology are highly uncertain since the accuracy of the pup count multiplier is affected by temporal and spatial variation in natality and survival rates, sex ratios, and age structure. The use of multipliers to estimate Steller sea lion population size is based on older studies and likely merit updating or revision (see Chapter 10).

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 (NMFS 2008a). Replicate surveys conducted in 1992 and 1994 confirmed NMFS understanding of Steller sea lion haul-out behavior patterns. The number of Steller 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 the small scale movement between sites within the same sub-area. Therefore, NMFS has reported the non-pup survey data as pooled counts of consistently surveyed sites within sub-areas to better reflect trend within the subarea. Estimating trend by analysis of individual haulouts ignores this movement and focuses attention on a single haulout which is known to be highly variable (much more so than a rookery) rather than the region in question. 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 contained in the RPA from 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).

3.1.3.1 Worldwide Population Trends

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 within the range of what is now recognized as 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 because the population in what is now known as the western DPS had declined while the population in what is now called the eastern DPS had increased. Loughlin *et al.* (1984) were not arguing that animals had moved but rather than there had been concurrent diverging population trends. 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. Through the 1990s, the eastern DPS increased at approximately 3% per year (Pitcher *et al.* 2007), while western DPS continued to decline at approximately 5% per year throughout its range (Burkanov and Loughlin 2005; NMFS 2008a). The worldwide population of Steller sea lions likely reached its smallest size (~105,000; NMFS unpublished) in 2000 when the overall decline of the western DPS stopped. In 2008, the worldwide population of Steller sea lion was estimated to total about 133,000 animals, with approximately 70,000 in the western DPS and 63,000 in the eastern DPS (V. Burkanov, personal communication for the western DPS in Russia; 1.4% per year increase [not significantly different than zero] between 2000 and 2008 for the western DPS in Alaska; continued 3.1% per year increase [significantly different than zero] in the eastern DPS).

3.1.3.2 Western DPS Status and Trends

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).

Alaska (U.S. portion of the range)

Trends in Adult and Juvenile (non-pup) Counts

Steller sea lions use 38 rookeries and hundreds of haul-out sites within the range of the western DPS in Alaska (Figures 3.4-3.6). 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)[6] [7].

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 3.1a[8]). 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). 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 3.1a). Approximately 110,000 adult and juvenile Steller 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.

Between 1990 and 2000, trend site 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 1970s (Figure 3.7). 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).

Counts of adult and juvenile Steller sea lions at all trend sites within the range of the western DPS in Alaska increased 12% between 2000 and 2008, and most of this increase occurred in the first four years (11% increase between 2000 and 2004; Table 3.1b). Non-pup surveys conducted in 2006 and 2007 (Fritz *et al*. 2008, DeMaster 2009) did not result in complete assessments of the population (Fritz *et al*. 2008, 2009). In the core of the western DPS range in Alaska (Kenai-Kiska), all of the 2000-2008 increase of 10% occurred between 2000 and 2004. In the larger Kenai-Attu region, counts increased 7% in the first four years, but then dropped slightly between 2004 and 2008. Consequently, the overall increase of 3% observed between 2004 and 2008 in the western DPS in Alaska was due entirely to a 35% higher count in the eastern Gulf of Alaska (E GULF; Table 3.1b).

- Non-pup counts in the following regions increased between 2000 and 2008: eastern AI, western GOA, and eastern GOA (Table 3.1b, Table 3.1c).

- Non-pup counts in the following regions decreased between 2000 and 2008: western AI, central AI, and the central GOA (Table 3.1b, Table 3.1c).

During the 2008 survey, NMFS counted greater numbers in the eastern GOA (western DPS) and fewer in southeast Alaska (eastern DPS) than expected based on the recent trends in both areas. Because the 2008

---

[6] For the western DPS of Steller sea lion in Alaska, count data have generally been combined and analyzed in six subareas (Figure 3.4), 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.

[7] Nelson (1887) reported on natural history collections taken in Alaska from 1877-1881. They estimated large numbers of Steller sea lions in the Pribilof Islands (over 25,000) and relatively low numbers throughout the Aleutian Island chain. This information seems to be based on conversations with Aleuts and their hunting experience as well as with westerners on the Pribilof Islands. Their methods are unclear and impossible to evaluate. In general, they indicate that there may have been some dense aggregations of sea lions but otherwise somewhat scarce (relative to the Pribilofs) throughout the Aleutians.

[8] 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.

survey was conducted early in the breeding season (early June), NMFS hypothesized that the unexpected results in the eastern Gulf of Alaska and southeast Alaska were due to animals from southeast Alaska foraging in the Prince William Sound area in late spring prior to returning to their southeast Alaska rookeries for breeding. This hypothesis was supported by data collected during the 2009 survey, which was conducted later in the breeding season in these areas. NMFS estimated that approximately 570 animals from southeast Alaska were counted on trend sites in the eastern Gulf of Alaska during the 2008 survey. Subtracting these from the 2008 western DPS non-pup trend site total reduced the percent differences between 2000 and 2008 from 14% to 12%.

Alternative Analysis of Regional Trends in Non-Pup Counts within the Alaskan Western DPS 1991-2008

NMFS (2008) uses six regions within the western DPS in Alaska for trend and status monitoring, three (eastern, central and western) within both the Aleutian Islands and Gulf of Alaska. Fritz *et al.* (2008) noted that groups of rookeries and haulouts within the central Aleutian Islands region, the largest of the six, had different trends, and that this spatial pattern was lost when counts were aggregated to the entire region. Furthermore, this suggested that sea lion populations were responding similarly within portions of the range and at finer scales than previously considered. Thus, AFSC (2010a) divided the Alaskan western DPS into 11 Rookery Cluster Areas (RCAs), (1-10 from west to east) and also utilized data from southeast Alaska (RCA 11). RCA boundaries as they relate to Steller sea lion Recovery Plan areas and fishery management areas are depicted in Figure 3.8. The RCA boundaries were determined based on demographic similarities of animals in groups of Steller sea lion sites, similarities in abundance trends among groups of sites, locations of Steller sea lion survey region boundaries, and the current locations of fishery management subarea boundaries.

- RCA 1 is the same as the western Aleutian Islands survey sub-region; non-pup counts declined at 7.2% per year between 2000 and 2008.
- RCA 2 is the western portion of the central Aleutian Islands survey subarea from Kiska Island to Amchitka Pass; non-pup counts declined at 4.4% per year between 2000 and 2008.
- RCA 3 encompasses the Delarof Island group, and Kanaga and Tanaga Islands in the central Aleutian Islands from Amchitka Pass to 177° W; non-pup counts declined at 1% per year between 2000 and 2008.
- RCA 4 is from Adak through Atka Islands in the central Aleutian Islands; non-pup counts declined between 1991 and 1994, increased from 1994 through 2004, and then declined through 2008
- RCA 5 is the eastern portion of the central Aleutian Islands from Amlia Island through the Islands of Four Mountains; non-pup counts declined between 1991 and 1996, then increased slowly through 2008.
- RCA 6 is the same as the eastern Aleutian Islands survey subarea; non-pup counts were stable through the 1990s, but increased at 3.4% per year from 2000 through 2008.
- RCA 7 is essentially the western Gulf of Alaska survey subarea but without Lighthouse Rocks; non-pup counts were stable through the 1990s, but increased at 4.7% per year from 2000 through 2008.
- RCA 8 is essentially the western portion of the central Gulf of Alaska survey subarea with the addition of Lighthouse Rocks; non-pup counts declined at ~6% per year through the 1990s and were stable from 2000 through 2008
- RCA 9 is essentially the eastern portion of the central Gulf of Alaska, but without the southwestern portion of the Kenai Peninsula; non-pup counts declined at ~6% per year through the 1990s and were stable from 2000 through 2008

- RCA 10 is essentially the eastern Gulf of Alaska survey subarea with the addition of the southwestern portion of the Kenai Peninsula; non-pup counts declined at ~9% per year through the 1990s and increased in the 2000s. Counts increased at 5.4% per year between 2000 and 2008.
- RCA 11 is the same as SE Alaska (the eastern DPS in Alaska); non-pup counts were stable between 1991 and 1996, and increased through 2002; the 2008 data utilized in NMFS-AFSC (2009) were compromised due to the early June survey date.

Regional Trends in Adult and Juvenile western Steller sea lion Counts

The AFSC examined regional trends in the western DPS Steller sea lion adult and juvenile (non-pup) counts at the 161 trend sites routinely surveyed since 1991. AFSC (2010c) compared three trend estimators derived from various methods (geometric Brownian motion models, linear regression of log counts, and combination Delta method and MCMC simulations). Regional trends were aggregated at three different spatial scales within the U.S. range (i.e., RCAs, sub-regions identified in the SSL Recovery Plan, and NMFS fishery management areas in the Aleutian Islands) using non-pup count data from 2000-2008. The results, including point estimates of the trend and associated 90th percentile confidence intervals are presented in AFSC (2010c) and the results for the most statistically robust method (geometric Brownian motion models conducted by Johnson 2010) are shown in Table 3.9. The results of the trend analysis for all three methods revealed the same spatial patterns (AFSC 2010c).

The general relationship between RCAs, sub-regions in the SSL Recovery Plan, and the three Aleutian Islands fishery management areas is shown below:

RCA	1	2	3	4	5	6	7	8	9	10
NMML	WAI	CAI				EAI	WGOA	CGOA		EGOA
NMFS	543	542		541						

Results of Johnson's analysis in AFSC (2010c) when aggregated by RCA, show a relatively steep population decline in RCA 1 (-7.2% per year) and a gradually slower decline to the east through RCA 4 (Table 3.9, Figure 5.1). Trends from RCA 5 through 7 were positive and gradually increasing from west to east. Further, non-pup counts were stable in RCAs 8-9, while to the east (RCA 10) and west (RCA 7), non-pup counts increased at between 4-5% per year (Table 3.9 Panel A, Figure 5.1).

For the Recovery Plan sub-region data, there was a gradual increase in the population trend from the western Aleutian Islands eastward through the western Gulf of Alaska, the central Gulf of Alaska was estimated to be stable, and the eastern Gulf of Alaska was estimated to be increasing at approximately 5% per year (Table 3.9 Panel B).

For the Aleutian Island fishery management areas, there was a gradual increase in population trend from a steeply declining population in 543 to a stable population in 541 (Table 3.9 Panel C). Overall, the estimated western DPS rate of population change was 1.4% per year (90% CI: -0.3%, 3.3%).

Trends in Pup Production

Pups have been counted less frequently than non-pups, but the overall trends since the late 1970s have been similar to counts of non-pups (Table 3.2, Figure 3.9). In both 2005 and 2009, all rookeries and major haulouts in Alaska were surveyed using high resolution vertical photography, permitting estimation of annual pup production within the state. In previous years, groups of rookeries were surveyed such that many years were required before a statewide estimate could be determined.

Between 2001/02 and 2009, rookery pup production declined 43% in the western and 7% in the central Aleutian Islands, but increased 47% in the eastern Aleutian Islands, and 23%, 6%, and 57% in the western, central, and eastern Gulf of Alaska, respectively, for an overall western DPS change of +14% (Table 3.2; Figure 3.9). This is equivalent to an increase of approximately 5 pups per rookery per year during the period 2001/02 through 2009. Analysis of recent regional and overall trends (Table 3.2; Figure 3.9) within the western DPS in Alaska indicates that:

- Pup production in the eastern Gulf of Alaska, eastern Aleutian Islands, western Gulf of Alaska and central Gulf of Alaska (listed in order of rates of increase from high to low) increased at rates of +5.8% (P=0.025), +5.3% (P=0.003), +3.3% (P=0.122), and +1.04 (P=0.390) per year from 2000 through 2009, respectively;
- Pup production in the western Aleutian Islands decreased at a rate of -7.13% (P=0.120) per year from 2000 through 2009; this includes the 2005 count from Attu/Cape Wrangell; in the central Aleutian Islands it decreased at a rate of -0.9% (P=0.093) per year from 2000 through 2009; and
- In the western DPS in AK overall increased at a non-significant rate of 1.74% per year (P=0.173) from 2000 to 2009.

The western DPS continues to show significant improvement in pup production in the core of its range, the eastern Aleutian Islands and western Gulf of Alaska. For instance, at both Clubbing Rocks and Ugamak Island, pup counts in 2009 were the greatest in over 20 years. In addition, South Rocks produced 60 pups in 2009, for the first time surpassing the 50 pup threshold traditionally used for rookery designation. There were increases in pup production between 2005 and 2009 at all rookeries in the central Gulf of Alaska except Chowiet Island, but since 2001/02, the number of pups has increased 6% in this area (Table 3.2). Pup counts in the eastern Gulf of Alaska between 2005 and 2009 increased by over 200 (+28%; Table 3.2). This increase occurred almost entirely at one rookery, Seal Rocks, which is the easternmost rookery in the range of the western DPS. NMML and ADF&G have proposed to obtain genetic samples from pups born on Seal Rocks and other rookeries at the eastern edge of the western DPS (as well as samples from pups born at northern rookeries in southeast Alaska) to investigate potential recent developments in population structure.

Pup production continues to decline in the western and central Aleutian Islands. Pup counts at four rookeries in these two sub-areas (Attu/Cape Wrangell and Buldir in the western, and Ayugadak and Amchitka/Column Rocks in the central) in 2005-2009 declined to less than 50. Buldir, with only 7 pups produced in 2008, may have ceased to function as a rookery. There is a boundary within the central Aleutian Islands at approximately 178°W (Tanaga Island) that separates declining rookeries to the west in the Near, Delarof and Rat Islands from stable or slightly increasing rookeries to the east in the Andreanof and Fox Islands (Figure 3.10). The 11 rookeries west of 178°W produced 268 fewer pups in 2009 than in 2005 (-17%) and now account for only 13% of rookery pup production within the Alaska western DPS, half the percentage that this region contributed in 1998.

Because NMFS was unable to survey any sites in the western Aleutian or Pribilof Islands in 2009, the best available information on pup production in these areas was collected in 2005 and 2008 (Fritz *et al.* 2008a;b). Consequently, the total pup production of 11,120 (DeMaster 2009) reported for 2009 includes counts from 2005 at Attu/Cape Wrangell in the western Aleutian Islands and at Walrus Island in the Pribilof Islands, and counts from 2008 at three other rookeries (Agattu/Cape Sabak, Agattu/Gillon Point, and Buldir) and one major haulout (Alaid) in the western Aleutian Islands. Pup production in the western Aleutians has declined steadily since the late 1990s. Consequently, data from 2005 and 2008 collected in this area likely over-estimate pup production in 2009.

Ratios of Pups to Non-Pups on Rookeries in 2009

The ratio of pups to non-pups provides a proxy of sorts for natality. Holmes *et al.* (2007) estimated that Steller sea lion natality rates in the central Gulf of Alaska declined 36% between the late 1970s and 2004 based on demographic modeling. They also showed that pup to non-pup ratios declined in the western

Gulf of Alaska and the eastern Aleutian Islands during this same period as evidence to suggest that declines in natality rates may not be limited solely to the central Gulf sea lion population. Pup to non-pup ratios based on data collected in 2009 are consistent with the interpretation that natality rates of western DPS Steller sea lions are lower than those in southeast Alaska (DeMaster *et al.* 2009). At the two largest and oldest rookeries in southeast Alaska (Forrester Complex and Hazy Island), the pup to non-pup ratio was 0.85 in 2009. Pitcher *et al.* (2007) reported a ratio of 0.75 in 2002. Rookery pup to non-pup ratios within the western DPS in AK ranged from 0.44 to 0.63 by sub-area in 2009, and averaged 0.57, or 33% lower than in southeast Alaska. While rookery pup to non-pup ratios are not estimates of actual female natality (since they include juveniles and males in the denominator), they provide insight into the relative birth rates of females within each region since females dominate rookery populations. For example, pup to non-pup ratios can be reduced because there are few pups per female, and because dependent juveniles from births in previous years are present with their mothers on the rookery. Both of these factors, however, would suggest reduced birth rates compared with rookeries with higher ratios.

The extent to which sub-adult males and other weaned juveniles haul out on rookeries will also affect pup to non-pup ratios and can vary between rookeries independent of differences in natality. The two southeast Alaska rookeries are likely near historical highs in pup production and density of animals on shore, which may inhibit juveniles and sub-adult males from hauling out on these rookeries compared to the smaller, less dense rookeries within the western DPS. Thus, lower pup: non-pup ratios documented for the western DPS may result from a larger fraction of juveniles and sub-adult males observed at rookeries in the western DPS than in southeast Alaska which could reduce pup to non-pup ratios independent of changes in female natality rates.

Pribilof Islands

The breeding population of Steller sea lions on the Pribilof Islands has largely disappeared and is not currently considered in the trend analyses for the western DPS. 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 Steller 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. He noted that the population was growing and that the Steller 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 in the late 1800s and early 1900s appears to have greatly reduced the size of the Pribilof Steller sea lion population. 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 that time. 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). However, between the 1960s and 2005 numbers on Walrus Island declined over 90%, to only 322 non-pups in 2001 and 29 pups in 2005 (Figure 3.6; NMFS 2008a). In the 2000s, Steller sea lions appear to have increased in abundance on an historic haulout at Dalnoi Point on St. George Island. Animals branded from other sites in the North Pacific Ocean have been observed at this site as well as females attending pups (Max Malavansky, pers. comm.).

Russia and Asia

Steller sea lions use 10 rookeries and approximately 77 haul-out sites within the range of the western DPS in Russia (Figure 3.11). Of these 77 haul-outs, three had been rookeries, but presently no breeding occurs there, 49 are active haul-out sites, 20 have been abandoned (no Steller 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 Steller sea lions at major rookeries and haul-outs in the Kuril Islands alone declined 74% (Merrick *et al.* 1990). By 1990, the total Russian population had declined by approximately 50% to about 13,000 (including pups) (Burkanov and Loughlin 2005). Between the early 1990s and 2004, the Russia and Asian population (including pups) increased slowly to ~ 16,000 overall (Burkanov and Loughlin 2005), and since then (through 2008), is thought to have increased to ~25,000 (V. Burkanov, personal communication).

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 3.3 and 3.4, Figure 3.12). Steller sea lion abundance in the sub-areas that are most similar to the Alaskan western DPS - Commander Islands, eastern Kamchatka and the western Bering Sea (Baker *et al.* 2005) - increased in the 1970s and 1980s, but declined significantly in the 1990s and have remained at low levels through 2008 (Burkanov and Loughlin 2005, V. Burkanov, personal communication). In the western Bering Sea, there are no rookeries; numbers of non-pups have plunged over 98% since 1982 and now total only about 100 individuals (Table 3.3). By contrast, Steller sea lion numbers in the ‘Asian’ stock as defined by Baker *et al.* (2005) – Sea of Okhotsk, Kuril Islands and Sakhalin Island - increased considerably between 1990 and 2008, particularly on Tuleny Island. The overall increase in Steller sea lion numbers in the Russian western DPS between 1990 and 2008 is due entirely to increases in the “Asian” portion as defined by Baker *et al.* (2005). The rate of increase for this sub-region is +4.3% per year and significantly different from zero (Devin Johnson, 2010, pers. Comm., Alaska Fisheries Science Center, Seattle, WA).

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 Steller sea lions will continue to be supported by U.S. research institutions for the foreseeable future.

3.1.3.3 Eastern DPS Status and Trends

Within the eastern DPS, 13 rookeries and about 85 major haulout sites currently exist from Cape Fairweather, Alaska, to Año Nuevo Island, California. Populations associated with 12 of these rookeries have either increased or stabilized at relatively high levels in recent years. 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 (Figure 3.13). 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 (Figure 3.14).

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.

In 1979, 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 Rock and Biali Rocks, appear to have developed into rookeries. 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, Steller sea lions used 30 major haulouts, plus several other sites for brief periods each year, probably in conjunction with seasonal prey concentrations.

In 2009, Steller sea lion pup production in Southeast Alaska totaled 7,462 pups, with 7,443 counted at the five major rookeries where 5,510 were counted in 2005. The increase of 1,933 in rookery pup production since 2005 equates to approximately 97 more pups per year at each of the southeast Alaska rookeries. Pup production in this region increased at the rates of +5.0% per year since 1996 and +3.6% per year since the late 1970s; between 2001/02 and 2009, rookery pup production increased 50% in southeast Alaska, equivalent to an increase of approximately 62 pups per rookery per year.

Overall, the eastern DPS increased at over 3% per year between 1982 and 2009, more than doubling in Southeast Alaska, British Columbia, and Oregon (Table 3.5 and Figure 3.15; Pitcher *et al.* 2007). The robustness of the observed positive trend for the eastern DPS over the past 25-30 years was confirmed by Bayesian trend analyses conducted by Goodman (see Appendix 3 in NMFS 2008a). Surveys of pups and non-pups conducted in British Columbia in 2006 by Canada's Department of Fisheries and Oceans (Olesiuk *et al*. 2008), and in southeast Alaska, California and Oregon in 2009 by NMFS, indicate that the overall eastern DPS trend described by Pitcher *et al.* (2007) continued through 2009. As indicated above, this trend is particularly strong in the northern portion of the eastern DPS in southeast Alaska and British Columbia. More detail regarding status and trends of Steller sea lion populations within the eastern DPS is available in NMFS (2008), Olesiuk *et al.* (2008), Fritz *et al.* (2008), and DeMaster (2009).

3.1.4 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, natality, fecundity; 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).

3.1.4.1 Survival

Causes of pup mortality include drowning, starvation caused by separation from the mother, disease, parasitism, predation, crushing by larger animals, biting by other Steller 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 in the central GOA 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 Steller 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 Steller 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; Pendleton *et al.* 2006).

Holmes and York (2003) and Holmes *et al.* (2007) modeled changes in Steller sea lion vital rates that were consistent with observed trends in pup and non-pup counts, and in the juvenile fraction of the population in the central GOA. They concluded that juvenile survivorship declined sharply between the late 1970s and late 1980s. Pendleton *et al.* (2006) came to a similar conclusion from their analysis of observations of animals branded as pups on Marmot Island in 1987-88, data which are completely different from those analyzed by Holmes *et al.* (2007). Juvenile survival increased through the 1990s such that by 1998-2004, it was almost back to late 1970s levels (Holmes *et al.* 2007; Figure 3.16), a conclusion that is also consistent with preliminary analyses from the current (since 2000) western DPS pup branding program (Figure 3.16 NMFS-AFSC unpublished data). Current (in the 2000s) survival rates of juveniles within the range of the western DPS as determined by brand-resight analyses or from modeling are similar to those estimated for pups branded on Forrester Island (eastern DPS) in the mid-1990s (Pendleton *et al.* 2006), when the population in southeast Alaska was increasing at approximately 3% per year.

Adult survival may be greater now (in the 2000s) than it was in the mid-1970s (Holmes *et al.* 2007). Pendleton *et al.* (2006) also found that adult survival of pups branded at Marmot in 1987-88 (or during the 1990s and early 2000s) was greater than in the 1970s (York 1994) as evidenced by the differences in slopes of the cumulative survival curves in Figure 3.16. Fay and Punt (2006) also concluded that adult and juvenile survival increased through the 1990s and 2000s, but that reproductive rates had likely declined.

To date, all specific information about changes in survival of adult and juvenile Steller sea lions in Alaska applies only to the populations breeding from southeast Alaska (eastern DPS) through the eastern Aleutian Islands (170°W) in the western DPS. There is no information specific to the declining Steller sea lion populations breeding in the central and western Aleutians regarding temporal changes in survival.

3.1.4.2 Reproduction and Growth

For the purposes of this Biological Opinion, an understanding of the reproductive cycle of Steller sea lions is essential to an analysis of various sources of impacts on the population, particularly given indications of reduced natality in portions of the species' range. For mature females, the reproductive cycle includes mating, gestation, parturition, and nursing or post-natal care. The reproductive success of an adult female is determined by a number of factors within a cycle and over time through multiple cycles. The ability for an adult female to successfully complete this cycle is dependent on multiple physiological and environmental factors.

Steller sea lions are polygynous; a single male may mate with multiple females. Males establish territories in May in anticipation of female arrival (Pitcher and Calkins 1981). Mating occurs on land (or in the surf or intertidal zones), thus males are able to defend territories and thereby exert at least partial control over access to adult females and mating privileges. Thorsteinson and Lensink (1962) found that 90% of males holding territories on rookeries in the western Gulf of Alaska were between 9 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.

The pupping and mating season is relatively short and synchronous, probably due to the strong seasonality of the environment and the need to balance aggregation for reproductive purposes with dispersion to take advantage of distant food resources (Bartholomew 1970). In late May and early June, adult females arrive at the rookeries, where pregnant females give birth to a single pup (twinning is rare). Viable births begin in late May and continue through early July. The sex ratio of pups at birth is approximately 1:1, though biased toward slightly greater production of males (e.g., Pike and Maxwell 1958, Lowry *et al.* 1982, NMFS 1992b).

Pupping occurs throughout the Steller sea lion range between the Aleutian Islands and California, with a median pupping date of 12-13 June (Bigg 1985, Merrick 1987). Pupping tends to be synchronous within individual rookeries with 90% of pups born within a 25-day period (Pitcher *et al.* 2001). Pitcher *et al.* (2001) found the earliest mean pupping dates at Forrester Island (southeast Alaska) and the latest mean pupping dates at Ano Nuevo Island (California). Mean date of birth became progressively later both north and south of Forrester Island. They hypothesized that timing of births at rookeries is determined through selection of periods when weather conditions are generally favorable for pup survival and when adequate prey are predictably available near rookeries for lactating females. The most likely explanation for temporal variability at individual rookeries is variable nutritional status of reproductive females (Pitcher *et al.* 2001).

Detailed information on Steller sea lion reproduction has historically been obtained from examinations of reproductive tracts of dead animals. These studies have shown that female Steller sea lions reach sexual maturity at 3-6 years of age and may produce young into their early 20s (Mathisen *et al.* 1962, 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). Adult females normally ovulate once each year, and most breed annually (Pitcher and Calkins 1981) 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, Pitcher *et al.* 1998). Females typically breed about 11 days after giving birth, undergoing delayed implantation wherein the blastocyst implants after about 3.5 months. Due to this delay, the metabolic demands of a developing fetus are not imposed on the female until well into fall and winter (Winship and Trites 2003).

In samples collected in the Gulf of Alaska in the mid-1980s, Calkins and Goodwin (1988) found that 97% of females aged 6 years and older had ovulated. Ninety-two percent of females 7-20 years old were pregnant when they were collected in October during early implantation. The pregnancy rate of sexually mature females collected during April-May (late gestation) was only 60%, indicating that a considerable amount of intrauterine mortality and/or premature births occurred after implantation. Estimates of near-term pregnancy rates of all adult females were 67% from a collection of females taken from 1975-1978 and 55% from a similar collection during the mid-1980s (Pitcher *et al.* 1998), the difference was not statistically significant between periods (P = 0.34), yet the statistical power to detect the difference was less than 0.50. However, the difference in pregnancy rates of the lactating females between the 1970s (63%) and 1980s (30%) was significant (P = 0.059).

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, Fay and Punt 2006; Holmes *et al.* 2007), but may have increased in the late 1990s and 2000s in some areas (Winship and Trites 2006). Holmes *et al.* (2007) estimated that natality rates in the central GOA were 36% lower in the period 1998-2004 than in the mid-1970s (Figure 3.16). In addition, they present time series of pup and non-pup counts in the western GOA and eastern Aleutians that suggest that reductions in natality between the mid-1970s and 2005 may be more widespread within the western DPS than the detailed modeling of the central GOA population indicated.

The results from the published modeling studies related to continued declines in natality were questioned in Maniscalco *et al.* 2010. Maniscalco *et al.* estimated rates of reproduction as part of a longitudinal study of predominantly naturally marked females at the Chiswell Islands (in the eastern Gulf of Alaska); their estimated rates (0.69) were higher than those reported by Holmes *et al.* (2007) for the central Gulf population. The authors reported an estimated natality rate of 0.69, which they noted was comparable to the natality rate of Steller sea lions prior to the decline in the 1980s. However, this is not strictly true as Maniscalco *et al.* (2010) does not include mortality prior to the first census (i.e., after birth). Horning and Mellish (2010) used a birth pulse Leslie population matrix model parameterized with SSL pup counts and annual survival rates, and with mortality estimates derived from their LHX tag data, to estimate natality for the eastern GOA; their estimate under stable or increasing population levels for the eastern GOA region was 0.63 (0.3-1.0).

Declines in female Steller sea lion reproductive performance may have been, and may still be, linked to body condition or growth. 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 stress (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 some females from giving birth each year (Pitcher *et al.* 1998). During the 1970s and 1980s, 97% of sexually mature females in the western DPS were pregnant in early gestation. However, the percentage of those females that carried their pregnancy to late gestation fell to 67% during the 1970s and to 55% in the 1980s (Pitcher *et al.* 1998). 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).

Merrick *et al.* (1995) compared pup sizes at different sites where Steller sea lion populations were either decreasing or increasing, to determine if decreased pup size or growth was correlated with decreasing population trend. Their results were not consistent with this hypothesis; rather, they found that pups about two to four weeks of age weighed more at western, declining rookeries in the Aleutian Islands and GOA than at eastern, stable or increasing rookeries in southeast Alaska or Oregon. While western DPS 2-4 week-old pups weighed more than those in the eastern DPS, they were not disproportionately heavy for their length (Fadely and Loughlin 2001). These size differences may arise through different growth rates, as no significant differences have been found among neonatal mass among rookeries (Brandon and Davis 1999, Adams 2000). Brandon and Davis (1999) and Adams (2000) found that pups at rookeries in areas of decline grew faster than pups from southeast Alaska. As there were no differences in milk or energy intake among pups at these rookeries, differences in growth rates may be attributable to differences in pup activity (Adams, 2000), time spent fasting between suckling bouts, or other physiological costs (Brandon *et al.* 2005).

The observed differences above indicate that at least this phase of reproduction may not be affected by whatever factors are limiting natality; that is, if females are able to complete their pregnancy and give birth, then the size of those pups does not appear to be compromised (Davis *et al.* 2006). Possible alternative explanations for the observed size differences are that pups were measured at different ages (i.e., pups in the GOA and Aleutian Islands may have been born earlier and therefore were older when weighed), or that over time, harsher environmental conditions in the Aleutian Islands or the GOA have selected for larger pup size.

Pup condition, measured as the ratio of observed body mass to that expected based on length, seems to be a reasonable index of condition related to survival (Trites and Jonker 2000). For the pups aged between 2 and 4 weeks, there was no general relationship between pup condition and pup numbers or magnitude of decline at rookeries, though the poorest average pup conditions during the late 1990s were associated with areas of greatest decline (Fadely and Loughlin 2001). There also was evidence that pup condition was poorest during weak depressions of the Aleutian Low, and better when the Aleutian Low was stronger.

After birth, mothers nurse their pups for 3-12 days before starting a series of trips to sea, which range in duration from 7-62 hours depending on geographic location (Higgins *et al.* 1988, Hood and Ono 1997, Brandon 2000). Pup gender does not appear to influence maternal attendance patterns (the cycle of time at sea and time on shore), but mothers increase their time at sea as pups get older (Higgins *et al.* 1988). Pups remain on the rookery for the first few weeks of life while females forage at sea (Gentry 1970, Higgins *et al.* 1988, Hood and Ono 1997, Trites and Porter 2002) and enter the water 2 to 4 weeks after birth to play around the periphery of the rookery (Sandegren 1970). Pups presumably disperse from the rookery with their mother 2 to 3 months after birth (Calkins and Pitcher 1982, Merrick *et al.* 1988, Raum-Suryan *et al.* 2004).

Female Steller sea lions and their pups adopt a strategy of central place and multiple central place foraging to deal with the temporal and spatial distribution of prey resources (Raum-Suryan *et al.* 2004). As pups get older, it is believed they make independent trips away from haulout sites while their mothers are at sea (Trites and Porter 2002). The length of the nursing period may be an important indicator of the female's condition and ability to support her pup, and the pup's condition at weaning (and hence, the likelihood that the pup will survive the post weaning period).

Timing of weaning is not well understood, but many are weaned during the first year and some animals may nurse for up to 3 years. (Pitcher and Calkins 1981, Porter 1997, Loughlin 1998, Trites and Porter 2002, Trites *et al.* 2006). Studies based on physiological development (Richmond *et al.* 2005, 2006), changes in fatty acid profiles of pup blubber (Beck *et al.* 2007), and changes in movements and dive characteristics (Loughlin *et al.* 2003, Fadely *et al.* 2005, Rehberg 2005) indicate that weaning typically occurs after 9 to 12 months of age. According to the growth data of Calkins and Pitcher (1982), Steller sea lions would be expected to wean at an age of 11 months; assuming a median pupping date of mid-June, is an age reached in mid-May. A weaning age of 11 months was also used in analyses of comparative mammalian weaning by Lee *et al.* (1991).

Porter (1997) distinguished metabolic weaning (i.e., the end of nutritional dependence of the pup or juvenile on the mother) from behavioral weaning (i.e., the point at which the pup or juvenile no longer maintains a behavioral attachment to the mother). He also suggested that metabolic weaning is more likely a gradual process occurring over time and more likely to occur in March-April, preceding the next reproductive season. Raum-Suryan *et al.* (2004) found that changes in round trip distance and duration occurred from April to June for young-of-the-year and older individuals, possibly indicating that annual timing of weaning may be less variable than age of weaning.

Recent studies of weaning, however, suggest that pups do not always wean before the next breeding season. York *et al.* 2008 used stable isotope ratios from female sea lion teeth to examine changes in the age of weaning. They found that except for the period of the regime shift of 1975-1976, the age of weaning increased over time. Overall, approximately 60% of the female pups weaned in their first year, 30% in their second year and 8% in their third year. Similarly, Trites *et al.* 2005 used observations of nursing pups and yearlings to argue that in southeast Alaska about one half of the female pups weaned in their second year, and most male pups weaned in their second year. In a study using satellite telemetry to compare foraging bout behavior between the eastern and western stock, Call *et al.* (2007) used significant increases in time spent at sea to suggest decreases in maternal dependence. They found that this apparent change occurred approximately 10 months later in individuals from Prince William Sound (western DPS) and southeast Alaska (eastern DPS) than in the Aleutian Islands and central Gulf of Alaska.

It is clear that the transition to nutritional independence may occur over a period of months as the pup begins to develop essential foraging skills, and depends less and less on the adult female. The length of the nursing period may also vary as a function of the condition of the adult female. The nature and timing of weaning is important because it determines the resources available to the pup during the more demanding winter season and, conversely, the demands placed on the mother during the same period. A bioenergetic model suggested that a 10 year old female nursing a pup in the spring would have to consume twice as much energy as a same age female without a pup (Winship 2000). The maintenance of the mother-offspring bond may also limit their distribution or the area used for foraging.

Relatively little is known about the life history of Steller sea lions during the juvenile years between weaning and maturity. Female growth is asymptotic, and reaches 87% of the asymptote during their third year (Winship *et al.* 2001), a size typically associated with puberty in female pinnipeds (Laws, 1956). The available literature indicates an overall female reproductive (birth) rate on the order of 55% - 70% or greater (Pike and Maxwell 1958, Gentry 1970, Pitcher and Calkins 1981, Maniscalco *et al.* 2010). York (1994) derived age-specific fecundity rates based on data from Calkins and Pitcher (1982). These rates illustrate a number of important points and assumptions. First, the probability of pupping is rare (about 10%) for animals 4 years of age or younger. Second, maturation of 100% of a cohort of females occurs over a prolonged period which may be as long as 4 years (starting at age 3 or 4). Third, the reported constancy of fecundity extending from age 6 to 30 indicates that either senescence has no effect on fecundity, or our information on fecundity rates is not sufficiently detailed to allow confident estimation of age-specific rates for animals older than age six. Given the small size of the sample taken, the latter is a more likely explanation for such an assumption. Holmes *et al.* (2007) reanalyzed the Calkins and Pitcher (1982) pregnancy data and included reproductive senescence in their life table of the 1970s central Gulf of Alaska Steller sea lion population.

Male growth is also asymptotic, but constant until about year 6 and thus males grow at a greater rate for a longer period than do females (Winship *et al.* 2001). Males also reach sexual maturity at about 3 - 8 years old, but do not have the physical size or skill to obtain and keep a breeding territory until they are nine years of age or older (Pitcher and Calkins 1981). A sample of 185 territorial males from Marmot, Atkins, Ugamak, Jude, and Chowiet Islands in 1959 included animals 6 - 17 years of age, with 90% from 9 - 13 years old (Thorsteinson and Lensink 1962). Males may return to the same territory for up to 7 years, but most return for no more than 3 years (Gisiner 1985). During the breeding season, males may not eat for 1 to 2 months. The rigors of fighting to obtain and hold a territory and the physiological stress of the mating season reduces their life expectancy. Males rarely live beyond their mid-teens, while females may live as long as 30 years.

In summary, an examination of reproduction and growth yields considerable evidence which suggests that while declines in the western Steller sea lion population in the 1980s were associated with decreased juvenile survival, the slower declines of the 1990s and the lack of a robust rate of increase in abundance in the 2000s in this population are associated with decreased reproductive success at least in some areas:

- Birth rates for Steller sea lions in the central Gulf of Alaska in the period 1998-2004 declined 36% from those estimated in the mid-1970s (Holmes *et al.* 2007, York 1994, Holmes and York 2003).
- Young females collected in the 1970s were larger than females of the same age collected in the 1980s (Calkins *et al.* 1998). Given that decreased size translates into delayed maturity and decreased body condition for reproduction, the lifetime reproductive success of females collected in the 1980s was inferred to be lower than those collected in the 1970s.
- Female pregnancy rates appeared to decline between the 1970s (67%) and the 1980s (55%), consistent with the hypothesis that reproductive effort in the 1980s was compromised Pitcher *et al.* (1998). This decline suggests a high rate of fetal mortality and/or an indication of stress (possibly nutritional) experienced by individual females.
- Late season pregnancy rates in lactating females declined between the 1970s (63%) and the 1980s (30%), indicating a decreased ability in females to support a fetus and successfully complete consecutive pregnancies (Pitcher *et al.* 1998).

It is important to note that inferences regarding changes in natality for Steller sea lions from the western or central Aleutian Islands are constrained by a lack of life history data for these regions. Maniscalco *et al.* (2010) was initially published as an Alaska SeaLife Center Technical Report (Maniscalco, Hennen, and Parker 2009) (MHP) which was reviewed and critiqued by several scientists at the Alaska Fisheries Science Center. In one critique, Holmes (2009) notes that MHP's analysis suffers from pseudo-replication (the unit of replication should be female, not year) and ignores those females not sighted at Chiswell after their last visit. She claims that high fecundity females are thus over-represented, and reanalyzes MHP's data to conclude that the probability of pupping per female at Chiswell is 0.52. In another critique, Johnson (2009) developed a model to estimate natality rate at Chiswell using MHP data assuming heterogeneity in natality and survival, with survival rate independent of natality rate; he noted some females are 'quality' producers and some are 'poor'. Johnson's model results indicate Chiswell natality is on average about 0.54 (0.46-0.63) instead of MHP's 0.54-0.64. The AFSC (2009) compiled additional comments on MHP and noted that MHP's analysis suffers from some bias (e.g., assuming Chiswell Is. represents the whole GOA) and unclear modeling (AFSC references an improved model by Johnson [2009] that accounts for resight probability and apparent survival). AFSC also notes MHP did not account for stillbirth and late term abortions in their modeling. Improving on MHP, AFSC (2009) determined that Chiswell Is. natality was more likely 10% lower than MHP and with a wider confidence interval. Based on these comments, Maniscalco *et al.* (2010) conducted additional modeling and evaluated the AFSC critiques, and subsequently published their revised Chiswell Is. SSL natality study. They asserted that some females may be more fit than others, and those that gave birth in a given year were more likely to survive and give birth in the subsequent year (0.851) than those that did not (0.777). They estimated natality at 0.69 for the Prince William Sound/Kenai Fjords area of the eastern Gulf of Alaska, which they believe is indicative of a stable or increasing population (Maniscalco *et al.* 2010).

3.1.4.3 Demographic Modeling and Population Variability

Population sizes change as a function of births, deaths, immigration, and emigration. Changes in population size over time are referred to as population growth rates. Extensive studies have been conducted to evaluate the relative contribution of births, deaths, immigration and emigration to observed changes in western Steller sea lion population growth rates.

The observed decline in western DPS population size is not explained by emigration from the western DPS to the eastern DPS. During the non-reproductive season, some Steller sea lions may move between the western and eastern populations (Pitcher and Calkins 1981); however, over the past two decades, the amount of growth observed in the eastern population is equivalent to only a small fraction of the losses in the western population. Genetic studies provide evidence of discrete population structure between the eastern and western DPSs (see section 3.1.1.2). Recently, adult females from the western DPS have given birth at two newly established rookeries in the range of the eastern DPS (see section 3.1.1.2) which indicates a possible emerging shift in population structure. This is likely a recent phenomenon given the consistent confirmation of discrete mtDNA profiles in the eastern and western portions of the Steller sea lion's range until 2007 (see section 3.1.1.2). Thus, the historic declines observed in the western DPS are primarily attributable to changes in birth and death rates. As mentioned above, modeling (York 1994, Holmes and York 2003) and mark-recapture experiments (Chumbley *et al.* 1997) indicate that the most likely problem leading to the decline in the 1980s was decreased juvenile survival, but lower reproductive success was almost certainly a factor contributing to the slower declines of the 1990s and the lack of a robust recovery observed since 2000 (Holmes and York 2003, Pendleton *et al.* 2006, Holmes *et al.* 2007). Durban *et al.* (2010) recently note the large number of killer whales in the GOA and central and eastern Aleutian Islands and speculate that they may have been an important factor in the decline in parts of the AI region. And Kruse and Huntington (2009) note that intentional shooting of SSLs throughout the Bering Sea, AI, and GOA regions in the 1970s and 1980s also may have contributed to the SSL decline but it does not appear to have been a leading cause of unexplained SSL mortality during 1974-1990. Survivorship of adults and juveniles has increased since the early 1980s (Holmes *et al.* 2007).

York (1994) concluded from her life table analysis that the population decline observed in the 1980s at Marmot likely was primarily owing to a large drop in juvenile survivorship compared to the 1970s, a conclusion also reached by Pascual and Adkison (1994). Holmes and York (2003) and Holmes *et al.* (2007) extended these analyses of central Gulf of Alaska sea lions through 2004 and added an index of juvenile recruitment to the model. Their results, along with those of Fay (2004) and Fay and Punt (2006), 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 an erosion in fecundity (birth rates plus pup mortality through 1 month) that began in the late 1970s and early 1980s (Holmes and York 2003) and continued through 2004 (Holmes *et al.* 2007).

Fay (2004) and Winship and Trites (2006) broadened the geographic scope of these analyses by estimating time series of vital rates for metapopulations[9], or 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 fecundity, 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 fecundity in the central Gulf of Alaska (similar to Holmes and York 2003, Fay 2004 Holmes *et al.* 2007), but not elsewhere in the range of the western DPS.

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. Hennen (2006) also stated that the timing of the regime shifts in the area could be responsible for the patterns seen in his analysis.

Two additional points are worth mentioning under this subheading. First, Springer *et al.* (2003) hypothesized that the primary factor driving the steep decline of the western Steller sea lions in the 1980s was killer whale predation (see section 4.2.3). While the publication of this hypothesis has been challenged by several authors (e.g., DeMaster *et al.* 2006, Trites *et al.* 2006), there is little disagreement that killer whale predation is an important factor in understanding the population dynamics of the western population. Further, the NRC (2003) reported that the likely cause of the steep decline of the western stock was from more than one factor (a conclusion reached by many authors), where predation by killer whales was one of those factors. They further noted that allocating primacy among these factors associated with observed declines in the 1980s and 1990s was not possible. Finally, they concluded that the recovery of the western population was most likely to be significantly impacted by killer whale predation, and was much less likely to be impacted by competition from commercial fisheries. Horning and Mellish (2010) observed killer whale predation on tagged juvenile SSLs from the Kenai Fjords/PWS region of the eastern GOA. Based on data collected post-mortem from LHX tagged western DPS Steller sea lions from the PWS/KF region, 8 juvenile SSLs were confirmed dead, at least 7 were killed by predators. From their data, Horning and Mellish (2010) speculate that over half of juvenile Steller sea lions are consumed by predators before age 4 (in this region of the eastern GOA).

---

[9] A metapopulation is a group of several local populations linked by immigration and emigration (Gotelli, 2000).

Second, there is a general agreement among Steller sea lion scientists that the impact of unregulated and unreported anthropogenic mortality in the 1980s (i.e., the period of steep decline in abundance) has been underestimated (Atkinson *et al.* 2008; Alverson 1992). Unfortunately, it is also recognized that the magnitude of this bias cannot be reliably estimated with available data. This is particularly relevant to population viability modeling, where the steep decline in abundance observed in the western population of Steller sea lions in the 1980s has to be incorporated into the modeling exercise as a potentially recurrent disaster (Goodman 2008). Absent this observation, it is likely that the target population for recovery of the western population of Steller sea lion would be considerably less than 100,000 animals.

3.1.4.4 Population Viability and Extinction Risk

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 (York *et al.* 1996, Gerber and VanBlaricom 2001, Winship and Trites 2006, NMFS 2008a. 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 Steller sea lion populations in the future. A summary of existing publications on population viability modeling is reported in NMFS (2008a).

There is some degree of consistency between the predictions of all four sets of PVA models (York *et al.* 1996, Gerber and VanBlaricom 2001, Winship and Trites 2006, NMFS 2008a) 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, Steller 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 PVAs 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). Although 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, the Recovery Plan was charged with addressing recovery of the entire species. 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.

Finally, Boyd (2010) reported on simple empirical indicators to evaluate the risk of extinction to a population and to document how this risk may change over time as new data become available and management measures are implemented. Boyd further noted that the current number of SSLs in Alaska and British Columbia is between 130,000 and 150,000 animals and that this number has been stable since the 1990s in spite of regional differences in population trends. He argues that an internal re-distribution of juveniles could explain the observed regional patterns in abundance and that the risk of extinction of SSLs in this area has declined to a level where the population should no longer be considered endangered. Finally, he notes that the declining trend in risk of extinction is consistent with the interpretation that management actions "taken since 1990 have probably been effective." While no formal review or rebuttal of these findings have been published, it is important to note that under US law and regulations applicable to the ESA, the Agency must evaluate the potential impact of the proposed action on the population structure recognized in the ESA listing process (i.e., a western DPS of SSL distinct from an eastern DPS). Therefore, evaluations of the risk of extinction that confound the dynamics of the two recognized DPSs are not applicable to this opinion.

3.1.5 Terrestrial Habitat Use

Steller sea lions use a variety of terrestrial habitats. Haulouts and rookeries tend to be preferentially located on exposed rocky shoreline and wave-cut platforms (Ban and Trites 2007, 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 and Trites 2007). When not on land, Steller sea lions are seen near shore and out to the edge of the continental shelf and beyond.

Female Steller 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 Steller 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. Steller 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, Steller sea lions display strong site fidelity to specific locations from year to year, although brand-resight studies show some animals both disperse from rookeries or immigrate from other rookeries (e.g., Burkanov 2010). 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).

Steller sea lion seasonal distribution, haulout use and haulout-specific foraging patterns are influenced by seasonal concentrations of prey. From 2001-2004, researchers counted sea lions of all age classes monthly at 28 terrestrial sites in Southeast Alaska using aerial surveys (Womble *et al.* 2009). Four distinct seasonal distributional patterns were discovered associated with the seasonal availability of prey near the terrestrial sites. In December, 55% of the sea lions studied were at terrestrial sites near over-wintering herring aggregations, whereas in May, a similar percentage (56%) were at sites near forage fish spawning aggregations. In July, 78% of the Steller sea lions in the study area were found at terrestrial sites near summer migratory corridors of salmon, while in September, 44% of sea lions were at sites near autumn migratory corridors of salmon. Womble *et al.* concluded that a reasonable Steller sea lion foraging strategy in southeast Alaska is to prey upon herring aggregations in winter, spawning aggregations of forage fish in spring, salmon in summer and autumn, and pollock and Pacific hake throughout the year.

3.1.6 Marine Habitat Use

Knowledge of movements and individual foraging patterns are essential to understanding how such patterns may be influenced by the availability of prey resources (Raum-Suryan *et al.* 2004). Steller sea lions consume a variety of demersal, semi-demersal, and pelagic prey, indicating a potentially broad spectrum of foraging styles probably based primarily on availability.

Inferences on Steller sea lion at-sea habitat use and movements are based on data collected from animals monitored with telemetry devices and from an opportunistic sighting database referred to as the Platforms of Opportunity (POP). Various telemetry studies have provided detailed information on movements of both adult females and juveniles. Telemetry studies have indicated that trip duration and distance vary seasonally, but rarely exceed 20 hours (h) and 20 km, respectively (Merrick and Loughlin 1997, Loughlin *et al.* 2003, Raum-Suryan *et al.* 2004, Rehberg 2005, Fadely *et al.* 2005, AFSC 2010b). Most locations at sea appear to be associated with onshelf waters <100 m deep (Fadely *et al.* 2005).

Gregr and Trites (2008) examined Steller sea lion foraging behavior, haulouts and rookery locations, bathymetry, and seasonal ocean climate to develop hypotheses relating SSL life-history traits and physical variables to its marine habitat. They developed a series of habitat models predicting the probability of sea lions occurring within a 3 × 3 km^2 grid in the Gulf of Alaska and the Bering Sea. They focused on adult female foraging zones around Steller sea lion sites to examine the efficacy of the currently-designated critical habitat for the western DPS. They used telemetry, oceanographic, and platform of opportunity data sets to develop and validate a model that would predict the most important foraging areas for adult females based on known foraging distances from sites during winter and summer. They determined that Steller sea lions rarely forage beyond 20 nm, and they noted that juvenile and adult female Steller sea lions particularly forage relatively close to rookeries and haulouts.

Overall, the 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.

A recent analysis of opportunistic sightings of Steller sea lions (the POP) yielded results consistent with the limited telemetry information available for the western and central Aleutian Islands. Boor (2010) derived spatially-explicit quantitative estimates of Steller sea lion at-sea habitat use by standardizing effort for the opportunistic sightings. The POP data span a 43 year period, 1958-2000, and contain 13,037 sightings representing 109,323 individual Steller sea lions. Boor (2010) also analyzed seasonal patterns of at-sea sightings. Steller sea lion encounter rates were high along the continental shelf break throughout the Gulf of Alaska and in the Bering Sea. High encounter rates also occurred in the Bering Sea’s Aleutian Basin between Bower’s Ridge and the Bering Sea continental shelf and in the offshore region to the south and east of Attu and Agattu Islands (Boor 2010). The availability of prey at these sites, far from rookeries and haulouts, may be crucial in allowing sea lions to take advantage of distant food sources. Offshore encounter rates were high in the breeding season (May through August) south and east of Attu and Agattu

Islands; however, there were no ship-days recorded in this region in the non-breeding season, confounding interpretation as to whether Steller sea-lions use this habitat in the non-breeding season. It is important to note that the offshore sightings south and east of Attu and Agattu Islands in the POP database were all observed in the 1980s when the abundance of Steller sea lions was higher than present. Though, very few ship-days were recorded in the region before and after the 1980s. It is unknown as to whether or not habitat use is similar today, however NMFS assumes at-sea use inferred from telemetry and POP information summarized in AFSC (2010b) and Boor (2010) reflect the at-sea use of Steller sea lions in the respective regions as they are the best data available.

To address the question of whether competition exists between the Steller sea lion and BSAI or GOA groundfish fisheries, it is necessary to evaluate region-specific foraging success. For a foraging Steller sea lion, the net gain in energy and nutrients is determined, in part, by the availability of prey or prey patches it encounters within its foraging distribution. Competition occurs if the fisheries reduce the availability of prey to the extent that Steller sea lion condition, growth, reproduction, or survival is diminished. Detail on Steller sea lion foraging behavior is provided below to contribute to an understanding of foraging patterns and potential interactions with available prey.

3.1.6.1 Adult Female Foraging Behavior

Due to logistics associated with capturing and handling, as well as recent litigation regarding the value of research involving capturing and handling, limited information exists on the foraging ecology of adult female Steller sea lions[10]. 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.

Numerous studies from both Alaska and the Kuril Islands, Russia have shown that during the breeding season, adult female Steller sea lions generally spent about half their time at sea on relatively brief foraging trips, where diving is shallow, brief, and frequent (Merrick *et al.* 1990, Merrick *et al.* 1994, Merrick and Loughlin 1997, Loughlin *et al.* 1998, Brandon 2000, Trites and Porter 2002, Milette and Trites 2003). Behavior also appears to change seasonally; satellite-tagged adult female Steller sea lions from the central Gulf of Alaska through the eastern Aleutian Islands were found to spend more time at sea, dive deeper, and have greater home ranges during winter than summer (Merrick and Loughlin 1997). These behaviors may reflect reproductive status or changes in prey availability and distribution resulting from seasonal variability or both (Merrick and Loughlin 1997). Outside of the breeding and pupping season, movements may be less constrained although animals still return to coastal haulouts to rest. In addition, Raum-Suryan *et al.* (2004) reported that females had a significantly greater mean trip duration than males and suggested females either were more selective than males when searching for prey, or they had to spend more time attaining prey resources because they had less diving capabilities than males.

Attendance cycles for lactating females (consisting of one trip to sea and one visit on land) averaged about 3 days in winter and 2 days in summer (Trites and Porter 2002, Milette and Trites 2003, Trites *et al.*

---

[10] In May 2005, NMFS issued six scientific research permits authorizing research on Steller sea lions. However, in July 2005, the Humane Society of the United States (HSUS) challenged the validity of the permits, alleging among other things that by failing to prepare an EIS before issuing the permits, NMFS had violated NEPA. On May 26, 2006, the District Court in Washington D.C. ruled that NMFS should have prepared an EIS under NEPA before issuing the permits in 2005. The judge held that the analysis contained in the EA did not support the finding of no significant harm to the environment. Subsequently, NMFS completed an EIS and research permitting continued in 2007.

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 2 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).

Trites *et al.* 2005 compared observations of females at sites in both the eastern and western DPS and found no difference in foraging bout lengths between the two populations. However, Andrews *et al.* (2002) used stomach temperature telemetry and satellite telemetry to monitor the behavior of four lactating Steller sea lions from the central Aleutian Islands and five lactating sea lions from areas near Forrester Island, southeast Alaska. Time spent at sea, trip duration, trip distance, and mean time from departure to first prey ingestion for females from the Aleutian Islands were shorter than those for females from southeast Alaska. The percent of a trip spent submerged did not differ significantly between the two regions. Additionally, dives performed by sea lions from the central Aleutian Islands were shorter and shallower, but more frequent than those by sea lions from southeast Alaska. Because fish surveys of the two areas were conducted simultaneously with data collection for sea lions, Andrews *et al.* (2001) were able to demonstrate a correlation between prey availability, foraging success, and pup growth.

In Southeast Alaska, adult females with pups have been found to make relatively brief foraging trips, while those with yearlings or without pups were much longer in duration (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).

3.1.6.2 Pup and Juvenile Foraging Behavior

Telemetry has been an important tool for investigating the movements, foraging behavior, habitat selection, and development of juvenile Steller sea lions. The need to understand the behavior of juveniles has focused research effort in recent years and resulted in a relatively large sample data set for animals less than 3 years of age (Loughlin *et al.* 2003, Raum-Suryan *et al.* 2004, Rehberg 2005, Pitcher *et al.* 2005, Fadely *et al.* 2005, Briggs *et al.* 2005, Call *et al.* 2007). 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). Mean dive depth and duration increases with age (Pitcher *et al.* 2005, Fadely *et al.* 2005, Briggs *et al.* 2005) and is predicted to increase in a positive relationship with body mass up to about 10 years of age (Pitcher *et al.* 2005).

Pups appear to be capable of traveling up to 120 km from their natal rookery by the age of 2 months (Raum-Suryan *et al.* 2004), and capable of traveling more than 400 km by 5 months of age (Raum-Suryan *et al.* 2002). Juveniles tend to disperse greater distances than pups (Raum-Suryan *et al.* 2002) and their swimming ability is comparable to that of adults (Loughlin *et al.* 2003). Time spent at-sea by juveniles has been found to increase with age, but Steller sea lions from the central and eastern Aleutian Islands and central Gulf of Alaska spent more time at-sea at earlier ages than did Steller sea lions tracked in Prince William Sound and Southeast Alaska (Call *et al.* 2007). In addition, Briggs *et al.* (2005) found that eastern Gulf of Alaska and Prince William Sound juveniles in the winter made a majority of their dives during the day and switched to night time feeding during the summer. Merrick and Loughlin (1997) found that most of the pups tracked during winter made relatively short trips to sea (mean distance 30 km), but one moved 320 km from the eastern Aleutians to the Pribilof Islands. 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.

Additional studies on immature Steller sea lions indicate three types of movements: 1) transits between land sites with a mean distance of 66.6 km; 2) long-range trips (more than 15 km and greater than 20 hours); and 3) short-range trips (less than 15 km and less than 20 hours) (Loughlin *et al.* 2003). Long-range trips started around 9 months of age and likely occurred most frequently around the time of weaning while short-range trips happened almost daily. 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. Likewise, Raum-Suryan *et al.* (2004) 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.

Frid *et al.* (2008) used satellite-linked time-at-depth recorders to investigate the behavior of juvenile Steller sea lions from Prince William Sound, and concluded that during the winter in Prince William Sound, juvenile Steller sea lions may (a) under-utilize walleye pollock, a predictable resource in deep strata, due to predation risk from Pacific sleeper sharks, and (b) under-utilize inshore aggregations of Pacific herring due to risk from other predators such as killer whales. However, as noted in section 3.1.12, there is little evidence to support the hypothesis that sleeper sharks are significant predators on Steller sea lions.

3.1.6.3 Foraging Behavior: Unpublished Dive-Filtered Telemetry Data

To investigate foraging behavior, an analysis of juvenile Steller sea lion dive locations was completed using satellite telemetry data obtained from 2000 to 2005 following the methods used by NMFS (AFSC 2010b). The previous analysis was based on summarized telemetry data collected from 63 Steller sea lions by scientists from the Alaska Fisheries Science Center (AFSC) during 2000 to 2003. The current analysis was updated with data from satellite tag deployments by ADF&G and AFSC since the time of the previous analysis, with updates to text, tables, and graphic representations (NMFS 2010b). Results from the current analysis were used to update tables II-6, II-7, II-9 (Table 3.10 and 3.11).

AFSC staff captured and equipped 23 juvenile Steller sea lions with satellite linked time depth recorders (SLTDRs) between 2003 and 2006. No additional Steller sea lions have been captured and monitored using satellite telemetry since that time due to permitting and logistical constraints. Additionally, 32 Steller sea lions captured by ADF&G from 2000-2002 were also included. A total of 116 animals (63 previous, 53 new) ranging in age from 3-26 months old at time of capture were used in this updated analysis (Table 3.12).

The previous analysis in NMFS (2003) used 10,006 dive associated locations from the 63 animals. In this updated analysis all locations that were transmitted from land were removed (NMFS 2006b). This reduced the 10,006 locations from the previous NMFS (2003) dataset to 8,141 at-sea locations. Also, the previous analysis used locations that were wet at the time of transmission, determined through land/sea status data only; this analysis uses both land/sea and timeline data to determine if a location was wet or dry at the time of transmission.

A total of 65,150 locations from all 116 animals were extracted from the database for processing, and 14,441 (22.17%) were used in the new analysis presented in the new tables (Table 3.13) and figures (Figures 3.17- 3.21). The remaining 50,709 locations were removed for a variety of reasons; namely, they either did not fit the dive criteria, were on land locations, were determined to be dry at the time of transmission, were calculated prior to deployment, were invalid due to duplicity, time of transmission, poor quality, or other errors (Table 3.13).

Compared to the NMFS (2003) analysis, results suggest a slightly decreased proportion of dive-associated locations within the 0-10 nm zone, and increased use of habitats >20 nm from shore or nearest listed haulouts or rookeries. In addition, sample sizes for winter locations are doubled, Prince William Sound was added as a new area (Figure 3.17), and 17 new deployments were made in the central Aleutian

Islands (Figure 3.21). Distributions of proportions of dive-associated locations were similar whether based on distance from shore or distance from the nearest listed haulout or rookery, but it is important to note that these distance measures are not directly comparable. That is, though a location may have been >20 nm from a nearest listed haulout or rookery, that location could have been anywhere between adjacent to shore to >20 nm from the nearest point of land (Figure 3.17). Figures 3.17-3.21 are all based on distances of dive-associated locations from the nearest listed haulout or rookery. Most locations >20 nm from a listed haulout or rookery fell outside critical habitat.

Juvenile Steller sea lions >10 months old had a greater proportion of dive-associated locations in zones >10 nm than did 3-10 month olds in both summer and winter. Juveniles >10 months old also showed nearly equal use of the 10-20 nm zone during summer and winter, but a much greater use of habitats >20 nm in summer compared to winter periods.

Regional differences in habitat use were evident (Table 3.11, Figures 3.17-21). In particular, dive-associated locations in the central-western Aleutian Islands area showed a much greater use of habitats >20 nm from the nearest listed haulout or rookery than in other areas, and 22% of the >20 nm zone locations in summer were outside of critical habitat (Table 3.11, Figure 3.21). Most of these locations were in the Bering Sea (Figure 3.21). However, it should be noted that these telemetry data are difficult to interpret for the following reasons: (1) samples sizes are very small. That is, in RCAs 1 and 2, there are only data from 3 individuals in each RCA, while in RCA 8 there are only data from four individuals, (2) no sea lions were captured and instrumented in RCAs 1, 2, 7 or 8. In other words, all of the telemetry data reported in these RCAs or sub-regions had to move into the specific area from elsewhere, (3) zones 1, 2, and 8 were only entered exclusively by males, and (4) there are no telemetry data from adult females. Nonetheless, these data must be considered the best available data and indicate foraging use patterns by juvenile animals outside of critical habitat in RCAs 1, 2, and 8.

3.1.7 Diving Physiology and Development

Fundamental to an evaluation of the effects of commercial fisheries on Steller sea lion populations is an understanding of the physiological adaptations that underlie the Steller sea lion's role in the Bering Sea and Gulf of Alaska ecosystems, specifically diving ability and the development of independent foraging. While foraging, swimming and diving behavior are controlled by a compromise between the necessity to breathe at the surface and to submerge to seek and consume prey. In addition to the abundance and distribution of prey, the time a Steller sea lion spends submerged will depend upon physiological adaptations for maximizing time underwater. This will be a result of the how fast oxygen stores are utilized (i.e., metabolic rate), and how much oxygen is stored in the body, and the conflicting demands of diving and exercise (Castellini 1991, Boyd 1997).

Overall, studies have indicated that dives of juvenile Steller sea lions are short and shallow (Merrick and Loughlin 1997, Loughlin *et al.* 2003, Pitcher *et al.* 2005, Rehberg 2005). Merrick and Loughlin (1997) satellite-tagged 5 young of the year and 15 adult female Steller sea lions from the central Gulf of Alaska through the eastern Aleutian Islands and found that young of the year Steller sea lions exerted less foraging effort and made shorter, shallower dives than adult females. However, older juveniles dove more frequently and deeper and spent more time at sea than younger juveniles. Because young of the year Steller sea lions were diving within their calculated aerobic dive limit (cADL) and did not appear constrained physiologically, the authors suggested that it may require time for young Steller sea lions to develop appropriate diving behaviors and knowledge of prey resources during development.

Young Steller sea lions do not have the same capacity to stay submerged or dive as deep as adults, which affects their ability to acquire prey and thus their choice of foraging strategies. During a dive, approximately 47% of a Steller sea lion's oxygen stores are in blood, with 35% in muscle and the remainder in the lungs (Kooyman 1985). Oxygen storage ability develops as sea lions mature due to increases in blood volume, muscle myoglobin and body mass (Horning and Trillmich 1997). Changes in diving activity likely correspond to an increase in Hct, Hb, and blood oxygen stores, which are similar to adults by approximately 9 months of age (Richmond *et al.* 2005, 2006). The oxygen-carrying capacity of muscle at approximately 17 months of age appears similar to adult females (Richmond *et al.* 2006).

A number of studies provide further evidence for dive depth and duration changing with age. Loughlin *et al.* (2003), who used a combination of SLTDRs and satellite depth recorders (SDRs) to monitor the diving behavior of juvenile Steller sea lions. Seven to ten month old sea lions tagged in Alaska typically had a mean dive duration of <1 minute and a mean dive depth of approximately 10 m. These parameters nearly doubled by the time Steller sea lions reached one year of age, when they appeared to be as capable as adults in their movement and diving behavior. Dive duration and depth did not differ between males and females in this study.

Pitcher *et al.* (2005) also found that diving performance increased as Steller sea lions developed, documenting with SDRs that young of the year Steller sea lions were capable of diving to nearly 100 m, yearlings to 200 m, and older juveniles to greater than 400 m. Mean dive duration increased with age, with maximum mean durations reaching approximately 4 minutes by 1 year of age, 5 minutes by 2 years of age, and 6 minutes by 3 years of age. On average, females appeared to dive deeper than males as they became older, but durations of dives were longer for males.

Fadely *et al.* (2005) observed similar trends for immature Steller sea lions (5 to 21 months of age) tagged with SDRs in the eastern Aleutian Islands, but also found that time at depth and dive rate (number of dives per time spent at sea within a 6 hour period) increased throughout the first 17 months of age. An increase in diving activity also coincided with increases in sea surface temperature and chlorophyll-a. Age differences in diving activity were more evident during winter months when juveniles dove more frequently, deeper, and spent more time at depth than did pups. However, between 1-2 years of age there was an apparent leveling of dive ability as measured by dive rate and time at depth.

3.1.8 Diet Preferences

Diet studies are central to understanding all other aspects of Steller sea lion life history, and particularly significant in evaluating the effects of commercial fisheries on Steller sea lion populations. Diet differentiation at regional and local scales across the range of the western DPS served as the earliest indicator of habitat requirements and metapopulation structure since confirmed by genetic analysis (O'Corry-Crowe *et al.* 2006; Sinclair and Zeppelin 2002, Sinclair *et al.* 2005). In this section, we provide an overview of Steller sea lion diet preferences. Readers are referred to the following documents for additional detailed information on the feeding ecology of Steller sea lions: the initial Steller Sea Lion Recovery Plan (NMFS 1992), revised Steller Sea Lion Recovery Plan (NMFS 2008a), and previous Section 7 consultations under the ESA (NMFS 1998, 2000, 2001, and 2003).

Steller sea lions are generalist predators that eat a variety of fishes and cephalopods distributed from nearshore demersal to epi-pelagic habitats; occasionally, other marine mammals and birds are consumed as well (Gentry and Johnson 1981, Pitcher 1981, Pitcher and Fay 1982, Calkins 1988, Calkins and Goodwin 1988, Daniel and Schneeweis 1992, NMFS 2000, Sinclair and Zeppelin 2002, Womble and Sigler 2006, Gende and Sigler 2006, Waite and Burkanov 2006).

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 Steller 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.* 2006d, Womble and Sigler 2006). 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, and capelin). There are also significant regional differences in the occurrences of some species (e.g., Atka mackerel are primarily in the Aleutian Islands, and arrowtooth flounder occur in the Gulf of Alaska).

Typically, many Steller sea lion 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).

Considerable effort has been devoted to describing the diet of Steller sea lions in the Gulf of Alaska, Aleutian Islands, and Bering Sea (Table 3.14; NMFS 2000). In the mid 1970s and mid 1980s, Pitcher (1981) and Calkins and Goodwin (1988) 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). Stomach content analysis from animals in Kodiak in the 1970s revealed walleye pollock as the most important prey in fall, winter spring, while in summer the most frequently eaten prey were small forage fishes (capelin, herring, and sand lance) (Merrick and Calkins 1996). In southeast Alaska, the most commonly identified prey items were walleye pollock, Pacific cod, flatfishes, rockfishes, herring, salmon, sand lance, skates, squid, and octopus (Calkins and Goodwin 1988, Trites *et al.* 2006d).

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). Prey occurrence of pollock, Pacific cod, and herring were higher in the 1980s than in the 1950s -1970s in stomach content samples for both eastern and western Steller sea lion populations (NMFS 2000; their Table 3.15 and Figure 3.22). 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.

Based on scat collection, Atka mackerel, pollock, and salmon have been found to be the most commonly reported prey items both in the Russian Far-East and in the central and western Aleutian Islands (Sinclair and Zeppelin 2002, Waite and Burkanov 2006). At the far western end of the Steller sea lion range, collections made in the Kuril Islands in 1962 identified Atka mackerel, sand lance, rockfish, and octopus as important foods (Panina 1966); in collections near Hokkaido, Japan from 1994 to 1996, pollock, Pacific cod, saffron cod, cephalopods, and flatfish were the main prey (Goto and Shimazaki 1998). Waite *et al.* (2005) report on diets from SSLs in Russia based on scats collected in the breeding season noting that, overall, Russian SSLs preyed on (FO, in order of high to low occurrence) Atka mackerel (65.7), pollock (32.4), salmon (29.9), and sculpins (25.7) and 20 other species at lower frequencies. In a recent study in the Kodiak Archipelago, the most frequent Steller sea lion prey were found to be Pacific sand lance, walleye pollock, arrowtooth flounder, Pacific cod, salmon, and Pacific herring (McKenzie and Wynne 2008).

Since 1990, additional information on Steller sea lion diet in Alaska has been obtained through scat analysis (Table 3.16 and Figure 3.23; Merrick *et al.* 1997, NMFS 2000, Sinclair and Zeppelin 2002, NMFS 2006b, Womble and Sigler 2006). These studies have shown that pollock continue to be a dominant prey in the Gulf of Alaska, eastern and central Aleutian Islands and that Atka mackerel was the most frequently occurring prey in the western Aleutian Islands scats. Pacific cod is also an important prey in winter, especially in the Gulf of Alaska and eastern Aleutian Islands. Salmon was eaten most frequently during the summer months in the Gulf of Alaska, and eastern and central Aleutian Islands, but in the western Aleutian Islands salmon was most frequently eaten in winter. Cephalopods continue to rank as important prey in the summer in the western Aleutian Islands. Results based on scats also indicated a wide variation in prey; 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 (Sinclair and Zeppelin 2002, Womble and Sigler 2006).

Trites and Calkins (2008) examined scat collected from a rookery and adjoining bachelor male site in Southeast Alaska to determine if the diet was similar. They found that salmon and herring dominated the summer scats of lactating females, while pollock and rockfish dominated the scat of breeding-age males and as such, scat collected at male haulouts could not be used as a proxy for female diet.

Tollit *et al.* (2009) used DNA and hard parts analysis techniques to examine SSL scats from BC and the AI regions. Using DNA techniques, salmon, cod, herring, and sole/flounder were most predominant in the BC samples; in the AI region it was Atka mackerel, sole/flounder, salmon, and pollock. Using hard parts, herring, salmon, elasmobranchs, hake, pollock, and cod were predominant in the BC samples, for the AI using this technique it was sole/flounder, pollock, sand lance, Atka mackerel, sculpins, and salmon.

The diet of Steller sea lions based on scats mirrors findings based on stable isotope analyses of blood including east to west regional differences in foraging (Kurle and Sinclair 2003, Kurle and Gudmundson 2007). The general pattern of increasing nitrogen isotope values among adult female Steller sea lions since the 1960s supports conclusions that a reduction in forage fishes and increase in demersal and semi-demersal groundfish has occurred in the North Pacific ecosystem (Conners *et al.* 2002, Hobson *et al.* 2004, York *et al.* 2008).Beck *et al.* (2007) measured blubber fatty acid profiles of young SSLs through transition from maternal dependence noting some differences in time of weaning and maternal diets between SE AK and PWS, but they could not identify the sources of fatty acids nor specific dietary components.

3.1.9 Prey Quality

Prey quality is an important factor which may change both seasonally and geographically and vary in relevance with the sex and life stage of the predator (Rosen 2009). For instance, while pollock is a staple for both sexes of juvenile and adult Steller sea lions, it may be particularly important to growing juveniles that require a high protein diet (Fritz and Hinckley 2005). In contrast, adult females may emphasize high fat foods during periods of lactation in order to maintain both adequate milk supply and body condition.

Steller sea lions are believed to encounter (on average) a prey field in the Aleutian Islands that has a lower energy density than that in SE AK (Winship and Trites 2003). However, eating prey with a relatively low energy density alone does not necessarily have negative health consequences for SSLs as long as they are readily available in predictable nearby locations (Rosen 2009, Sigler *et al.* 2009). Schaufler *et al.* (2006) examined geographical variation in Steller sea lion prey quality between the western and eastern DPSs (see also Logerwell and Schaufler 2006). Overall, the mean energy density for 22 forage species from southeastern Alaska was greater than that of 15 species from the Aleutians. Arrowtooth flounder, sandfish and squid had significantly higher energy density in southeastern Alaska than the Aleutians. Pacific cod, on the other hand, had a significantly higher energy density in the Aleutians, as did rockfish. Pollock from both regions had similar estimated energy densities, which is of particular interest because some of the sharpest declines in Steller sea lion populations have occurred in areas where pollock dominates the diet and pollock is a major component of the diet of both stocks.

The nutrient content of walleye pollock may have an impact on Steller sea lions, particularly as the energetic value appears to be relatively low during important feeding periods. Kitts *et al.* (2004) examined the seasonal changes in proximate nutrients of pollock collected in the Bering Sea. Mean energy density (dry mass) of pollock peaked in October then declined and remained low throughout winter. Energy recovery occurred in the summer months (post-spawning) with strong recovery observed in female fish caught in July. Contrary to whole fish carcass energy contents, both total protein and moisture contents were at their highest levels in winter (January) when total crude lipid content was at its lowest ($p < 0.05$). This trend gradually declined to its lowest levels in the fall when lipid content was high.

Although forage fish are generally considered to have higher lipid contents and energy densities than groundfish, not all groundfish species have energy densities lower than forage fish. Logerwell and Schaufler (2005) investigated lipid, protein, moisture, and ash compositions for several significant Steller sea lion prey species from the Gulf of Alaska, Bering Sea, and Aleutian Islands. Pacific herring, sand lance, and rockfish were found to have high average energy densities (>6 kilojoules per gram [kJ/g wet weight), while Atka mackerel, capelin, salmon, sandfish, pollock, Pacific cod, squid, skates, and rock sole had intermediate average energy densities (3-6 kJ/g wet weight). Smooth lumpsuckers and snailfish were found to have low energy densities (<3 kJ/g wet weight). Rockfish had high energy densities similar to that of Pacific herring, while some forage fish such as capelin had intermediate energy densities, similar to those of pollock or Pacific cod.

An estimation of North Pacific prey species' energetic value is complicated by seasonal and spawn-state variations, as well as some species-dependent differences such as gender and geographical region. Logerwell and Schaufler (2005) found that Atka mackerel had a gender-related variation in average energy density during the summer spawning season, with males having energy densities higher than females ($p<0.05$). Geographic comparisons indicated that pollock collected from the eastern Bering Sea (non-spawning season) had higher energy densities than those in the Aleutian Island region, which were in turn higher than those from the Gulf of Alaska.

3.1.10 Nutritional Requirements

Knowledge of nutritional requirements for Steller sea lions is crucial in evaluating the effects of commercial fisheries on the species. 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). Nutritional requirements for free-ranging Steller sea lions have not been measured, however it is clear that Steller sea lion 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 lion 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).

Data from captive animals have revealed that females increase their daily requirement by approximately 30% when they became sexually mature and produced pups (Kastelein *et al.* 1990). This study also showed that average daily food consumption increased with age (from 4 to 6 kg per day for 1 year olds to 10-13 kg per day at age 5), with males generally eating more than females. Captive feeding experiments with 1 to 2 year olds indicate that the daily maximum digestive limit of Steller sea lions is equivalent to about 14 to 16% of their body weight (Rosen and Trites 2004), and adult, non-pregnant, non-lactating pinnipeds are believed to require 6 to 10% of their body weight in food per day (Keyes 1968).

Large Steller sea lions have a relatively large stomach capacity, an adaptation that likely allows them to feed at infrequent intervals. Kastelein *et al.* (1990) reported that after a day of fasting, captive Steller sea lions ate meals that were about 25% larger than their daily average. In addition, the amount of food found in Steller sea lion stomachs has usually been on the order of one-fourth of their average daily requirements, suggesting that meal sizes may at times be much larger (Kastelein *et al.* 1990). The stomach of a 311 kg Steller 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).

Based on bioenergetic modeling, juvenile animals are predicted to have higher mass-specific food requirements than adults, while lactating females need to consume about 70% more food on average if her pup is entirely dependent on her for energy during its first year of life (Winship *et al.* 2002). In this study, the mean predicted food requirement of an average Steller sea lion consuming an average Alaskan diet was estimated at 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).

Steller sea lions, at least adult females and juveniles, are unlike most marine mammals that store large amounts of fat to allow periods of fasting. Steller sea lions need more or less continuous access to food resources throughout the year (Williams 2006a) as described in Figure 3.24, a schematic of the Steller sea lion life cycle with an emphasis on reproduction. The sensitivity of sea lions to competition from fisheries may be higher during certain times of the year. Reproduction likely places a considerable physiological or metabolic burden on adult females throughout their annual cycle. Following birth of a pup, the female must acquire sufficient nutrients and energy to support both herself and her pup. The added demand may persist until the next reproductive season, or longer, and is exaggerated by the rigors and requirements of winter conditions. The metabolic requirements of a female that has given birth and then become pregnant again are increased further to the extent that lactation and pregnancy overlap and the female must support her young of the year, the developing fetus, and herself (Williams 2006b).

3.1.11 Infectious Disease and Toxins

Disease and injury can occur as part of natural mortality and in natural populations is the mechanism by which many animals die. Relevant to understanding the current status of Steller sea lions and their critical habitat, important questions for Steller sea lions are 1) whether disease agents currently have the potential to prevent or slow recovery through increased mortality or decreased reproductive output, 2) whether disease is reducing the conservation value of critical habitat (e.g., by harming prey and/or contaminating prey), and 3) whether human actions are affecting the rate, type and incidence of disease, or are likely to do so in the future.

3.1.11.1 Disease and Parasites

Currently there is not evidence indicating that infectious disease was a predominant factor in the decline of Steller sea lions, or is currently having an effect large enough to impede recovery. However, available evidence indicates that the potential for such population-level impacts exists. While infectious disease occurs naturally in all animal populations, it occasionally can result in mortality levels large enough to have population consequences. In addition, some pathogens are known to result in reproductive loss, either through spontaneous abortions (e.g., *Brucella*) (Brown and Bolin 2000), embryonic or fetal re-absorption, or through rendering the female infertile.

Härkönen *et al.* (2006) summarized that phocine distemper virus (PDV) caused two epidemics in waters off of Northern Europe resulting in the deaths of an estimated 23,000 European harbour seals in 1988 (Dietz *et al.* 1989, Heide-Jørgensen *et al.* 1992) and more than 30,000 deaths in 2002 (Harding *et al.* 2002, Jensen *et al.* 2002). Other marine mammal species have been affected by other viruses, such as canine distemper virus and dolphin morbillivirus (Härkönen *et al.* 2006). Recently, Goldstein *et al.* (2009) confirmed the presence of phocine distemper virus (PDV) in wild caught and salvaged sea otters in areas of Prince William Sound, the Kodiak Archipelago and the eastern Aleutian Islands. Steller sea lions are now vulnerable to potential exposure to this virus.

Burek *et al.* (2006) reviewed and compared available serological data from published and unpublished sources, as well as from more recent (1997-2000) collections to determine if there was evidence indicating that infectious disease may have contributed to the decline of Steller sea lions in the western DPS. Data from the western DPS was compared with that from the eastern DPS. These authors reported that antibodies from the 1970s to the early 1990s were detectable for *Leptospira interrogans, Chlamydophila psittaci, Brucella spp.,* phocid herpesvirus-1, and calciviruses. Serum samples collected from 1997-2000 were tested for antibodies to these agents as well as to marine mammal morbilliviruses, canine parvovirus, and canine adenovirus-1 and -2. Due to inconsistencies in the data, the study was inconclusive regarding changes in antibody prevalence to these agents during the decline of Steller sea lions. Results showed that Steller sea lions had been exposed to phocid herpesviruses, caliciviruses, canine adenovirus, and *C. psittaci* in regions of both increasing and decreasing Steller sea lion abundance; the authors concluded that these agents were unlikely to have been the primary cause of the population decline but may have contributed to it or may impede recovery. There was no convincing evidence of significant exposure to morbilliviruses, *Brucella*, canine parvovirus, or *L. interrogans*.

Additional detail from studies in which samples from Steller sea lions were tested for exposure to infectious disease agents are available in earlier papers (e.g., regarding phocid herpesvirus, and phocine and canine distemper viruses: Barlough *et al.* 1987, Zarnke *et al.* 1997, Sheffield and Zarnke 1997; regarding morbilliviruses, canine parvovirus, *Brucella, Toxoplasma*, and influenza A: Sheffield and Zarnke 1997, Burek *et al.* 2003). Examination and necropsy of dead Steller sea lions has shown some occurrences of hepatitis, Chlamydia, myocarditis, endometritis, tumors, and pneumonia (Gerber *et al.* 1993). In Alaska, 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. Preliminary results indicate higher levels of stress hormones (i.e., haptoglobin) in the eastern DPS, where population densities are higher and high prevalence of hookworm parasites have been found (Rea *et al.* 2010). For further discussion of disease agents in pinnipeds, including the effects of parasites, readers may reference the Recovery Plan (NMFS 2008a).

3.1.11.2 Toxic Substances and Contaminants

No events have been recorded that support the possibility of acute toxicity causing substantial mortality of Steller sea lions (other than the Exxon Valdez oil spill in 1989, which occurred well after the Steller sea lion decline was underway) (Calkins *et al.* 1994). Likewise, no toxicological studies have been performed on Steller sea lions to determine possible effects of organochlorines, though 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).

Heintz *et al.* (2006) investigated organochlorines 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 organochlorine compounds (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, it is unlikely that OCs are the primary factor for recent declines in population and natality observed in the western DPS.

Flame retardant polybrominated diphenyl ethers (PBDEs) are compounds added to plastics, textiles, clothing, electronic circuit boards and other materials. 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). Few studies have looked at PBDE exposure and associated health effects in marine mammals; this remains a significant gap in our understanding of impacts of pollutants on Steller sea lions (Barron *et al.* 2003). Hong *et al.* (2005) examined polychlorinated biphenyl (PCB) levels in SSL tissues obtained from samples collected from the Bering Sea and PWS. They found the levels below immunotoxic and physiological toxic thresholds.

Some studies to date indicate that heavy metals are unlikely to have been a significant factor in the decline of the western DPS of Steller sea lion. Castellini (1999) found that zinc, copper, and metallothionien levels were comparable between sea lion pups sampled from both the western and eastern DPS, and were lower than for captive Steller sea lions. 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. Holmes *et al.* (2008) reported some differences in metal concentrations, and their potential toxicity, in tissues of western and eastern SSL pups, but concluded that, overall, there were few differences in metal concentrations between populations. They did report that western SSLs had statistically higher mercury in kidney and liver tissues and lead in liver tissues.

Other studies provide more concern over potential effects of contaminant levels in SSLs. Myers *et al.* (2008) studied organochlorine (OC) contaminants in blood of SSL pups from western AK and Russia. Average OC concentrations were significantly higher in the blood of Russian animals compared to western Alaska (for PCBs and DDTs, pb0.001) and in both areas females had higher concentrations than males. They imply that SSLs may be at risk for population effects from OC contaminants. In 2008, Myers studied OCs (e.g.,DDT, PCBs) in free ranging and captive SSLs. He reported that these contaminants have the potential to adversely affect SSLs because of their toxicity as accumulated in body tissues. In his thesis published posthumously, Myers indicated that his data suggest that concentrations of OCs in Steller sea lions may be high enough to cause endocrine or reproductive dysfunction and could potentially impact fertility or fecundity, stating: “Therefore, OC contaminants can not be dismissed as a contributing source to either the decline or the failure of the Steller sea lion population to recover.”

In a recent study investigating possible regional and age-related differences in mercury exposure in Steller sea lions in Alaska, Castellini *et al.* (2009) measured total mercury in hair during live capture/release studies, as well as both methyl mercury and selenium in liver samples from dead animals found opportunistically in the field. Young Steller sea lions from western populations (Aleutians, Kodiak, Prince William Sound) appeared to have higher levels of mercury exposure than those from the eastern population (Southeast Alaska); across all age classes, very young pups seem to have the highest—and also the most variable—exposure to mercury. Exposure likely occurs *in utero,* with a high proportion of mercury taking the form of methyl mercury (Castellini *et al.* 2009).

Castellini *et al.* (2009) also found that mercury concentrations decreased with age, with older pups, young of the year and yearlings showing significantly lower levels of total mercury than young pups. Newborn pups exposed to mercury appear to excrete it through their fur, as natal hair is shed and replaced by new hair growth. In addition, mercury is not lipophilic and therefore probably not present in high values in milk. Further, selenium, which binds to mercury, was found in sufficient concentrations in the liver to potentially protect animals from the toxic effects of methyl mercury. These findings indicate that Steller sea lions may have several mechanisms which allow them to process and/or sequester mercury, thereby minimizing harmful effects. However, further study is necessary to provide insight into these interactions.

Atkinson *et al.* (2008) noted that the "ability to assess fully the impact of contaminants on the wSSL population decline or their poor recovery is hampered by the lack of contaminant studies of SSL, but also by a lack of knowledge on the actual mechanisms by which contaminants influence SSL health.. They concluded that contaminants could possibly be an adverse factor regarding the future recovery of this stock.

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 polychlorinated biphenyls (PCBs) and mercury (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 effects of chronic exposure to toxins is still not well enough understood to relate observed toxin levels to population effects (impacts on vital rates and population trends) in the western DPS of Steller sea lion.

3.1.12 Predators

Steller sea lions are eaten by transient killer whales (*Orcinus orca*) in both the western and eastern DPSs. Although transient killer whale predation on Steller sea lions in the North Pacific has received substantial attention and study in recent years, major limitations in the available data result in substantial uncertainty, especially when trying to determine the historic impacts of killer whale predation. The available information on transient killer whale populations and feeding ecology within the range of Steller sea lions, and the likely impact of killer whale predation on Steller sea lions is briefly discussed here and given greater treatment in Chapter 4.

Evidence indicates that killer whale predation has been, and likely still is, a source of natural mortality in Steller sea lions. Pups and juveniles appear to be the most vulnerable age-class, although there is a high level of uncertainty associated with the available evidence. Isotope analyses and field observations indicate that the primary prey species of killer whales are at lower trophic levels than Steller sea lions, yet some groups of killer whales may specialize on Steller sea lions. Recent field studies of predation rates by GOA killer whales, a group known to specialize on Steller 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. 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 2003, the National Research Council (NRC) reported on possible factors in the historical SSL population decline and the potential role of fisheries. They noted that multiple factors were likely involved, but due to gaps in available data, identification of specific causes is unlikely, but that a weight of evidence analysis indicates the decline is most consistent with a top-down forcing scenario. The NRC (2003) report indicated top down factors are more probable than hypotheses invoking nutritional stress. Top down sources of mortality, including killer whale predation and human-caused mortality, appear to pose the greatest threat to the current population of SSLs.

Springer *et al.* (2003) suggested a hypothesis that could explain the SSL decline (and the decline of other pinnipeds and sea otters) in the North Pacific. Their hypothesis involved the ecological consequences of large industrial whaling in the 1950s to 1970s causing a sequential decline in pinnipeds and sea otters beginning in that period and continuing through the 1990s. The mechanism was killer whale predation that previously concentrated on the great whales that shifted to other sources of prey when whales were decimated in this region. Killer whales turned to SSLs, harbor seals, and sea otters as prey, sequentially driving each population to low levels. Their hypothesis was based on observations and publications of many, bioenergetic studies by the authors, knowledge of killer whale feeding ecology, and the coincidence of events since the end of World War II in the ecology of the North Pacific. Criticized by many (DeMaster *et al.* 2006; Mizroch and Rice 2006; Trites *et al.* 2007; Wade *et al.* 2007), Springer *et al.* (2003) remains an oft-referenced hypothesis on the likely importance and the impact of killer whale predation on SSLs and the degree to which killer whales may have either caused the decline in the western DPS, or currently suppresses the SSL population. In Springer *et al.* (2008), the authors disagreed with their critics. Estes *et al.* (2009) further comment on the likely sequential nature of killer whale predation on marine mammals in the North Pacific.

In recent years, studies of juvenile SSLs collected as part of Alaska SeaLife Center research and returned to the wild have found evidence for high levels of juvenile Steller sea lion mortality, presumably from killer whales, in the Kenai Fjords/PWS region in the eastern portion of the range of the western DPS. Horning and Mellish (2009, 2010a, 2010b) report that, based on data collected post-mortem from implanted life history tagged wSSLs from the PWS/KF region, 8 juvenile SSLs were confirmed dead, at least 7 of which were killed by predators. Their data show that over half of juvenile SSLs in this region are consumed by predators before age 4. They speculate that predation on SSLs, rather than low natality, may be the primary impediment to recovery of the wSSL in the PWS/KF region.

Guenette *et al.* (2007) used Ecopath and Ecosim to examine predation by killer whales in both the AI region (where SSLs declined) and SE AK (where SSLs increased). They report that killer whale predation likely contributed to the SSL decline in the central and western subareas of the AI but was not the primary cause. They note that killer whale predation, however, could have become a significant source of mortality during the 1990s when SSL numbers were low. Their work also suggests that killer whales may have been a large source of SSL mortality in SE AK in the 1960s when SSL abundance there was low, but ceased to control population numbers in the region in the 1980s and 1990s.

Population structure and seasonal movements of killer whales were studied using mark-recapture data in the Gulf of Alaska (GOA) and AI regions by Durban *et al.* (2010 in press). Between the central GOA and the central AI, abundance of transient killer whales was estimated to be 345 individuals. Durban *et al.* (2010) note the high potential level of predation pressure that comes from this high a number of killer whales. A hot spot of killer whales is in the eastern AI region near Unimak Pass, probably due to the seasonal presence of abundant prey (pinnipeds and cetaceans). They noted that throughout the study area, not all individual killer whales remain in specific areas at all times, resulting in temporal and spatial variation in predation pressure on coastal marine mammals.

Kuker and Barrett-Lennard (2010) reexamined data on sea otter mortalities and population decline in the AI region and the causative killer whale hypothesis, and found no empirical data to support that killer whales caused the decline. They conclude that the hypothesis is not disproved either, and suggest additional factors such as shark predation (increases noted in the region), other predation, disease, and contaminants (high in sea otters), and research should focus there. Maniscalco *et al.* (2008) studied SSL pup mortality using remote video at Chiswell Island in the PWS region. Pup mortality up to 2.5 months post partum averaged 15.4%, with causes varying greatly across years (2001-2007). They noted that high surf conditions and killer whale predation accounted for over half the mortalities. They speculate that even at this level of pup mortality, the Chiswell Island SSL population has increased. Between 2002 and 2007, Permyakov and Burkanov (2009) observed killer whale (ecotype uncertain) approaches (n=105) to a Russian SSL rookery, but SSL responses were minor; in one case a mass rush into the sea occurred with the SSLs tightly grouped together. They noted that predation events were very rare; killer whales seemed to show little interest in SSLs at this rookery during the breeding season.

Steller sea lions may also be attacked by sharks, though little evidence exists to indicate that sharks are actually preying upon 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. Salmon shark populations have increased since 1990, but these sharks are considered piscivorous and have not been reported to prey on Steller sea lions.

The Pacific sleeper shark (*Somniosus pacificus*) is common in the Gulf of Alaska, Aleutian Islands, and Bering Sea (Orlov 1999); however; no scientific evidence presently exists to suggest that sleeper sharks actively prey on Steller sea lions. 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 fish and cephalopods were the dominant prey (Sigler *et al.* 2006). Tissues of marine mammals were found in 15% of the shark stomachs, but no Steller sea lion tissue was detected. Overall, the study concluded that Steller sea lions are unlikely prey for sleeper sharks. A companion study documented that shark and sea lion home ranges overlapped (Hulbert *et al.* 2006); 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.

Mammal-eating killer whales and/or predation from other sources can have considerable impact on SSL populations, particularly when a sub-region is comprised of only small numbers of SSLs. Williams *et al.* (2004) examined the energy needs of killer whales and the potential caloric energy provided by various prey. Using bioenergetics and demographic modeling, Williams *et al.* (2004) reported that fewer than 40 killer whales could have caused the recent Steller sea lion decline in the Aleutian archipelago, and a pod of five could suppress a low population. Springer *et al.* (2003) similarly noted the impact of small numbers of killer whales on a depressed SSL population. Durban *et al.* (2010, in press) (see above) surmised that currently a small number of killer whales has the potential to limit recovery of the depressed SSL population in the central AI region. Guenette *et al.* (2007) similarly noted the significant source of SSL mortality from killer whales during the 1990s when SSL numbers were low. These observations, modeling efforts, and energetics research collectively point to the imposing potential impact of killer whale or other sources of predation on SSLs when SSL abundance is low, often called the predator pit concept. These studies give some insights into predator impacts on SSLs, but are based on limited observation or are based on calculations or modeling.

3.1.13 Competitors

Steller sea lions forage on a variety of marine prey that are also consumed by other marine mammals (e.g., northern fur seals, harbor seals, humpback whales), marine birds (e.g., murres and kittiwakes), and marine fishes (e.g., pollock, arrowtooth flounder). To some extent, these potential competitors may partition the prey resource so that little direct competition occurs. For example, harbor seals and northern fur seals may consume smaller pollock than Steller sea lions (Fritz *et al.* 1995). Competition may still occur if the consumption of smaller pollock limits the eventual biomass of larger pollock for Steller sea lions, but the connection would be difficult to demonstrate. Such competition may occur only seasonally if, for example, fur seals migrate out of the area of competition in the winter and spring months. Similarly, competition may occur only locally if prey availability or prey selection varies geographically for either potential competitor. Finally, competition between Steller sea lions and other predators may be restricted to certain age classes, because diet may change with age or size. Further discussion of the impacts of competitors is provided in Chapter 4.

3.1.14 Nutritional Stress in Steller Sea Lions

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 fecundity, reduced body size, higher juvenile and adult mortality, increased predation risk) (Trites and Donnelly 2003, NMFS 2000).

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 considered one of several hypotheses to explain the rapid decline of the western DPS of Steller sea lion (NMFS 2000), and has been the subject of considerable debate (e.g., NRC 2003, Trites and Donnelly 2003, Fritz and Hinckley 2005, Springer *et al.* 2003, Trites *et al.* 2006). 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 recent studies of Steller sea lions from the western DPS (Table 3.17). Yet, if survival in the 1980s was greatly reduced, then it is likely that affected animals are under-represented in current samples. 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 3.17). Therefore, if nutritional stress is acting on the western DPS, then we must look for evidence for/against chronic nutritional stress as opposed to acute nutritional stress (Trites and Donnelly 2003, Rosen *et al.* 2006).

Rosen (2009) recently reviewed and evaluated data from laboratory studies of nutritional stress in pinnipeds to test if the nutritional stress hypothesis could explain the decline of Steller sea lions. He concluded that strong evidence existed for biologically meaningful differences in the nutritional quality of major prey species, and that Steller sea lions can partly compensate for low-quality prey by increasing consumption. The study acknowledged, however, that the ability to increase consumption is physiologically limited, particularly in young animals. The overall impacts of nutritional stress are complex, the study concluded, and are dependent upon season, prey quality, age and the duration and intensity of the nutritional stress event. The effects of a nutritional stress episode are also affected by variation in body condition in the period leading up to the nutritional stress event.

3.1.14.1 Evidence of Nutritional Stress During the Rapid Decline – the 1980s

It is highly likely that no single cause fully explains the rapid decline in Steller sea lion abundance observed in the 1980s (NRC 2003). Nonetheless, 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). 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).

A reduction in average body size of Steller sea lions occurred during the period of rapid decline in the 1980s, another indication that the western DPS may have been nutritionally compromised (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 Steller 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).

Calkins *et al.* (1998) found that the largest reductions in female body size (mass, length, girth) between collections of animals from the 1980s and 1970s occurred for animals 1-8 years of age, and suggested nutrition as a main factor influencing body length. Winship *et al.* (20010 found that 90% of asymptotic length is achieved at age 4 for females and age 8 for males. Calkins and Pitcher (1982) reported that SSLs reach asymptotic length in their first 8 years of life. Calkins *et al.* (1998) backdated 8 years from their mid-1980s sample to determine the break point for the reduction in size—the late 1970s, or just after the 1977 regime shift. Ages of Steller 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 Steller 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.

Trites *et al.* (2008) compared Steller sea lion skull-growth curves from animals that grew before and after the 1976/77 regime shift, and found that females were significantly larger after this time period in the western DPS, as were males in the western DPS. No difference in body size over time was detected in the eastern DPS. The authors suggest that pups in the western DPS were nursed longer and remained longer with their mothers following the regime shift to survive in an environment of abundant, low-quality prey.

What this reflects, in fact, is that females who are giving birth less often, on average, have the capacity to nurse individual pups longer.

Overall, the changes in morphological indices from animals in the wild (Pitcher *et al.* 2000) are 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.

3.1.14.2 Evidence for Nutritional Stress 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 Steller sea lions involved comparing individual animals from the western and eastern DPS. 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.

After analysis of blood chemistry and hematology in pups less than 1 month of age, Rea *et al.* (1998) found that pups in southeast Alaska had elevated beta-HBA concentrations suggesting they underwent longer periods of fasting than seen in pups in other areas. Their findings did not indicate that pups from areas of population decline were nutritionally stressed. 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).

Results from numerous studies in the 1990s suggest that adult females at rookeries in the declining population did not have difficulty finding prey during the summer, an opposite from expected finding based on comparative studies between the western and eastern DPS of Steller 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), and dive behavior as an indicator of foraging effort during early pup rearing (Rehberg *et al.* 2009). Furthermore, no apparent difference was observed between average winter attendance cycles of females from the declining western DPS and increasing eastern DPS 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 Steller sea lions captured in a longitudinal survey in Alaska from 2000-2003, and found that growth rates for juveniles were higher in the western DPS than for the eastern DPS. This finding suggests that juveniles in the western DPS are not nutritionally stressed, a result in agreement with the relatively high survival estimated from branded animal observations (L. Fritz, AFSC, pers. comm.., 28 May 2010).

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 nutritional stress was not evident in adult females or pups from the western DPS. Whether this was due to inherent biases in the study design or other confounding factors 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 Steller 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) assuming the prey field remained constant. Despite poor knowledge of the underlying mechanisms, morphological (Williams unpublished data) and survey (Fritz and Stinchcomb 2005) data indicate a trend towards improvement in conditions of Steller sea lions in the western DPS relative to conditions in the late 1970s and 1980s, while other demographic evidence (Holmes *et al.* 2007, Holmes and York 2003, Fay 2004, Fay and Punt 2006) suggests a lingering chronic impact (low fecundity) that could affect the ability of the western DPS to recover.

3.1.14.3 Energetic Demands and 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) 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 Steller sea lions based on the notion that pollock are a low quality food and the western population of Steller sea lions has not been able to sustain itself with a larger fraction of its diet comprised of pollock. This has become 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.

Low energy prey such as pollock is part of normal Steller sea lion diets. Winship and Trites (2003) concluded that the key difference between the diets of increasing and decreasing Steller sea lion populations in the North Pacific is the overall amount of low energy prey consumed by Steller 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*. 2006d), 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).

Logerwell and Schaufler (2005) reported energy density for Steller sea lion prey species collected in the Aleutian Islands, eastern Bering Sea, and Gulf of Alaska. Sand lance, herring, and rockfish were high in lipid and thus energy density, while species such as pollock, salmon, Atka mackerel, and Pacific cod were intermediate. This study noted the importance of these data as they were the first collected on adult and juvenile pollock and adult Atka mackerel from the Aleutian Islands during the non-spawning season. They reported the following energy density rankings (from highest to lowest)(Logerwell and Schaufler 2005): Pacific herring, sand lance, rockfish, Atka mackerel, surf smelt, capelin, salmon, sandfish, adult pollock, yellow Irish lord, juvenile pollock, Pacific cod, squid, skate, rock sole, smooth lumpsucker, and snailfish.

A number of short-term diet manipulation studies on captive pinnipeds have been conducted to determine the effect of nutritional status on Steller 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). A study on mature harbor seals suggested that digestibility may be more dependant on prey species and less dependant on nutrient composition of any particular type of prey (Stanberry 2003), while showing that 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. Fadely *et al.* (1994) found that California sea lions maintained mass equally well on a diet of pollock or herring.

The maximum weight that a Steller sea lion can digest per day on a sustainable basis appears to be about 14-16% of their body mass (Rosen and Trites 2004). 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 low energy prey diet was found to necessitate food intake levels that apparently exceeded the physiological digestive capacities of young animals (Rosen and Trites 2004); in contrast, older Steller 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 Steller sea lions compared to young animals (Winship *et al.* 2002).

Rosen *et al.* (2006) also found that Steller 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. Similarly, Calkins, Castellini *et al.* (2005) conducted 3-year feeding experiments with 3 young adult sea lions and concluded that Steller sea lions were able to compensate for lower quality prey (without reaching satiation as described by Rosen and Trites 2004) and that seasonal changes in sea lion physiology was more likely to impact body condition than prey quality. In another experiment with captive juvenile Steller sea lions, Calkins, Mellish *et al.* (2005) fed a group of animals a diet of exclusively pollock and another group a mixed diet of fish and cephalopods; with no significant differences between groups in body mass, they concluded no negative consequences to free-ranging Steller sea lions from an exclusive pollock diet.

Based on additional captive studies, there appears to be a marked seasonal effect on both body mass and composition of juvenile Steller sea lions (Rosen and Trites 2002, Kumagai 2004, Kumagai *et al.* 2006). Juveniles maintained on a low-lipid pollock diet lost significantly more body lipid reserves during periods of 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 Steller sea lions fed Atka mackerel lost more of their lipid energy reserves, the Steller sea lions fed herring lost more lean body mass (e.g., muscle). If sea lions in the wild are similarly restricted in their energy intake, it could have 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). Rosen and Kumagai (2008) found that Steller sea lions that were fed restricted diets during spring, summer, fall, and winter lost an average of 10% of their initial body mass, indicating that the degree of change after food restriction was significantly affected by season. They also found that both rates of body mass loss and changes in cortisol were greatest in the winter, suggesting that Steller sea lions may be more strongly impacted by short-term, reduced energy intake during winter than during other seasons.

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 (2-6 week) changes in the diets of juvenile pinnipeds held in permanent captivity. As noted above, Calkins, Mellish *et al.* (2005) evaluated the effects of diet on free-ranging juvenile Steller sea lions held in temporary captivity. All animals gained weight on both pollock and on a mixture of fish and cephalopod 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.

The longest study conducted to date was by Castellini (2002) and Calkins, Castellini *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 Steller 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 adult sea lions could compensate for low energy prey by increasing their ingestion if sufficient quantity was available. They found that changing seasonal physiology is likely to have more impact on body condition than quality of prey (Calkins, Castellini *et al.* 2005)

Overall, as a result of the above-described studies, there is widespread agreement that adult Steller sea lions can compensate for lower energy prey by increasing the amount of food they eat. In addition, while there are some differential effects between high-lipid and low-lipid (or low-protein and high-protein) prey on sea lion body composition (e.g., decline in body fat proportion on low-lipid diet), adult Steller sea lions are able to maintain weight and healthy body condition when they consume sufficient prey to meet their energy demands, while juvenile Steller sea lions, due to smaller stomach capacity, apparently cannot if they consume only low-energy density prey (Rosen 2009). This result from captive feeding experiments suggests that in areas with a high proportion of gadid in the diet (e.g., central Gulf of Alaska, eastern Aleutian Islands), juvenile survival and condition should also be low. However, survival of juvenile sea lions in the 2000s in the eastern Aleutian Islands and central Gulf of Alaska has improved considerably since the 1980s (Figure 3.16), while juvenile condition based on animals captured and instrumented at-sea has been high (Fadely *et al.* 2005, Lander *et al.* 2009, NMML unpublished); both of these findings are contrary to the results predicted by the juvenile nutritional stress hypothesis proposed Trites and Rosen (2003) and summarized by Rosen (2009). Gadids (primarily pollock) have very likely been an important component of a healthy sea lion diet for decades (Calkins *et al.* 2005, Fritz and Hinckley 2005, Rosen 2009, Sigler *et al.* 2009).

3.1.14.4 Research Challenges

A critical 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. Figure 4.25 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. Given that no measurements have been made for adult Steller sea lions in the Alaska portion of the western DPS since the 1990s (other than numbers of individuals from population counts), changes in body condition, reproductive success, and 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 (York 2003, Fay 2004, Fay and Punt 2006, Holmes *et al.* 2007).

To date, the focus of nutritional research has been on the effects of nutritional status on individual Steller 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 system-wide and/or localized availability of prey affects Steller sea lion foraging ecology). A series of critical data gaps remain regarding the determination of 1) in which parts of the range have rates of natality continued to decline, 2) whether this decline 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.

3.1.14.5 Summary of Nutritional Stress

Nutritional stress may result from the inability of Steller sea lions to acquire sufficient prey to meet the energetic demands, especially during reproduction or seasonal growth. Steller sea lions in the 1970s and 1980s exhibited 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 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. More recently, morphological and survey data indicate a trend towards improving conditions for Steller sea lions in the western DPS, while demographic evidence from the 2000s suggests a lingering chronic impact (low fecundity) that may affect the western DPS’s ability to recover.

Findings indicate that juveniles are susceptible to nutritional stress due to their high metabolic requirements, potential consumptive limitations (Rosen and Trites 2003), and limited foraging abilities. In terms of acute nutritional stress, there is no indication at any time (1970s–2005) of emaciated juveniles or adults, of a decrease in pup body size, or of lactating females spending more time searching for prey (Table 3.17). 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 may have been due to chronic poor nutrition among other causes. 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 4-6 weeks of lactation, and (2) pregnant females with and without pups may have experienced chronic nutritional stress after leaving the rookery, as evidenced by decreased pregnancy rates of lactating females and decreased natality rates overall. There have not been any adult female Steller sea lions captured and handled since the late 1990s, due to a change in focus during the period from 2000 – 2006 on juveniles, and on the termination of required permits to handle adult females since 2006. Further, it is possible that chronic nutritional stress also may have delayed the age of sexual maturity in adult females, as well as increasing the average period of dependency of pups. These mechanisms would also be expected to reduce pup production in the population.

3.2 Steller Sea Lion Designated Critical Habitat

On August 27, 1993 NMFS published a final rule to designate critical habitat for the Steller sea lion (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 first Steller Sea Lion Recovery Team, independent marine mammal scientists, and the public (Figures 3.2 and 3.3).

Designated critical habitat for Steller sea lions (both eastern and western DPSs) includes 1) a terrestrial zone that extends 3,000 ft (0.9 km) landward from the baseline or base point of each major rookery and major haulout, 2) an air zone that extends 3,000 ft (0.9 km) above the terrestrial zone, measured vertically from sea level, 3) an aquatic zone that extends 3,000 ft (0.9 km) seaward in State and Federally managed waters from the baseline or basepoint of each major haulout in Alaska that is east of 144° W long, 4) an aquatic zone that extends 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° W long, and 5) three special aquatic foraging areas in Alaska; the Shelikof Strait area, the Bogoslof area, and the Seguam Pass area. Specific coordinates for these protected areas can be found in the regulations at 50 CFR § 226.202).

3.2.1 Description of Critical Habitat

Steller sea lions require both terrestrial and aquatic resources for survival in the wild. Land sites used by Steller sea lions are referred to as rookeries and haulouts. Rookeries are used by adult males and females for pupping, nursing, and mating during the reproductive season (late May to early July). Haulouts are used by all size and sex classes but are generally not sites of reproductive activity. The continued use of particular sites may be due to site fidelity, or the tendency of Steller sea lions to return repeatedly to the same site, often the site of their birth. Presumably, these sites were chosen by Steller sea lions because of their substrate and terrain, the protection they offer from terrestrial and marine predators, protection from severe climate or sea surface conditions, and the availability of prey resources.

Steller sea lion critical habitat is listed in 50 CFR §226.202 (all major Steller sea lion rookeries are identified in Table 3.18 and major haulouts in Table 3.19 along with associated terrestrial, air, and aquatic zones). NMFS recognizes that more accurate locations for the sites listed in 50 CFR §226.202 are available. Advances in technology and repeated surveys to these areas have resulted in more precise and accurate location estimates. NMFS intends to update these locations as soon as practicable. However, the current inaccuracy in some of the locations in 50 CFR §226.202 does not substantially diminish the utility of those designations, rather, more accurate locations would aid those citizens attempting to navigate or fish near these listed sites.

Two kinds of marine foraging habitat were designated as critical: (1) areas immediately around rookeries and haulouts, and (2) three aquatic foraging areas where large concentrations of important prey species were known to occur.

First, areas 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 foraging 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 must 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. Young animals are almost certainly less efficient foragers and may have relatively greater food requirements, which 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 animals, and juveniles, and those areas were deemed essential to protect.

Second, three marine areas were chosen based on 1) at-sea observations indicating that Steller sea lions commonly used these areas for foraging, 2) records of animals killed incidentally in fisheries in the 1980s, 3) knowledge of Steller 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 Steller 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 Steller sea lion foraging (Perez and Loughlin 1991).

3.2.2 Essential Features of Critical Habitat

The regulations at 50 CFR §424.12(b) outline those physical and biological features which should be considered when designating critical habitat for listed species, which include space for individual and population growth; food, water, air, light, minerals, or other nutritional or physiological requirements; cover or shelter; sites for breeding, reproduction, rearing of offspring; and habitats that are protected from disturbance or are representative of the historic geographical and ecological distributions of a species.

In general, the physical and biological features of critical habitat essential to the conservation of Steller sea lions are those items that support successful foraging, rest, refuge, and reproduction. The August 27, 1993 final rule to designate critical habitat for the Steller sea lion (58 FR 45269) describes essential aquatic (foraging areas) and terrestrial features (rookeries and haulouts) of critical habitat and the rational behind the regulatory definition of critical habitat.

3.2.2.1 Essential Features of Marine Critical Habitat

Prey resources are the most essential feature of marine critical habitat for Steller sea lions (see 58 FR 45269). Marine areas may be used for a variety of other reasons (e.g., social interaction, rafting or resting), but foraging is the most important Steller sea lion activity that occurs when the animals are at sea. A discussion of Steller sea lion foraging patterns and prey use has been discussed in Sections 3.1.6-3.1.10. While many of the important physical and biological elements of Steller sea lion critical habitat can be identified, most of those features (particularly biological features) cannot be described in a complete and quantitative manner.

The status of critical habitat is best described as the status of the important prey resources contained within those areas. Walleye pollock, Atka mackerel, Pacific cod, arrowtooth flounder, rockfish, herring, capelin, sand lance, other forage fish, squid, and octopus are important prey items found in Steller sea lion critical habitat. Due to the dynamic nature of aquatic ecosystems and fish, NMFS was unable to describe the specific attributes of prey within critical habitat at the time of listing. Thus, prey resources were described in general, and are constantly re-assessed to determine their conservation value to Steller sea lions. These fishery resources are evaluated annually and that description is contained in the SAFE reports produced by the Council and NMFS.

Understanding the status and trends of fish species known to be prey items essential to Steller sea lions is a crucial aspect in understanding the quality of critical habitat and, in turn, potential impacts to Steller sea lion populations. Although this consultation addresses the entire federal groundfish fishery, as well as parallel fisheries, the following four fish species are considered to be principal prey species for Steller sea lions in the western DPS: Atka mackerel, walleye pollock, Pacific cod, and arrowtooth flounder. This determination results from species found to make up at least 10% of the Steller sea lion diet (Sinclair and Zeppelin 2002).

We include here a brief summary of status for these principal prey species from the 2009 SAFE reports, including historical ABC and TAC data where available (see Tables 3.20-3.27). It is important to note that certain locations are more significant than others for Steller sea lions relative to prey, depending how heavily they rely on a particular species as a component of their diet. For instance, Atka mackerel make up less of the diet for Steller sea lions in the Gulf of Alaska than they do in the Aleutian Islands, while arrowtooth flounder in the Gulf of Alaska are more important to sea lions than they are in the Aleutian Islands. Further information on these species can be accessed by referencing the SAFE reports in their entirety (NPFMC 2009).

Atka Mackerel

Atka mackerel (*Pleurogrammus monopterygius*) is the most prevalent species found in the diet of Steller sea lions in the western and central Aleutian Islands region (Sinclair and Zeppelin 2002) and in the Russian subpopulation (Waite *et al.* 2005). The species is widely distributed along the continental shelf across the North Pacific Ocean and Bering Sea from Asia to North America. On the Asian side they extend from the Kuril Islands to Provideniya Bay (Rutenburg 1962); moving eastward, they are distributed throughout the Commander and Aleutian Islands, north along the eastern Bering Sea shelf, and through the Gulf of Alaska to southeast Alaska. During periods of high recruitment in the Aleutian Islands, it is thought that juvenile Atka mackerel may move into the Gulf of Alaska under favorable conditions (Ronholt 1989, Lowe *et al.* 2005). Recently, Atka mackerel have been detected by the summer trawl surveys primarily in the Shumagin (Western) area of the Gulf of Alaska.

The Aleutian Islands biomass estimate from the 2006 Aleutian Islands bottom trawl survey was 728,935 mt, down 18% relative to the 2004 survey estimate. The decrease in biomass in the 2006 survey is largely a result of a decrease in biomass found in the western area (376,414 mt in 2004 down to 100,693 mt in 2006), despite a large increase in the eastern area. Relative to the 2004 survey, the 2006 biomass estimates are down 73% in the Western area, up 3% in the Central area, and up 44% in the eastern area. Areas with large catches of Atka mackerel in the most recent 2006 survey included Seguam Pass, Tanaga Pass, Kiska Island, and Stalemate Bank.

Most of the Gulf of Alaska Atka mackerel biomass (96%, 98%, and 99.6% in 2005, 2007, and 2009 respectively) is distributed within the Shumagin area of the western GOA. Atka mackerel were encountered in 24% of the hauls conducted in the Shumagin area in the 2009 survey. The 2009 estimate of Atka mackerel biomass in the Shumagin area is 135,100 t.

In the GOA, from 1977 to 1984 and in 1990, up to 11% of the annual Atka mackerel harvest was caught within 20 miles of all Gulf of Alaska Steller sea lion rookeries and major haulouts, reflecting the offshore distribution of the fishery. In 1991-1993, however, the fishery moved closer to shore, and this percentage increased to 82-98%, almost all of which was caught between 10-20 nm of Steller sea lion rookeries on Ogchul and Adugak Islands (near Umnak Island), and Atkins and Chernabura Islands in the Shumagin Islands. Currently, however, Steller sea lion protection measures prohibit directed fishing for Atka mackerel in the GOA; there has not been a directed Gulf of Alaska Atka mackerel fishery since 1996. This fishery is now prosecuted mostly by cooperatives using larger trawl vessels (2% is allocated to jig gear) and in platoon groupings to spread effort geographically and temporally.

Walleye Pollock

Walleye pollock (*Theragra chalcogramma*) is a semi-pelagic schooling fish widely distributed in the North Pacific Ocean. Concentrations in areas and depths are dependent on season. Pollock in the Gulf of Alaska are managed as a single stock independently of pollock in the Bering Sea and Aleutian Islands. In the Bering Sea (2009 SAFE report) the 2009 unfished spawning biomass was estimated to be 4,934,000 mt, the female spawning biomass 1,443,000 mt (projected to be 1,830,000 mt in 2010), and a maximum 2010 ABC (Tier 1b) of 813,000 mt. The 2009 Bogoslof pollock echo integration-trawl (EIT) survey resulted in the lowest estimate of biomass (110,000 mt) in the region since the EIT survey began in 1988. No fishery is recommended on the Bogoslof stock. In the Aleutian Islands, spawning biomass (F100) is 300,000 mt, with the maximum permissible ABC for 2010 under Tier 3b of 33,064 t. The maximum permissible ABC under Tier 5 using the 2006 bottom trawl survey for both years is 15,670 t. In the GOA, the model estimate of spawning biomass in 2010 is 184,567 t, which is 29.8% of unfished spawning biomass (based on average post-1977 recruitment) and below B40% (248,000 t). Recommended ABC for 2010 was 77,150 mt.

The fishery for pollock in the Gulf of Alaska is entirely shore-based with approximately 90% of the catch taken with pelagic trawls. During winter, fishing effort targets pre-spawning aggregations in Shelikof Strait and near the Shumagin Islands. Fishing in summer is less predictable, but typically occurs on the east side of Kodiak Island and in nearshore waters along the Alaska Peninsula. Since 1992, the Gulf of Alaska pollock TAC has been apportioned spatially and temporally to reduce potential impacts on Steller sea lions.

The Aleutian Islands fisheries management area is comprised of the federal and Alaska State managed waters off the Aleutian Islands chain from 170°W to 170°E longitude. The 1991 through 2002 Aleutian Islands bottom trawl surveys consistently found a relatively even distribution of pollock throughout the Aleutian Islands fishery management area. In contrast, the 2004 and 2006 Aleutian Islands trawl surveys reveal an apparent easterly shift in the distribution pollock. The earlier bottom trawl surveys estimated that on average 35% of the total Aleutian Islands pollock biomass was located in the area between 170°W and 174°W longitude. The 2004 and 2006 surveys estimated 85% and 73% of the total Aleutian Islands pollock biomass, respectively, occurred in this eastern-most area of the Aleutian Islands fisheries management area.

There is considerable uncertainty in the Aleutian Islands pollock assessment, in part due to an incomplete understanding of pollock stock structure and because no fishery or survey age data have been available since 2006. The stock assessment for this species indicates that the spawning stock biomass dropped to near 20% of virgin biomass in 1999, but began to increase in following years due to a moratorium on directed pollock fishing between 1999 and 2005. The lack of age data since 2006 makes the more recent trends in abundance from the stock assessment model rather uncertain, but the model suggests a stabilization to slight increase. The estimated female spawning biomass projected for 2010 is 97,486 t. The projected total age 3+ biomass for 2010 is 307,046 t.

Pacific Cod

Pacific cod (*Gadus macrocephalus*) is a transoceanic species, occurring at depths from shoreline to 500 m. The southern limit of the species' distribution is about 34° N latitude, with a northern limit of about 63° N latitude. Pacific cod is distributed widely over the eastern Bering Sea (EBS) as well as in the Aleutian Islands (AI) area. Tagging studies (e.g., Shimada and Kimura 1994) have demonstrated significant migration both within and between the EBS, AI, and Gulf of Alaska (GOA). Juveniles occur mostly over the inner continental shelf at depths of 60 to 150 m. Adults occur in depths from the shoreline to 500 m, although occurrence in depths greater than 300 m is fairly rare.

Estimates of total abundance (both in biomass and numbers of fish) obtained from the trawl surveys in the BSAI indicate that biomass increased steadily from 1978 through 1983, and then remained relatively constant from 1983 through 1988. The highest biomass ever observed by the survey was the 1994 estimate of 1,368,120 t. Following the high observation in 1994, the survey biomass estimate declined steadily through 1998. The survey biomass estimates remained in the 596,000-619,000 t range from 2002 through 2005. Survey biomass estimates dropped after 2005, producing all-time lows in 2007 and again in 2008. The 2009 biomass estimate was slightly higher than the 2008 estimate.

In the GOA, the highest biomass ever observed by the survey was the 2009 estimate of 752,651 t, and the low point was the preceding (2007) estimate of 233,310 t. The 2009 biomass estimate represented a 223% increase over the 2007 estimate. In terms of population numbers, the record high was observed in 2009, when the population was estimated to include over 573 million fish. The 2005 estimate of 140 million fish was the low point in the time series.

Arrowtooth Flounder

The arrowtooth flounder (*Atheresthes stomias*) is a relatively large flatfish which occupies continental shelf waters almost exclusively until age 4, but at older ages occupies both shelf and slope waters. Migration patterns are not well known; however, there is some indication that arrowtooth flounder move into deeper water as they grow, similar to other flatfish. Spawning occurs in deep waters (>400m) in winter along the continental shelf break.

Arrowtooth flounder have a wide-spread bathymetric distribution in the Bering Sea/Aleutian Islands region and are believed to be at a high level, primarily as a result of a series of above average year-classes spawned from 1995-2003, and minimal commercial harvest. Biomass point estimates for arrowtooth flounder from the standard survey area in the eastern Bering Sea and Aleutian Islands region indicate that between 1982-1994, arrowtooth abundance increased eight-fold to a high of 570,600 mt. The population biomass remained at a high level from 1992-1997. Following this, population biomass declined, then increased in 2005 to peak at 757,685 mt. The 2009 survey point estimate was lower at 453,559 mt.

Modelling estimates indicate that arrowtooth flounder total biomass increased more than four-fold from 1976 to the 2009 value of 1.09 million mt. In the Aleutian Islands, arrowtooth flounder comprised an average (of survey years 2002, 2004, 2006) of 2%, 2%, and 11% of the estimated biomass in Regulatory Areas 543, 542, and 541, respectively. Trends are level in Areas 543 and 542, but showed an increase in Area 541 over the three survey years.

For the BSAI region, the projected age 1+ total biomass for 2010 is 1,120,160 mt. The projected female spawning biomass for 2010 is 807,100 mt. The recommended 2010 ABC is 156,300 mt.

In the Gulf of Alaska, arrowtooth flounder are currently one of the most abundant groundfish species. Trophic studies suggest they are an important component in the dynamics of the Gulf of Alaska benthic ecosystem. The recommended 2010 ABC in the Gulf of Alaska is 215,882 mt. The projected female spawning biomass for the GOA for 2010 is 1,253,210 mt.

3.2.2.2 Essential Features of Terrestrial Critical Habitat

In this section we describe important Steller sea lion habitat areas based on usage patterns. This includes the determination of important sites not previously designated as critical habitat under the ESA, a review of rookeries, and a description of the seasonal usage of both ESA and non-ESA designated sites.

Long-used rookery sites were likely selected by Steller sea lions for a variety of reasons, including substrate and terrain, protection from land-based and marine predators, protection from harsh wave or surf conditions, and local availability of prey. Successful reproduction for the species depends on the availability of rookery sites where animals can aggregate for sufficiently long periods of time to give birth, mate, and raise their young until the young are able to survive at sea. As the reproductive period requires at least several months, food supplies in the vicinity of the rookeries must be sufficient to meet the energetic needs of animals involved in reproduction (adult females and males and pups). Once the reproductive season and the need for social aggregation is over, and pups have gained sufficient competence at sea, then animals (including mothers with pups) may not disperse to other haulout sites. Throughout the remainder of the year, the local availability of prey remains a crucial factor (probably the most important factor) in determining their movements and distribution. Mothers with dependent pups are still likely to be constrained in their foraging distribution. All pups are susceptible because they have limited reserves compared to adult animals. Pups in the process of weaning are likely poor foragers that may be susceptible to reductions in prey availability. Pups are likely dependent on nearshore prey resources while they make the difficult transition to independent foraging. Juveniles, older but still immature, must continue to develop their foraging skills over time, but probably remain particularly sensitive to reductions in available prey. Like other, older animals, they may range more widely, but their distribution and haulout patterns must be determined, in large part, by the availability of prey.

The foraging success of these animals, whether based on rookeries or haulouts, is determined by their ability to balance the gains from foraging with the costs of daily activities, including the act of foraging itself (i.e., energy balance). If the prey resources around rookeries and haulouts are inadequate for their needs (potentially reduced or depleted), then they are forced to increase the time and energy expended to find sufficient prey. As a result, they are more likely to fail in securing the resources necessary for growth, reproduction, and survival. Population recovery will likely depend upon increased reproduction and juvenile survival.

Determination of Important Terrestrial Sites

In the 1998 Biological pinion (NMFS 1998), NMFS identified nineteen Steller sea lion sites which were not designated as critical habitat but which required special management measures in order to avoid jeopardizing the western DPS. The determination was based on historical population counts in which at least one recorded count of non-pup Steller sea lions exceeding certain criteria during the breeding (greater than 200 non-pups from May-August) or non-breeding seasons (greater than 75 non-pups from September-April; NMFS 1998; their Table 2.26). The nineteen sites were not designated as critical habitat, but were included for management purposes as part of the RFRPA process (NMFS 1999).

The analysis was repeated again in March 2006 (NMFS 2006b) and updated through breeding season non-pup surveys conducted in 2006 and 2007 (Fritz *et al.* 2008). The criteria for breeding season haulout use remained the same (>200 non-pups), but the non-breeding season threshold count was raised from 75 to 100 non-pups. This was based on the work of Sease and York (2003), who found that non-breeding season counts were approximately half those of breeding season counts. The threshold number of 200 was used previously during critical habitat designation to determine which haul-outs were “major” based largely on counts conducted during the breeding season. Therefore, the 200 non-pup breeding season count was retained as the threshold, and the non-breeding season count of 100 was used to identify major non-breeding season haulouts. Thus, the reference to “major” haulouts and rookeries throughout this section refer to those listed in critical habitat based on the thresholds described above. Additional haulouts and rookeries not originally designated as critical habitat, but considered necessary to provide protection for the recovery and conservation of the species, are referred to here as “important” sites.

Analysis of non-pup count data collected through 2007 indicates that Samalga Island and Amchitka/Cape Ivakin (listed in bold in Table 3.28) could be removed from the list of 19 important sites because both sites had only one breeding season count > 200 animals. Ugamak/Round (in italics in Table 3.28) failed to meet either criteria since 1990, but should be retained as an important site for management purposes, given that it is an integral part of the Ugamak Island rookery complex and represents a significant terrestrial site within that complex. An additional 21 haul-out sites were identified as meeting the criteria for an important site. However, only six sites met the criteria since 1990 and should be included as important sites:

- ELIZABETH/CAPE ELIZABETH had 112 non-pups in March 1993
- FLAT had 174 non-pups in Dec 1994 and 125 in March 1999
- UNGA/ACHEREDIN POINT had 264 non-pups in June 2004
- UNIMAK/OKSENOF POINT had 269 non-pups in June 2007
- TAGALAK had 150 non-pups in March 1999
- SEMISOPOCHNOI/TUMAN POINT had 154 non-pups in March 1993

The remaining fifteen sites technically met the criteria, but all had only 1 or 2 counts that met the criteria and all but two occurred prior to 1966. Therefore, the following sites may not currently be important sites:

- USHAGAT/ROCKS SOUTH (breeding 1985)
- UGAIUSHAK (breeding 1956)
- AKUN/AKUN HEAD (non-breeding 1960)
- AKUTAN/NORTH HEAD (non-breeding 1957)
- EGG (non-breeding 1957)
- UNALASKA/CAPE STARICHKOF (non-breeding 1960 and breeding 1977)
- UNALASKA/SPRAY CAPE (non-breeding 1960)
- CARLISLE (breeding 1960 and breeding 1965)
- AMLIA/CAPE MISTY (breeding 1959)
- IKIGINAK (breeding 1959)
- IGITKIN/SW POINT (breeding 1959)
- SKAGUL/S. POINT (breeding 1959)
- GARELOI (breeding 1960).

- USHAGAT is the island on which USHAGAT/SW is located, and the latter is both an ESA-listed haul-out and an RFRPA site
- AMATIGNAK is the island on which AMATIGNAK/NITROF POINT is located, and the latter is both an ESA-listed haul-out and an RFRPA site.

The sites listed in Table 3.29 meet the minimum thresholds of non-pup counts in the breeding and non-breeding seasons since 1990, are not ESA-listed critical habitat sites, and are also considered to be important sites. The list includes 23 sites: 17 of the original 19 important sites (NMFS 1999; Table 3.28), plus 6 additional sites identified in NMFS (2006b) and in Fritz *et al.* 2008.

Determination of Important Rookeries

A site has previously been designated as a "rookery" when a minimum number of pups have been born and certain demographic and behavioral characteristics have been observed including: bulls defending territories occupied by adult females with pups, a low proportion of juvenile animals, and sub-adult males occupy the area outside of defended territories. It is important to identify these rookery sites such that appropriate management can be applied to rookeries which are more vulnerable to stressors during the summer pupping and breeding season.

For this analysis, rookeries were defined as sites with a pup count of at least 50 since 1978. In support of this value, the age and sex composition of the Steller sea lion population occupying these rookeries was compared with that on haulouts based on analysis of medium format photographs taken in 2004 (NMFS 2006b). Based on the analysis (NMFS 2006b), five new sites[11] in the range of the western DPS should be considered rookeries for conservation purposes (NMFS 2006b): Chiswell Islands, Jude Island, Kanaga/Ship Rock, Lighthouse Rocks, and Ushagat/SW. For the eastern DPS in southeast Alaska, two new sites, Biali Rocks and Graves Island, are now considered rookeries. White Sisters also was established as a new rookery in Southeast Alaska in the early 1990s.

Four sites in the range of the western DPS previously designated as rookeries should be considered as haulouts because none have had a pup count >50 (Agligadak, Semisopochnoi/Pochnoi, Semisopochnoi/Petrel, Amchitka/East Cape) (NMFS 2006b). The remaining 34 ESA-listed rookeries should retain their rookery status, and the five sites listed above should be added for a total of 39 rookeries in the western DPS. The eastern DPS rookery count should currently number five with the new sites from above. These changes are articulated in Table 3.30.

Determination of Terrestrial Seasonal Usage Patterns

As stated above, the selection of important sites and seasons is based on the requirement to provide the protection necessary for recovery and conservation of the species. The analysis evaluates important sites and seasons based on seasonal counts from 1990-2005. The following general standards were used to make determinations about the important sites described above. Designated critical habitat sites that did not meet the criteria were labeled as "neither" in the seasonal column (Table 3.31).

Summer haulout: > 200 non-pups in at least 1 year since 1990

---

[11] These 5 sites are designated critical habitat haulouts under the ESA (50 CRR part 226.202). Although the critical habitat designation includes a determination of haulout or rookery for each site, no specific action is required for haulouts in the ESA designation (whereas no-entry zones exist to 3 nautical miles for rookeries in the western DPS). However, the type of site is important when considering whether the habitat is being conserved under Section 7 of the ESA. Therefore, accurate description of whether a site is a haulout or rookery is important and must be updated occasionally as usage patterns change due to population demographics and environmental changes.

Winter haulout: > 100 non-pups in at least 1 year since 1990
Summer rookery: > 50 pups in at least 1 year since 1975 (and had > 200 non-pups since 1990)
Winter rookery: > 100 non-pups in at least 1 year since 1990 (site must also be classified as summer rookery)

Seasonal differences were a result of variations in sea lion counts in summer and winter surveys (Sease and York 2003), smaller aggregations and wider dispersal in winter, more time at sea in winter and less time at haulouts where they would be counted. The list in Table 3.31 includes all of the sites designated as critical habitat, the additional 23 sites identified above, and 34 sites which did not meet either seasonal criteria during this time period.

While these are the best available data for the purpose of identifying important haulout sites, the breeding and non-breeding seasonal counts summarized here could under-represent the importance of some haulout sites to Steller sea lions and, on that basis, should not be considered conservative. Finally, factors other than the decline could have altered the distribution patterns of Steller sea lions and the relative importance of their haulouts. However, the best available scientific and commercial data are not sufficient to describe such a change in haulout patterns as a result of changes in oceanographic parameters or changes in composition of the prey community. As described above, the existing data on haulout patterns is sufficient to indicate some hauling sites that have been or are currently important to Steller sea lions. It is not sufficient to detect shifting patterns of use that could be attributed to any general factor.

3.3 Humpback Whale

3.3.1 Species Description and Distribution

The humpback whale (*Megaptera novaeangliae)* belongs to the Order Cetacea, suborder Mysticete. The mysticete are baleen whales, named for the comb-like plates (baleen) descending from the roof of the mouth that are used to filter prey. Humpback whales are in the family of rorquals, the Balaenopteridae.

Humpback whales are distributed worldwide in all ocean basins. Two types of migrations may be distinguished: (1) within-season movement through a portion of the summer range, presumably in order to find or follow concentrations of prey; and (2) long-distance migrations between summering and wintering areas (NMFS 1991). Although humpback whales travel to follow prey, they also exhibit a high degree of site fidelity to feeding areas by segregating into discrete feeding aggregations, between which little interchange occurs (Waite *et al.* 1999, Calambokidis *et al.* 2001, Witteveen *et al.* 2004, Calambokidis *et al.* 2008). The rate of interchange between Alaska feeding areas (i.e., southeast Alaska, Prince William Sound, the Gulf of Alaska, Kodiak Island, Yakutat Bay, and the Bering Sea) has been found to be less than 1% (Mizroch *et al.* 2004).

Most humpback whales occur in the temperate and tropical waters of the northern and southern hemispheres in the winter (from 10°-23° latitude). During this period, breeding and reproductive activities are their principal focus; during the warmer months, humpback whales move to northern latitudes where feeding is the principal activity. The historic feeding range of humpback whales in the North Pacific included coastal and inland waters around the Pacific Rim from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Nemoto 1957, Tomlin 1967, Johnson and Wolman 1984).

Traditionally, efforts to define humpback whale feeding aggregations and migratory routes have relied on mark-recapture techniques using photo-id or genetics; however, both methods require individuals to be sampled at both breeding and feeding grounds (Witteveen *et al.* 2009a). Recent analysis of stable carbon and nitrogen isotopes has been used to examine population structure and feeding ecology, where isotopic signatures in the tissues of whales reflect the ratio of heavy to light isotopes in their foods. Beyond defining distinct feeding aggregations, geographic variability in stable isotope ratios of both humpback whales and their prey can be used to understand more about foraging ecology and prey use in humpback whale habitat areas (Witteveen *et al.* 2009a). Using this technique, researchers found evidence for 6 isotopically distinct feeding groups in the North Pacific. Further results show promise in increasing understanding of humpback whale life history in the North Pacific by assigning breeding whales to their high-latitude feeding destinations through stable isotope analysis (Witteveen 2009b).

To date, three management units (populations) of humpback whales are recognized in the North Pacific, migrating between their respective summer/fall feeding areas to winter/spring calving and mating areas as follows (Calambokidis *et al.* 1997, Baker *et al.* 1998):

> 1) the California/Oregon/Washington and Mexico population, which are found winter/spring in coastal Central America and Mexico and migrate to the coast of California to southern British Columbia in summer/fall (Calambokidis *et al.* 1989, Steiger *et al.* 1991, Calambokidis *et al.* 1993);
>
> 2) the Central North Pacific population, which are found winter/spring in the Hawaiian Islands and migrate to northern British Columbia/southeast Alaska (including Glacier Bay) and Prince William Sound west to Kodiak in summer/fall (Baker *et al.* 1990, Perry *et al.* 1990, Calambokidis *et al.* 1997); and
>
> 3) the Western North Pacific population, which occurs in winter/spring off Japan and, based on Discovery Tag information, probably migrate to waters west of the Kodiak Archipelago (the Bering Sea and Aleutian Islands) in summer/fall (Berzin and Rovnin 1966, Nishiwaki 1966, Darling 1991).

Currently, these populations continue to be distinguished as three management units for conservation and management purposes, as defined in the Stock Assessment Reports produced by NMFS. However, recent data from the Structure of Populations, Levels of Abundance, and Status of Humpbacks (SPLASH) project conducted between 2004-2006 indicate that population structure and migration patterns are much more complex than previously understood. Though many life history questions remain about migratory pathways, routes and destinations, the overall pattern from the research findings showed that coastal wintering regions of the western (Asia) and eastern (mainland Mexico and Central America) North Pacific were the primary wintering areas for the lower latitude coastal feeding regions. The wintering areas off Hawaii and Revillagigedo Archipelago were the primary wintering regions for the more central and northern latitude feeding areas (Calambokidis *et al.* 2008).

For the purposes of this Biological Opinion, the effects of the proposed action on humpback whales will be assessed for two of these three management units based on their distributional overlap with the action area: the Central North Pacific population and the Western North Pacific population. However, given that this consultation occurs under the authority of Section 7 of the ESA, NMFS will ultimately be making final conclusions on effects to the humpback whale as a species as listed under the ESA.

3.3.1.1 Western North Pacific Population

The continental shelf of the Aleutian Islands and Alaska Peninsula were once considered the center of the North Pacific humpback whale population (Berzin and Rovnin 1966; Nishiwaki 1966). The northern Bering Sea, Bering Strait, and the southern Chukchi Sea along the Chukchi Peninsula appear to form the northern extreme of the Western North Pacific population's range (Nikulin 1946, Berzin and Rovnin

1966). However, sightings of humpback whales in the Bering Sea were most frequent south of Nunivak Island and east of the Pribilof Islands (Berzin and Rovnin 1966; Braham *et al.* 1977; Nemoto 1978; Braham *et al.* 1982; Leatherwood *et al.* 1983).

Surveys in the central-eastern and southeastern Bering Sea in 1999 and 2000 resulted in updated information about the distribution of humpback whales in these areas (Moore *et al.* 2002). The only sightings of humpback whales in the central-eastern Bering Sea were southwest of St. Lawrence Island; animals co-occurred with a group of killer whales and a large aggregation of Arctic cod. A few sightings occurred in the southeast Bering Sea, primarily outside Bristol Bay and north of the eastern Aleutian Islands (Moore *et al.* 2002). In a NMFS survey cruise in 2001 and 2002 of the central and eastern Aleutian Islands, humpback whales were most common in the area between Samalga and Unimak Islands (Sinclair *et al.* 2005). Of the 259 individuals seen, only 3 were west of Samalga. These sightings clearly demonstrate that the Aleutian Islands and Bering Sea remain important feeding areas (NMFS 2006a). In addition, a NOAA survey conducted in 2005 found numerous humpback whales north of the central Aleutian Islands, reinforcing the idea that the Bering Sea is an important foraging habitat (Angliss and Allen 2009).

Data from the SPLASH research effort has suggested the likely existence of unknown wintering areas (those that have not been previously described) for humpbacks which feed around the Aleutians and in the Bering Sea. These animals were not well-represented in any of the winter sampling areas, indicating that they must be using a different and unidentified winter location. Although a reasonable assumption for this breeding ground would be a region in the eastern central North Pacific, the location is uncertain given the complexities revealed through the SPLASH study around migratory pathways (Calambokidis *et al.* 2008).

3.3.1.2 Central North Pacific Population

Humpback whales in the Central North Pacific population are believed to summer throughout southeast Alaska, the central and western portions of the GOA, including Prince William Sound, around Kodiak Island (including Shelikof Strait and the Barren Islands), and along the southern coastline of the Alaska Peninsula. In Prince William Sound, prior to 1991, humpback whales have congregated near Naked Islands, in Perry Passage, near Cheega Island, in Jackpot, Icy and Whale Bays, in Port Bainbridge and north of Montague Islands between Green Island and the Needle (Hall 1979, 1982; von Ziegesar 1984; von Ziegesar and Matkin 1986). The few sightings of humpback whales in offshore waters of the central GOA are usually attributed to animals migrating into coastal waters (Morris *et al.* 1983), although use of offshore banks for feeding is also suggested.

Most humpback whales in the Central North Pacific population spend the winter months in the waters off Hawaii where they breed, give birth to and nurse their calves. The whales undertake the northward migration to Alaska waters in late winter and generally arrive on the feeding grounds in May, remaining into November before returning to the waters off Hawaii. Some animals, however, remain on the feeding grounds year-round. Humpback whales do not feed while on the wintering grounds off Hawaii.

3.3.2 Listing Status

Humpback whales were listed as endangered under the ESA in 1973 (16 USC 1531 *et seq.*) due to the reduced population size that resulted from significant commercial whaling harvest. At the time of this listing, the population was considered to be in danger of extinction in all or a portion of its range. Historically, both aboriginal and early commercial harpoon whalers harvested an unknown number of humpback whales. Much greater harvest pressure occurred in the 20th Century when these animals were subject to heavy commercial exploitation during modern whaling operations. Prior to 1905, there were an estimated 15,000 humpback whales in the entire North Pacific; by 1966, following commercial harvest, the population was estimated to be between 1,000 and 1,200 animals. Measures to protect the humpback whales in the North Atlantic were first taken in 1946 with the establishment of the regulatory International Whaling Commission (IWC), and a ban on non-subsistence hunting followed in 1955. In 1965, the IWC banned the commercial hunting of humpback whales in the Pacific Ocean**.** Soviet whalers, however, continued to harvest humpback whales until 1980 (Perry *et al.*1999). Currently, some illegal whaling continues although actual harvest levels are unknown. Humpback whales are also protected by the Convention on International Trade in Endangered Species of Wild Flora and Fauna and the MMPA.

3.3.3 Population Status and Trends

There are clear indications of increasing trend for humpback whales found in the North Pacific basin. The past annual abundance estimate for the entire North Pacific population (Central, Eastern, and Western) was 6,010 animals (Calambokidis *et al.* 1997). More recent abundance estimates are now available for humpback whale populations throughout the North Pacific based on data collected under the SPLASH program through photo-identification studies of humpback whales in wintering areas and summer feeding areas. The current SPLASH-based population estimate for the entire North Pacific is approximately 18,000-20,000 individuals. A total of 18,469 quality fluke-id photographs were taken in over 27,000 approaches of humpback whales, in an international effort involving 50 research groups and more than 400 researchers in 10 countries. Over 50% of the population was estimated to winter in Hawaii, with large numbers also overwintering in Mexico. Numbers of whales wintering in Asia and Central America were low (1,000 or less). Average abundance estimates for feeding areas ranged from approximately 100-700 for Russia, 6,000-14,000 for the Bering Sea and Aleutians, 3,000-5,000 each for the Gulf of Alaska and the combined Southeast and British Columbia area, 200-400 for Southern British Columbia-Northern Washington, and 1,400-1,700 for California-Oregon (Calambokidis *et al.* 2008).

3.3.3.1 Western North Pacific Population

Until recently, the Western North Pacific humpback whale population was previously estimated at about 394 animals (Calambokidis *et al.* 1997). However, in light of the low geographic coverage of sampling effort for the western North Pacific population, this number was likely an underestimate of the stock's true abundance (Angliss and Allen 2009). Currently, results from the SPLASH project estimate a population of approximately 6,000-14,000 for the Bering Sea and Aleutians, and 100-700 for Russia.

Waite *et al.* (1999) identified 127 individual humpback whales in the Kodiak Island area between 1991 and 1994 and estimated there were 651 whales in this region (95% CI:356-1,523). This study also estimated that 200 humpback whales regularly feed in Prince William Sound. Meanwhile, photo-identification studies initiated to the west of Kodiak Island from 1999 to 2002 have identified 171 individual humpback whales, which resulted in a mark-recapture estimate of 410 (95% CI: 241-683). It is not known how many animals occurring to the west of Kodiak Island belong to the Western or Central North Pacific stock, but matches between animals photographed west of Kodiak Island and animals photographed in Hawaii, offshore Mexico, coastal Mexico, and Japan clearly indicate that overlap between stocks occurs in this area (Witteveen *et al.* 2004).

In the BSAI, the humpback whale population was dramatically reduced by commercial whaling. The humpback whale population is believed to have increased since whaling ceased, although the rate of increase is unknown. Brueggeman *et al.* (1987) did not sight humpback whales in the North Aleutian and St. George Basin Outer Continental Shelf planning areas to the north and west of the Alaska Peninsula. Similarly, Stewart *et al.* (1987) did not observe humpback whales during aerial surveys on or near areas hunted by vessels from the Akutan whaling station in the eastern Aleutians. Braham *et al.* (1977) saw 14 humpback whales in the northern Bering Sea in August 1976, and Braham *et al.* (1982) documented 25 humpback whales between 1958 and 1978 between Unimak Pass and the Pribilof Islands in the southern

Bering Sea (NMFS 2006). A research cruise in 1994 found humpback whales in scattered aggregations (57 sightings) throughout the 2,050 nm study area south of the Aleutian Islands (Forney and Brownell 1996). Sightings of humpback whales also occurred during the survey conducted in the eastern Bering Sea in 2000; these sightings resulted in an estimated abundance of 102 (95% CI: 40-262). It is unknown whether these animals belong to the central or western North Pacific stock of humpback whales (Angliss and Allen 2009).

3.3.3.2 Central North Pacific Population

Humpback whales in the Central North Pacific show a high degree of fidelity to feeding areas. This fidelity is maternally directed; that is, whales return to the feeding areas where their mothers first brought them as calves (Martin *et al.* 1984, Baker *et al.* 1987). Humpback whales in the Central North Pacific population typically show fidelity to either the southeast Alaska or the Prince William Sound feeding areas. Photographs taken from 1979-1996 indicate that less than 1% of the individual whales photographed in these areas moved between areas (Mizroch *et al.* 2004). Due to the lack of interchange between the southeast Alaska area and the Prince William Sound, Kodiak and Shumagin Islands feeding areas, it is unlikely that a reduction in the population in southeast Alaska would be augmented by animals that use other feeding areas. Thus, NMFS is considering whether the southeast Alaska feeding area, and possibly other feeding areas in the North Pacific, warrant formal designation as separate stocks under the MMPA (Angliss and Allen 2009).

Prior to results from the SPLASH study, abundance estimates indicated that the Central North Pacific stock consisted of approximately 4,005 individuals and had been increasing in abundance since the early 1980s (Angliss and Outlaw 2005; Mobley, Jr. *et al.* 1999; Baker and Herman 1987). This estimate uses Darroch's (1961) method, utilizing only data from wintering areas and averaging the 1991-92, 1992-93, and 1991-93 winter release-recovery information (Calambokidis *et al.* 1997). The current best adult survival rate in the Central North Pacific population is estimated at 0.963 (95% CI: 0.944-0.978) based on over 31 years of data from over 11 research groups (Mizroch *et al.* 2004).

In Prince William Sound from 1977 to 2001, 315 individual humpback whales identified using fluke photo-identification methods (von Ziegesar 1992, Waite *et al.* 1999, von Ziegesar *et al.* 2004). In the Northern British Columbia region (primarily near Langara Island), 275 humpback whales were identified from 1992 to 1998 (G. Ellis, pers. comm., reported in Angliss and Lodge 2004). Straley *et al.* (2002) analyzed data for the northern portion of southeast Alaska from 1994-2000 and indicated an annual abundance of 961 humpback whales. However, this estimate for southeast Alaska is known to be a minimum estimate because there was little to no photo-identification effort in the lower half of southeast Alaska (south of Frederick Sound) (Angliss and Allen 2009). In addition, current results from the SPLASH study indicate that the humpback whale population in the combined feeding areas of southeast Alaska and Northern British Columbia is approximately 3,000-5,000 animals at present (Calambokidis *et al.* 2008).

In the Northern British Columbia region, 850-1,000 individuals were identified in 2003, significantly more than the previously reported 275 animals from 1992-1998 (J. Ford, pers. comm. in Angliss and Outlaw, 2005). In Hawaii, Baker and Herman (1987) estimated the entire stock to be 1,407 individuals from data collected between 1980 and 1983. Between 1993 and 2000, aerial surveys were conducted throughout the main Hawaiian Islands which resulted in estimates that grew from 2,754 in 1993 to 4,491 in 2000 (Mobley *et al.* 2001).

Humpback whales in southeast Alaska have been intensively studied in Glacier Bay National Park and Preserve. The Park Service has monitored humpback whales in the bay every year since 1985 to document the number of individuals, residence times, spatial and temporal distribution, feeding behavior and interactions with vessels (Neilson and Gabriele 2008). This monitoring program covers most of Glacier Bay and Icy Strait. The overall number of whales in the combined study area of Glacier Bay and Icy Strait has generally increased over the last 20 years. In 2008, 157 whales were observed, and fifteen mother/calf pairs were documented in Glacier Bay/Icy Strait with a calculated crude birth rate of 9.6% (Neilson and Gabriele 2008).

Estimates of population growth vary among researchers and across regions for humpback whales in the central North Pacific population. Mizroch *et al.* (2004) estimated the central North Pacific stock to be increasing at a rate of 10% per year. Mobley *et al.* (2001) estimated an annual population increase of 7% based on aerial surveys conducted from 1993 – 2000 across the main Hawaiian Islands. The rate of increase between 1979 and 2000 for the humpback whale population in southeast Alaska has been calculated as ~0.088 (Angliss and Lodge 2004). The currently acknowledged rate of increase for the central North Pacific population of humpback whales is 7% per year (Angliss and Allen 2009), while humpback whale population growth for the North Pacific as a whole is estimated to be increasing at 5% per year (Calambokidis *et al.* 2008).

3.3.4 Life history information

3.3.4.1 Reproductive Biology

Humpback whale calving in the northern hemisphere generally takes places between January and March. Age at sexual maturity has been estimated to range from 4 to 9 years in females, and annual pregnancy rates have been estimated at about 0.40-0.42 (NMFS unpublished and Nishiwaki 1959). The calving interval for females, though variable, appears to range from 2-3 years. For some females, however, calving may take place on an annual or multi-year basis (up to 5 years) (NMFS 1991). Gestation averages about 12 months and lactation generally lasts close to a year. One female in southeast Alaska was observed with a calf for three consecutive summers, while another was seen with a calf for two summers in a row. Although the specific timing of separation may vary, the majority of calves are weaned after one year (Perry *et al.* 1999). In the North Pacific, the annual reproduction rate in wintering areas has been estimated at 0.58 calves per year; in summering areas, this rate was estimated at 0.38 calves annually (Perry *et al.* 1999).

3.3.4.2 Feeding Ecology and Behavior

Humpback whales feed in coastal waters near shore and exhibit a wide range of foraging behaviors. They feed singly or in groups using several different feeding strategies to capture their prey. Some of the common feeding behaviors in southeast Alaska include lunge-feeding conducted by individual animals, non-synchronized diving behavior, and bubble-net feeding. Bubble-net feeding generally involves an assemblage of animals diving near an aggregation of prey, releasing bubbles to concentrate the prey and surfacing through the bubbles to capture the prey. On each lunge, each whale in the group appears to maintain the same position, indicating an organized feeding structure during such maneuvers (Alaska-BC Whale Foundation 1996). Little is known about the underlying social structure of such foraging groups, although research indicates that animals associated with one another through foraging appear to have enduring social bonds and may represent combinations of individuals performing compatible tasks (Sharpe 2002). Ongoing investigations into feeding behavior in southeast Alaska are examining the relatedness of humpback whales in cooperative social groups through genetic analysis (Alaska-BC Whale Foundation 1996).

Humpback whales feed mainly on small schooling fishes, euphausiids, and other large zooplankton. Fish prey species in the North Pacific include Pacific herring (*Clupea harengus pallasi*), capelin (*Mallotus villosus*), juvenile walleye pollock (*Theragra chalcogramma*), and sand lance (*Ammodytes hexapterus*).

Humpback are also known to feed on eulachon, Atka mackerel, Pacific cod, saffron cod, arctic cod, juvenile salmon, and rockfish. In the waters west of the Attu Islands and south of Amchitka Island, Atka mackerel were preferred prey of humpback whales (Nemoto 1957). Invertebrate prey include euphausiids (krill), mysids, amphipods, shrimps, and copepods (Wing and Krieger 1983, Krieger and Wing 1986, Krieger 1986). The productive temperate waters off Alaska have historically contained large numbers of herring schools and krill patches in inland coastal waters in predictable locations. Humpback whales in southeast Alaska, although not limited to these areas, return to specific feeding locations such as Frederick Sound, Chatham Strait, North Pass, Sitka Sounds, Glacier Bay, and Point Adolphus. Adult animals typically consume up to 3,000 pounds per day, and generally only forage while on the feeding grounds 6 to 9 months of the year. Should the animals not get enough food during the time spent in Alaska, compensation will not occur in other locations or at other times of the year.

3.3.4.3 Diving Behavior

Maximum diving depths recorded for humpback whales are approximately 150 meters (but usually less than 60 meters), and dives may last up to 21 minutes (Hamilton 1997, Dolphin 1987). In southeast Alaska, dive times for feeding humpback whales average 2.8 minutes. For non-feeding whales, dive times average 3.0 minutes, and for resting whales dive times average 4.3 minutes (Dolphin 1987). Most humpback whale dive depths are probably relatively shallow due to the fact that their prey is generally found at depths above 300 m (NMFS 2002).

3.3.4.4 Vocalizations and Hearing

Hearing in marine mammals is a function of the level of sounds that marine mammals can hear in the absence of ambient noise (hearing thresholds); the ability of the animal to discriminate between different frequencies and intensities; effects of masking (the ability to distinguish signal from ambient); and individual variability. Humpback whales communicate at and respond to low frequency noise, generally in the range of 12 Hertz (Hz) to 22 kiloHertz (kHz). Frankel (1994) estimated the source level for singing humpback whales at between 170-175 dB. On the breeding grounds, male humpback whales sing long, complex songs that range in frequency from 25 to 5000 Hz and can reach intensities of up to 181 decibels (dB) (Winn *et al.* 1970, Thompson *et al.* 1986). Thompson *et al.* (1979) estimated source levels of singing whales to average 155 dB and range from 144 dB to 174 dB. These songs appear to have an effective range of six to 12 miles (10 to 20 km). Humpback whales appear to produce a wide variety of sounds during the breeding season, while fewer sounds are produced on the summer feeding grounds. Feeding groups produce distinctive sounds ranging from 20 Hz to 2 kHz, with median durations of 0.2-0.8 seconds and source levels of 175-192 dB (Thompson *et al*. 1986). Sounds produced on the feeding ground can be characterized as loud, trumpet-like calls, and appear to be used to herd schooling fish and attract other whales to the feeding activity (D'Vincent 1985, Sharpe and Dill 1997, Alaska-BC Whale Foundation 1996).

Anatomical evidence also indicates that baleen whales are adapted to hear low-frequency sounds (Ketten 1998). Observations of whale responses to low frequency sound sources also support this (Richardson and Greene 1993, Richardson *et al.* 1995). Migrating gray whales would avoid a sound source 50% of the time when the received level was 116-124 dB (Malme *et al.* 1984, 1983). However when similar noises were played to feeding humpback whales, they showed no response at received levels up to 120 dB (Malme *et al.* 1985). The results of these studies indicate that prolonged exposure to man-made sounds at received levels greater than 120 dB will elicit a behavioral response from baleen whales (Frankel and Clark 1998). Few studies of humpback whale response to vessels have included specific sound levels where behavioral responses occurred.

3.3.5 Natural Causes of Mortality

3.3.5.1 Disease

As with any wild mammal population, a multitude of infectious (viral, bacterial, parasitic, or mycotic) or toxicological (heavy metal, organochlorine) diseases may affect marine mammals. Toxins are known to affect humpback whales, but the effects of disease on their population trends are unknown. In 1987 and 1988, 14 humpback whales died in Cape Cod Bay from paralytic shellfish poisoning (PSP) as a result of ingesting dinoflagellate saxitoxin-infected Atlantic mackerel (NMFS 1991; Perry *et al.* 1999). This incident is the only natural mass mortality on record. There are no records of such poisonings occurring in Alaska with this species. In addition, humpback whales are known hosts for the parasite *Crassicauda boopis,* a nematode that may cause mesenteric arteritis, occlusion of the blood vessels draining the kidneys, congestive kidney failure, and death (NMFS 1991).

3.3.5.2 Predators

Killer whales prey upon humpback whales, although such attacks are observed relatively infrequently. In Alaska, 15-20% of the photographically identified humpback whales bear scars of killer whale attack (Perry *et al.* 1999), although the two species are also observed feeding in close proximity without aggressive interactions (NMFS 1991). All three ecotypes of killer whale (resident, transient, and offshore) are known to inhabit the same waters as humpback whales in many parts of Alaska, yet predation appears to be relatively rare. Younger animals may be more vulnerable to this type of predation during migration when group size is smaller than in summering or wintering areas (Perry *et al.* 1999). Apparent shark bites have also been documented on adult animals, and observed rake marks on the fins and flippers of calves have indicated attacks by false killer whales (NMFS 1991).

A recent study examined the incidence of rake marks from killer whales on humpback whale flukes to assess predation pressure throughout the North Pacific (Steiger *et al.* 2008). After analyzing 3,650 fluke photographs from 16 summering and wintering areas collected from 1990-1993, the authors found varying percentages of rake mark scarring among regions. The highest rates occurred off Mexico and off feeding areas in California, suggesting that attacks occur at or near the wintering grounds in the eastern North Pacific. The prevalence of rake marks indicated that killer whale predation has the potential to be a major source of mortality; however, the location of attacks is inconsistent with the hypothesis that humpback whales seek tropical waters to avoid predation. The authors conclude that attacks in the wintering grounds are made primarily on calves (Steiger *et al.* 2008).

3.3.6 Humpback Whale Critical Habitat

Critical habitat has not been designated for humpback whales anywhere throughout their range.

3.4 Sperm Whale

3.4.1 Species Description and Distribution

Sperm whales (*Physeter macrocephalus*) are distributed in all of the world’s oceans and throughout most Alaskan waters (Angliss and Lodge 2004). Several authors have recommended three or more stocks of sperm whales in the North Pacific for management purposes (Kasuya 1991, Bannister and Mitchell 1980). However, the IWC’s Scientific Committee designated two sperm whale stocks in the North Pacific: a western and eastern stock (Donovan 1991). The line separating these stocks has been debated since their acceptance by the IWC’s Scientific Committee. For stock assessment purposes, NMFS recognizes three discrete population centers of sperm whales: (1) Alaska, (2) California/Oregon/Washington, and (3)

Hawaii. For the purposes of this Biological Opinion, based on their overlap with the action area, we will assess impacts to the Alaska (North Pacific) sperm whale population.

Sperm whales are found throughout the North Pacific and are distributed broadly from tropical and temperate waters to the Bering Sea as far north as Cape Navarin. Mature female and immature sperm whales of both sexes are found in more temperate and tropical waters from the equator to around 45°N throughout the year. These groups of adult females and immature sperm whales are rarely found at latitudes higher than 50°N and 50°S (Reeves and Whitehead 1997). Sexually mature males join these groups throughout the winter. During the summer, mature male sperm whales are thought to move north into the Aleutian Islands, GOA, and the Bering Sea.

Sightings surveys conducted by the AFSC in the summer months between 2001 and 2006 have found sperm whales to be the most frequently sighted large cetacean in the coastal waters around the central and western Aleutian Islands (AFSC unpublished data). Acoustic surveys detected the presence of sperm whales year-round in the Gulf of Alaska although they appear to be more common in summer than in winter (Mellinger *et al.* 2004). These seasonal detections are consistent with the hypothesis that sperm whales migrate to higher latitudes in summer and migrate to lower latitudes in winter (Whitehead and Arnbom 1987).

Sperm whales are rarely found in waters less than 300 m in depth. They are often concentrated around oceanic islands in areas of upwelling, and along the outer continental shelf and mid-ocean waters. Because they inhabit deeper pelagic waters, these whales generally remain offshore in the eastern AI, GOA, and the Bering Sea. Their distribution generally does not include the broad continental shelf of the Eastern Bering Sea.

Areas where sperm whales are consistently seen and heard are assumed to be important to their survival (NMFS 2006c). Areas used infrequently or for short periods may also be linked to population fitness. Habitat characterization involves, among other things, descriptions of prey types, densities, and abundances, and of associated oceanographic and hydrographic features. Inter-annual variability in habitat characteristics, and in habitat use, are important components of habitat characterization. Researchers in many different areas have begun to explore the correlations between sperm whale occurrence and habitat features (Jaquet and Whitehead 1996; Jaquet *et al.* 1996, 1998; Waring *et al.* 1993; Davis *et al.* 1998; Hooker *et al.* 1998).

3.4.2 Listing Status

Sperm whales have been protected from commercial harvest by the IWC since 1981, although the Japanese continued to harvest sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). Sperm whales were listed as endangered under the ESA in 1973 (16 USC 1531 *et seq.*) due to the reduced population size that resulted from significant commercial whaling harvest. At the time of this listing, the population was considered to be in danger of extinction in all or a portion of its range.

Approximately 258,000 sperm whales in the North Pacific were harvested by commercial whalers between 1947 and 1987 (Hill and DeMaster 1999). In particular, the Bering Sea population of sperm whales (consisting mostly of males) was severely depleted (Perry *et al.* 1999). Catches in the North Pacific continued to climb until 1968, when 16,357 sperm whales were harvested. Catches declined after 1968 through limits imposed by the IWC. Measures to protect sperm whales were first taken in 1946 with the establishment of the regulatory International Whaling Commission (IWC). Sperm whales are also protected by the Convention on International Trade in Endangered Species of Wild Flora and Fauna and the MMPA.

3.4.3 Population Status and Trends

Current estimates for population abundance, status, and trends for the Alaska population of sperm whales are not available (Hill and DeMaster 1999). However, the number of sperm whales in the eastern North Pacific has been estimated to be 39,200 animals (Barlow and Taylor 1998). During 2001 and 2002 research cruises by NMFS in the central and eastern Aleutian Islands, 56 individual sperm whales were seen in waters west of Samalga (Sinclair *et al.* 2005).

The density of sperm whales (individuals per 1,000 km^2) has been estimated for five large study areas in the North Pacific: 1.36 in the eastern tropical Pacific (Wade and Gerrodette 1993, as corrected by Whitehead 2002); 1.16 in the western North Pacific (Kato and Miyashita 1998, as corrected by Whitehead 2002); 1.08 off the U.S. West Coast; 3.4 to 4.2 in the eastern temperate Pacific (Barlow and Taylor 2005); and 2.82 in the Hawaiian EEZ (Barlow 2006). Collectively, these surveys cover the majority of sperm whale habitat in the North Pacific. If the mean density in those areas (2.33 per 1,000 km^2) were extrapolated to the 80 million square kilometers in the entire North Pacific, the sperm whale population would be about 187,000. Using Whitehead's (2002) global average of sperm whale density (1.40 per 1000 km^2), the North Pacific would have approximately 112,000 sperm whales. Given these extrapolations, a practical working range for estimates for sperm whale abundance in the entire North Pacific might be 100,000-200,000 (NMFS 2006c).

3.4.4 Life History Information

3.4.4.1 Reproductive Biology

Female sperm whales take about 9 years to become sexually mature (Kasuya 1991, as cited in Perry *et al.* 1999). Male sperm whales take between 9 and 20 years to become sexually mature, but will require another 10 years to become large enough to successfully compete for breeding rights (Kasuya 1991). Adult females give birth after about 15 months gestation and nurse their calves for 2 B3 years. The calving interval is estimated to be about four to six years (Kasuya 1991). The age distribution of the sperm whale population is unknown, but sperm whales are believed to live at least 60 years (Rice 1978). Estimated annual mortality rates of sperm whales are thought to vary by age, but previous estimates of mortality rate for juveniles and adults are now considered unreliable (IWC 1980, as cited in Perry *et al.* 1999).

3.4.4.2 Feeding Ecology and Behavior

Sperm whales are known for their deep foraging dives (in excess of 3 km). They feed primarily on mesopelagic squid, but also consume octopus, other invertebrates, and fish (Tomilin 1967, Tarasevich 1968, Berzin 1971). Perez (1990) estimated that their diet in the Bering Sea was 82% cephalopods (mostly squid) and 18% fish. Fish eaten in the North Pacific included salmon, lantern fishes, lancetfish, Pacific cod, pollock, saffron cod, rockfishes, sablefish, Atka mackerel, sculpins, lumpsuckers, lamprey, skates, and rattails (Tomilin 1967, Kawakami 1980, Rice 1986b). Sperm whales taken in the GOA in the 1960s had fed primarily on fish. Daily food consumption rates for sperm whales ranges from 2 - 4% of their total body weight (Lockyer 1976b, Kawakami 1980).

Sperm whales' capacity to produce multipulsed, long duration, nondirectional signals or "clicks" appears to be a key foraging mechanism for the species. This biosonar is thought to originate in the nose of the sperm whale and has been recorded at levels up to 236 dB re 1 micropascal. At these source levels, sperm whale clicks are the loudest biologically-produced sound ever recorded (Mohl *et al.* 2003). In addition, during foraging dives, sperm whales also appear to produce "creaks", a series of rapid-click buzzes interspersed among regular series of clicks (Miller *et al.* 2004). Dtags attached to sperm whales in the

Gulf of Mexico and the Ligurian Sea by Miller *et al.* (2004) supported the hypothesis that creaks are produced during prey capture as an echolocation signal adapted for foraging.

Vision may also play a central role in sperm whale predation. One hypothesis maintains that sperm whales locate their prey visually in silhouette or by searching for bioluminescence produced by prey movements (Fristrup and Harbison 2002). This is based in part from studies that document the importance of vision in mesopelagic communities, namely, in midwater fish and invertebrates. An alternate theory maintains that sperm whales create a zone of stimulated bioluminescence around their mouths which attracts those squid, fish and invertebrates that are also visual predators (Fristrup and Harbison 2002).

Directed studies of sperm whale feeding ecology are challenging. The whales are usually distributed far offshore (at least in areas where the shelf is wide) and feed at considerable depth on cephalopod species which are themselves difficult to sample and study. Most of what is known about sperm whale feeding has come from examinations of stomach contents of killed whales. Whitehead *et al.* (1989) and Smith and Whitehead (1993) used observed defecation rate as an index of "feeding success" in sperm whales near the Galápagos and related this index to oceanographic conditions (NMFS 2006c).

Male sperm whales are known to take sablefish off longline gear in the GOA (Rozell 2004). Further discussion of this behavior can be found in the Environmental Baseline section for this species.

3.4.5 Natural Causes of Mortality

Sperm whales are vulnerable to predation by killer whales (Pitman *et al.* 2001). In 1997, Pitman *et al.* (2001) observed a pod of nine sperm whales under attack by a herd of approximately 35 killer whales 130 km off the coast of central California. The attack appeared to be a wound and withdraw strategy on the part of the killer whales. As defense, sperm whales formed a rosette with heads together and tails facing out. Over four hours, one sperm whale from the group was killed and eaten, while the rest received serious, and possibly fatal, wounds. Several other encounters between killer whales and sperm whales were also witnessed. The authors conclude that killer whale predation may have been an important selective factor in sperm whale evolution and ecology, which in turn may have influenced the development of complex social behavior and distribution patterns.

The papilloma virus is also a potential source of natural mortality for sperm whales (Lambertson *et al.* 1987). Overall, however, little is known about rates of natural mortality in sperm whale populations from predation or disease.

3.4.6 Sperm Whale Critical Habitat

Critical habitat has not been designated for sperm whales anywhere throughout their range.

3.5 Fin Whale

3.5.1 Species Description and Distribution

Fin whales are widely distributed in the world's oceans. Within the U.S. waters in the Pacific, fin whales are found seasonally off the coast of North America and in the Bering Sea during the summer. Recent information on seasonal fin whale distribution has been inferred from the reception of fin whale calls by bottom-mounted, offshore hydrophone arrays along the U.S. Pacific coast, in the central North Pacific, and in the western Aleutian Islands. High levels of fin whale calls have been recorded along the U.S. Pacific coast beginning in August/September and lasting through February, suggesting that these may be important feeding areas during the winter (Moore *et al.* 1998, 2006, Watkins *et al.* 2000, Stafford *et al.* 2007). Call rates peaked during fall and winter in the central North Pacific and the Aleutian Islands and occurred at a lower rate during summer months in the Gulf of Alaska (Stafford *et al.* 2007). Seasonal differences in recorded call rates are generally consistent with the results of aerial surveys conducted to document seasonal whale distribution. However, it is not known whether these differences in call rates reflect true seasonal differences in whale distribution, differences in calling rates, or differences in oceanographic properties (Moore *et al.* 1998). Fin whale calls have also been recorded in Hawaiian waters in all months except June and July (Thompson and Friedl 1982; McDonald and Fox 1999). Sightings of fin whales in Hawaii are extremely rare: documented sightings occurred in 1976 (Shallenberger 1981), 1979 (Mizroch *et al.* 2009) and 1994 (Mobley *et al.* 1996).

Surveys in the central-eastern and southeastern Bering Sea in 1999 and 2000 and in coastal waters of the Aleutian Islands and the Alaska Peninsula from 2001 to 2003 resulted in new information about the distribution and relative abundance of fin whales in these areas (Moore *et al.* 2000, 2002; Zerbini *et al.* 2006). Fin whale abundance estimates were nearly five times higher in the central-eastern Bering Sea than in the southeastern Bering Sea (Moore *et al.* 2002), and most sightings in the central-eastern Bering Sea occurred in a zone of particularly high productivity along the shelf break (Moore *et al.* 2000).

3.5.2 Listing Status

The fin whale has been listed as endangered under the ESA since its passage in 1973. Although most populations were depleted by modern whaling in the mid-twentieth century, there are still tens of thousands of fin whales worldwide. Commercial whaling for this species ended in the North Pacific in 1976. Fin whales are also protected by the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES), the MMPA (“depleted” status), and the International Whaling Convention (IWC). Critical habitat has not been designated for this species in waters off Alaska, California, Oregon, and Washington. A Final Recovery Plan was issued by NMFS in 2010 (75 FR 47538 August 6, 2010).

For management purposes, three stocks of fin whales are currently recognized in U.S. waters in the Pacific: 1) Alaska (Northeast Pacific [NEP]), 2) California/Washington/Oregon, and 3) Hawaii. New information from Mizroch *et al.* (2009) suggests that this structure should be reviewed and updated, if appropriate, to reflect current data. The NEP stock of fin whales is likely to be affected by the Federal and parallel groundfish fisheries off Alaska.

3.5.3 Population Status and Trends

The total North Pacific fin whale population before whaling began, has been estimated at 42,000–45,000, based on catch data and a population model (Ohsumi and Wada 1974; Omura and Ohsumi 1964). Of this, the “American population” (i.e., the component centered in waters east of 180° W longitude) was estimated to be 25,000–27,000. There were an estimated 8,000–11,000 fin whales in the eastern North Pacific in 1973 (Ohsumi and Wada 1974). From a crude analysis of catch statistics and whaling effort, Rice (1974) concluded that the population of fin whales in the eastern North Pacific declined by more than half, between 1958 and 1970, from about 20,000 to 9,000 “recruited animals” (i.e., individuals longer than the minimum length limit of 50 ft).

Reliable estimates of current and historical abundance for the entire NEP fin whale stock are currently not available. Population surveys, resulting abundance estimates, and their regional significance and limitations are reviewed in Allen and Angliss (2009). Modern surveys include portions of the range south of the Aleutian Islands in 1994 (Forney and Brownell 1996), central-eastern Bering Sea in July-August 1999 and in the southeastern Bering Sea in June-July, 2000 (Moore *et al.* 2000, 2002), and coastal waters of western Alaska and the eastern and central Aleutian Islands in July-August 2001-2003 (Zerbini *et al.*

2006). Although the full range of the NEP stock of fin whales in Alaskan waters has not been surveyed, a rough estimate of the size of the population west of the Kenai Peninsula can be attained by summing estimates from Moore *et al.* (2002) and Zerbini *et al.* (2006). Using this approach, the provisional estimate of the fin whale population west of the Kenai Peninsula is 5,700 individuals. This is a minimum estimate for the entire stock as surveys covered only a small portion of the stock's range.

Current Population Trend

Zerbini *et al.* (2006) estimated that the fin whale population in coastal waters south of the Alaska Peninsula (Kodiak and Shumagin Islands) increased at annual rate of 4.8% (95% CI: 4.1-5.4%) from 1987-2003. This is the first available estimate of population growth for North Pacific fin whales and is consistent with other estimates of population growth rates of large whales. This estimate should be used with caution, however, due to uncertainties in the initial population estimate for the first trend year (1987) and due to uncertainties about the population structure of the fin whales in the area. Also, the study represented only a small fraction of the range of the NEP stock.

3.5.4 Life History Information

3.5.4.1 Reproductive Biology

Most fin whale reproductive activity, including mating and births, takes place in the winter season (November to March; peak December/January) (Haug 1981; Mitchell 1974), although "out-of-season" births do occur off the eastern United States (Hain *et al.* 1992). The gestation period is probably somewhat less than a year, and fin whale calves are nursed for 6–7 months (Haug 1981; Gambell 1985).

The average calving interval has been estimated at about two years, based on whaling data (Christensen *et al.* 1992). In unexploited populations, the interval may be somewhat longer. Agler *et al.* (1993) used photo-identification data to estimate an average interval of 2.7 years for fin whales in the Gulf of Maine although they acknowledged that this value was probably biased upward by incomplete sighting histories. If certain females calved in "missed" years (i.e., years in which they were not photo-identified in the study area), the mean interval could have been as low as 2.24 years (Agler *et al.* 1993). Ohsumi's (1986) analysis of age at sexual maturity for a large sample of fin whales killed in the eastern North Pacific from the mid-1950s to 1975 showed a marked decline with time. According to Ohsumi, the average age at attainment of sexual maturity declined from 12 to 6 years in females and from 11 to 4 years in males. This change was interpreted by Ohsumi as a density-dependent response to heavy exploitation of the population. Sigurjønsson (1995) gave the range of pregnancy rates for the species (proportion of adult females pregnant in a given year) as 0.36–0.47.

3.5.4.2 Feeding Ecology and Behavior

Although there may be some degree of specialization, most individuals probably prey on both invertebrates and fish, depending on availability (Watkins *et al.* 1984; Edds and Macfarlane 1987; Borobia *et al.* 1995). Sergeant (1977) suggested that euphausiids were the "basic food" of fin whales and that they took advantage of fish when sufficiently concentrated, particularly in the pre-spawning, spawning, and post-spawning adult stages on the Continental Shelf and in coastal waters. In the North Pacific overall, fin whales apparently prefer euphausiids (mainly *Euphausia pacifica*, *Thysanoessa longipes*, *T. spinifera*, and *T. inermis*) and large copepods (mainly *Calanus cristatus*), followed by schooling fish such as herring, pollock , and capelin (Nemoto 1970; Kawamura 1982).

3.5.5 Natural Causes of Mortality

Injury and suffocation from ice entrapment is not known to be a cause of natural mortality of fin whales in the North Pacific as it is in the North Atlantic (NMFS 2010). Killer whales presumably attack fin whales at least occasionally; however, there is little evidence of such predation from the North Pacific (Tomilin 1967). Shark attacks presumably occur on young or sick fin whales, although such events have not been documented.

4 ENVIRONMENTAL BASELINE

The Environmental Baseline is an analysis of the effects of past and ongoing human-caused and natural factors leading to the current status of the species or its habitat and the ecosystem within the action area. Environmental baselines for Biological Opinions include past and present impacts of all state, federal or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions that are contemporaneous with the consultation in process (50 CFR 402.02). Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

The Environmental Baseline represents the impacts of a suite of prior and contemporaneous human activities, and natural phenomena, on threatened and endangered species and on designated critical habitat. Environmental Baselines as are not "lines" defined in time, but are base conditions for the individuals, populations, and constituent elements (within critical habitat) that occur within an action area. The purpose of an Environmental Baseline is to describe the response of listed resources that have been exposed to physical, chemical, biotic, and ecological stressors (and subsidies) directly or indirectly caused by the suite of activities. This suite of activities represents subsidies (for example, additional opportunities for sea lions to eat by catching fish caught in nets) and stressors (for example, being shot or harassed by fishermen, having less food available in the water column, having to spend more time searching for food, experiencing disturbance on rookeries and haulouts from overflights, etc.). The set of subsidies and stressors associated with the activities form a response regime. These response regimes might change from place to place (for example, there could be one response regime in the western Aleutians and another in the Central Gulf of Alaska) and over time.

When we develop the Environmental Baseline (this chapter), we assess the base condition of the listed individuals and the populations those individuals comprise. We also assess the base condition of designated critical habitat.

For Steller sea lions, information used in previous Biological Opinions was reviewed and used where appropriate in our analysis of the Environmental Baseline. We also used new and different information on Steller sea lions from studies that were completed or published since the earlier Biological Opinions. The new information used included recent studies on Steller sea lion population trends and vital rates, physiology, movements, foraging behavior and diet, predation, and an analysis of Steller sea lion population trends compared to fisheries and oceanographic data. We also revisited earlier studies using a report and a compendium of Steller sea lion literature published between 2000 and early 2006 contracted by the NPFMC (Loughlin and Taggart 2006). That report resulted in a compilation of nearly all available literature (reports, gray literature, peer-reviewed journal articles, abstracts of symposium papers and posters, and other documents) on Steller sea lions published through early 2006, most of which were not available since the last Biological Opinion had been prepared. In addition, NMFS (2008) completed a revised recovery plan for Steller sea lions which provides additional information on historical stressors and those stressors likely to be acting today, and includes demographic and threats criteria.

4.1 Ecosystem Dynamics in the Action Area

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), Mueter *et al.* (2004), Guenette and Christensen (2005), Mundy (2005), Weingartner *et al.* (2002), and Weingartner *et al.* (2009).

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 Steller sea lions include coastal geomorphology, air and water temperatures, wind speeds, wave conditions, freshwater inflow, tides, currents, etc. A few recent studies have addressed how such factors may influence Steller 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 Steller sea lions are not static, but rather show variations on several time scales (Schumacher and Alexander 1999, Mueter *et al.* (2004), and 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, Polovina 2005, 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 (NRC 2006). The precise effects on Steller sea lions of all factors have been impossible to determine, but have likely been substantial, variable over time, and both top-down and bottom-up in nature. This human modification of marine mammals and fish stocks has occurred against a complicated pattern of climate and oceanographic variability, both natural and human-catalyzed as well as human use of coastal waters and nearby lands that has increased the potential for disturbance and pollution. The history of the Steller sea lion, their prey, and the ways in which both natural and anthropogenic forces have affected both is extremely complex.

Ecosystem models are available for the Aleutian Islands, Eastern Bering Sea and southeast Alaska. Other models are being developed in the GOA, (e.g., under the GLOBEC Program) (Weingartner *et al.* 2002). Some models can be used to gain insight into the combined effects that fishing, predation, ocean climate change, and interspecies interactions have had on Steller sea lions and their ecosystems as a whole (e.g., Trites *et al.* 1999, Guenette 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 that they are potentially affected by predation by killer whales (particularly when Steller 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. Ecosystem-level effects of fishing are evident (NRC 2006) as is climatic variability and human fostered global warming.

4.1.1 Environmental Variability

On a large spatial and temporal scale, the major mode of physical variability in the North Pacific has been identified as the Pacific Decadal Oscillation (PDO), which was described in the 1990s (Francis and Hare 1994, Mantua *et al.* 1997), but as of late 2003 may no longer be considered oscillatory (Bond *et al.* 2003).This may be a coupled ocean atmosphere phenomenon (some argue that it’s a true coupled system oscillation like ENSO (Latif and Barnett 1996), but the physical mechanisms are largely undescribed) which results in sea surface temperature (SST) and sea level pressure (SLP) anomalies and altered circulation in the entire North Pacific ocean. Clear "regime shifts" with fundamentally different SST and SLP patterns in space manifested in the atmosphere ocean system do appear on a decadal scale, in particular in 1946-47 and in 1976-77. The U.S. West Coast, eastern tropical Pacific, and the Gulf of Alaska shelf were warmer and the Central North Pacific was cooler post 1977 compared with the decades before (Zhang *et al.* 1997). Fall/winter coastal freshwater runoff influences the subsequent winter/spring temperature distribution in the coastal Gulf of Alaska, and new data have revealed anomalously low water temperatures in the northern Gulf in winter/spring of 2007 and 2008, and perhaps even 2009 (Janout *et al.* 2010), which may have had important influence on biological productivity in recent years.

An additional regime shift has been identified to have occurred in 1925 (Mantua *et al.* 1997). The decadal variability in the mid latitudes may be related to and definitely interacts with the better understood tropical atmosphere ocean variability that results in the El Nino Southern Oscillation (ENSO), which has an inter-annual timescale. However, it has been pointed out that there have been other reversals in the patterns of SST and SLP which are just as dramatic from a physical standpoint as those in the early 1940s and late 1970s, but which did not persist as long and therefore have not been termed regime shifts in retrospect (e.g.,1957-58, Zhang *et al.* 1997). "Regime shift" may therefore be interpreted as not a purely physically defined phenomenon, but one which requires an associated biological shift to be described in order to receive "official" recognition. Therefore, it is important to look at the type and spatial scale of physical forcing, as well as its persistence with respect to biological communities, because it seems feasible that species with certain life histories would respond to any multi-year shift in physical conditions while others would require at least decadal variability to respond, and the interaction between these species throughout the responses would also contribute to ecosystem dynamics.

Localized physical characteristics of the Gulf of Alaska continental shelf ecosystem are important to understanding the spatial and temporal variability in the biological communities as well, especially since many of its environmental parameters do not appear to display any decadal signal over the past 50 years (Stabeno *et al.* 2004). Circulation in the central GOA consists of the cyclonic Alaska Gyre that includes the Alaska Current and the Alaskan Stream, and the eastward-flowing North Pacific Current at the southern boundary of the GOA (Weingartner (2005). On the shelf, Weingartner (2005) notes three domains based on water mass structure and circulation characteristics: the Alaska Coastal Current on the inner shelf, the shelf break front along the outer shelf, and a mid-shelf region; boundaries separating these domains are dynamic. Freshwater runoff from coastal landmasses influence Gulf circulation, and the large runoff from Southeast Alaska combined with tidal and storm processes likely establish a flow field that links the marine shelf/slope systems of the northern and southern GOA (Weingartner *et al.* 2009). Perhaps the two most important broad circulation features in the coastal Gulf of Alaska are the Alaskan Stream and the Alaska Coastal Current (Reed and Schumacher 1987, Weingartner 2005).The Alaskan Stream runs relatively narrow and fast along the shelf break from the Northern GOA off Cape St. Elias towards the Aleutian Islands in the west. The position and strength of this current and its interaction with bottom topography is thought to alter the nutrient supply to phytoplankton on the shelf, along with seasonal wind driven cross shelf (Ekman) transport and entrainment due to freshwater runoff (Parsons 1987, Sambrotto and Lorenzen, 1987). Recent information indicates that the Alaska Stream is relatively steady within a season, but exhibits variability on interannual timescales (Hermann *et al.* 2002). The Alaska Coastal Current is a weaker flow in general, running parallel to the Alaska Stream closer to shore and through Shelikof Strait, but it is seasonally quite variable due to changes in freshwater runoff, which usually peaks in September-October (Stabeno *et al.* 2004). The Alaska Coastal Current transports large quantities of freshwater, and may be an important source of freshwater for the Bering Sea shelf and the Arctic Ocean (via transport through Bering Strait) (Weingartner *et al.* 2005).

Runoff also changes surface salinity and therefore water column stratification on the GOA shelf seasonally and locally, contributing to spatial and temporal variation in productivity. Vertical flow of water from surface to bottom (downwelling) and deep waters to surface (upwelling) can maintain or disrupt the flow of nutrients to the better lit surface waters where marine plants (phytoplankton) reside—therefore, downwelling and upwelling are important processes for biological production (Valiela 1995, Mann and Lazier 1991). Both seasonal downwelling and upwelling occur locally on the GOA continental shelf as a result of the interactions of these currents, runoff, and seasonally as well as locally varying winds (Stabeno *et al.* 2004). In general, downwelling dominates the system during the winter seasons, and (sporadic) upwelling predominates during the summer (Parsons 1987), although the duration and strength of summer upwelling varies locally with the wind field, so that some areas of the shelf may only experience upwelling regimes for 1 to 2 months of the year (Reed and Schumacher 1987). On the northern Gulf of Alaska shelf, upwelling not attributable to broad-scale physical forces may also be caused by localized wind stress curl (Hermann pers comm. 2005).

In addition, mesoscale (~200 km diameter) eddies form as a result of both bottom topography (e.g., the Sitka eddy) and the interaction of the Alaska Stream and Alaska Coastal Current (Reed and Schumacher 1987, Hermann *et al.* 2002, Ladd *et al.* 2005, Janout *et al.* 2009). These eddies are most common in spring and are often anticyclonic (Hermann *et al.* 2002), therefore producing localized downwelling where they occur. Thus the physical conditions on the Gulf of Alaska continental shelf are complex and variable at several temporal and spatial scales, so we might expect considerable spatial and temporal variation in the biological community due to physical forcing alone, in addition to variability imposed by biological dynamics. Yakutat eddies that propagate westward along the GOA slope may have higher concentrations of chlorophyll than adjacent waters (Janout *et al.* 2009).

Changes in the Gulf of Alaska continental shelf assemblage of benthic invertebrate and fish predators, including groundfish, invertebrates, and salmon, are likely attributed to climate regime shifts (Orensanz *et al.* 1998, Andersen and Piatt 1999, Mantua *et al.* 1997, Francis *et al.* 1998, Hare and Mantua 2000). The proposed mechanism for climate change forcing the observed change in productivity at higher trophic levels often involves “bottom up” forcing due to a change in phytoplankton and zooplankton production in response to changed physical condition such as mixed layer depth and temperature (Francis *et al.* 1998). There have been several studies which have modeled a lower trophic level response to changes in mixed layer depth and temperature associated with climate change: Polovina *et al.* (1995) used the 1985 Evans and Parslow model, and Haigh *et al.* (2001) used a more complex combination of the Evans and Parslow (1985) model and the Fasham (1995) model which included a detrital loop to evaluate the response of pelagic plankton communities to physical changes associated with decadal climate oscillations. However, none of these studies address the Gulf of Alaska shelf ecosystem specifically, where the observed changes in shrimp and groundfish productivity have occurred. The lack of a clear PDO signal in the physical conditions on the continental shelf (Stabeno *et al.* 2004) makes physically mediated bottom-up forcing arguments difficult to support by the mechanisms listed above for the open oceanic Gulf of Alaska. Further, no direct evidence of increased primary and secondary productivity within the shelf ecosystem has been identified in relation to the 1977 regime shift, in part because the time series are inadequate to address the question. However, knowing that large scale physical shifts have occurred, and finding that fishing mortality contributes relatively little to some groundfish stock’s total mortality and production, the regime shift paradigm finds more and more support through correlative analyses despite a modest supply of mechanistic connections.

4.1.2 Climate and Biological Regime Shifts

There is evidence for past climate regime shifts[12] and ecosystem responses to those shifts in the EBS and GOA (mid-1940s, 1977 and 1989), although evidence for a recent climate regime shift (1999) is unclear.

---

[12] Atmospheric scientists often refer to decadal-scale changes in the climate as climate regime shifts. This type of regime shift is different than a biological regime shift. There are observed decadal-scale changes in some biological components

Based on basin-wide North Pacific climate-ocean indices, there appear to have been major climate/ocean regime shifts in the mid-1940s and in 1976/77, and a minor climate regime shift in 1988/89 (Boldt 2005a, Hare and Mantua 2000, King 2005). For the earlier climate regime shifts, the mid-1940s and 1977, the pattern of sea surface temperature spatial variability implied a west-east dipole (sea surface temperature gradient trends west-east) (Boldt 2004, Bond *et al.* 2003). After 1989, the pattern of spatial variability was dominated by a second pattern of sea surface temperature variability, which implied a north-south dipole. At regional scales the responses to these basin-scale changes may not be as coherent (Boldt 2004). Given the variability in the indices since 1998, there is some uncertainty if there was a climate regime shift in the late 1990s (Rodionov *et al.* 2005).

It is important to note that regimes cannot be characterized by only two possible states (King 2005). It is currently not possible to reliably predict when a regime shift will occur. There are multiple physical and ecological processes underlying regime shifts that are currently not well understood. Different statistical models fitted to data provide divergent predictions of future conditions (King 2005).

In addition to decadal-scale climate regime shifts global temperatures are increasing and are expected to have profound impacts on arctic and sub-arctic ecosystems. See section on Global Climate Change.

Some investigators have argued that natural 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 Steller sea lions 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.

However, available evidence indicates that the magnitude of at least some components of the potential environmental change that may be facing the North Pacific and Bering Sea ecosystems as a result of anthropogenic release of carbon dioxide (CO_2)(e.g., global warming and ocean acidification), coupled with the restrictions to Steller sea lion alternative terrestrial habitat use due to human settlement, is probably unprecedented (see below), recognizing that this impact may require decades to become detectable.

4.1.2.1 Bering Sea

The Bering Sea has shown three multidecadal regimes in surface air temperatures (SAT) fluctuations: 1921-1939 (warm), 1940-1976 (cold), and 1977-2005 (warm) (Rodionov *et al.* 2005). More recently in 2006 - 2009, conditions in the Bering Sea have been cooler than the previous six years, which were very warm. The Bering Sea was subject to a change in the physical environment and an ecosystem response after 1977, influenced by shifts in Arctic atmospheric circulation in the early 1990s, and persistent warm conditions since 2000 (see Tables 2 and 3 in Boldt 2005b). A major transformation, or regime shift, of the Bering Sea occurred in atmospheric conditions around 1977, changing from a predominantly cold Arctic

---

of the North Pacific, and these are often referred to as biological regime shifts. Climate regime shifts may be observed in the physical conditions of the ocean and may affect the biology; however, the mechanisms by which the biology might be influenced are largely unknown. In this analysis we have attempted to distinguish between climate and biological regime shifts.

climate to a warmer subarctic maritime climate as part of the Pacific Decadal Oscillation (PDO) (see Tables 2 and 3 in Boldt 2005b). Weather data beginning in the 1910s and proxy data (e.g., tree rings) back to 1800 suggest that, except for a period in the 1930s, the Bering Sea was generally cool before 1977, with sufficient time for slow growing, long-lived, cold-adapted species to adjust. A specific Arctic influence on the Bering Sea began in 1989, as a shift in polar vortex winds (the Arctic Oscillation – AO) reinforced the warm Bering Sea conditions, especially promoting an earlier timing of spring meltback of sea ice. During 2000 - 2005, the climate patterns resulted in southwesterly wind anomalies and, hence, very warm atmospheric conditions in the Bering Sea (Stabeno and Overland 2007). The winds also resulted in a decreased ice extent, earlier ice melt, and warm ocean temperatures (Stabeno and Overland 2007). In 2006-2008, climate patterns have resulted in colder atmospheric conditions, cooler ocean temperatures and more sea ice that has persisted longer relative to 2000-2005 conditions (Stabeno and Overland 2007, Wang *et al.* 2007). A comprehensive report (National Academy of Sciences [NAS] 1996) indicated that a combination of fishing and the 1977 shift in physical forcing caused a major reorganization of the marine ecosystem on the Bering Sea shelf over the following decade. Surveys show an increase in the role of pollock as an energy source to the ecosystem. The NAS (1996) report hypothesizes that fishing of large whales increased the availability of planktonic prey, fishing on herring reduced competition, and fishing on flatfish reduced predation. The modeling study of Trites *et al.* (1999) noted that the increase in pollock biomass could not be explained solely by trophic interaction from these removals, and favored environmental shifts as an explanation. While the physical shift after 1976 was abrupt and pollock biomass increased rapidly, the ecosystem adjustment probably took a prolonged period as relative biomass shifted within the ecosystem.

Some responses to the climate shift in 1989 were observed in the Bering Sea. Northern rock sole recruitment was relatively high in the mid-1980s due to favorable onshore larval advection (Wilderbuer *et al.* 2002), but the AO shift to weaker winds after 1989 reduced these favorable conditions and recruitment was lower (Overland *et al.* 1999). In five of seven years during 2001-2007, transport was again onshore towards favorable nursery areas, with corresponding above average recruitment of northern rock sole (Wilderbuer and Ingraham 2007). Biodiversity measures (richness and evenness) of groundfish, excluding pollock, also appeared to shift in the late 1980s; indices decreased throughout the 1980s and were stable in the 1990s (Hoff 2003). Jellyfish, which share a common trophic level with juvenile pollock and herring, may have played a role in the ecosystem adjustment as their biomass increased exponentially beginning in the late 1980s, but decreased to lower levels in 2001-2007 (Tables 2 and 3 in Boldt 2007).

As global temperatures increase, impacts of those temperatures will likely occur in the Bering Sea (see discussion of global warming below). Warm conditions tend to favor pelagic over benthic components of the ecosystem (Hunt *et al.* 2002, Palmer 2003). Cold water species, i.e. Greenland turbot, Arctic cod, snow crab, are no longer found in abundance in the southeast Bering Sea, and the range of Pacific walrus is moving northward. While it is difficult to show direct causality, the timing of the reduction in some marine mammal abundance levels suggests it is due to some loss of their traditional Arctic habitat. Although physical conditions appear mostly stable over the last decade, the warmest water column temperatures occurred in 2001 to 2005 on the southeast Bering Sea shelf, despite considerable year-to-year variability in the AO and PDO.

The overall climate change occurring in the Arctic, as indicated by warmer atmospheric and oceanic temperatures and loss of 15% of sea ice and tundra area over the previous two decades, is hypothesized to make the Bering Sea less sensitive to the intrinsic climate variability of the North Pacific. Indeed, when the waters off the west coast of the continental U.S. shifted to cooler conditions after 1998, the subarctic did not change (Victoria pattern), in contrast to three earlier PDO shifts in the 20th Century.

4.1.2.2 Aleutian Islands

Climatic conditions have varied between the east and west Aleutian Islands. Around 170 °W: to the west there was a long term cooling trend in winter between 1956 and 2002, while to the east conditions change with the PDO (Rodionov *et al.* 2005). This location is also near the first major pass between the Pacific and Bering Seas for currents coming from the east. Biological conditions in the Aleutian Islands have changed since the 1980s, and it is too soon to discern if there was a change associated with the 1998 climate regime shift. Pollock and Atka mackerel productivity do not appear to vary on a decadal-scale. Pacific ocean perch population dynamics vary on a decadal-scale. For example, Pacific ocean perch survival changed at the approximate times of climate regime shifts, 1975 and 1989. However, there is not enough information on the early life history of Pacific ocean perch to define a mechanism for the observed variations.

4.1.2.3 Gulf of Alaska

Evidence suggests there were climate regime shifts in 1977 and 1989 in the North Pacific. Ecosystem responses to these climate shifts in the Gulf of Alaska (GOA) were strong after 1977, but weaker after 1989. Initially it was hypothesized that there was also a climate regime shift in 1998/99. However, evidence for this shift is unclear. Variation in the strength of ecosystem responses to climate shifts may be due to the geographical location of the GOA in relation to the spatial pattern of climate variability in the North Pacific. Prior to 1989, climate forcing varied in an east-west pattern, and the GOA was exposed to extremes in this forcing. After 1989, climate forcing varied in a north-south pattern, with the GOA as a transition zone between the extremes in this forcing.

There were both physical and biological responses to climate regime shifts in the GOA. However, the primary reorganization of the GOA ecosystem occurred after the 1977 climate shift. After 1977, the Aleutian Low intensified. This resulted in a stronger Alaska current, warmer water temperatures, increased coastal rain, and, therefore, increased water column stability. The optimal stability window hypothesis suggests that water column stability is the limiting factor for primary production in the GOA (Gargett 1997). A doubling of zooplankton biomass between the 1950s-1960s and the 1980s indicates production was positively affected after the 1977 climate regime shift (Brodeur and Ware 1992). Recruitment and survival of salmon and demersal fish species also improved after 1977 (Tables 4 and 5 in Boldt 2005b). Catches of Pacific salmon in Alaska increased, recruitment of rockfish (Pacific ocean perch) increased, and flatfish (arrowtooth flounder, halibut, and flathead sole) recruitment and biomass increased. Individual groundfish stock survival and recruitment indices showed inconsistent responses to the 1977 shift. However, combined standardized indices of groundfish survival and recruitment indicated that overall groundfish productivity increased after the 1977 climate shift (Mueter *et al.* 2007). There are indications that shrimp and forage fish, such as capelin, were negatively affected after 1977, as survey catches declined dramatically in the early 1980s (Anderson 2003; Tables 2 and 3 in Boldt 2005b). The decline in marine mammal and seabird populations, observed after 1977, may have been related to the change in forage fish availability (Piatt and Anderson 1996).

After 1989, water temperatures were cooler and more variable in the coastal GOA, suggesting production may have been lower and more variable. After 1989, British Columbia (BC) salmon catches and survival were low and Queen Charlotte Island (northern BC) herring declined. Salmon catches in Alaska, however, remained high. Groundfish recruitment and survival, as measured by combined standardized indices, showed inconsistent responses to the 1989-climate shift (Mueter *et al.* 2007). Groundfish biomass trends that began in the early 1980s continued, with increases in flatfish biomass. By the late 1980s arrowtooth flounder, rather than walleye pollock, were the dominant groundfish. Large groundfish biomass estimates resulted in negative recruit per spawning biomass anomalies of demersal fish.

Initially, there was some indication that the GOA ecosystem may have weakly responded to the suspected 1998 climate regime shift. Increased storm intensity from 1999 to 2001 resulted in a deeper mixed layer depth in the central GOA, and coastal temperatures were average or slightly below average. After 1998, coho survival increased in southern BC, shrimp catches increased in the northern GOA (but have since declined again in 2003), and the 1999 year class of both walleye pollock and Pacific cod was strong in the northern GOA.

4.1.3 Changes in Biological Productivity

Conners *et al.* (2002) present an analysis of bottom trawl survey data from 1963-2000. Three index areas with good survey coverage through the full time span were selected; one area includes Steller sea lion critical habitat north of Unimak Island. A robust index of median catch per unit effort (CPUE) was used as an indicator of regional groundfish abundance. Time series for total catch and for several major groundfish groups showed substantial increases in the early- to mid-1980s in all three index areas (Figure 4.1). Time series for walleye pollock, Pacific cod, rock sole, flathead sole, cartilaginous fishes (skates) and benthic invertebrates showed substantial increasing trends. The timing of change in trawl CPUE is consistent with effects of the strong regime shift observed in climate indices in 1976-1977. The similarity in trends both across the region and across both commercial and unexploited groups suggests that a widespread reorganization of benthic and demersal food webs may have taken place. There is little evidence of similar biological responses to smaller climate shifts in the 1990s. These results are also consistent with recently documented shifts in ecosystem dynamics resulting from changes in ice cover and thermal structure in the eastern Bering Sea. This analysis indicates that there was a much higher biomass of groundfish at all three sites during 1980-2000 than in 1960-1980. These results provide strong evidence against the hypothesis that the decline of Steller sea lions was due entirely to a decrease in overall productivity of the eastern Bering Sea (NMFS 2006b).

The NMFS bottom trawl survey does not effectively sample pelagic forage fishes such as capelin, herring, and eulachon, which are important prey fish for sea lions. Data from inshore surveys in the Gulf of Alaska (Anderson and Piatt 1999) suggest that abundance of these species declined dramatically following the 1976-77 regime shift. There are no data available on whether a similar decline occurred in the Bering Sea. It is entirely feasible that the reorganization in food webs indicated in the retrospective study also affected pelagic food webs or the balance of demersal/pelagic production. There does not appear to have been a substantial decline in overall productivity in the EBS, but there could well have been a substantial shift in how production is distributed through the food web.

From 1954 to 1998, Eastern Bering Sea (EBS) summer zooplankton biomass data, collected by the Hokkaido University research vessel T/S Oshoru Maru and re-analyzed by Hunt *et al.* (2002) and Napp *et al.* (2002), showed no discernable trends in any of the four EBS geographic domains (Napp *et al.* 2002; Figure 41 in Boldt 2005). The updated time series, however, depicts a strong decrease in biomass during 2000-2004. What is remarkable is that the decrease occurred in all four domains (see Figure 41 in Napp and Shiga 2005). Part of the decrease in biomass over the middle shelf may be due to recent decreases in the abundance of *Calanus marshallae*, the only “large” copepod found in that area (Napp, in prep.). It is not clear what might be the cause of declines in other regions.

Annual surplus production (ASP) indices, the sum of new growth and recruitment minus deaths from natural mortality, suggest high variability in groundfish production in the EBS and a decrease in production between 1978 and 2005 (Figure 110 in Mueter 2007). Production in the GOA was much lower on average, less variable, and decreased slightly from 1978 to 2005 (Mueter 2007). Because trends in ASP indices are largely driven by variability in walleye pollock in the EBS, the results suggest a strong, significant decrease in aggregate surplus production of all non-pollock species from 1978 – 2005 (Figure 112 in Mueter 2007). The declines in production may be a density-dependent response to observed increases in biomass because theory suggests that surplus production will decrease as biomass increases above B_{MSY}, which has been the case for a number of flatfish species (e.g., rock sole, flathead sole) and rockfish species (Pacific ocean perch, northern rockfish; Mueter 2007). This may be indicative of a "top-down" phase in a larger ecosystem cycle (e.g., the Oscillating Control Hypothesis described for the eastern Bering Sea in Hunt *et al.* (2002), or the shift from bottom-up to top-down control described in Bailey (2000). As shown in Figure 4.2, while the overall biomass of the main groundfish in the Bering Sea has increased since the late 1970s, the populations have also aged and grown larger; this trend is particularly pronounced in the 1990s. This aging population would be expected to have a decreased ratio of production to energy consumption, although this does not take into account possible importance of contributions of high natality individuals in the larger sizes.

4.1.4 Steller Sea Lion Prey Response to Climate and Regime Changes

4.1.4.1 Recruitment Response to Regime Changes

Eastern Bering Sea Pollock

To evaluate EBS pollock recruitment relative to a suite of putative regimes, sets of years were included within the integrated stock assessment model to provide estimates of uncertainty. For the period 1963-1976 the average age-1 recruitment appears to be substantively lower than that for all other periods (Figure 4.3). The coefficients of variation for these estimates were relatively low (except for the cohorts from 1999-2005; Table 4.1). In conclusion, there appears to be evidence of higher recruitment post 1976 compared to the earlier period. Evidence of significant differences from subsequent putative regimes is apparently lacking based on an analysis of data through 2004. A recent assessment (Ianelli *et al.* 2007) showed the time period between 2004-2006 to have extremely low recruitment, with 2004 and 2005 showing the lowest assessed recruitment since 1965, which has given rise to concerns that we may be entering a period of low recruitment associated with a warmer, ice-free southeastern Bering Sea. However, preliminary data (Ianelli *et al.* 2007) shows that 2007 recruitment was closer to 1990s levels. The extremely warm, ice-free years in 2002-2005 may have contributed to low zooplankton biomass and low pollock survival during 2004-2005. Cold years occurred from 2006-2007 with extensive ice cover in the Bering Sea, although ice remained at historic lows in the Arctic in general. Determining the future of the Bering Sea pollock stock will require the close examination of climate models to predict whether 2002-2005 conditions were anomalously warm compared to future expectations or whether those years represent an expected future state of the climate with correspondingly poor conditions for pollock.

Gulf of Alaska pollock

Recruitment of pollock in the Gulf of Alaska is highly variable on multiple time scales (Dorn *et al.* 2005). On an interannual time scale, recruitment of Gulf of Alaska pollock is more variable (CV = 1.07) than Eastern Bering Sea pollock (CV = 0.64). Among North Pacific groundfish stocks with age-structured assessments, GOA pollock ranks third in recruitment variability after sablefish and Pacific Ocean perch (http://www.afsc.noaa.gov/refm/stocks/estimates.htm). Unlike sablefish and Pacific Ocean perch, pollock have a short generation time (<10 yrs), so that large year classes do not persist in the population long enough to have a buffering effect on population variability. High recruitment variability implies a large environmental component to forcing, since biotic factors such as density dependence or predation tend to change more gradually. On decadal time scales, there is also variability in pollock recruitment. Mean recruitment increased by approximately five times from the 1960s to the 1970s, then declined in the 1980s, declined further in the 1990s, but stabilized in the early 2000s (Table 4.2).

In the Gulf of Alaska, climatic regime shifts occurred in 1977 and in 1989 based on persistent changes in PDO. As noted earlier, the term climatic regime shift refers to persistent changes in atmospheric conditions and the physical condition of the ocean, not to the biological response to those changes. Although correlation analyses (or other related approaches) can be used to relate climate forcing to biological response, often the mechanistic link must be hypothesized because environmental data are not available at appropriate temporal and spatial scales (Baumann 1998).

There are several hypotheses about how the 1977 regime shift might have affected pollock recruitment in the GOA. First, the shift from cool temperatures to warm temperatures may have favored better larval pollock survival through one or more indirect mechanisms (Bailey 2000). A second hypothesis is that the spring zooplankton bloom shifted earlier in the year, favoring winter spawners such as pollock (Andersen and Piatt 1999). A final hypothesis is the optimal stability “window” (Gargett 1997), which hypothesizes that changes in strength of the Aleutian Low associated with the 1977 regime shift affected water column stability, resulting in an increase in primary production in coastal areas of the Gulf of Alaska. While all of these hypotheses seem reasonable, oceanographic time series in the Gulf of Alaska are too short to establish observational proof. Further, Stabeno *et al.* (2004) did not find a strong PDO signal in physical conditions of coastal waters of the Gulf of Alaska, raising questions about the importance of basin-scale climatic patterns in physical forcing at spatial and temporal scales important to pollock recruitment.

A more important question is whether the pattern of pollock recruitment changed after the regime shift. Although pollock recruitment shows a clear pattern of increase and decline over the period 1959-2004, there are no obvious changes occurring immediately after the 1977 or the 1989 regime shift (Figure 4.4). The 1970s stand out as a decade of very strong recruitment for GOA pollock, but five out of the eight strong year classes (> 1.0 billion age-2 recruits) in the 1970s occur prior to 1977. In the twenty-seven years since 1980, strong year classes have recruited to the population every six years on average. However, no years with more than one billion age 2 recruits have been observed since 1991, and 2003-2005 were the second, third, and fourth lowest recruitments observed over the 1961-2007 time series. Average year classes were observed in 2006 and 2007, and are predicted for 2008 and 2009 (Dorn *et al.* 2007). Hollowed *et al.* (2001) found that GOA pollock exhibited higher incidence of strong recruitment during years when El Niño conditions propagated into the Gulf of Alaska, but did not find a relationship between the 1977 phase change in PDO and pollock recruitment. Support for the hypothesis that the climatic regime shift in 1977 resulted in improved conditions for pollock recruitment is not compelling.

Pacific cod

In the EBS Pacific cod model (Thompson *et al.* 2007), recruitment estimates are obtained for each year class from 1974 through 2006, and in the GOA Pacific cod model (Thompson *et al.* 2006), recruitment estimates are obtained for each year class from 1964 through 2004. In both the EBS and GOA models, the effects of the 1976-1977 regime shift are modeled explicitly by estimating separate median recruitment levels for the two portions of the time series.

The EBS Pacific cod model estimates median numbers at age 0 for the pre-1977 and post-1976 regimes at values of 149 million fish and 800 million fish, respectively (i.e., the pre-1977 median is 81% lower than the post-1976 median). Of the 13 pre-1977 cohorts, none of the point estimates exceed the post-1976 median, but the 95% confidence interval overlaps the post-1976 median in 2 cases (1974 and 1976). Of the 30 post-1976 cohorts, none of the point estimates fall below the pre-1977 median, and the 95% confidence interval overlaps the pre-1977 median in only 2 cases (1981 and 1987).

Given the structure of the EBS Pacific cod model and the existing data, there is a low likelihood that the pre-1977 median recruitment was as high as the post-1976 median. That is, there is a 95% chance that the pre-1977 median was at least 76% lower than the post-1976 median, and a 99% chance that the pre-1977 median was at least 73% lower than the post-1977 median.

The GOA Pacific cod assessment estimates median numbers at age 0 for the 1964-1976 and 1977-2004 time periods at values of 87 million fish and 273 million fish, respectively (i.e., the pre-1977 median is 68% lower than the post-1976 median). Of the 13 pre-1977 cohorts, none of the 95% confidence intervals overlap the post-1976 median, and of the 28 post-1976 cohorts, none of the 95% confidence intervals overlap the post-1976 median.

The software used to create the GOA Pacific cod model differed from that used to create the EBS Pacific cod model. One of the differences pertains to the manner in which the pre-1977 median recruitment was estimated. Unfortunately, this difference means that the type of statistical comparison between pre-1977 and post-1976 medians described above for the EBS model cannot be conducted for the GOA model. Moreover, a rigorous statistical evaluation of the potential existence of regime shifts other than the 1977 shift is not possible in either the EBS model or the GOA model. As a first approximation, a simple, "difference between two means" test can be used to evaluate the existence of the 1977 regime shift in the GOA. The same test can also be applied to hypothesized 1988-1989 and 1998-1999 regime shifts in both the EBS and GOA. In all cases, however, it should be emphasized that some assumptions inherent in the test are being violated (e.g., the variances associated with the individual estimated recruitments are not equal). The results of these tests are shown below:

EBS
The 1988-1989 shift is insignificant.
The 1998-1999 shift is insignificant.

GOA
The 1976-1977 shift is significant.
The 1988-1989 shift is insignificant.
The 1998-1999 shift is insignificant.

On the basis of the above, it appears that the difference in median/mean recruitment before and after the 1976-1977 regime shift is statistically significant at any reasonable level of significance (e.g., Type 1 error level of 1% or 5%) in both the EBS and GOA, but the differences in mean recruitment before and after the hypothesized 1988-1989 or 1998-1999 regime shifts are not. These results are similar to those obtained by Boldt and Conners (2004), with the exception that the assessment results available to Boldt and Conners did not include estimates of pre-1977 cohorts.

Atka mackerel

It is unclear to what extent if any, that recruitment of Atka mackerel follows expectations of good vs. bad environmental conditions based on regime shift theory. Until we understand the mechanisms, processes and environmental linkages that contribute to successful recruitment, we cannot know how recruitment is related to regime shift theory. The recruitment history of Atka mackerel is characterized by variable but fairly good recruitment throughout the time series of stock assessment estimates. The strong 1999 year class is most notable followed by the 1988, 1977 and 2001 year classes (Lowe *et al.* 2007). The most recent stock assessment estimates above average (greater than 20% of the mean) recruitment from the 1977, 1986, 1988, 1992, 1995, 1998, 1999, 2000, and 2001 year classes (Lowe *et al.* 2007). Based on basin-wide North Pacific climate indices, there appears to have been a major regime shift in 1976/77, and a minor regime shift in 1988/89 (Boldt 2005, Hare and Mantua 2000, King 2005). There is some uncertainty if there was a regime shift in 1999 given the variability in environmental indices since 1998 (Rodionov *et al.* 2005). These hypothesized regime shifts coincide with the three strongest Atka mackerel year classes, however, it should noted that the mechanisms which produce successful recruitment are unknown.

In an analysis by Boldt *et al.* (2004), climate regime-scale variability in recruit per spawner time series was not detected in groundfish (pollock, cod, and Aleutian Islands Atka mackerel). The conclusion from this analysis was that the survival of groundfish does not appear to be related to decadal-scale climate variability as defined by the hypothesized 1976/77, 1988/89, or 1998 years of regime shifts. In a more recent analysis, it was shown that individual groundfish stock survival and recruitment indices showed inconsistent responses to the 1977 shift; however, standardized indices of survival and recruitment for all groundfish combined indicated that overall groundfish productivity increased after the 1977 climate shift (Mueter *et al.* 2007).

4.1.4.2 Response of Major Pollock Spawning Aggregations

A comparative approach was used within the stock assessment to evaluate whether fishing impacts or other factors, such as environmental changes, were likely to have caused observed patterns of recruitment and biomass. Over the last 12 years, harvest rates in the three areas show good contrast (Ianelli *et al.* 2005a, 2005b, Dorn *et al.* 2005). The Bogoslof area has barely been fished at all during this period, but has shown the greatest percent decline (Figure 4.5). The continued decline in survey biomass after major fishery impacts ceased in 1991 is contrary to what would be expected if fishing within the Bogoslof area was the primary factor controlling stock abundance. Harvest rates have been similar between the GOA and EBS, and are low compared to fisheries for other gadids (Brander 2003). Survey biomass has been stable to slightly increasing in the EBS, but has declined in Shelikof Strait. The differing survey trends under similar fishing impacts is also contrary to what would be expected if fishing were the primary factor controlling stock abundance in the EBS and GOA.

An important question is why pollock abundance has declined in the Gulf of Alaska if pollock have been consistently harvested at less than F_{MSY}. This question was explored by “replaying” the population dynamics without fishing. The simplest approach is to replay the population dynamics with the same recruitment time series. This approach does not take into account the potential impact of fishing on recruitment due to changes in stock biomass (potentially fewer recruits at low stock size, or more cannibalism on pre-recruits at high stock size). To evaluate the potential impact of higher spawning biomass, we also replayed the stock dynamics with a rescaled recruitment time series based on a stock-recruit relationship (NMFS 2006b).

Results, based on a single species perspective, showed that a significant decline of pollock abundance from the peak in the1980s would have occurred even without fishing (Figure 4.6). This suggests that other factors such as environmental variability may be a more significant driver for the stock abundance. Another explanation is that Gulf of Alaska pollock are extremely unproductive; however analysis of available stock-recruit data suggests that this alternative has relatively low probability (Dorn *et al.* 2003). Other factors include ecosystem dynamics which were considered above.

The relationship between both Bogoslof and Shelikof spawning aggregations and larger regional populations is not well established. There is no evidence that these aggregations are genetically distinct populations, and some exchange likely takes place between these aggregations and pollock populations in other parts of the eastern Bering Sea and the Gulf of Alaska. The extent of exchange is unknown. One possibility for observed pattern of decline in the Bogoslof area and in Shelikof Strait is a change in spatial patterns of spawning. Winter surveys of spawning aggregations in other parts of the Gulf of Alaska provide evidence a significant amount of pollock spawn outside of Shelikof Strait (Dorn *et al.* 2003, Dorn *et al.* 2005). Attempts have been made to identify environmental factors influencing the spawning migration into Shelikof Strait, but so far models with environmental variables have poor predictive power (Boldt *et al.* 2002). However, it is also possible that fishing may have impacted the Shelikof Strait spawning aggregation, but this is not predicted by the single species models which generally assume no negative impacts of removing large pre-spawning fish.

Based on the assessment results, recruitment variability is highest in the Bogoslof area (CV = 1.96), high in the Gulf of Alaska (CV = 1.07), and relatively stable in the eastern Bering Sea (CV = 0.64). The recruitment time series for the Bogoslof area is notable for an exceptionally strong 1978 year class that was still the most abundant year class at age 14 in the 1992 survey. High recruitment variability suggests a strong environmental component to forcing and a highly dynamic environment. The range of recruitment variability for pollock in the Gulf of Alaska, the Bogoslof area and the eastern Bering Sea is consistent with the observation that the Gulf of Alaska and Aleutian Island ecosystems are more open, dynamic systems than the eastern Bering Sea shelf.

Pollock have a relatively short generation time (<10 yrs[13]), so that large year classes do not persist in the population long enough to have a buffering effect on population variability. Therefore, the typical pattern of biomass variability for pollock stocks with high recruitment variability will be sharp increases due to strong recruitment, followed by periods of gradual decline until the next strong year class recruits to the population. Pollock in the Bogoslof area and in the Gulf of Alaska are more likely to show this pattern than other groundfish stocks in the North Pacific due to the combination of a short generation time and high recruitment variability.

A simulation model was used to evaluate stock biomass variability under the current harvest policy for Gulf of Alaska pollock. Simulations were conducted using a stock recruitment relationship such that $F_{MSY} = F_{35\%}$ and modeled recruitment variability and autocorrelation based on historical patterns. A graph of 1000-year subsample of a simulation run demonstrates that even for a harvest policy appropriate to stock productivity, variability around mean stock size will be large (Figure 4.7). A typical pattern of variability consists of a sharp increase in stock size due to the recruitment of one or more strong year classes, followed by a sustained decline. The observed decline in pollock abundance in the Gulf of Alaska does not appear unusual in the 1000-year subsample. These patterns can be obtained with a stationary stock-recruit relationship without invoking “regime shifts” or decadal shifts in stock productivity.

4.1.4.3 Response of Aleutian Islands Pollock to Environmental Changes and Fishing Prohibitions

Fishing for Aleutians Island pollock was prohibited in 1999 under the Steller sea lion conservation measures and was allowed again outside of critical habitat in 2005. The long-term biomass trend for AI pollock had been decreasing until about 1999. Given the extensive closure area for this stock, it provides a unique opportunity to evaluate the effects of prohibiting fishing and observing how environmental conditions may naturally impact recruitment in the absence of fishing pressure.

Although the 2000 and 2002 summer bottom trawl surveys purport an increase in the pollock biomass in the Aleutian Islands area from the 1997 estimate (Table 4.3), these surveys have CV’s equal to 28% and 38%, respectively) and unreliable indices of abundance given the variability in vertical distribution of Alaska pollock (Barbeaux *et al.* 2007). The 2004 summer bottom trawl survey shows a decline in abundance from 175,000 t in 2002 to 130,000 t in 2004, but the variance in the 2004 estimate (CV = 78%) is substantially higher than any previous estimate. These data are therefore insufficient to reliably discern abundance trends post-1999.

If the bottom trawl survey pollock abundance estimates were accurate and precise (q = 1.0), the catch levels estimated for the 1990’s fishery (Table 4.4) would be unsustainable since under this assumption, the catch to biomass ratio would be between 28% and 75%. Indications are that the summer bottom trawl

---

[13] Generation time is defined in this opinion as the average age of all reproductive females in the population; definition is from the Revised Recovery Plan and is based on SSL life tables in Calkins and Pitcher (1982) and York (1984). Average age of first reproduction is about 5 years of age for sea lions.

survey assesses only one component of the pollock stock in the Aleutian Islands and that this component may not include that taken by the fishery during winter. Also, the large catches during the early and mid-1990s primarily consisted of the 1978 year class and later the 1989 year class (Table 4.5). The 1978 year class was only surpassed in catch weight by the 1989 year class in the 1995 fishery, but still remained a significant proportion of the catch through 1998 (7%). Pollock recruitment processes that led to the 1978 year-class event throughout the Aleutian Islands and eastern Bering Sea are poorly understood. The high variability in Aleutian Islands pollock recruitment is likely due to environmental conditions. The degree to which Aleutian Islands pollock abundance depends on movement from the EBS is also unknown. While it is possible that the EBS fishery causes some interception of potential Aleutian Islands recruitment, the exploitation rates within the EBS appear to be at sustainable levels (single species). It may be that the Aleutian Islands pollock stock depends on extremely favorable recruitment conditions such as that observed from 1978 and 1989.

In March and April 2006, 2007 and 2008 cooperative acoustics research surveys were conducted in the Aleutian Islands to assess the abundance of pollock in the region during spawning. Results show that, in the area surveyed, pollock biomass was lower than that available during the 1990s. Importantly, this study provides direct observation of localized abundance levels that have long been considered important for Steller sea lion conservation concerns. Current stock assessment models only deal with highly aggregated data and provide highly aggregated (and uncertain) results. In addition, the observed distribution and behavior under fishing suggests that a high catch-per-unit-effort could be achieved in this area even at low levels of abundance (NMFS 2006b). The pollock were concentrated on the shelf break and became more concentrated as the experimental fishery progressed in 2006. Such “hyperstability” in catch rates highlights the potential risk of interpreting commercial data (e.g., CPUE) alone. For example, depletion experiments may not be valid if the stock shows this stability in catch rates even though the actual biomass is small and being depleted, because the experiment would come to a completely opposite conclusion. These results are preliminary and further analysis is required before they can be considered conclusive. However, its does highlight the sensitivity of the Aleutian Islands to fishing pressure and the potential for long term impacts on the stock which may not be detectable from the single species perspective or through depletion experiments.

4.1.4.4 Changes in the Distribution of Important Prey

To evaluate changes in fish distribution for the eastern Bering sea (pollock and Pacific cod) station-specific CPUE data from NMFS summer bottom-trawl surveys were used. Average CPUE was computed by 1° longitude by 0.5° latitude quadrangles and contoured to evaluate annual changes. Results indicate that interannual spatial variability is high for both pollock and Pacific cod, but with Pacific cod having a somewhat broader but lower density overall than pollock (Figures 4.8-13). Pollock summer bottom-trawl concentrations within Steller sea lion critical habitat area show considerable variability with some years having relatively low densities (e.g., 1982, 1988, 1991, 1997-1999, 2006) and other years having high concentrations in critical habitat (Figures 4.8-10). In recent years, (since 2000) moderate densities of pollock have been consistently present in critical habitat (with the exception of 2006). For Pacific cod, the relative density in critical habitat was higher in the 1980s compared to later years (Figures 4.11-4.13).

To further summarize these densities relative to changes in fish distribution, central concentrations of pollock and Pacific cod were computed and mapped by year to ascertain if certain groups of years were different than others. This involved computing the CPUE-weight average location of pollock and Pacific cod. Results show that the centers of pollock distributions by year were more variable over years (spreading northwest to southeast) than those for Pacific cod (Figure 4.14). The 1993-1995 surveys show a tendency for pollock to be most dense towards the southeastern part while other years are mixed. This can be attributed to the relatively high abundance of the 1989 year class in the south-eastern regions.

Pacific cod shows a marked north-northwest shift in distribution during the period 2000-2005. This is consistent with the lower densities of Pacific cod observed in the southeastern regions during these years. The summer NMFS bottom-trawl survey data were also evaluated for CPUE patterns within and outside of Steller sea lion critical habitat areas. Mean values of CPUE were computed inside the critical habitat and compared to the mean CPUE outside of this region over time. Figure 4.15 shows a high degree of inter-annual variability of the relative CPUE inside Steller sea lion critical habitat compared to outside, especially for pollock (top panel).

The distribution of winter spawning pollock have shifted in the Bogoslof Island management district from near Bogoslof Island during the late 1980s and early 1990s to closer to Samalga pass and north-east of Umnak Island (Figure 4.16). Such relatively fine-scale shifts in spatial distribution suggests that environmental conditions may have changed to favor spawning habitat closer to the Aleutian Islands chain than in the past.

Shifts in distribution of pollock in the Aleutian Islands appear to coincide mostly with a connection to the "Aleutian Basin" stock (as indexed by the Bogoslof region) and that of the EBS shelf region. Specifically, the highest recorded historical catches occurred during winter months in the eastern most part. These pollock were thought to be comprised of mainly the 1978 and 1989 year classes and currently pollock abundance in the eastern region of the Aleutian Islands remains low, despite limited directed pollock fishing in this region since 1999. The changes in distribution of Pacific cod in the Aleutian Islands rely on summer biennial bottom trawl surveys and these show a high degree of variability.

Atka mackerel in the Aleutian Islands region also shows considerable variability over time based on summer bottom trawl surveys (Figures 4.17 and 4.18). As with many groundfish species, Atka mackerel is particularly prone to having high variance estimates, especially when broken down to finer management areas. This is due to the patchy distribution of this species. Nonetheless, trends for Atka mackerel in general suggest increased abundances throughout their range, particularly in the eastern and western management areas. This abundance pattern extends in recent years into the GOA where directed fishing for Atka mackerel is prohibited.

Pacific cod in the GOA also show a high degree of CPUE variability (Figure 4.19). Pacific cod in general are thought to be relatively mobile groundfish species based on the tagging studies of Shimada and Kimura (1994). Results from evaluating GOA survey patterns are consistent with a mobile species. Apparent long-term shifts in GOA Pacific cod abundances from summer survey data are difficult to ascertain.

GOA pollock spatial distributions have been evaluated regularly, particularly for patterns of spawning concentrations. For example, the annual winter Shelikof Strait surveys of spawning pollock have traditionally been considered to represent the majority of the GOA stock. Modeling efforts of the population have shown that about 67% of the pollock spawning occurs in the Shelikof region (on average) and about 20% in the Shumagin Islands region with the balance in other locations. There are trends in these data that suggest the Shelikof Strait spawning contribution has been below average for a number of years (2002-2005; Dorn *et al.* 2005). Relative pollock biomass in the GOA during the summer shows variability among regions (Figure 4.20). In some years the majority of the biomass appears in the Kodiak Island region while in other years, the Shumagin region appears to have the highest levels.

4.1.5 Changes in the Carrying Capacity for Western DPS Steller Sea Lion

Populations can experience abrupt and dramatic declines because of dramatic reductions in environmental carrying capacity (Odum 1971). 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. Large, natural variability often masks the effects of human activity on natural ecosystems and populations. Because of the complex relationships between wild populations, their physical environment, and their ecological relationships, it is extremely difficult to assign a population's decline to a single cause.

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). Changes in the North Pacific fish community structure stemming from the regime shift in 1976-77 may have been substantial enough to result in a dominance of pollock and other gadids. However, it is unclear whether this environmental variability and the associated diet shifts were outside the limits of natural variability in the history of Steller sea lions in the North Pacific and were principal factors in their population decline. Gadids have been and are likely to continue to be a principal component of the diet of sea lions (Table 4.6). The 1976-77 regime shift likely affected species differently (Section 4.1.4.1). In an analysis by Boldt *et al.* (2004), climate regime-scale variability in recruit per spawner time series was not detected for pollock, Pacific cod, and Atka mackerel. It is likely that the groundfish community changes, with some regimes or time periods more/less favorable for sea lions. Another shift may have occurred in 1989 and 1998 (Hare and Mantua 2000, Bond *et al.* 2003).

In addition to the environmental changes, the removal of prey by many fisheries increased markedly in the 1980s and could have contributed to changes in carrying capacity, possibly in non-linear and unpredictable ways (Calkins 1998, Goodman *et al.* 2002, NRC 2003, 1996). As these groundfish fisheries expanded, numerous investigators expressed concern about the effects of the expanded fisheries on populations of pinnipeds and seabirds in the North Pacific Ocean (Alverson 1991, Ashwell-Erickson and Elsner 1981). Several populations of seabirds and pinnipeds declined from the early to mid-1980s. As a result, scientists and fishery managers began to debate the relative roles of the regime shift and the groundfish fisheries on trophic relationships in the BSAI and GOA (Lowry *et al.* 1982, Alaska Sea Grant 1993, NRC 2003).

4.1.6 Global Climate Change and Ocean Acidification

4.1.6.1 Global Climate Change

There is growing concern about global climate change. Global air and ocean temperatures during this century and before are warming (IPCC 2007a, 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).

Key relevant findings from the most recent report of the Intergovernmental Panel on Climate Change (IPCC) (2007b) include (bold text in original):

> **"Warming of the climate system is unequivocal, as is now evident from observations of increases in global average air and ocean temperatures, widespread melting of snow and ice and rising global average sea level**…"
>
> '…Eleven of the last twelve years (1995-2006) rank among the twelve warmest years in the instrumental record of global surface temperature (since 1850). The 100-year linear trend (1906-2005) of 0.74 [0.56 to 0.92]°C1 is larger than the corresponding trend of 0.6 [0.4 to 0.8]°C (1901-

> 2000) given in the Third Assessment Report (TAR) (Figure SPM.1). The temperature increase is widespread over the globe and is greater at higher northern latitudes."
>
> "Observational evidence from all continents and most oceans shows that many natural systems are being affected by regional climate changes, particularly temperature increases."
>
> Anthropogenic warming over the last three decades has likely had a discernible influence at the global scale on observed changes in many physical and biological systems.

These impacts, and others, are projected to accelerate during this century. Solomon *et al.* (2009) concluded that:

> "…the climate change that takes place due to increases in carbon dioxide concentration is largely irreversible for 1,000 years after emissions stop…Among illustrative irreversible impacts that should be expected if atmospheric carbon dioxide concentrations increase from current levels near 385 parts per million by volume (ppmv) to a peak of 450–600 ppmv over the coming century are…inexorable sea level rise. Thermal expansion of the warming ocean provides a conservative lower limit to irreversible global average sea level rise of at least 0.4–1.0 m if 21st Century CO_2 concentrations exceed 600 ppmv and 0.6–1.9 m for peak CO_2 concentrations exceeding ≈1,000 ppmv. Additional contributions from glaciers and ice sheet contributions to future sea level rise are uncertain but may equal or exceed several meters over the next millennium or longer.

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 Steller 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 Steller 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). In addition, benthic-feeding walrus, bearded seals, gray whales and diving sea-ducks such as Spectacled eider are all potentially threatened by these changes, as are Arctic Native communities whose traditional subsistence culture has relied on these ice-associated mammals and birds for subsistence for thousands of years. Recently, the

USFWS has listed polar bears as threatened under the Endangered Species Act. Separately, NMFS in response to a petition to list ice seals has made a determination for ribbon seals that a listing is not warranted and for spotted seals that only the Asian DPS should be listed at this time. Evaluations regarding ESA listing status for ringed and bearded seals are pending. Similarly, USFS is evaluating the merits of a petition to list walrus under the ESA.

Higher latitude ecosystems have 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 Steller 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, Steller 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 Steller sea lion habitat amount or availability is uncertain, but at the projected rate it is unlikely to have a significant effect for many years.

4.1.6.2 Ocean Acidification

Ocean acidification is a threat to ocean ecosystems related to global climate change and CO_2 emissions. The IPCC (2007a): summarized that;

> The uptake of anthropogenic carbon since 1750 has led to the ocean becoming more acidic…Projections based on SRES scenarios give a reduction in average global surface ocean pH of between 0.14 and 0.35 units over the 21st century. While the effects of observed ocean acidification on the marine biosphere are as yet undocumented, the progressive acidification of oceans is expected to have negative impacts on marine shell-forming organisms…and their dependent species.

Anticipated pH changes will likely exceed current regional and seasonal variability (Interacademy Panel [IAP] 2009). The Royal Society (2005) pointed out that “Other calcifying organisms that may be affected are components of the phytoplankton and the zooplankton, and are a major food source for fish and other animals.”

The potential effects and the specific timeframes for effects of ocean acidification on North Pacific ecosystems specifically are uncertain. However, in June 2009, the Interacademy Panel on International Issues (IAP 2009) stated that:

> “The high CO_2 waters in polar and upwelling regions such as the eastern Pacific and Bering Sea for example, will experience low pH more rapidly than other regions…The ocean chemistry changes projected will exceed the range of natural variability, which is likely to be too rapid for many species to adapt to. Many coastal animals and groups of phytoplankton and zooplankton

> may be directly affected with implications for fish, marine mammals and the other groups that depend on them for food…The Impacts of these changes on oceanic ecosystems…cannot yet be estimated accurately but they are potentially large…Although some species may benefit, most are adapted to current conditions and the impacts on ocean biological diversity and ecosystem functioning will likely be severe.

4.1.7 Removals of Large Whales and other Marine Mammals

4.1.7.1 Harvest of Northern Fur Seals

Commercial harvests of marine mammals in the Bering Sea began with the industrial harvest of northern fur seals in the Pribilof Islands in the late 1700s. The size of the fur seal population on the Pribilofs was estimated at 2.5 million animals (Kenyon *et al.* 1954). From its beginning until about 1835, commercial harvests of these fur seals were "extravagant, wasteful, and largely unrecorded" (Kenyon *et al.* 1954). By 1803, about 800,000 skins had accumulated in storehouses on the Pribilofs, 700,000 of which "were thrown into the sea as worthless."

By 1834, the northern fur seal population had declined to less than 1,000,000 animals, which resulted in a seven-year ban on killing fur seals to allow the population to recover. From the 1840s to the 1860s, the harvest of fur seals increased from 10,000 animals per year to about 75,000 animals. In 1868, when the U.S. first occupied the Pribilof Islands, 242,000 fur seals were harvested. From 1870 to 1909, commercial companies from the U.S. conducted the fur seal harvest accompanied by the onset of pelagic sealing.

The practice of pelagic sealing was not selective and resulted in the death of a high percentage of pregnant, female fur seals. From the 1860s to about 1911, more than 950,000 fur seals were taken by pelagic sealers. At the same time, more than 2,900,000 fur seals were taken on the Pribilof Islands. `The combination of pelagic sealing and land-based sealing dramatically reduced the size of the fur seal population: by 1897, the fur seal population had been reduced to about 400,000 animals; by 1911, it had been reduced to about 215,000 animals. Because the takes were greatly reducing the fur seal stock, Great Britain (for Canada), Japan, Russia, and the United States ratified the Treaty for the Preservation and Protection of Fur Seals and Sea Otters in 1911. The treaty prohibited pelagic sealing and required a reduction in the taking of seals on the land.

From 1912 to the mid-1950s, the population slowly increased to about 1,500,000 animals with a harvest of about 60,000 male seals each year. In the early 1950s, biologists realized that the fur seal population had ceased to grow and agreed to experiment with increasing the harvest of male fur seals and begin another harvest of female fur seals in the hope that the fur seal population would increase further. In 1953, the harvest of female fur seals began with the death of about 850 female fur seals. This harvest peaked in 1957, with 47,413 animals. From its discovery until the mid-1950s, more than 7.8 million fur seals were taken in commercial harvests. In 1957, the signatories of the 1911 Treaty ratified a new agreement, the Interim Convention on the Conservation of North Pacific Fur Seals, for the conservation, research, and harvesting of fur seals. About 18,000 female fur seals were killed each year from 1963 to 1968.

When this experiment ended, more than 300,000 female fur seals had been killed in an attempt to increase the productivity of the population and, as a result, the size of the commercial harvest (Kenyon *et al.* 1954). The harvest did not increase the population's productivity as expected; instead, pup production on St. Paul Island declined by 7% per year from 1975 to 1983 and production on St. George declined by 6% per year from 1973 to 1990. From 1950 to 1988, the fur seal population declined by over 50% (to about 1 million animals).

The authority of the 1957 Convention was extended in 1963, 1969, 1976 and 1980. Under the terms of the 1980 extension, the Convention expired on October 14, 1984. In consultation with the U.S. Departments of State and Justice, and the Marine Mammal Commission, the United States declined to sign an extension. It was determined that no commercial harvest could be conducted under existing domestic law and, therefore, the commercial harvest on St. Paul Island was terminated. Management of the fur seals then reverted to the MMPA. Accordingly, on July 8, 1985, NMFS issued an emergency interim rule to govern the subsistence taking of fur seals for the 1985 season under the authority of section 105(a) of the Fur Seal Act. A final rule was published on July 9, 1985.

On June 17, 1988, NMFS declared the stock of northern fur seals on the Pribilof Islands to be depleted under the MMPA. The MMPA defines a species, population, or stock as depleted if it falls below its optimum sustainable population (OSP). The lower bound of OSP for northern fur seals is thought to be at least 60% of the carrying capacity level. The Pribilof Islands population was designated depleted because it declined to less than 50% of levels observed in the late 1950s, and no compelling evidence suggested that carrying capacity has changed substantially since the late 1950s.

The Pribilof Islands population has continued to decline since the depleted listing. Between 1998 and 2004 estimated pup production declined at 6.2% per year (SE = 0.78%, P = 0.01) on St. Paul Island, and at 4.5% per year (SE = 0.45%, P = 0.01) on St. George Island. The 2004 estimate of pup production on St. Paul Island is comparable with the level observed in 1921, while on St. George it is below the level observed in 1916. Recent satellite telemetry studies indicate lactating female and juvenile male northern fur seals are central place foragers while in the Bering Sea. These studies also suggest separation of Bering Sea foraging areas defined by the central breeding area of departure for fur seals (Robson *et al.* 2004).

Changes in the quantity or quality of available prey may also influence the health and fitness of individual fur seals. Important fur seal prey includes pollock, small schooling fish, and gonatid squid. The importance of any particular prey category depends on the sampling location and may be related to biases in the method used to assess prey importance. Walleye pollock and squid are important fur seal prey in the eastern Bering Sea with the addition of Pacific herring, Pacific sand lance, and capelin in the Gulf of Alaska and Pacific Ocean. The abundance has changed for major fish species across the entire range of fur seals. Whether and what extent fish abundance was affected by fishing or environmental change is unknown. How alteration of fish abundance influences population trends of the eastern Pacific stock is also unknown. The complexity of ecosystem interactions and limitations of data and models make it difficult to determine specific effects on the fur seal population.

NMFS completed a conservation plan under the MMPA for northern fur seals in December 2007. The plan reviews and assesses the known and possible factors influencing northern fur seals in Alaska; it also contains pertinent information on fur seals breeding in California and Russia. Natural factors influencing the population include predation, parasitism, disease, and environmental change. Human-related factors influencing the population include subsistence harvests, direct and indirect effects of commercial fishing, marine debris, poaching, pollution, vessel and aircraft traffic, tourism, coastal development, noise, and oil and gas activities.

4.1.7.2 Harvest of Large Cetaceans

By the late 1800s, commercial whaling had severely reduced the population of bowhead whales in the Bering and Chukchi Sea and had left the eastern North Pacific right whale population nearly extinct. The modern era of pelagic whaling in the North Pacific began in 1946, and from 1952-1962 the whaling era using factory ships began with two operating in 1953 and seven by 1963 (Mizroch *et al.* 2009), and this type of whaling extended eastward to the American side of the Pacific. In 1963, the arrival of seven factory ships from Japan and USSR to whaling grounds in the north Pacific partially resulted from the protection of blue whales in the Antarctic and strict quotas on other Antarctic species. These pelagic whalers concentrated on humpback whales, fin whales, and sei whales in the late 1960s. In 1968, whalers in the North Pacific focused on hunting sperm whales and took between 8,000 to 10,000 per year during that period. There were 49,936 reported fin whales kills in the North Pacific during the years 1911 to 1985 (Mizroch *et al.* 2009) and many thousands of blue, sei, humpback, and sperm whales.

Native Alaskans harvested whales in the eastern North Pacific for many years prior to the arrival of commercial whalers in the 19th century. The Inuit of the Bering Sea coast of Alaska have been whalers for centuries. Aboriginal whaling took place in three main areas in the eastern North Pacific (1) the west and northwest coasts of Alaska, (2) the Aleutian Islands and the Alaska Peninsula, and (3) the coasts of Vancouver Island and Washington.

The Aleuts of the Aleutian Islands and the Alaska Peninsula hunted whales with hand-thrown spears. They likely harvested humpback whales, gray whales and possibly right whales. Along the coast of British Columbia and Washington, whales were hunted by Nootka, Makah, Quilleute, and Quinault tribes, who targeted gray and humpback whales, and possibly right whales. The number of whales that were taken in these fisheries is unknown (Scarff 1986).

As noted above, several papers by Springer, Estes, Williams and others describe an hypothesis that links declines in abundance of several marine mammal populations in Alaska to the harvests of the great whales in the middle of the 20th Century (e.g., Steller sea lion, northern fur seal, harbor seal, and northern sea otter). A series of papers have rebutted parts or all of the hypothesis (e.g., DeMaster *et al.*, 2006, Trites *et al.* 2006, Mizroch and Rice 2006, Wade *et al.* 2007). Also as noted above, the original Springer *et al.* (2003) group has published their reaffirmation of this work (Springer *et al.* 2008), noting it remains a hypothesis and as such provides a view point on the potential impacts of such a large scale “adjustment” to the predator-prey dynamics in the North Pacific ecosystem.

4.2 Natural Factors Affecting the Status of Steller Sea Lions in the Action Area

4.2.1 Climate and Oceanography

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 “junk food” hypothesis, the quantity of prey available to Steller sea lions was thought to be high overall in the 1980s, 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 Steller sea lions (i.e., both juveniles and adults). However, results of subsequent feeding experiments, mathematical models, and field observations suggested that adult Steller 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, Rosen 2009), assuming ample amounts of gadid prey are available. 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 Steller 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 Steller sea lion food habits did not support the hypothesis that the regime shift triggered changes in the prey community that would, on their own, have been deleterious to Steller sea lions. In addition, feeding experiments at the Alaska SeaLife Center have shown no negative consequences to captive juvenile Steller sea lions fed only pollock (Calkins, Mellish *et al.* 2005). 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 adult Steller sea lion body composition when animals are able to consume sufficient quantities of prey to meet their energy demands (Rosen and Trites 2004, 2005, Rosen 2009).

It is likely that Steller sea lions may have lived through many climate/biological regime shifts in the last few millenia. What may be different about this most recent shift (1977-78) is the (1) coincident development of extensive fisheries targeting the same prey that sea lions depend on, (2) changes in marine mammal biomass that might influence foraging behavior of mammal-eating killer whales, and (3) significant unreported and unregulated shooting or bycatch of sea lions in proximity to commercial fishing operations. As noted above, the cause of the sea lion decline need not be a single factor and may differ between different regions and periods of time. In addition, strong environmental influences on Gulf of Alaska and Bering Sea ecosystems could increase the sensitivity of Steller sea lion populations to fisheries or changes in those ecosystems resulting from fisheries.

Given an 80% reduction in the western DPS of Steller sea lions, a general lack of robust population growth, and the lack of evidence suggesting sustained high levels of anthropogenic removals, it is likely that the environmental carrying capacity has been reduced either through natural environmental changes or human induced changes. Given the equivocal data surrounding the dietary needs of Steller sea lions, the consequences of climate regime shifts, and massive population declines, it is highly unlikely that natural environmental change has been the sole underlying cause for the decline of Steller sea lions. Therefore, this consultation looks to this and other possible causes of the decline recognizing that environmental change is an important component in this equation, and may combine with other factors to contribute to the past decline in abundance of the western population of Steller sea lion and the current lack of recovery.

4.2.2 Disease, Parasites, and Toxic Substances

The effects of disease, parasites, and toxins on Steller sea lions were discussed in detail in Section 3. Available serologic evidence does not support the likelihood that a disease epidemic occurred during the Steller sea lion decline of the late 1970s and 1980s; however, due to sampling limitations the possibility cannot be excluded completely (Atkinson *et al.* 2008). Although Steller 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, but the potential for those agents to cause disease among Steller sea lions has not been documented. Disease and parasitism are common in all pinniped populations and have been responsible for major die-offs (e.g., Osterhaus *et al.* 1997), but such events are usually relatively short-lived and provide more evidence of morbidity or mortality. The potential for parasitism to have a population level effect on Steller sea lions is largely unknown, although these effects could become significant if combined with other stresses. Moles and Heintz (2008) studied forage fishes that are prey for Steller sea lions, noting that several of the more abundant forage fishes in the Aleutian Islands harbor a greater proportion of nematode and acanthocephalan parasites, some of which are potentially pathogenic and capable of infecting sea lions. The ramifications of disease and parasitism remain a concern, both as primary and secondary problems, but do not appear to be significant impediments to recovery at this time or on the basis of the information currently available.

At present, there is not enough information to determine what role, if any, exposure to contaminants plays in the health, survival and recovery of Steller sea lion populations (Atkinson *et al.* 2008). In the revised Steller sea lion Recovery Plan, NMFS (2008) concluded that toxic substance exposure posed a medium risk to the recovery of the western DPS of Steller sea lions and concluded also that there is a high level of uncertainty associated with that statement. Research focused on this topic could help our understanding of factors that are impacting recovery in the western DPS.

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 (e.g., mercury), 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. Air borne transport of contaminants to the North Pacific may have occurred and may still be a vector for contamination, but there currently are no data substantiating sources or levels of such contaminants in this marine ecosystem. Some contaminants studies have been conducted on both Steller sea lions and other pinniped species and are briefly reviewed below by toxic category.

Steller sea lions in many areas may have been directly exposed to crude and refined petroleum products due to the many small and some large or very large oil spills that have occurred throughout their range. Unlike the Exxon Valdez oil spill, after most smaller spills, and some large spills, there is very little study of ecological effects. However, large and very large spills (as defined by MMS in the 2003 Cook Inlet Oil and Gas Lease Sale EIS 2003) have occurred in waters off of California (e.g., the Santa Barbara spill), the coast of Washington (e.g., the heavy fuel oil spill caused by the grounding of the U.S.S. General M.C. Meiggs), Prince William Sound (the Exxon Valdez oil spill), the Kodiak archipelago, and other locations in their range. A complete review of this topic is beyond the scope of this document. We consider this topic further in the section below on oil and gas development.

Steller 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 Prince William Sound. Furthermore, Steller 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). 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). However, this study was limited by a lack of pre-spill data on key health and other parameters that would have allowed more focused studies of sublethal effects. Further, as the decline in the western DPS was underway at the time of Exxon Valdez oil spill, researchers were not able to discriminate spill effects, and, hence, the kinds of detailed studies related to chronic effects that were conducted on some other species were not undertaken on sea lions.

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 Steller 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.

OC concentrations have been significantly different among Steller sea lions in some regions (Myers and Atkinson 2005, Hoshino *et al.* 2006, Myers 2008), 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.

Few studies of the effects of known OCs have 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 Steller sea lion decline. Newer OC studies by Myers (2008) and Myers *et al.* (2008) were reviewed in the previous section; these researchers noted potential adverse effects of OCs in the marine environment on Steller sea lions.

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. Thus, low-level mercury exposure is evident in pups and females, but the long-term effect mercury or even methylmercury has on Steller sea lions is unclear (Beckmen *et al.* 2002). Based on recent data on methylmercury in salmon, Beckmen *et al.* (2002) suggest that exposure at low levels is cause for concern because methylmercury is a powerful neurotoxicant that acts synergistically with polychlorinated biphenyls. Studies on non-human primates have demonstrated both reproductive effects (Burbacher *et al.* 1984, Burbacher *et al.* 1988; Mohamed *et al.* 1987) at methylmercury levels that do not cause overt toxicity, as well as developmental effects due to exposure of relatively low levels of exposure in utero. Recent unpublished findings by Castellini *et al.* (2008) suggest that significant portion of very young pups in the western DPS have total mercury levels higher than those which the Environmental Protection Agency (EPA) suggests may indicate exposure sufficient to produce toxic effects. Such contaminant load in young pups should reflect load transferred from their mothers. Holmes *et al.* (2008) also report on mercury levels in SSL pups, noting that the western SSL had statistically higher mercury levels in kidney and liver tissues and lead in liver tissues than in the eastern DPS. Not enough is currently known about the potential effects of this or other contaminants on the recovery of 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.

Steller sea lions have shown various levels of toxic substances including heavy metals and organochlorines with generally higher levels in the most western portions of the range including Russia. In general, these concentrations of 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 on Steller sea lions specifically 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. Steller 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, but that it is unlikely to be having substantial impacts on the population. Further study is needed to test the response of individual Steller sea lions to these concentrations of toxins.

4.2.3 Impacts of Killer Whale Predation

4.2.3.1 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, Krahn *et al.* 2007). 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. Krahn *et al.* (2007) confirmed the existence of this ecotype in a study of stable isotope ratios, persistent organic pollutants, and fatty acids in 84 killer whales in the North Pacific. 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 Steller 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 Stock Assessment Report [SAR]). 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 few 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. Analyses of photo-identification and genetic data from within the coastal survey area from Kenai Fjords to Tanaga Pass provide insights into abundance and movement patters of transient killer whales (Durban *et al.* 2010). Specifically, (1) approximately 255 to 487 whales were estimated to have occurred in the study area, (2) the largest number of killer whale encounters occurred in the eastern Aleutian subarea, (3) a smaller number of animals was estimated (i.e., 251 [95% CI = 97 – 644]) to be present in the study area in July and August, and (4) none of the individuals documented in the western Gulf of Alaska were subsequently observed in the Aleutian Islands, and none of the individuals documented in the central Aleutians were sighted in the adjacent eastern Aleutian subarea. The authors further noted the importance of spatial segregation and site fidelity in evaluating the intensity and distribution of predation pressure on marine mammal prey.

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).

4.2.3.2 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-identification, which provide an estimate of the total number of individual killer whales in the region surveyed across the entire survey period. It should be noted that the former methodology (line transect surveys) provides an estimate of abundance that is a snapshot at a given point in time, whereas, the latter method (mark-recapture) provides an estimate of the total number of animals that used a given area over specific period of time. Therefore, in general line transect estimates of abundance will be lower than estimates of abundance based on mark-recapture. Which method provides for a better indicator of exposure to predation is not possible to ascertain with available data.

Line transect ship surveys were conducted in summer 2001-2003 in coastal waters of the western Gulf of Alaska and the Aleutian Islands to estimate abundance of killer whale ecotypes, which were distinguished based upon morphological and genetic data (Zerbini *et al.* 2006). Although calculations were performed on two data sets that differed in the methods used to survey whales, Zerbini *et al.* (2006) reported that the more accurate counts of resident killer whales were 991 (379-2585) which were at least four times greater than those of transient killer whales (200 [81-488]). Residents were most abundant near Kodiak Island, around Umnak and Unalaska Islands in the eastern Aleutians, and in Seguam Pass in the central Aleutians. In contrast, transient killer whale sightings were found at higher densities south of the Alaska Peninsula between the Shumagin Islands and the eastern Aleutians. They also noted that only two sightings of ‘offshore’-type killer whales were recorded during the surveys, one northeast of Unalaska Island and the other south of Kodiak Island. These were the first estimates of abundance of killer whale ecotypes in the Aleutian Islands and Alaska Peninsula. More recently updated data were provided by Durban *et al.* (2010) of 345 transients between the central GOA and the central AI region. The density of transients appears to vary regionally, with higher densities from the Shumagin Islands through the eastern Aleutian Islands. However, the minimum count of transients in this area from the combined NMFS and North Gulf Oceanic Society (NGOS) photo-identification catalogues is currently 314 whales (Angliss and Outlaw 2005), and preliminary mark-recapture estimates for transients based on photo-identification data is on the order of 345 (Durban *et al.* 2010). Current abundance estimates and photo-id cataloguing only refer to coastal waters within approximately 30 nm 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.

The diet of AT1 transients, a pod of transients often associated with Prince William Sound, 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). Stomach contents of two stranded carcasses contained a harbor seal, Dall's porpoise, and Steller sea lion remains (Heise *et al.* 2003). 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 Steller 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[14]).

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 sub-population of Steller sea lion has been increasing overall (with some rookeries showing stability), and likely has been increasing for many years under similar numbers of killer whales. 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[15] 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 prey were sea lions. Based on their observations they estimated that 59 Steller sea lions were consumed over the summer seasons of 2002-2005. However, based on estimates of killer whale caloric requirements (Williams *et al.*2004), they would

---

[14] 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 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. These are limited sample sizes, but the data available do not support the idea that all GOA transients eat sea lions exclusively.

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

have expected loss due to orca predation of 103 sea lions over this period. Maniscalco *et al.* (2007) 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. Therefore, 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).

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

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 Chapter 3), 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. From these data, Williams *et al.* (2004) reported that an average adult killer whale would require 2 - 3 Steller 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 Steller sea lion per day (approximately 160 per year) would be needed to satisfy the killer whale's metabolic needs. 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 Steller sea lions (York 1994). The annual number of Steller sea lions eaten increases 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. Despite the conservative estimates of energetic needs and the 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 caloric 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 sea lions lost per year at the height of the decline. Furthermore, predation losses to a single pod of five killer whales could theoretically prevent the present Steller sea lion population from recovering.

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 Steller 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 have examined the hypothesis and conclusions from those studies have varied:

> 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.* (2006c) showed that populations of seals, Steller 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. They additionally note that killer whales are

> unquestionably a significant predator of seals, Steller sea lions and sea otters — but not because of commercial whaling.
>
> Wade *et al.* (2007 argued that available data do not support the Springer *et al.* (2003) hypothesis. They noted 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. 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. They summarized that: a) large whale biomass in the Bering Sea did not decline as much as suggested by Springer *et al.* (2003); b) much of the reduction occurred 50-100 years ago, well before the declines of pinnipeds and sea otters began; and c) thus the need to switch prey starting in the 1970s is doubtful. They compiled data showing that, with the sole exception that the documented sea otter decline followed the decline of pinnipeds, the reported declines were not in fact sequential. Given this, Wade *et al.* (2007) concluded that it is unlikely that a sequential megafaunal collapse from whales to sea otters occurred.
>
> Kenney (2007), 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."

In 2008, Springer *et al.* replied to many of the above criticisms. They cite additional papers and assert there is accumulating evidence for the importance of top–down forcing processes, especially those driven by large vertebrate consumers. They cite additional papers on killer whale predation and its role in the decline of sea otters in Alaska, lending support of the hypothesis of prey switching and resultant sympatric pinniped declines. Springer *et al.* (2008) also note that demographic/energetic models have demonstrated the vulnerability of some pinniped populations to relatively small changes in the diet of transient killer whales (Williams *et al.* 2004). And they are additionally convinced of the logic of their hypothesis based on the compelling coincidence of the North Pacific's multispecies collapse and the large reduction in whales by post WWII industrial whaling. They conclude "…we do not regard this hypothesis as well-tested, but believe it is well reasoned and well supported."

4.2.3.4 Direct Impact of Killer Whales on Steller Sea Lions

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 2001-2003. 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.

A recent summary of survey data for transient killer whales from the eastern GOA (Kenai Peninsula area) to the central Aleutian Islands (Wade and Durban 2010) provides estimates from both line transect surveys and mark-recapture studies. In this overall area, they report the highest estimates of transient killer whales in the eastern Aleutians (176 animals [130-252]) from mark-recapture studies in 2004 to 2010; line transect survey data for the eastern Aleutians for the period 2001-2003 indicate and estimate of 88 (20-273) killer whales. No data are available for the western Aleutians, but for the central Aleutian Islands sub-region, Wade and Durban (2010) report estimates of 87 (19-391) from line transect data and 90 (48-184) from mark-recapture data. In the eastern and central Aleutian Islands combined, Wade and Durban (2010) report estimates of 175 killer whales (39-764) using line transect methods and 266 killer whales (178-436) from the mark-recapture data. Only line transect data are available for the central GOA and western GOA providing estimates of 27 (4-179) and 51 (12-227) killer whales, respectively.

Williams *et al.* (2004) hypothesized that a population of 170 mammal-eating killer whales could have caused the observed decline of western Steller sea lions in the 1980s. Their critical assumption was that the killer whales would have preyed exclusively on Steller 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.* (2009) reviewed observations of killer whale predation on marine mammals since the 1960s. 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. Since 2001, 11% of the observed kills of Steller sea lions in the BSAI by killer whales (AFSC unpublished data). Matkin *et al.* (2005) report 4% of observed kills were of SSLs 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.* 2007*,* 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 Steller sea lion natural mortality rates within the western DPS. Horning and Mellish (2009) implanted satellite-linked post mortem data transmitters (LHX) in 21 juvenile Steller sea lions. Data recovered from 4 of 5 detected mortalities indicated precipitous drops in temperature which were likely associated with acute death at sea by trauma, probably predation. Based on these observations, they estimated survival rates in year 2 and year 3 equal to 0.59 and 0.87, respectively. NMFS estimate of cumulative survival to age 3 from resights of animals branded as pups in the Prince William Sound area is 0.42 for females, 0.33 for males, and 0.38 for sexes combined. NMFS' estimate for Prince William Sound males is not statistically different from the estimate of Horning and Mellish, whose sample was composed largely of males. Of the three areas that NMFS has sea lion survival data from pup branding, juvenile female sea lion survival rates (cumulative to age 3) were lowest in the PWS area (0.42) where Horning and Mellish conducted their study, and higher near Kodiak (0.47) and the eastern Aleutian Islands.

For a stable population of Steller sea lions in the western DPS in Alaska, the average (over all age-classes) annual natural mortality rate is about 10-20% – this reflects about 5,000 - 10,000 animals dying each year due to trampling, senescence, disease, killer whales, etc. This important fact is often overlooked, when 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 Steller sea lions each year. Maniscalco *et al.* (2007), conclude that although these killer whales were observed to eat Steller sea lions exclusively (in their limited study area and time), killer whale predation accounted for only about a quarter of the annual natural mortality.

Durban *et al.* (2010) analyzed the mark-recapture database for transient killer whales in the Gulf of Alaska and Aleutian Islands and noted that, although killer whales may be relatively abundant in some areas, and may have the predation potential to cause declines in Steller sea lion populations, there is considerable spatial segregation, site fidelity, and temporal variation in killer whale abundance such that not all of the population may be present at one time or in one location, features which must be considered when evaluating killer whale impacts on their marine mammal prey populations.

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. For example, lower age-specific rates of growth and reduced pregnancy rates were reported in the 1980s relative to the 1970s (Calkins *et al.* 1998, Pitcher *et al.* 1998). This indicates carrying capacity for Steller 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 the only cause of the decline and there is no easily identified direct link between killer whale predation and a decline in natality (although see Springer *et al.* 2009 and Frid *et al.* 2008). In addition, the eastern DPS has increased at approximately 3% per year for at least 20 years while co-existing with a similar population of transient killer whales in an environment historically exposed to commercial whaling and environmental change (Trites *et al.* 2006).

Nonetheless, there is ample literature to suggest that in some areas, particularly areas of low Steller sea lion abundance (e.g., the central Aleutian Islands), killer whale predation can be an important factor in either causing continued declines or contributing to a lack of a robust recovery (see Williams *et al.* 2004, Williams 2006, Guinette *et al.*, 2007, Heise *et al.* 2003, Durban *et al.* 2010).

4.2.4 Inter-Specific Competition for Prey Resources

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 increased arrowtooth flounder abundance, 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.

Arrowtooth flounder have been discussed as possible competitors of Steller sea lions for common prey items, although there are few studies of this competition. This “competition” may be exacerbated by the currently very high abundance of arrowtooth flounder in both the Bering Sea and the Gulf of Alaska, but may be particularly of potential importance in the Gulf of Alaska given the high biomass estimates of arrowtooth flounder in the Gulf. And there is currently little fishing pressure on arrowtooth, particularly given its abundance. Arrowtooth are not a target species for Bering Sea/Aleutian Island fisheries although some harvest is reported as bycatch. In the Gulf of Alaska, well over 20,000 mt of arrowtooth have been harvested in the late 2000s. Arrowtooth are important predators of pollock in the Gulf and the Bering Sea, and in warmer years in the Bering Sea, favoring increased biomass of arrowtooth flounder, increased predation pressure from arrowtooth may lead to declines in eastern Bering Sea pollock (Zador and Aydin 2009).

Biomass estimates of arrowtooth flounder from AFSC surveys on the Bering Sea continental shelf have shown a consistent increasing trend since 1975. Since 1982, biomass point estimates indicate that arrowtooth flounder abundance has increased eight-fold to a high of 570,600 mt in 1994. The population biomass remained at a high level from 1992-97, and in the 1997-2000 bottom trawl surveys the Bering Sea shelf population biomass had declined to 340,000 mt, 60% of the peak 1994 biomass point estimate. Beginning in 2002 the shelf survey estimate increased again and peaked in 2005 at a biomass of 757,685 mt. In recent years (2006-2008) the estimates declined slightly but were still at high levels, and in 2009, survey estimates were lower at 453,559 mt (Wilderbuer *et al.* 2009). In the Gulf of Alaska, Turnock and Wilderbuer (2009) report modeled biomass of arrowtooth flounder increased from 331,298 mt in 1961 to a high of 2,187,450 mt in 2006 and a slight decrease in biomass to 2009 at 2,155,780 mt.

Wilderbuer *et al.* (2009a) report that arrowtooth flounder are an important ecosystem component as a predator of other fishes targeted commercially and of fishes that are prey for Steller sea lions. This may be relevant from an ecosystem perspective, as in recent years' stock assessments, arrowtooth flounder have been increasing rapidly in abundance. In the Bering Sea, nearly half of the adult diet is comprised of juvenile pollock (47%) followed by adult pollock (19%) and euphausids (9%). This is in contrast to their diet in the Gulf of Alaska, where pollock are a smaller percentage of their forage base, which instead consists primarily of shrimp, but this varies based on size of arrowtooth. Interestingly, there is a reverse relationship as well, as modeling results reported in Wilderbuer *et al.* (2009) show the effect of decreasing pollock (adults or juveniles) is to increase arrowtooth flounder in the model rather than decrease it, suggesting that the role of pollock as a predator on arrowtooth flounder (potentially limiting their population growth) may be greater than the importance of pollock as prey, at least for small perturbations of pollock. In the Gulf of Alaska, Turnock and Wilderbuer (2009) note that the majority of the prey by weight of arrowtooth larger than 40 cm was pollock; the percent of pollock in the diet of arrowtooth flounder increases for arrowtooth sizes greater than 40 cm.

The diets and distribution of many other marine mammals and birds also overlap those of the western DPS of Steller sea lion. 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 (in terms of biomass) 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, Springer *et al.* 2008). 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 a substantial increase in central North Pacific humpbacks between the early 1980s and early 1990s (Baker and Herman 1987, Calambokidis *et al.* 1997), and late 1990s (Calambokidis and Barlow 2004). 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 whale species are piscivorous and fulfill their annual consumption needs on high latitude feeding ground, including waters found within critical habitat of the western DPS of Steller sea lion. Substantial seasonal feeding aggregations of humpback (Waite *et al.* 1998, Witteveen 2003), fin, and minke 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, sand lance, smelts, small pollock) (Pitcher 1981, Sinclair and Zeppelin 2002) or by the prey of Steller 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 Steller sea lion prey species feed. As populations of piscivorous cetaceans recover, this potential interaction would be expected to strengthen.

4.2.5 Status of Important Steller Sea Lion Prey Resources in the Action Area

4.2.5.1 Walleye Pollock

Recruitment

Walleye pollock, *Theragra chalcogramma*, is a marine fish species that is highly fecund, producing millions of eggs per individual spawner, and which has highly variable mortality rates in early life (Bailey and Ciannelli 2007). A consequence of this reproductive strategy (producing lots of young with high expected mortality) is fluctuating annual recruitment levels (the number of young fish entering the population each year). The instability of fluctuating year classes must be buffered by the averaging effect of many age classes in the population. Because most of the oldest fish have been removed from the population by the fishery, the abundance of walleye pollock in the Gulf of Alaska is driven by recruitment. Although the recruitment process of walleye pollock in the Gulf of Alaska is one of the better studied systems in the world, admittedly there is still much that is not well-understood.

Pollock is an opportunistic species that has a broad range and has adapted to different environments. On the other hand, the population is limited by finding and adapting to local conditions that favor successful spawning (maximizing reproduction) and survival (minimizing mortality) of the early life stages. Local populations of pollock respond differently to shifting environmental regimes, as warming periods have seen those stocks at the southern margins of the pollock distribution falter or fail (Bailey *et al.* 1999). In the center of its distribution of mass in the eastern Bering Sea, pollock have been favorably impacted by periods of environmental warming (Hollowed *et al.* 2001, Quinn and Niebauer 1995). Delayed springtime blooms may negatively influence recruitment in the Bering Sea (Mueter *et al.* 2006). However, another recent study correlates cool temperatures in the Bering with increasing recruitment (Megrey pers. comm.). In the Gulf of Alaska the situation appears more complex, as pollock have been initially favored by a warm environmental regime (e.g., stock increase in the late 1970s and mid 1980s) but negatively impacted afterwards (Hollowed *et al.* 2001), possibly in association with an increase of predator biomass. However, conflicting with other findings, a recent study has tentatively and weakly linked cool springtime SST with increasing recruitment (Hollowed pers. comm.). These conflicting findings illustrate the difficulty in relating environmental indices near the birth of the cohort to highly variable recruitment 3-4 years later.

Pollock spawn once per year, in an event that involves individual pairing and courtship (Baird and Olla 1991), and that is highly concentrated in space and time (Kendall and Picquelle 1990). Given the fragility of eggs and larvae to environmental conditions, and their concentration in space and time, the survival of a whole year class is vulnerable to the vagaries of the ocean and weather, such as storms passing through Shelikof Strait, the major spawning site. On the other hand, pollock dynamics are buffered partly by multiple spawning stocks, spawning in different locales, compensatory mortality and by multiple age groups in the population. Spawning in different locations moderates the effects of temporal variation in habitat suitability by taking advantage of spatial variation. While the long life span of pollock is an adaptation that tempers the high variation in year class strength, a high abundance of predators on adults, as well as commercial fishing that removes older age groups, reduces the age-span over which mean abundance is averaged (and perhaps other aspects of the contribution of older fish to the population's viability). Consequently, the population will be more dependent on fewer age groups, hence contributing to overall stock variability (Longhurst 2002).

The spawning regions of pollock are noted for mixing of coastal and nutrient-laden oceanic waters and stratification of the water column, leading to enhanced productivity; these conditions favor the survival of early life stages of pollock. In the Gulf of Alaska pollock typically spawn during the last week in March and first week in April, in the Shelikof Strait (Ciannelli *et al.* 2007). In this area, mixing of the Alaska Coastal Current, the Alaska Stream and coastal water, along with springtime increases in sunlight and water column stratification leads to an intense spring bloom and reproduction of zooplankton. Zooplankton prey of pollock larvae are further concentrated by physical features, such as eddies and fronts (Napp *et al.* 1996), leading to favorable feeding conditions. Late larvae and juvenile pollock are advected toward favorable nursery areas, such as the waters around the Shumagin Islands.

Mortality rates of pollock eggs and young larvae are very high, ranging from 4% to 40% per day, but decline as the larvae develop. In fact, larval condition can vary from year to year and by location, and high percentages of larvae in the ocean have sometimes been observed in poor feeding condition (Theilacker *et al.* 1996). Studies have shown that egg and early larval development and survival are suboptimal at temperatures below about 0^{o} and above 10^{o}-13^{o}C (Hamai *et al.* 1971, Nakatani and Maeda 1984; Blood 2002). Extremely high and low temperatures can be lethal to eggs and larvae, but generally for the Gulf of Alaska population, which is in the central part of its distribution (4^{o}- 6^{o}C springtime SST), higher temperatures tend to favor better survival of early stages, perhaps through one or more indirect mechanisms (Bailey 2000). Optimal prey levels for successful feeding depend on many different conditions, including larval size, temperature, light levels, turbidity and turbulence (Porter *et al.* 2005), but generally they range between 20 and 40 prey/liter (Theilacker *et al.* 1996). At high levels of abundance, pollock may deplete their prey (Duffy-Anderson *et al.* 2002) leading to slower growth and higher mortality. At later stages, predation on juveniles is an important source of loss to the population. Large juvenile and adult pollock prey heavily on small juvenile pollock, and other piscivorous fishes, including halibut, cod, arrowtooth flounder and flathead sole contribute significantly to mortality of juvenile pollock (Livingston 1993).

An evolving perspective of the recruitment of pollock is that it is a complex process, influenced by both high frequency changes in the environment of young fish stages and by bounding effects of low frequency changes in the ecosystem (Bailey *et al.* 2005). As a consequence recruitment is caught in the push-pull between these scales. Larval mortality is highly variable and subject to many interacting high frequency factors (such as storms and prey availability), with feedback and non-linearity (Bailey *et al.* 2004). Larvae show sophisticated behaviors involving choice and decisions when confronted with multiple and perhaps conflicting stimuli (Olla *et al.* 1996). For example, they avoid turbulence by descending (Davis 2001), taking them out of the photic zone and into colder water where growth is less optimal and prey are less abundant (Kendall *et al.* 1994). Under normal circumstances, these conditions are associated with poor feeding and high mortality. However, prey are driven deeper by turbulence and if there is bright daylight, these conditions are then optimal for feeding (Porter *et al.* 2005). Thus, environmental factors driving recruitment are governed by complex relationships. On the other hand, although juveniles also show complicated behaviors in response to the environment (e.g., Sogard and Olla 1996), they are less impacted by small-scale physics, and juvenile mortality seems to be more stable and predictable, occurring largely as a result of predation and density-dependent mechanisms. The role of density-dependent mechanisms also seems to be influenced by environmental factors (Ciannelli *et al.* 2004). Environmental and ecosystem structure shifts may also have indirect effects on pollock survival, such as causing changes in the operation of density-dependent mechanisms. For example, Ciannelli *et al.* (2004) found that the level of density-dependent mortality in juvenile pollock increases when water temperature and predation intensity are high. The build-up of predators in the community represents a low frequency, slowly changing pattern with lagged effects. Changes in ecosystem structure may be related to the relative stage in life history when recruitment is determined (i.e., larval versus juvenile control) (Bailey 2000). Therefore, control points may change from year to year, and depend on longer term changes in the environment and community structure, such as those occurring with environmental and biological regime shifts. General patterns in recruitment have been well-described by models incorporating stochastic mortality related to environmental conditions during the larval period and by deterministic factors and constraints during the juvenile period (Ciannelli *et al.* 2004, Ciannelli *et al.* 2005). It should be noted that although we have a fairly good understanding of how small-scale factors affect survival of early life stages, knowing how these factors combine and interact over larger and longer space and time scales (scale up), thus determining how pollock populations respond to the environment, fluctuating and shifting prey and predator abundances, and to self-regulation is a difficult problem.

ABC as Recommended in the Most Recent Stock Assessments

Eastern BS pollock fell into Tier 1 of the ABC/OFL definitions for 2007, because reliable point estimates of biomass (B), B_{MSY}, and a reliable pdf of F_{MSY} are available through the age-structured stock assessment model (Ianelli *et al.* 2007). The year 2008 spawning biomass was estimated to be 1,380 thousand mt (at the time of spawning, assuming the stock is fished at Tier 1b level). This is below the B_{MSY} value of 1,876 thousand mt. Under Amendment 56, this stock therefore qualifies under Tier 1b and the harmonic mean value of the pdf of F_{MSY} is considered a risk-averse policy since reliable estimates are available. The exploitation-rate type value that corresponds to the F_{MSY} level was applied to the "fishable" biomass for computing ABC levels. The OFL's and maximum permissible ABC values for Tier 1b were thus:

Tier	Year	Max ABC	OFL
1b	2008	1,170 thousand mt	1,443 thousand mt
1b	2009	976 thousand mt	1,204 thousand mt

Given the rapidly declining stock and the recent increases in harvest rates, the assessment authors argued that it would be prudent to consider harvest levels that would (1) provide stability to the fishery; (2) provide added conservation to an important prey species of the endangered stock of Steller sea lions; and (3) provide extra precaution due to uncertain stock removals in Russian waters. The authors conceded that the degree to which the ABC should be adjusted downwards is difficult to quantitatively justify. The maximum permissible ABC under Tier 1b appeared too high to the stock assessment authors given the continued decline and the lower abundances of older fish seen in the population in recent years. For stability in catches and an added level of precaution given the uncertainty in recent recruitment trends, the authors recommended an ABC of 1.0 million mt in 2008 for the BS pollock stock. The Council set ABC and TAC at 1.0 million mt for the 2008 fishery.

GOA pollock fell into Tier 3 of the ABC/OFL definitions, which require reliable estimates of biomass, $B_{40\%}$, $_{F30\%}$, and $F_{40\%}$. Under the definitions and current stock conditions, the overfishing rate is the fishing mortality rate that reduces the spawner stock biomass to 35% of its theoretical unfished level (the $F_{35\%}$ rate). The model estimate of spawning biomass in 2008 was 145,101 mt, which is 26% of unfished spawning biomass (assuming average post-1977 recruitment) and below $B_{40\%}$ (221,000 mt), thereby placing Gulf of Alaska pollock in sub-tier “b” of Tier 3. The author’s 2008 ABC recommendation for pollock in the Gulf of Alaska west of 140° W longitude (W/C/WYK) was 53,590 mt, a decrease of 16% from the 2007 ABC, but close to the projected catch in 2007. This recommendation was based on a more conservative alternative to the maximum permissible F_{ABC} introduced in the 2001 SAFE. The OFL in 2008 was 72,110 mt. In 2009, the recommended ABC and OFL were 71,580 mt and 95,940 mt, respectively. For GOA pollock in this subarea, the Council set ABC and TAC at 51,940 mt for 2008 and 41,620 for 2009, respectively.

Aleutian Island pollock also fell into Tier 3 of the ABC/OFL definitions. The projected year 2008 female spawning biomass (*SB08*) was estimated to be 82,250 mt, above the $B_{40\%}$ value of 51,450 mt, thus placing Aleutian Island pollock in Tier 3a. The projected total age 3+ biomass for 2008 was 197,280 mt. The maximum permissible 2008 ABC based on $F_{40\%} = 0.196$ is 28,160 mt and OFL based on $F_{35\%} = 0.244$ was 34,040 mt. The Council set TAC at 19,000 mt for 2008.

4.2.5.2 Pacific Cod

Recruitment

Pacific cod (*Gadus macrocephalus*) are demersal gadids that commonly occur in the Gulf of Alaska, Bering Sea, and Aleutian Islands. Little is known about the recruitment process in this species, though events occurring during the egg, larval, and juvenile stages of fish life history are thought to be major regulators of recruitment to the adult populations. Interannual recruitment variability is high in this species, due in part, to the high natality of females (hundreds of thousands to millions of eggs per female), high rates of cumulative mortality among early life history stages, and considerable interannual variation in growth rates.

Pacific cod spawn primarily February through June, and eggs are demersal and weakly adhesive. Larvae hatch out at approximately 3-4 mm and are pelagic, occurring at approximately 50 m (Rugen and Matarese 1988). Larvae are most abundant in the pelagic environment April-June (Matarese *et al.* 2003). Laboratory studies have shown that Pacific cod larvae hatch out from eggs between 16-28 days post fertilization, with peak hatching on day 21 (Abookire and Piatt 2005). Laboratory studies on the development of external morphology and digestive function of Pacific cod larvae indicate that an ecological turning point may to occur at approximately 9 mm total length (TL) (Yoseda *et al.* 1993). This developmental state was associated with significant changes in feeding morphology and also high mortality in that study. It should be noted that flexion in this species begins at approximately that time (10-15 mm standard length [SL]), suggesting that improvements in swimming ability may have ecological consequences.

Climate-induced trophic restructuring is well-documented for a variety of species and marine systems (Duffy-Anderson *et al.* 2005), and it is likely that recruitment in Pacific cod is similarly influenced. Alterations in climate influence ecosystem biota through a variety of co-occurring and synergistic processes (climate, seasonal timings and couplings, predation, feeding, transport), but of the one major factors is likely bottom-up forcing. Factors that affect hydrography influence zooplankton availability and ultimately fish abundance, though the explicit mechanism is as yet unresolved. Pacific cod larvae are opportunistic feeders that primarily consume copepod nauplii and copepodites (Takatsu *et al.* 2002), and consequently depend on zooplankton availability for growth and survival. Because Pacific cod larvae rely on a supply of zooplankton prey for food, climate-induced variations in ocean circulation that modulate the supply of zooplankton available could significantly impact feeding, growth, and survival in this species. Oceanographic features that act to concentrate zooplankton and larvae together, such as eddies and fronts, may enhance feeding opportunities for Pacific cod larvae. Factors that break down prey-larval associations, such as storms, mixing, and significant turbulence, could lead to increased mortality among larvae.

Geographical variations in larval size are also likely related to interannual variations in local meteorological oceanographic conditions. Pacific cod larvae may be vulnerable to density-dependent regulation in the late-larval stage due to prey limitation and associated slow growth. Work with Atlantic cod (*Gadus morhua*), has shown that rapid growth increases survivorship, and that selection for fast growth is enhanced in slow-growing cohorts (Meekan and Fortier 1996). Variations in larval density may also contribute to differences in year-class strength in this species (Duffy-Anderson *et al.* 2002), though the affects of this form of pre-recruitment mortality may be comparatively small relative to other forcing factors.

As early juveniles, Pacific cod move toward the bottom and become demersal. Nursery habitats for juvenile Pacific cod are the shallow Alaskan coastal waters, where Pacific cod occur in highest abundances at moderate depths (15-20 m) (Abookire and Piatt 2005). Juvenile Pacific cod appear to have fairly specific habitat requirements, and they may have an affinity for structure. Juvenile cod have been shown to be associated with eelgrasses (Laur and Haldorson 1996), sea cucumber mounds (Abookire and Piatt 2005), and macroalgae. Consequently, Pacific cod may be sensitive to small-scale variations in spatial heterogeneities, and density-dependent recruitment in Pacific cod between age-0 and age-1 could be influenced by the availability and/or extent of nursery habitat (Fraser *et al.* 1996).

Juvenile cod diets in the Gulf of Alaska consist of small calanoid copepods, larval barnacles and crabs, mysids, worms, and gammarid amphipods, which suggest that Pacific cod feed on benthic and epibenthic as well as pelagic prey (Abookire *et al.* 2007). Juvenile cod demonstrate shifts in habitat preference with length, which may be related to changes in either foraging opportunity or predation vulnerability. Regardless, variations in growth and or survival associated with differences in habitat use could affect overall recruitment.

Trophodynamic (species interactions) shifts in the North Pacific ecosystem could also influence recruitment in Pacific cod. Bailey (2000) has shown that recruitment control of walleye pollock, another North Pacific gadid, shifted from the larval to the juvenile stage in the Gulf of Alaska, primarily due to increased predation by arrowtooth flounder on immature pollock. Pacific cod juveniles may be similarly vulnerable to the effects of increased groundfish predation, resulting in additional density-dependent regulation during the juvenile stage, which would be superimposed on that associated with habitat limitation.

The shifting distributions of adult Pacific cod throughout the year indicate seasonal migrations. Pre-spawning Pacific cod occur primarily over the inner and middle shelves of the Bering Sea (<30-100 m depths) in summer. In winter (January-March) Pacific cod appear to aggregate in major spawning areas between Unalaska and Unimak Islands in the eastern Aleutian Islands and near the Shumagin Islands. Postspawning dispersal occurs in summer when Pacific cod move from deep off-shelf waters to shallower depths on the eastern Bering Sea shelf (Shimada and Kimura ,1994). Pacific cod may be disproportionally vulnerable to the effects of fishing since they form large spawning aggregations and demonstrate some spawning site fidelity. Adult Pacific cod are opportunistic feeders and eat both invertebrate (shrimp, crabs, squid) and vertebrate prey (piscivory) (Yang 2003).

ABC as Recommended in the Most Recent Stock Assessments

In 2007, a major effort was undertaken to consider a large number of alternative assumptions in the EBS assessment model. This effort included a technical workshop that reviewed 40 model configurations and resulted in 40 suggestions for further investigation, examination of an enormous array of alternative models before selecting four models to present in the preliminary SAFE report, and examination of another enormous array of alternative models before selecting four models to present in the final SAFE report (Thompson *et al.* 2007). OFL for 2007 was 207,000 mt and ABC was recommended at 176,000; 2008 OFL and ABC recommendations were the same. The Council set TAC for 2007 at 170,720 mt and the same for 2008.

In the GOA, using the Tier 5 formula, the 2008 ABC was set at 66,493 mt, based on a 2007 trawl survey biomass of 230,310mt.

4.2.5.3 Atka Mackerel

Recruitment

The recruitment history of Atka mackerel is characterized by variable but fairly good recruitment throughout the time series of stock assessment estimates. The strong 1999 year class is most notable followed by the 1988, 1977, and 2001 year classes (Lowe *et al.* 2007). The most recent stock assessment estimates above average (greater than 20% of the mean) recruitment from the 1977, 1986, 1988, 1992, 1995, 1998, 1999, 2000, and 2001 year classes (Lowe *et al.* 2007). Given the history of variable, but widespread consistent recruitment for BSAI Atka mackerel, it is more likely that recruitment is largely driven by environmental factors than fishery management measures. Based on basin-wide North Pacific climate indices, there appears to have been a major regime shift in 1976/77, and a minor regime shift in 1988/89 (Boldt 2005, Hare and Mantua 2000, King 2005). There is some uncertainty if there was a regime shift in 1999 given the variability in environmental indices since 1998 (Rodionov *et al.* 2005). These hypothesized regime shifts coincide with the three strongest Atka mackerel year classes, however, it should noted that the mechanisms which produce successful recruitment are unknown.

ABC as Recommended in the Most Recent Stock Assessments

In 2008, BSAI Atka mackerel were placed into Tier 3a of the ABC/OFL definitions, which requires reliable estimates of biomass, $B_{40\%}$, $F_{35\%}$, and $F_{40\%}$. Under the definitions and current stock conditions, the overfishing fishing mortality rate is the $F_{35\%}$ rate which was estimated to be 0.398 for Atka mackerel and equated to a yield of 71,400 mt (Lowe *et al*. 2007b). The maximum allowable fishing mortality rate for ABC (F_{ABC}) is the $F_{40\%}$ rate which was estimated to be 0.331 for Atka mackerel in 2008, which translated to a yield of 60,700 t (Lowe *et al*. 2007b). The 2008 TAC was set equivalent to ABC at 60,700 t.

Gulf of Alaska Atka mackerel fall into Tier 6 of the ABC/OFL definitions, which defines the overfishing level as the average catch from 1978 to 1995, and that ABC cannot exceed 75% of the OFL. The average annual catch from 1978-95 is 6,200 mt; thus ABC cannot exceed 4,700 mt. The current ABC recommendation from the stock assessment is equal to the maximum prescribed under Tier 6; however, the stock assessment suggests that prudent management is still warranted and reiterated the rationale as given in the past for a TAC to provide for anticipated bycatch needs of other fisheries, principally for Pacific cod, rockfish and pollock, and to only allow for minimal targeting (Lowe *et al*. 2007a). The 2007 and 2008 TACs for GOA Atka mackerel were 1,500 mt.

4.2.5.4 Pacific Herring

Pacific herring (*Clupea pallasi*) is a marine fish species with moderate fecundity producing thousands of eggs per individual spawner (Paulson and Smith 1977). Pacific herring spawn in the spring in near shore regions throughout the Gulf of Alaska and the Bering Sea. In the GOA, major spawning locations occur near Sitka, Alaska and Prince William Sound (Williams and Quinn 1998, Hulson *et al.* 2007). In the

Bering Sea, major spawning concentrations can be found in Bristol Bay near the village of Togiak and in Norton Sound (Funk and Rowell, 1995; Williams and Quinn 1998). In the Bering Sea, herring migrate to one of two major wintering grounds located in northern and southern regions of the outer domain (Tojo *et al.* 2007). Comparison of recruitment time series of Pacific herring across the northwest Pacific reveals that this species exhibits episodic recruitment events that show some evidence of synchrony at a regional scale (Williams and Quinn 2000a).

Several hypotheses have been advanced to explain recruitment trends in Pacific herring. Quinn *et al.* (2001) and Marty (2003) reported that disease may have contributed to the decline in spawning biomass of Pacific herring in Prince William Sound. Disease may have resulted in a weakened condition due to inadequate energy stores resulting from poor feeding conditions (Hulson *et al.* 2007). An alternative hypothesis suggests that large-scale shifts in climate forcing can influence a variety of oceanographic factors including: timing of production, metabolic rate, larval transport, prey availability, and probability of encounters between predator and prey. Climate shifts have been recorded in the North Pacific in 1977, 1989 (Hare and Mantua, 2000). The most recent shift in atmospheric forcing occurred in 1998 with spatially differing impacts on ocean conditions in the Gulf of Alaska and Bering Sea (Bond and Overland, 2004). Intra-species competition and predator prey interactions are may also influence recruitment patterns for Pacific herring.

Shifts in large scale atmospheric forcing appears to influence the structure of marine fish communities in the western central Gulf of Alaska ecosystem through its role in determining the timing of peak production. Species that spawn in the winter will be favored by periods of early peak production, while species that spawn in the spring and summer will be favored by periods of delayed production (Mackus *et al.* 1997, Anderson and Piatt 1999).

Environmental forcing can influence a variety of oceanographic factors governing survival during the early life history period. Tanasichuk and Ware (1987) found that temperature affected fecundity and egg size. Alderdice and Hourston (1985) found temperature influenced embryonic survival rates. Williams and Quinn (2000b) found supplementing a Ricker type spawner recruit relationship with sea surface or air temperature produced an improvement to forecasts of Pacific herring recruitment. Climate shifts can influence major transport corridors for Pacific herring. Wespestad (1991) found recruitment trends of Togiak region were related to local wind conditions.

Detailed studies of Prince William Sound reveal that interspecific competition for common prey resources can result in complex recruitment patterns (Norcross *et al.* 2001). In Prince William Sound, Cooney (1993) noted that pollock predation could explain some of the observed trends in juvenile salmon and Pacific herring survival. They suggested that years of high copepod abundance were associated with high juvenile salmon survival because pollock relied on an alternative prey resource.

At finer spatial scales prey resources for forage fish may be prey limited leading to resource partitioning to minimize competition between forage fish species that occupy similar habitats. Willette *et al.* (1997) examined the diets of juvenile walleye pollock, Pacific herring, pink salmon and chum salmon in Prince William Sound. Their study revealed that two species pairs (walleye pollock and Pacific herring, and pink and chum salmon) exhibited a high degree of dietary overlap. This finding suggests that in PWS, competition for food resources may occur within these pairs when food abundance is limited. Foy and Norcross (1999) found water transported into Prince William Sound influenced the spatial and temporal distribution of prey for age-0 Pacific herring within Prince William Sound resulting in fine scale partitions in the condition of age-0 Pacific herring within the sound.

Competition for prey and oceanographic factors influencing prey availability can influence the probability of over-wintering survival for juvenile herring. Juvenile herring rely on fat resources acquired during the summer growing season during the winter (Foy and Paul 1999). Interspecies competition for common prey can produce complex recruitment patterns.

4.3 Impacts of Human Activities on Steller Sea Lions

4.3.1 Subsistence Harvests of Steller Sea Lions

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 Steller sea lions in Alaska. Haynes and Mischler (1991) examined historical data on subsistence uses and summarized the limited data on contemporary harvest levels and uses. Available information indicated that Steller sea lions were being harvested at several villages on the Bering Sea, in the Aleutian Islands, in Prince William Sound, the Kodiak Archipelago, and Lower Cook Inlet (Haynes and Mishler 1991). The most recent year for which subsistence data have been summarized is 2007. During 1992-2007, harvest data were collected through systematic retrospective interviews with hunters in at least 60 coastal communities throughout the range of Steller sea lions in Alaska (e.g., Wolfe *et al.* 2005, 2008, 2009). Results show the annual take (i.e., harvest plus struck and loss) decreasing substantially from about 550 sea lions in 1992 to less than 200 in 1996 followed by annual takes between 164 and 217 from 1997 to 2007 (see Table III-1 in NMFS 2008a). Wolfe *et al.* (2009) give the following totals for the reported take in subsistence hunts by year: 1992: 549; 1993: 487; 1994:416; 1995: 339; 1996:186; 1997:164; 1998:178; 1999: no data given; 2000: 171; 2001: 198; 2002:185; 2003:212; 2004: 216; 2005:218; 2006:187; 2007:217). Confidence intervals on these estimates are wide (e.g., the 95% confidence interval in 2007 of 147 to 324 Steller sea lions taken). Reported strike and loss rates are also relatively high. For example in 2007, 23.6% of the total reported take was struck but lost whereas 76.4% were harvested.

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. In 2007, the reported ratio was about equal (juveniles: 50.8%, adults: 49.2%). The ratio of males to females harvested in 2007 was 2.6:1.8, below the 5-year average of 4:1 during the previous five years. However, there is high uncertainty about the sex ratio of animals taken, as sex is reported as “unknown” for 30% of the animals taken. The reported sex ratio of harvested animals has varied substantially in different years (e.g., in 2004, the ratio was 1:8). In 2007, a total of 25.6 adult females were harvested, representing about 15% of the total harvest of known sex and age. This percentage has also varied substantially in different years.

During 1992-1995, the greatest numbers of Steller sea lions harvested were in the Pribilof Islands, whereas during 1996-2004 the harvest was greatest in the Aleutian Islands. In 2007, the highest level of reported take occurred also in the Aleutian Islands (total = 61.3), the Pacific Rim (villages in Lower Cook Inlet, Prince William Sound, and Resurrection Bay) (total = 47.5), and the Pribilof Islands (total = 31.3) (Table 14 in Wolfe *et al.* 2009). Only 6.1 Steller sea lions were reported taken from a single village in southeast Alaska. While the North Pacific Rim grouping in Wolfe *et al.* (2009) includes villages in Resurrection Bay and lower Cook Inlet, no Steller sea lions were reported taken from Homer northward in Cook Inlet or in Resurrection Bay in 2007. 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 Steller 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 Steller sea lion take of about 25-35 per year from 1998-2002, followed by a reduced take of 18 Steller sea lions in both 2003 and 2004 (Zavadil *et al.* 2005). 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 Steller 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), Steller 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). Native hunters take some Steller sea lions in Chukotka, but the number killed is unknown.

Current subsistence harvests represent a large proportion of the potential biological removal level calculated for the western DPS of Steller sea lion (Angliss and Outlaw 2005). However, subsistence harvests account for only a relatively small portion of the Steller sea lions lost to the population each year through natural mortality and are primarily young males which reduces the impact on the recovery of the population.

4.3.2 Commercial Harvest of Steller Sea Lions

Currently, no commercial harvest for Steller sea lions exists in the United States, but Steller 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, Atkinson *et al*. 2008). 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, with an average of 631 animals 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.

4.3.3 Incidental Take by Fisheries

4.3.3.1 Western DPS: U.S. Waters

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 Steller 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 mortality incidental to commercial fisheries has declined to approximately 30 animals per year (see below).

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 Steller sea lions are covered by observers.

The minimum estimated mortality rate incidental to commercial fisheries in 2002 was 29.5 Steller 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).

In a recent stock assessment for the western DPS of Steller sea lion, Angliss and Allen (2008) reported an estimated mean annual mortality rate in the observed fisheries of 25.8 (CV = 0.60) Steller sea lions per year from this stock. This estimated combined mortality from the Bering Sea and GOA groundfish trawl and Gulf of Alaska longline fisheries with mortality estimates from the Prince William Sound salmon gillnet fishery. The latter fishery, however, was last observed in 1991. Angliss and Allen (2008) provided a minimum estimated mortality rate incidental to U.S. commercial fisheries of 26.2 animals based on the aforementioned observer data and stranding data. However, currently used stranding data likely underestimates the actual rate of entanglement since “because not all entangled animals strand and not all stranded animals are found or reported” (Angliss and Allen 2009).

4.3.3.2 Western DPS: Russian Waters

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. Quantitative information on Steller 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

During October-December 2002, observers recorded the incidental take of Steller sea lions during a herring trawl fishery in the western Bering Sea. Preliminary estimates of the total number of Steller 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 Steller sea lions may be from (i.e., Aleutian, Commander, and Kuril Islands, and Kamchatka). The majority, if not all, of these Steller sea lions were subadult males. Observers on vessels fishing for herring during the 2002-2003 season in the Bering Sea in Russia observed 15 Steller sea lions killed[16] (Waite and Burkanov 2006).

---

[16] The most common prey found in the stomachs of these incidentally caught sea lions was Pacific herring (FO=100%) and pollock (FO=76.9%) (Waite and Burkanov 2005).

4.3.3.3 Eastern DPS

In the most recent stock assessment for the eastern DPS, Angliss and Allen (2008) reported that fishery observers monitored four commercial fisheries during the period from 1990 to 2005 in which incidental take of Steller sea lions from this stock was reported. These fisheries were: the Gulf of Alaska sablefish longline fishery, the California (CA)/Oregon (OR) thresher shark and swordfish drift gillnet fishery, the WA/OR/CA groundfish trawl fishery, and the Northern Washington (WA) marine set gillnet fishery. In 2001-2003, one Steller sea lion was observed killed in each year in the WA/OR/CA groundfish trawl. There have been no observed serious injuries or mortalities incidental to the CA/OR thresher shark and swordfish drift gillnet fishery in recent years (Angliss and Allen 2009, citing Carretta 2002, Carretta and Chivers 2003, Carretta and Chivers 2004).

No Steller sea lion mortalities have been reported by observers in the Gulf of Alaska sablefish longline since 2000 (Perez unpubl. ms. as cited in Angliss and Allen 2009). A mean estimated annual mortality level of 0.8 was calculated based on the level of the aforementioned observer takes in combination with a mortality that occurred in an unmonitored haul (see Table 5 of Angliss and Allen 2009). No data are available after 1998 for the northern Washington marine set gillnet fishery. While mortalities have been associated with drift gillnet and set gillnet fisheries in Washington and Oregon in previous decades, none have been reported by observers monitoring these fisheries this decade (Angliss and Allen 2009). Based on available data, Angliss and Allen 2009) provided a minimum total annual mortality of eastern DPS Steller sea lions in both Canadian and U.S. waters of 1.4 (stranding data =0.6 and observer data = 0.8). It is likely the rate of entanglement is substantially underestimated (as with the western DPS, the rate of entangled animals is likely underestimated.

Angliss and Allen (2008) also reported that Steller sea lions were killed during commercial salmon farming operations in British Columbia. Between 1999-2003, an average of 47.75 Steller sea lions were killed annually in this fishery (Olesiuk 2004). Angliss and Allen reported that, as of 2004, aquaculture facility personnel are no longer permitted to shoot Steller sea lions.

4.3.4 Intentional and Illegal Shooting

In some areas Steller sea lions were deliberately shot 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 Steller sea lions was significantly less than during 1978 (Wynne 1990). However, the two studies are not directly comparable due to differences in methodologies and periods and locations sampled. During the 1960s, Steller 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). It is difficult to estimate this take. Steller sea lions found shot are not assumed to be illegal, as the animal may have been shot and lost by a subsistence hunter. More recently, Kruse and Huntington (2009) documented the estimated level of intentional shooting of SSLs in the central and western GOA, AI, and EBS regions (1974-1990) based on personal interviews with local fishermen and regional residents. This local and traditional knowledge of 36 individuals indicates shooting was fairly common but varied across time, region, fishery, and other factors.

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).

4.3.5 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 Steller 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 Steller 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 Steller 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. Observations by researchers in southeast Alaska indicate higher numbers of Steller sea lions entangled in fishing gear and other marine debris than previously estimated (Pitcher *et al.* 2007).

4.3.6 Impact to Water Quality due to Human Population Growth in the Action Area

As the size of human communities increase, there is an accompanying increase in habitat alterations for housing, roads, commercial facilities, and other infrastructure. The impacts of these activities on landscapes and the biota they support increases as the size of the human population expands. The Alaska population has increased by almost 50% in the past 20 years, most of that increase has occurred in the Cities of Anchorage and Fairbanks. Outside of the City of Anchorage, few of the cities, towns, and villages would be considered urbanized. Despite low levels of industrialization in the action area, some commercial and industrial facilities in the action area have had, or have the potential for significant, adverse effects on the terrestrial, coastal, and marine environments, primarily because of their potential effects on water quality.

Four superfund sites occur in the action area: Adak Naval Air Station (Aleutians West), Elmendorf Air Force Base (Borough of Anchorage), Fort Richardson Army Base (Municipality of Anchorage), and the U.S. Department of Transportation's Standard Steel and Metals Salvage Yard (Municipality of Anchorage).

The Naval Air Station at Adak covers about 64,000 acres on the Island of Adak near the western end of the Aleutian Island archipelago. Adak Island became a military base in 1942 and has been controlled by the U.S. Navy since 1950. In 1986, the Navy identified 32 areas that potentially received hazardous substances, including chlorinated solvents, batteries, and transformer oils containing PCBs over a period of 40 years. Investigations on the island focused on two areas: the Palisades Landfill and Metals Landfill. Disposals had stopped at the Palisades landfill in the 1970s and the landfill was covered. The Metals landfill contains a hazardous waste pile under the Resource Conservation and Recovery Act (RCRA) and a closure plan is being developed for the site. A land exchange between Aleut Corp., the U.S. Navy, and the Department of the Interior transferred most of the former Naval facilities to the Aleut Corporation in March 2004.

The cities of Kodiak and Unalaska both have wastewater treatment plants, as do the City of Anchorage and several cities in the Kenai borough. Most of the industrial facilities in the action area (outside of Anchorage and the Kenai Borough) are involved in seafood processing. Canneries or land-based processors occur at Adak, Anchorage, Chignik, Cordova, Dillingham, Egegik, Emmonak,, False Pass, Homer, Kenai, King Cove, King Salmon, Kodiak, Larsen Bay, Nikiski, Ninilchik, Nome, St. Paul, Sand Point, Savoonga, Seward, Soldotna, Togiak, Toksook Bay, Unalaska, Valdez, and Whittier.

In the 1970s, fish and shellfish waste discharged from mobile and shore-based processors at Kodiak, Dutch Harbor, and Akutan polluted coastal waters around those communities (Jarvela 1986). In 1976, waste was discharged at Dutch Harbor. In 1983, the shore-based Trident Seafoods plant at Akutan released cod and crab wastes into Akutan Harbor before the plant was destroyed by fire. Sonar surveys of Akutan Harbor identified a waste pile that was about 7 m thick and 200 m in diameter. In 1998, the list of impaired waters that was prepared by the Alaska Department of Environmental Conservation (ADEC) included water bodies in Cold Bay, Dutch Harbor, and Kodiak that had been impaired by seafood processing, logging operations, military materiel, or fuel storage. Although total maximum daily loads will not be developed for these facilities before this Biological Opinion is completed, the effects of these facilities appear to be localized and would not be expected to adversely affect threatened or endangered species under NMFS' jurisdiction.

As the human population expands, the risk of disturbance to listed species in the action area, especially Steller sea lions, also increases. Several studies have noted the potential adverse effects of human disturbance on Steller sea lions. Calkins and Pitcher (1982) found that disturbance from aircraft and vessel traffic has extremely variable effects on hauled-out Steller sea lions. Steller sea lion reaction to occasional disturbances ranges from no reaction at all to complete and immediate departure from the haulout area. The type of reaction appears to depend on a variety of factors. When Steller sea lions are frightened off rookeries during the breeding and pupping season, pups may be trampled or even abandoned in extreme cases. Steller sea lions have temporarily abandoned some areas after repeated disturbance (Thorsteinson and Lensink 1962), but in other situations they have continued using areas after repeated and severe harassment. Johnson *et al.* (1989) evaluated the potential vulnerability of various Steller sea lion haulout sites and rookeries to noise and disturbance and also noted a variable effect on Steller sea lions. Kenyon (1962) noted permanent abandonment of areas in the Pribilof Islands that were subjected to repeated disturbance. A major Steller sea lion rookery at Cape Sarichef was abandoned after the construction of a light house at that site, but has been used again as a haulout after the light house was no longer inhabited by humans. The consequences of such disturbance to the overall population are difficult to measure. Disturbance may have exacerbated the decline, although it is not likely to have been a major factor.

4.3.7 Disturbance

The possible impacts of various types of disturbance on Steller sea lions have not been well studied, yet the response by Steller 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 will cause hauled out Steller sea lions to go into the water, and can cause some animals to move to other haulouts (Calkins and Pitcher 1982, 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 Steller sea lions can observe the approach have less effect than fast approaches and a sudden appearance. Steller 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, Thorsteinson and Lensink 1962). When humans set foot on a rookery or haulout, the response by Steller 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, Kucey 2005). 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).

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. 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. The consequences of such disturbance to the overall population are difficult to measure. Disturbance may have contributed to or exacerbated the decline, although it is not likely to have been a major factor.

4.3.8 Impacts of Oil and Gas Development

Oil and gas leasing and exploration have been proposed or have occurred in many parts of the Steller sea lion range, including California, Cook Inlet, the Gulf of Alaska, the North Aleutian Basin, and Russia. Since the 1970s in these areas, the Minerals Management Service (MMS) has made blocks of the Outer Continental Shelf off Alaska available for oil and gas leases in Cook Inlet primarily, but also other locations in the Steller sea lion range. Except for two active leases in lower Cook Inlet, all of the leases in these areas have either expired or been relinquished. Lease sales are currently proposed in the MMS 5-year plan for Cook Inlet, but a lease sale for the north Aleutian Basin is no longer to be offered. On October 15, 1993, NMFS completed a Biological Opinion on the Cook Inlet lease sale (lease sale Number 149), which concluded that the lease and associated exploration activities were not likely to jeopardize the continued existence of any listed or proposed species, nor were they likely to destroy or adversely modify critical habitats. That Biological Opinion recognized the proximity of the lease area to important Steller sea lion rookeries and haulouts in Shelikof Strait, the use of the Strait by foraging Steller sea lions, and its value as an area of high forage fish production, but recognized the low probability of oil spills during exploration activities. NMFS (2003) reached the same conclusion after a programmatic consultation on the MMS proposed lease sales 191 and 199. In 1995, NMFS conducted a section 7 consultation with the MMS and concluded that the lease sale and exploration activities for the proposed oil and gas Lease Sale Number 158, Yakutat were not likely to jeopardize the continued existence of any listed or proposed species, nor were the activities likely to destroy or adversely modify critical habitats (NMFS 1995).

The State of Alaska also manages oil and gas leasing in Alaska. In 1896, oil claims were staked at Katalla approximately 50 miles south of Cordova. Oil was discovered there in 1902. An on-site refinery near Controller Bay produced oil for over 30 years. The refinery burned down in 1933 and was not replaced. Exploration in Cook Inlet began in 1955 on the Kenai Peninsula in the Swanson River area, and oil was discovered in 1957. Today, a number of active fields produce oil in Cook Inlet, all of which is processed at the refinery at Nikiski on the Kenai Peninsula. Estimated oil reserves in Cook Inlet are 72 million barrels of oil. Currently there are additional lease sales planned through 2012 for the Cook Inlet area (2 sales) and none for the North Aleutian Basin area. Based on a U.S. Court of Appeals decision, on April 6, 2010 MMS issued a revised leasing scenario for Alaska through 2012. MMS reported that the preliminary decision is to remove five sales from the schedule, sales 209 and 217 in the Beaufort Sea, 212 and 221 in the Chukchi Sea and 214 in the North Aleutian Basin, including Bristol Bay.

One of the biggest oil and gas developments in the world is being developed about 15 km offshore Sakhalin Island in the sea of Okhotsk (Sakhalin II) where massive oil field and gas field developments feed two 800 km pipelines to a liquefied natural gas plant, an oil export terminal, and waiting tankers.

Regarding oil and gas in the range of the eastern DPS, there are multiple active leases and oil-producing platforms in areas off parts of California, including the Santa Barbara channel, the Long Beach-San Pedro Basin area, and the Santa Maria Basin (maps available at http://www.mms.gov/omm/pacific/lease/maps.htm). Within the proposed action area, the MMS Draft Proposed Program for 2010-2015 includes proposed lease sales in the North Aleutian Basin (2 sales), Cook Inlet (2 sales), Southern California (2 sales), and Northern California (1 sale) (see schedule at http://www.mms.gov/5-year/PDFs/2010-2015/DPP%20FINAL%20(HQPrint%20with%20landscape%20maps,%20map%2010).pdf). As the MMS has not yet consulted with NMFS on these sales and associated activities, we do not consider their potential effects further, although we note that recently the opportunity for lease sales in the North Aleutian Basin have been canceled by Court order and subsequent Secretarial decision.

Oil spills are expected to adversely affect Steller sea lions if they contact individual animals, haulouts, or rookeries when occupied, or large proportions of major prey populations (MMS 2003). It is well-documented that exposure of at least some mammals to petroleum hydrocarbons through surface contact, ingestion, and particularly inhalation can be harmful. Surface contact with particularly the low-molecular weight fractions can cause temporary or permanent damage of the mucous membranes and eyes, and/or epidermis (see Section IV.F.3 in MMS 2006). Contact with crude oil can damage eyes. Corneal ulcers and abrasions, conjunctivitis, and swollen nictitating membranes were observed in captive ringed seals place in crude oil-covered water and in seals in the Antarctic after an oil spill. Corneal ulcers and scarring were observed in others captured in oiled areas (Monnett and Rotterman 1989) and in oiled otters brought to post-Exxon Valdes oil spill treatment centers. Ingestion of petroleum hydrocarbons can lead to subtle and progressive organ damage or to rapid death. Inhalation of volatile hydrocarbon fractions of fresh crude oil can damage the respiratory system, cause neurological disorders or liver damage, have anesthetic effects, and if accompanied by excessive adrenalin release, can cause sudden death (e.g., Geraci 1988). Many PAHs are teratogenic and embryotoxic in at least some mammals. Ingestion of oil by pregnant females can negatively affect the birth weight of their young. As summarized by MMS (2006:90): “There are few post spill studies with sufficient details to reach firm conclusions about the effects, especially the long-term effects, of an oil spill on free-ranging populations of marine mammals.”

Steller sea lions would be particularly vulnerable if large amounts of crude oil coated rookeries when young pups were on the rookeries or oil contaminated concentrations of prey. The extent to which Steller sea lions avoid areas that have been oiled is not known. Groups and individual Steller sea lions observed in Prince William Sound and the Gulf of Alaska after the Exxon Valdez oil spill did not appear to avoid oiled areas (Calkins and Becker 1990). Steller sea lions were sighted swimming in or near oil slicks, oil was seen near numerous haulout sites, and oil fouled the rookeries at Seal Rocks and Sugarloaf Island (Calkins *et al.* 1994). All of the Steller sea lions collected in Prince William Sound in October 1989 had high enough levels of metabolites of aromatic hydrocarbons in the bile to confirm exposure and active metabolism at the tissue level. As noted above, no evidence indicated damage caused to Steller sea lions from toxic effects of the oil (Calkins *et al.* 1994). However, studies after Exxon Valdez oil spill on Steller sea lions were hampered by a lack of baseline on key health, condition and population parameters to enable the type of detailed study needed to discern chronic effects. Because of the ongoing decline in the western DPS, it was determined it would be difficult or impossible to distinguish an oil spill effect and thus, the focused studies needed to determine if there were, or were not, long-term effects were not undertaken. However, it should be noted that the subarea most directly impacted by the EVOS (i.e., eastern GOA) has a sea lion subpopulation that is recovering at a robust rate.

4.3.9 Impacts of Research

We give further detail on effects of research in the Effects of the Proposed Action chapter of this document, and the reader is also referred to a recent Biological Opinion on Steller sea lion research (see http://www.nmfs.noaa.gov/pr/pdfs/permits/eis_opinion.pdf).

Steller sea lions have been killed for scientific research since the end of World War II (Thorsteinson and Lensink 1962, Calkins and Pitcher 1982, Calkins and Goodwin 1988, and Calkins *et al.* 1994). In 1959, 630 Steller sea lions bulls were killed in an experimental-commercial harvest and provided life history information (age, size, reproductive condition, food habits). Between 1975 and 1978, 250 Steller sea lions were killed in nearshore waters and on rookeries and haulouts of the GOA; their stomachs were removed and examined for food content, reproductive organs were preserved for examination, blood samples were taken for disease and parasite studies, body measurements were recorded for growth studies, skulls were retained for age determination, tissue samples were preserved for elemental analysis and pelage samples were taken for molt studies. In 1985 and 1986, 178 Steller sea lions were killed in the GOA and southeast Alaska to compare food habits, reproductive parameters, growth and condition, and diseases, with the same parameters from animals that were collected in the 1970s. The study was designed to address the problem of declining numbers of Steller sea lions in the North Pacific and particularly in the GOA. More recently, sixteen Steller sea lions were killed for a Natural Resources Damage Assessment study following the Exxon Valdez oil spill. Thus, in the aforementioned studies, a total of 1,074 Steller sea lions were intentionally killed for research.

Researchers have been conducting surveys and behavioral research on Steller sea lions for many decades. However, methods used during research, level of research being undertaken, and the number of people involved has changed over the years. Currently, information available in annual reports indicates that Steller sea lion populations are not adversely affected by this research, although individual animals may be disturbed and a small number of animals are accidentally killed. In 1998 for example, 48,000 Steller sea lions were estimated to have been disturbed by these investigations, 384 pups were captured, tagged, and branded, and there were no mortalities observed. In 1997, 31,150 Steller sea lions were approached by researchers, 14,550 were disturbed, 137 were captured, and 121 were tagged, but there were no known mortalities. The studies conducted in 1996 had similar effects, although one Steller sea lions died during the study. In 1995, 7,500 Steller sea lions were disturbed and none were reported to have died.

Disturbance of Steller sea lions could potentially occur, but is unlikely to occur, during aerial surveys. It is very likely to occur during capture of animals for branding, tagging, and sample collection, and may occur during close vessel approaches to rookeries and haulouts to observe branded animals. Steller sea lions are occasionally killed accidentally in the course of some types of scientific research activities. For example, Steller sea lions may be killed accidentally during anesthesia. Suffocation can result when animals are herded. Loughlin and York (2000) estimated that about three animals per year died due to research on the western DPS. However, the recent average is about 1-2 for the western DPS (NMFS unpublished data). The potential exists for additional unobserved mortality to occur following the completion of research activities. Data are not sufficient to derive reliable estimates. Pups are the age-class most vulnerable to disturbance from research activities.

On May 26, 2006, a District Court judge in Washington, D.C. issued an opinion and a court order relative to a law suit filed against NOAA by the Humane Society of the United States. The Humane Society argued that NOAA did not follow proper procedures under the National Environmental Policy Act before issuing permits to six entities to conduct Steller sea lion research in Alaska. The court sided with the Humane Society and directed NOAA to immediately vacate all six existing permits and prepare a full EIS, per NEPA requirements. A settlement agreement was reached in June 2006 which allowed limited "No Take", "Low Take Non-Invasive Activities", and Low Take Handling and Release of Captured Animals to continue while NMFS completed an EIS on the research program. As summarized in the proposed action, that EIS was finalized in 2007 and a Summary Document has been written to update the EIS.

4.3.10 Summary of Known Direct Non-Research Related Take of Steller Sea Lions

The information below represents the best estimate of the sum of direct human related mortality factors as developed by the Steller Sea Lion Recovery Team (Recovery Team) up to 2004 (NMFS 2006). We have no evidence that indicates that typical rates of take from 2004-2008 differ substantially from those in the 2000-2004 period except that rates of intentional illegal shooting may have declined. 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 Recovery Team extended and extrapolated the Trites and Larkin estimates for the PVA analysis in NMFS (2008; Appendix A). This resulted in the values below, where the cell entries are the accumulated number of deaths attributed to each cause over the interval. 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.

	Time Period				
Mortality Source	**1958-1977**	**1977-1985**	**1985-1989**	**1989-2000**	**2000-2004**
Non-subsistence harvest	45,178	0	0	0	0
Subsistence harvest	9,995	2,900	850	3,300	750
Shooting	12,716	8,277	1,870	2,200	1,000
Incidental catch and entanglement	28,191	14,461	2,255	330	150
Total	**96,080**	**25,638**	**4,975**	**5,830**	**1,900**

4.3.11 Early Environmental Observations

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). A number of early observations of the North Pacific ecosystem have been previously cited in this opinion, especially those relating to Steller sea lion food habits. Other reviews provide quotes from various early sources as well (Causey *et al.* 2005, NRC 2003). In this section we provide an overview of some of the commonly cited observations. 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. They do provide a sense of the variability in the ecosystem and should remind us that the environment is not static.

Nelson (1887) reported that Steller 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, Maschner *et al.* 2010). In reports from expeditions to the Pribilof and Aleutian islands, researchers found Alaskan pollock in the Bering Sea and neighboring waters south to Sitka and the Kurils to be “excessively abundant, swimming near the surface and furnishing the great part of the food of the fur seal” (Jordon and Evermann 1902).

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 are 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 ft 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 and that "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."
While 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).

4.3.12 Summary of Fishery Conservation Measures for Steller Sea Lions

This section describes the conservation measures that have been undertaken to reduce impacts of fisheries on Steller sea lions. These measures are part of the environmental baseline. It is unknown whether fishery conservation measures have been effective in reducing threats to Steller sea lions. Moderating declines and recent population increases following these measures have resulted in debates about cause and effect. Unlike direct impacts, indirect impacts through competitive interactions are impossible to unequivocally demonstrate with currently available data. The increasing Steller sea lion population trend is correlated with fishery conservation measures taken since the 1990s, but it is unknown whether the relationship is causal (Hennen 2006).

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). Fishery removals have the potential to reduce the availability of these species to Steller sea lions at a variety of spatial and temporal scales (NMFS 2000, 2001). Reduced prey availability can represent an acute or chronic threat to Steller sea lion populations (Trites and Donnelly 2003). 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 NPFMC and implemented by NMFS may have indirectly contributed to Steller sea lion conservation efforts (see below for a detailed list of actions).

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.

The implementation of conservation measures, in both the early 1990s and the late 1990s early 2000s, is correlated with a reduction in the rate of decline of the western DPS of sea lions. That is, there was a change in trend in abundance for the western DPS from a decline on the order of 5% per year in the 1990s to an increase on the order of 1.4% per year from 2000 to 2009 that is statistically significant (J. VerHouf, pers.comm, National Marine Mammal Laboratory, AFSC). However, the information necessary to determine if the conservation measures actually contributed to the change in population trend is not currently available.

A suite of fishery conservation measures was implemented in 2002 after being reviewed under an ESA section 7 consultation (NMFS 2001). These measures are described in detail in the 2001 Biological Opinion (NMFS 2001) and its Supplement (NMFS 2003). 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.

The 2002 measures provided increased protection for near-shore critical habitat areas based on an analysis that closely examined satellite telemetry data and on information on 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 Steller 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 (available at the time), and an analysis of the important foraging areas for Steller sea lions based on those data. Further, a summary of the federal fishery management measures that may have affected Steller sea lions is provided in NPFMC (2005a, 2005b) and described in this opinion in Chapter 2.

The following is a compilation of the conservation measures implemented by NMFS since the development of the BSAI and GOA groundfish FMPs. Further description of conservation actions are provided in Section 2.5.2. Biological opinions are described in Section 1.2.

1. In 1989, the Environmental Defense Fund and 17 other environmental organizations petitioned NMFS for an emergency rule listing all populations of Steller sea lions in Alaska as endangered and to initiate a rulemaking to make that emergency listing permanent.
2. On April 5, 1990, NMFS issued an emergency interim rule (55 FR 12645) to list the Steller sea lion as a threatened species under the ESA and established protective regulations as emergency interim measures to begin the recovery process. The rule established the following:
    - Monitoring of incidental take and monthly estimates of the level of incidental kill of Steller sea lions in observed fisheries.
    - Aggressive enforcement of protective regulations, especially as they relate to intentional, lethal takes of Steller sea lions.
    - Establishment of a Recovery Team to provide recommendations on further conservation measures.

- Prohibition of shooting at or within 100 yards of Steller sea lions (this did not apply to Alaska native subsistence hunting).
- Establishment of 3 nm "no-approach" buffer zones around the principle Steller sea lion rookeries in the GOA and Aleutian Islands.
- Reduction of incidental kill quota from 1,350 to no more than 675 Steller sea lions.

3. On November 26, 1990, NMFS issued the final rule to list the Steller sea lion as threatened under the ESA (55 FR 49204).
4. On January 7, 1991, NMFS issued a final rule to implement regulations to amend the BSAI and GOA FMPs that limited pollock roe-stripping and seasonally allocated the pollock TAC in the BSAI and GOA groundfish (56 FR 492). For BSAI fisheries, the pollock TAC was divided between an A (roe) season and a B season (summer-fall). In the GOA fisheries, the pollock TAC for the Central and Western (C/W) Regulatory areas was divided into 4 equal seasons. NMFS noted in the proposed rule (55 FR 37907, September 14, 1990) that "shifting fishing effort to later in the year may reduce competition for pollock between the fishery and Steller sea lions whose populations have been declining in recent years."
5. On June 19, 1991, NMFS issued an emergency interim rule to ensure that pollock fishing did not jeopardize the continued existence or recovery of the threatened Steller sea lion (56 FR 28112). The rule contained the following measures to protect Steller sea lions:

    - Allocated the pollock TAC for the combined western and central Regulatory areas equally between two subareas located east and west of 154°W,
    - Limited the amount of unharvested pollock TAC that may be rolled over to subsequent quarters in a fishing year, and
    - Prohibited fishing with trawl gear in the EEZ within 10 nm of 14 Steller sea lion rookeries.

6. On January 23, 1992, NMFS issued a final rule to implement amendments 20/25 to the BSAI and GOA groundfish FMPs (57 FR 2683). This replaced prior emergency rules, and extended some of the protections. The amendments contained the following protections:

    - Prohibited trawling year-round within 10 nm of 37 Steller sea lion rookeries in the GOA and BSAI,
    - Expanded the no-trawl zone to 20 nm for 5 of these rookeries from January 1 through April 15 each year,
    - Established 3 GOA pollock management districts, and
    - Imposed a limit on the amount of an excess pollock seasonal harvest that may be taken in a quarter in each district.

7. On January 7, 1993 NMFS released the final Steller Sea Lion Recovery Plan. Section 4(f) of the ESA requires that NMFS develop and implement plans for the conservation and survival of endangered and threatened species. NMFS appointed a Steller Sea Lion Recovery Team to draft the Recovery Plan in 1990. The draft Recovery Plan was released for public review and comment on March 15, 1991. NMFS responded to comments received and provided notice on January 7, 1993 that the final Recovery Plan was available (58 FR 3008).

8. On March 12, 1993, NMFS issued a final rule to implement a seasonally expanded no-trawl zone around the Ugamak Island Steller sea lion rookery in the eastern Aleutian Islands during the pollock roe fishery season in the BSAI (58 FR 13561). The expanded buffer zone around Ugamak Island was expected to better encompass Steller sea lion winter habitats and juvenile foraging areas in this portion of the southeastern Bering Sea shelf during the BSAI winter pollock fishery.
9. On July 13, 1993, NMFS issued a final rule to implement regulations (BSAI FMP amendment 28) that subdivided the Aleutian Islands sub-district into three subareas (Areas 541, 542, 543) (58 FR 37660). This action was taken because of concerns that concentrated fishery removals, particularly Atka mackerel, in the eastern Aleutian Islands could cause localized depletions. While dispersal of the Atka mackerel TAC was initiated to conserve fishery resources, it was also consistent with the conservation objectives for Steller sea lions.
10. On August 27, 1993, pursuant to the ESA (§1533(a)(3)(A)), NMFS designated critical habitat for Steller sea lions (58 FR 45269).
11. On November 1, 1993, NMFS initiated a status review of Steller sea lions to determine whether a change in classification to endangered was warranted (58 FR 58318). NMFS solicited comments and biological information concerning the status of Steller sea lions to be used in its review.
12. On November 29-30, 1994, NMFS convened the Steller Sea Lion Recovery Team specifically to consider the appropriate ESA listing status for Steller sea lions and to evaluate the adequacy of ongoing research and management programs. The Recovery Team recommended that NMFS list the Steller sea lion as two separate population segments, split to the east and west of 144°W. The Recovery Team recommended that the western population segment be listed as endangered and the eastern population segment be listed as threatened.
13. In October 1995, NMFS issued a proposed rule to list the western population of the Steller sea lion as endangered.
14. On May 5, 1997, NMFS reclassified Steller sea lions as two distinct population segments under the ESA (62 FR 24345). The population segment west of 144°W (near Cape Suckling, AK) was reclassified as endangered, while the population east of 144°W was maintained as threatened.
15. On March 17, 1998, NMFS issued regulations to create a separate forage fish category (Amendments 36/39 to the BSAI and GOA FMPs; 63 FR 13009). Directed fishing for forage fish was prohibited at all times in Federal waters of the BSAI and GOA. The intended effect of this action was to prevent the development of a commercial directed fishery for forage fish, a critical food source for many marine mammal, seabird, and fish species.
16. On June 11, 1998, NMFS issued a final rule to reallocate pollock TAC in the W/C Regulatory areas of the GOA by moving 10% of the TAC from the 3rd fishing season, which started on September 1, to the 2nd fishing season, which started on June 1 (63 FR 31939). This seasonal TAC shift was a precautionary measure intended to reduce the potential impacts on Steller sea lions.
17. On January 22, 1999, NMFS issued a final rule to spatially and temporally distribute the Atka mackerel TAC in the Aleutian Islands subarea. This was a precautionary approach to reduce the probability of localized depletions of Atka mackerel inside Steller sea lion critical habitat. The amendment implemented both spatial and temporal redistribution of the Atka mackerel TAC.
18. On January 22, 1999, NMFS published an emergency interim rule (64 FR 3437) implementing the RPA from the December 3, 1998 Biological Opinion which concluded that the pollock fisheries as proposed were likely to jeopardize the continued existence of the endangered western population of Steller sea lions and adversely modify its critical habitat. The rule created (1) temporal dispersion of fishing effort, (2) spatial dispersion of fishing effort, and (3) pollock trawl

exclusion zones around Steller sea lion rookeries and haulouts. On July 21, 1999, NMFS extended the emergency rule through December 31, 1999 (64 FR 39087), with revisions to include specifications for the B and C pollock seasons in the Bering Sea.

19. In October 1999, NMFS conducted additional analyses of the RPAs and developed revised final RPAs (RFRPAs) to be incorporated into the December 3, 1998 Opinion as compelled by a court order. The RFRPAs provided a detailed set of alternative management measures that would avoid the likelihood that the pollock fisheries would jeopardize the continued existence of the western population of Steller sea lions or adversely modify its critical habitat. Season dates, pollock catch percentages within critical habitat, and no pollock trawling areas were modified from the original RPAs.
20. On January 25, 2000, NMFS published an emergency interim rule (65 FR 3892) implementing the RFRPAs from the December 3, 1998, Biological Opinion as modified in October 1999. On June 12, 2000, NMFS extended the emergency interim rule through December 31, 2000 (65 FR 36795).
21. On August 9, 2000, NMFS closed all Steller sea lion critical habitat to all groundfish trawling to comply with a U.S. District Court Order (65 FR 49766, August 15, 2000).
22. On November 30, 2000, NMFS issued a Biological Opinion on the FMPs, which determined that the pollock, Pacific cod, and Atka mackerel fisheries were likely to jeopardize the continued existence of the western DPS of Steller sea lions and to adversely modify its critical habitat. It contained a RPA that included large fishery closure areas, harvest limits and seasonal distribution of harvest for the pollock, Pacific cod, and Atka mackerel fisheries. Before the RPA could be implemented, the President signed Public Law 106–554 on December 21, 2000, which contained a one year timetable to phase in the RPA. This year provided the Council with time to develop alternative conservation measures that would avoid jeopardy and adverse modification of critical habitat for Steller sea lions.
23. On January 1, 2001, in accordance with Public Law 106–554, the 2001 BSAI and GOA groundfish fisheries were initially managed in accordance with the fishery management plans and Federal regulations in effect for such fisheries prior to July 15, 2000 (i.e., prior to the trawling ban in critical habitat, thus lifting the prohibition).
24. On January 22, 2001 NMFS published an emergency interim rule (66 FR 7276) under the MSA which replaced the initial fishery management regime of 2001 as provided in Public Law 106–554, section 209(c)(4), effective on January 18, 2001 (and corrected and amended March 20, 2001 (66 FR 15656), March 29, 2001 (66 FR 17083 and 17087), July 2, 2001 (66 FR 34852), July 17, 2001 (66 FR 37167) August 22, 2001 (66 FR 44073), and September 20, 2001 (66 FR 48371)). The emergency interim rules contained a suite of management measures that phased in certain provisions of the RPA from the 2000 Biological Opinion. The July 17, 2001 emergency interim rule implemented the Steller sea lion protection measures that were developed by the Council's RPA Committee and forwarded to NMFS for review and implementation.
25. In July 2001, the parties to the litigation concerning the Biological Opinions and the RFRPA (1998 Biological Opinion and subsequent October 1999 revision) filed a joint status report and agreed to stay further litigation until completion of the 2001 Biological Opinion in October 2001. A subsequent joint status report dated November 1, 2001, agreed to continue the temporary stay of litigation until January 18, 2002, when a follow-up status report would be filed with the court.
26. In October 2001, NMFS issued a Biological Opinion in (2001 Biological Opinion), which determined that the Steller sea lion protection measures developed by the RPA Committee and the Council were unlikely to jeopardize the continued existence of the western DPS of Steller sea lions or adversely modify its critical habitat. These measures were implemented by emergency

interim rule (67 FR 956, January 8, 2002, amended and corrected 67 FR 21600, May 1, 2002, and extended 67 FR 34860, May 16, 2002 and corrected July 10, July 19, and October 18, 2002 (67 FR 45671, 47472, and 64315, respectively).

27. On January 2, 2003 NMFS issued a final rule (68 FR 204), which implemented the Steller sea lion protection measures reviewed in the 2001 Biological Opinion (and corrected May 8, 2003 (68 FR 24615)).
28. To ensure consistency with State closures (Alaska State waters) for Steller sea lion protection measures in the Pacific cod pot fishery, NMFS removed restrictions on using pot gear for directed fishing for Pacific cod by vessels named on a FFP in waters within 3 nm of Cape Barnabas and Caton Island (May 28, 2003, 68 FR 31629).
29. On December 20, 2004, NMFS issued a final rule (69 FR 75865) which implemented changes to the Steller sea lion protection measures in the GOA for the pollock and Pacific cod fisheries. The final rule adjusted Pacific cod and pollock fishing closure areas near four Steller sea lion haulouts and modified the seasonal management of pollock harvest in the GOA. The intent of the revisions was to maintain protection for Steller sea lions and their critical habitat while easing the economic burden on GOA fishing communities.
30. March 2008, NMFS finalized the Revised Recovery Plan for the eastern and western DPS of Steller Sea Lion. The 2008 Recovery Plan identified the reduction of threats to Steller sea lion habitat as a recovery priority and recommended actions for protecting and improving aquatic habitat such that the prey base is adequate to support recovered populations of Steller sea lions.

4.4 Summary of Direct Effects of Commercial Fisheries on Steller Sea Lions

Commercial fisheries can directly affect Steller sea lions in the BSAI, and GOA by capturing, injuring, or killing them in fishing gear or in collisions with fishing vessels, and if fishermen kill them intentionally. These impacts were described in detail above in Sections 4.3.3 (incidental take in commercial fisheries), 4.3.4 (intentional and illegal killing), and in 4.3.7 (disturbance). In general, the current level of direct impact to Steller sea lions is relatively small (summary in Section 4.3.10). However, it is likely that historical direct impacts influenced the rapid decline rate observed in the 1980s, but by the mid-1990s were no longer an important factor in the decline and lack of recovery. Vital rate analyses confirm the reduction in direct mortality (Holmes and York 2003, Holmes *et al.* 2007, Atkinson *et al.* 2008).

4.5 Indirect Effects of Commercial Fisheries on Steller Sea Lions: Habitat Based

The BS, AI, and GOA contain some of the most productive waters on earth. The continental shelf in the eastern Bering Sea is broad and supports large, standing stocks of groundfish. The GOA has a much narrower shelf and supports a smaller standing stock than the BS. The Aleutian Islands has a very narrow shelf with deep drop offs within 5-20 nm from shore and has recently been highlighted as a unique and potentially fragile ecosystem (Ladd *et al.* 2005, Hunt and Stabeno 2005, Stabeno *et al.* 2005). Since the 1950s, a complex international fishery has harvested numerous species; most of the fish harvested in this region are groundfish. The Bering Sea supports about 300 species of fish, most of which live on or near the bottom. About 24 of these species support commercial fisheries in the BSAI and GOA.

Commercial fisheries in the action area have gone through many cycles of development and collapse since they began in the 1800s and the focus of the fisheries has shifted many times since its beginning. A complete historical review of commercial fisheries is provided in NMFS (2000) and incorporated here by reference. Three time periods were outlined:

1. Early commercial fisheries from the 1800s to the 1950s,
2. Large scale growth of fisheries from the 1950s to the 1970s, and
3. Commercial fisheries in the action area from the 1970s to 2000.

The federal action under consideration in this Biological Opinion, continued authorization of the groundfish fisheries in the BSAI and GOA, is an ongoing action. Effects of these fisheries on the environment in the BSAI and GOA were described in previous Biological Opinions (NMFS 2000, 2001, 2003). Environmental effects described in past Biological Opinions and here in the baseline are expected to be ongoing. In examining whether a federal action is likely to jeopardize the continued existence of a listed species or adversely modify their designated critical habitat, NMFS must demonstrate that the listed species or their designated critical habitat are exposed to the effects of an action. To establish exposure we determine the spatial and temporal overlap between listed resources and the direct or indirect physical, chemical, and biotic stressors of an action. The overlap between groundfish fisheries and Steller sea lions and their designated critical habitat is well established through the extensive formal consultation history on these fisheries.

Figure 4.24 summarizes the pathways for Steller sea lions to be exposed to the effects of the federal action and Figure 4.25 summarizes the potential responses of Steller sea lions and designated critical habitat exposed to the effects of the federal action. The complete analysis of the effects is contained in Chapter 5. Effects of the federal action are considered in light of the environmental baseline which includes the past and present impacts of fisheries in the action area.

Management measures to address potential fishery effects on Steller sea lions between 1991 and 2002 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 Steller sea lions within critical habitat, and modified harvest control rules to reduce the likelihood of overall prey abundance being reduced to less than 20% of theoretical unfished levels. NMFS concluded that the suite of management measures implemented in 2003 (NMFS 2003) avoided jeopardy and adverse modification of critical habitat. However, since 2003, a substantial quantity of scientific literature on Steller sea lion biology, habitat, and fisheries effects has become available (for example, see summary by Loughlin and Tagart 2006) which must be considered in an evaluation of the extent to which the implemented conservation measures remain effective in avoiding jeopardy and adverse modification.

Fishing can affect the availability of prey on localized and ecosystem-wide scales (Trites *et al.* 2006e), which is of concern because of potential adverse impacts on the recovery of the western population of Steller sea lion (Lowry *et al.* 1982). Fisheries in Alaska are some of the largest in the world. In 2005, over 2 million metric tons of groundfish were caught in the BS, AI and GOA, which is equivalent to an overall annual harvest rate of approximately 10% of the specific species targeted in these fisheries (Table 2.7 and Figure 4.23). 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.* 2006e).

The most notable indirect effect of commercial fisheries on Steller sea lions is the removal of prey species which could alter the animal's natural foraging patterns and their foraging success rate; both of these effects could have further downstream results such as increased predation risk due to longer or different foraging patterns. Fisheries can also have indirect biological effects that occur when fisheries remove large numbers of target species and non-target species (incidental catch or bycatch) from a marine ecosystem. These removals can change the composition of the fish community with associated effects on the distribution and abundance of prey organisms. Fishery removals compete with other consumers that depend on target organisms for food. These biological interactions are generally termed cascade effects and competition. The ultimate impact to Steller sea lions from these types of modification to their prey resources could potentially include either acute or chronic nutritional stress (Trites and Donnelly 2003; see Section 3.1.14).

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, Williams 2005a,b; see Section 4.6 below). Due to the high energetic demands of Steller sea lions relative to terrestrial mammals and the large number of Steller sea lions seasonally concentrated on rookeries, this species may be vulnerable to reduced prey biomass and quality (Winship and Trites 2003, Williams 2005a). 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 survival and recovery.

A reduction in prey resources may result in a reduction in population growth rate. Specifically, reduced prey availability can lead to physiological responses by Steller 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 Steller sea lions. These potential impacts have generally been referred to as nutritional stress (see Section 3.1.14).

4.5.1 Important Steller Sea Lion Prey Species and Fisheries Which Potentially Affect Prey

Our knowledge of Steller sea lion prey use is largely through the collection and analysis of scat samples and historically through stomach contents (Sinclair and Zeppelin 2002; Tollit *et al.* 2009; Table 3.15). In NMFS (2000, 2003), 14 species (or species groups) were important prey in the BSAI and 15 species in the GOA (see Table III-1 in NMFS 2003) that could be affected by the federal groundfish fisheries. Sinclair and Zeppelin (2002) assess the importance of various species by area and season. Steller sea lions rely on a variety of prey resources with pollock, salmon, Atka mackerel, Pacific cod, cephalopods, sculpins, herring, sand lance, and arrowtooth flounder representing the more common species. Using prey DNA techniques, Tollit *et al.* (2009) noted that salmon, cod, herring, and sole/flounder were most predominant in samples from British Columbia and Atka mackerel, sole/flounder, salmon, and pollock in the Aleutian Islands. Using hard parts of prey items, Tollit *et al.* (2009) found that herring, salmon, elasmobranchs, hake, pollock, and cod were predominant in the British Columbia samples, while in the AI region using this technique it was sole/flounder, pollock, sand lance, Atka mackerel, sculpins, and salmon. Steller sea lions are opportunistic predators which rely on seasonal aggregations of prey resources in predictable locations and quantities (Womble and Sigler 2006, Gende and Sigler 2006, Sigler *et al*. 2009).

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; Table 3.21). The abundances of many of the primary prey species of Steller sea lions have undergone changes during the past 30 years (NRC 1996, 2003, NPFMC 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 (e.g., arrowtooth flounder increased while GOA pollock decreased). Several factors have been implicated in these changes in prey biomass for Steller sea lions: 1) natural or environmental variability, 2) anthropogenic (fisheries) effects, and 3) ecosystem disruption resulting in inter-specific 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.

Steller sea lion diet likely reflects the availability of prey and their ability to take advantage of it (Pitcher 1981, Calkins and Goodwin 1988, NMFS 2000, Sinclair and Zeppelin 2002, Trites 2003, Womble and Sigler

2006, Gende and Sigler 2006, Waite and Burkanov 2006, Trites *et al.* 2007, Trites and Calkins 2008, Sigler *et al*. 2009) as well as the energy density of prey (see Section 3.1.14.3). Although we are limited in the locations and times that we have sampled Steller sea lion diets (stomachs or scats), diet likely reflects local availability and vice versa. The Aleutian Islands represent a good example of this foraging pattern. In the central and western Aleutian Islands, the average frequency of occurrence of pollock in winter was only 12% while Atka mackerel appears to have been the primary food source for Steller sea lions (found in 55% of scats in winter and 96% in summer; Table 3.16). Sinclair and Zeppelin (2002) point out that although some of the food items had a low frequency of occurrence (FO) when averaged across all samples, some had higher occurrences when looked at during specific seasons or at specific sites (Sinclair and Zeppelin 2002, their Appendix 1). Specifically, areas within the eastern Aleutian Islands seem to be more dependent upon pollock with a FO of 53% in winter. In NMFS (2006c; their Table 9), the FO is provided for various haulouts near Adak in the central Aleutian Islands (from Sinclair and Zeppelin 2002; their Appendix 1). Pollock ranked among the top three prey species at both Kasatochi Island (summer) and at Ulak Island (summer), both of which are rookeries in the Central Aleutian Islands. Table 10 (of NMFS 2006b) describes the prey items found in scats at Adak, Amlia, and Kasatochi in 1999 and 2000, and Table 11 (of NMFS 2006b) describes scats at a variety of sites in the central Aleutian Islands since 2001. In general, Atka mackerel was the dominant prey item found, especially during the summer. Pollock was more important in the diet during the winter but was also found at some sites during the summer (NMFS 2006b; Tables 10 and 11, Figure 9). In samples collected during the winter of 2002, pollock was between 8% and 46% FO at Seguam and Silak (near Adak Island). In these samples pollock was much more important in the diet than the average values reported above and likely represent the local availability of prey as well as the variability in sampling times. Season appears to be an important consideration as pollock was most often in the diet of Steller sea lions during the winter.

In more recent work, as noted above, Tollit *et al.* (2009) examined DNA in samples of prey items in SSL scats; they found differences in predominant dietary elements when comparing prey hard parts versus prey DNA. In that study, DNA tests indicated that salmon, cod, herring, and sole/flounder were most predominant in the British Columbia samples; in the AI region it was Atka mackerel, sole/flounder, salmon, and pollock. Using hard parts, herring, salmon, elasmobranchs, hake, pollock, and cod were predominant in the British Columbia samples, while in the AI region using this technique it was sole/flounder, pollock, sand lance, Atka mackerel, sculpins, and salmon (Tollit *et al.* 2009).

NMFS (2006c) concluded:

> *In summary, pollock is an important prey item for Steller sea lions in the Aleutian Islands, especially in the eastern portion of the area and in other locations where pollock may be available in relatively small aggregations, especially in winter. Based on the differences in the occurrence of pollock in scat samples, pollock may be more important to Steller sea lions using the Atka Island/North Cape haulout than for animals using haulouts near Kanaga Sound. The variability of pollock in the diet of sea lions is likely to be linked to the availability of the prey and is likely to reflect similar patterns as the fishery. Harvest of pollock in the Aleutian Islands has been patchily distributed with some locally high harvest amounts due to dense aggregations of pollock nearshore during spawning. Due to the remoteness of the Aleutian Islands, scat is not frequently collected at many sites which further confounds our ability to draw a clear picture of prey utilization in these areas. From the best information available, pollock is likely to be an important component of Steller sea lion diet in the winter but not during the summer (Tables 10 and 11; Sinclair and Zeppelin 2002). Also from the 2001 Opinion, we know that the ratio of prey biomass available to the biomass consumed by sea lions is the lowest in the Aleutian Islands, and may be lower than what is optimal for their survival (NMFS 2003, their Table III-8). This indicates that sea lions in the Aleutian Islands may be more susceptible to perturbations in the prey field than other areas such as the eastern Bering Sea.*

Thus, we cannot assume that average FO over large areas is necessarily a good representation of important prey species at individual haulout sites, or a good representation of the accessibility of the prey field as experienced by Steller sea lions. This is especially true in regions where key prey species may be in relatively low abundance (due to their range). Therefore, although regional trends are important to understand the overall impact of fisheries on subpopulations, it is also important to look at prey needs at smaller scales where the local availability of prey, likely consumption by local Steller sea lions, and the potential for localized depletion may have specific consequences not discernible at larger scales (NRC 2003 Ortiz and Loggerwell, 2010).

An extensive body of analytical work has been developed on the potential competitive interactions between Steller sea lions and fisheries for pollock, Pacific cod, and Atka mackerel (e.g., Loughlin and Merrick 1989, Ferrero and Fritz 1994, Fritz *et al.* 1995, Fritz and Ferrero 1998, NMFS 2000, Fritz and Brown 2005, Wilson *et al.* 2003, Calkins 2006, NMFS 2006b, Matthiopolous *et al.* 2008, Hui *et al.* 2010, Ortiz and Loggerwell 2010, AFSC 2010a). These fisheries were the obvious starting place for analyses of interactions because their target species are the most prevalent items in the diet of Steller sea lions in the GOA and the BSAI (Sinclair and Zeppelin 2002; Table 3.21). However, there are other species targeted (and incidentally caught species) by the Alaska groundfish fisheries in the BSAI and GOA that are also consumed by Steller sea lions. The question of how much overlap actually occurs is highly relevant to determining the exposure of Steller sea lions to fisheries authorized by the State of Alaska and the federal FMPs for groundfish.

For the purposes of determining which fisheries (if any) overlap substantially with important Steller sea lion prey species, we utilized a simplified version of the qualitative criteria developed by Lowry *et al.* (1982) and used by NMFS (2000 and 2003). To determine the likelihood and relative severity of indirect effects of fisheries on marine mammals, Lowry established criteria based on each marine mammal's diet (with respect to species consumed, size, and composition of prey), feeding strategy, and the importance of the BSAI as a foraging area. This approach is diagrammed in the exposure analysis (Figure 4.24), and is a step-wise approach to determining how many Steller sea lions will be exposed to potentially adverse fishing effects. The response diagram is provided in Figure 4.25 and takes the exposed Steller sea lions in Figure 4.24 and tests their responses and ultimate changes in natality, growth, and survival rates.

The first step in the exposure analysis is to determine the important prey species for Steller sea lions and the directed fisheries which target these species. Steller sea lion food habits data from 1999 to 2005 (Table 3.21) and from Sinclair and Zeppelin (2002) were examined. Prey items which occurred in greater than 10% of the scats (FO) by area, season, and DPS-wide were determined to be prey species of importance. Atka mackerel, pollock, and Pacific cod fisheries stand out as the areas of greatest overlap in fisheries and important prey resources for Steller sea lions. Also, rock sole and arrowtooth flounder made the list but in much lower importance. Salmon was ranked fairly high, and was often higher than Pacific cod or pollock depending upon area and season. Therefore, based on directed fisheries (Tables 2.8 and 2.9), the fisheries of concern include:

- Pollock (trawl)
- Atka mackerel (trawl)
- Pacific cod (trawl, hook-&-line, pot)
- Rock sole in the BSAI (trawl), shallow water flatfish assemblage in the GOA (trawl)
- Salmon fisheries
- Arrowtooth (trawl)

Rock sole and arrowtooth flounder occurred in more than 10% of Steller sea lion scats collected from 1999-2005 in the eastern AI and central and western GOA (Table 3.16). Although there is a TAC specified for arrowtooth, the fishery is relatively small compared with other BSAI trawl fisheries. However, because arrowtooth flounder meet the criterion of important prey (>10% FOC) in Steller sea lion diets, this analysis includes the arrowtooth flounder fishery as a potential competitor with Steller sea lions, particularly in the central GOA (McKenzie and Wynne 2008). Arrowtooth flounder has not been fished as intensively as other target species. Arrowtooth flounder are apex predators and dominate the groundfish community biomass in the EBS and GOA. In 2009, arrowtooth flounder comprised 59% of the GOA apex predator guild, the next largest component was Pacific cod (13%) (Boldt and Jador 2009). Prior to 1986, pollock was the dominant groundfish species by biomass in the GOA. Pollock are the major prey item of arrowtooth flounder ≥40 cm long in the GOA. Thus, it is possible that the groundfish fishery has resulted in indirect ecosystem effects that have allowed various competitors of Steller sea lions, such as arrowtooth, to increase in abundance. This indirect pathway will be discussed further below in the food web dynamics section. Rock sole also is generally underutilized (based on ABC, TAC, and actual catch amounts relative to the ABC; see Tables 2.5 and 2.6), and therefore may influence the prey field of Steller sea lions through some form of competitive release.

In summary, based on best available scientific and commercial data, the fisheries as authorized under the FMPs potentially compete with Steller sea lions for common resources. Fisheries and Steller sea lions both consume pollock, Atka mackerel, Pacific cod, salmon, arrowtooth flounder, and rock sole. The high degree of overlap between pollock, Atka mackerel, and Pacific cod fisheries, and to perhaps a lesser degree with the other fisheries listed above, and the foraging needs of Steller sea lions support the hypothesis that competitive interactions may be occurring in the range of the western DPS on a number of spatial and temporal scales.

4.5.2 Description of Fishing Patterns and Catch: Inside and Outside Critical Habitat

Determining the nature and extent of commercial fisheries catch in the action area is an essential descriptive element of the baseline. This analysis looks at fishery catch on a number of levels relating to Steller sea lion foraging habitat and distribution as well as fishery management areas to assess the potential for competitive overlap between the groundfish fisheries and Steller sea lions.

Data Available for Assessing Spatial Fishing Patterns

In previous Biological Opinions on Alaska groundfish fisheries and Steller sea lions (e.g., NMFS 2000, 2001, 2003) NMFS described commercial fisheries removals in the action area based on catch estimates derived from an extensive array of data collected by the Groundfish Observer Program (NOAA, Fisheries Monitoring and Analysis Division) operating in the BS, AI, and GOA. Observer-collected data include the type and amount of fish caught, position of gear deployment and retrieval, date, processor type, gear type, detailed catch composition, and biological samples used to describe the age structure of the catch for many species. These data are a sample of the Federal Groundfish fishery with coverage rates established by 50 CFR 679.50.[17] Once deployed on fishing trip, observers follow statistically rigorous sampling procedures published annually and monitored by the AFSC (http://www.afsc.noaa.gov/FMA/).

Consistent with previous Biological Opinions, NMFS extrapolated fishery observer data to infer the spatial distribution of the fishery removals inside and outside of Steller sea lion critical habitat. Observer-collected data on the latitude and longitude of gear deployment and retrieval have

---

[17] In October 2010, the Council took final action to modify the design of the groundfish observer program. The Council’s motion would define operations requiring 100% observer coverage. For operations with <100% coverage requirements, coverage levels would not be specified in regulation but determined according to a randomized sampling plan to be developed by NMFS.

historically provided the finest scale information available on the spatial distribution of the fisheries.

Observer data are a sample of the entire fishery, since not all vessels carry observers on all trips, nor are all hauls sampled for catch composition. For instance, no vessels smaller than 60 feet in length overall (LOA) are required to carry an observer, and vessels between 60 and 125 LOA must only carry an observer on 30% of their trips unless they are part of certain management programs (e.g., Amendment 80). As a consequence, the use of observer data to estimate total catch underepresents the catch of smaller vessels that generally fish closer to shore than larger vessels. This would affect the estimated spatial distribution of the catch of species that have a coastal distribution and are fished by small vessels (e.g., Pacific cod catches in the GOA by the longline fleet) more than species that have a broader distribution and are fished by larger vessels (e.g., walleye pollock catches in the EBS by the trawl fleet and Pacific cod in the Amendment 80 trawl fleet). However, despite its incompleteness and potential bias, the observer data contain detailed information on catch location and composition available in no other catch data set for the time period of interest (1991 through 2008). The analysis in this Biological Opinion focuses on harvest patterns from 1999-2008.

NMFS assumed that the distribution of the observed catch represented the unobserved catch. A series of expansion factors, or ratios of total to observed catch, were computed to estimate the distribution of total catch from the distribution of observed catch. Categories used to compute observed catch expansion factors are listed in Appendix III.

Observed catches were obtained by querying the groundfish fishery observer databases maintained at the AFSC. Observed catch of each species in each observer record (e.g., haul) was multiplied by the appropriate year-species-area-gear-processor expansion factor to estimate total catch, which was then assigned to the observed haul location. For CDQ catches in the BSAI, observed catches were used directly since all vessels fishing CDQ are required to carry observers and 100% of the catch during CDQ operations is sampled for catch composition. Therefore, total catch for the BSAI consists of both observed, unexpanded CDQ hauls and observed, expanded non-CDQ fishery hauls. Estimated catches were assigned to Steller sea lion critical habitat regions using ArcGIS. The resulting Expanded Observer Dataset (EOD) was queried and summarized to provide a detailed estimate of catch within specific geographic areas, RCAs (see below), and critical habitat zones.

In 2002-2003, NMFS implemented a satellite-linked VMS requirement for some fisheries to better assess the nature and extent of fishing activities in the action area, and, in part, to provide the means to overcome some of the bias associated with the issues noted above. In 2007, NMFS began developing a fisheries harvest database that would integrate spatial data acquired from both onboard observers and data on vessel movements acquired by the VMS. This VMS-Observer Enabled Catch-In-Areas (VOE-CIA) database uses a spatial approach to incorporate both vessel location and observer data into catch estimates for the action area (Appendix II).

Catch estimates based on the EOD and the VOE-CIA for 2003-2008 are compared in Appendix II. [18] This comparison confirms that while the EOD underestimates catch in some nearshore areas, these two approaches to estimate the spatial distribution of catch produce values that are generally in agreement, especially when viewed on a regional geographic scale. However, both methods are estimated with error and NMFS attempted to quantify this error by comparing EOD and VOE-CIA estimates of total catch by fishery (species and area) with NMFS' official Catch Accounting database (described in Chapter 2).

---

[18] The dates when the observer data were retrieved from the catch accounting system for building these tables varies by area and year, and may result in some modest differences between what is reported in the tables and information that is currently available in the catch accounting system. See Appendix V for a comparison of the differences.

Appendix V provides the percent difference between the EOD and the Catch Accounting database and the VOE-CIA database. The EOD are considered to comprise the best available information on the spatial distribution of catch for comparisons of catch inside and outside of Steller sea lion critical habitat for the period of interest for this Biological Opinion since they are the only spatial data available prior to 2003. Other data used in this Biological Opinion for assessing groundfish fishing patterns and rates include the Catch Accounting database, the VOE-CIA database, and data from the Stock Assessment and Fishery Evaluation (SAFE) reports to apply the best data available for the respective analyses.

NMFS (2000, 2001, and 2003) provided a description of fishing patterns from 1991 through 2001 in two subset regions, the BSAI and the Gulf of Alaska. For those regions, the amount of catch of key Steller sea lion prey species was delineated into various zones around sites recurrently used by sea lions – rookeries principally – based on perceived intensity of use by foraging animals while away from these sites (i.e., 0-3 nm, 3-10 nm, 10-20 nm, and several specific offshore foraging areas) that comprise designated critical habitat. Prey species of primary interest were pollock, Pacific cod, and Atka mackerel due to the potential for competitive overlap between the groundfish fisheries and Steller sea lions. The analysis in this Biological Opinion follows a similar approach but further delineates the BSAI into 2 regions: the Bering Sea and Aleutian Islands. This geographic separation into three regions (GOA, BS, and AI) provides a somewhat finer scale examination of catch in the action area relevant to Steller sea lion population trends and foraging activities within critical habitat (Figure 4.26). A fourth species of concern, arrowtooth flounder, has been added to those considered previously because recent food habits studies have shown this species contributes 10% or more (frequency of occurrence basis) to the diet of Steller sea lions in some regions of the action area (McKenzie and Wynne, 2008). Catch data presented in previous Biological Opinions have been updated and new data up to and including 2008 have been added. The analysis in this Biological Opinion focuses on fishery harvest patterns and data from 1999-2008.

To further examine potential relationships between catch and Steller sea lion population trends, a more geographically detailed catch analysis was made possible through the delineation of 11 areas in the BS, AI, and GOA regions; 10 within the action area and one encompassing Southeast Alaska for purposes of discussion (Figure 4.27). The bounds of the ten regions used in this analysis were chosen primarily on grouped similarities in trends of Steller sea lion population parameters, knowledge of movement patterns of Steller sea lions on rookery sites within each region, and on availability of suitable biomass estimates for the key prey species; these groupings are referred to as RCAs. Regions 1-5 are equivalent to the western and central Aleutian Islands Steller sea lion survey areas (NMFS 2008a), the Aleutian Islands fishery management areas (541-543), and the Aleutian Islands groundfish trawl survey area. Region 6 is equivalent to the eastern Aleutian Islands SSL survey area (NMFS 2008a), the eastern Bering Sea fishery management and groundfish survey areas, the SE Bering Sea portion of the Aleutian groundfish trawl survey area, and portions of the western Gulf of Alaska fishery management (610) and groundfish trawl survey areas. Regions 7-11 encompass the remainder of the Gulf of Alaska from Unimak Pass through SE Alaska (133-165°W), which includes: the western, central and eastern Gulf of Alaska, and SE Alaska Steller sea lion survey areas; the majority of the Gulf of Alaska fishery management areas (part of 610 and all the area from 620-650); and the majority of the Gulf of Alaska groundfish bottom trawl survey areas.

Using a GIS, the estimated catch was assigned to one of the BS, AI, or GOA regions for the large-scale analysis and to the RCA regions for a finer scale analysis. The resulting EOD for 1991-2008 was used to generate summary tables and figures for the following section describing the fisheries catch on a large regional scale (BS, AI, and GOA regions). Because there are a large number of figures and tables, we have placed them into Appendix III for ease of reference. Similarly, we have placed the figures and tables associated with the finer-scale RCA analysis (see 4.5.2.5, below) into Appendix IV.

4.5.2.1 Spatial Implementation of the Fisheries in the BS, AI, and GOA

A summary of fisheries catch data for the BS, AI, and GOA (Figure III-1) is provided in Tables III-1 to III-3 (see Appendix III for all similarly named tables and figures). Total catch (State parallel, State guideline harvest level, and federally-managed catch) per year and the amount of catch inside Steller sea lion critical habitat of pollock, Pacific cod, Atka mackerel, and arrowtooth flounder are shown in Figures III-2 through III-5, while Figures III-6 through III-17 show spatial and temporal details of catch of the four species. Figure III-18 shows trends in biomass, total catch, and catch within critical habitat for pollock in the EBS and GOA from 1977 through 2008.

In the BS, the removals of pollock from critical habitat ranged between 600,000-800,000 mt from 1991 to 1998 and then declined in 1999-2000 as a result of management measures and critical habitat closures imposed in those years (Table III-1). Following implementation of new management measures in 2001-2002 and increases in the pollock TAC, pollock catches in critical habitat returned to 1991-1998 levels (Figure III-2; top panel); while the proportion of pollock caught in critical habitat declined, the amount caught in critical habitat in 2002-2007 exceeded that caught in 1999 and 2000. From 2001-2006 the total catch in the BS remained steady at or near historic highs (near the 1,500,000 mt cap), while biomass estimates for the region declined steadily – markedly so beginning in 2007 (Figure III-18). During this period TAC was set well below the maximum permissible ABC in each of these years. In contrast to the entire BS, the catch of pollock in BS critical habitat has declined, which may be as much related to a redistribution of fish and fishing effort to the north (Figures 1.2 and Figure 1.3 in Ianelli *et. al.* 2009) as well as it is to any management measures implemented in recent years (see below). The 2008 decrease in catch of BS pollock was, in part, a response to a lower ABC (reflecting a decline in pollock biomass which began in 2003). The fact that the actual catch was below the TAC (by about 10,000 mt) was partly due to poorer fishing conditions (low pollock abundance) in the southeast BS (much of which is critical habitat). Subsequently, biomass estimate have dropped below B_{MSY} which by FMP amendment 56, reduced the harvest rate (to promote rebuilding). The 2009 and 2010 ABC reflects these downward adjustments and the total catch levels have been set to just over 800,000 mt (Ianelli *et al.* 2009).

Total catch in the BS Pacific cod fishery increased slightly between 1998 and 2002, largely leveling off through 2005, then declining in the subsequent 2 years. The proportion caught in critical habitat by the Pacific cod trawl fishery increased slightly through 2005. However, catch by the Pacific cod pot and longline fisheries in critical habitat has generally declined since 2006 (Figure III-2, middle panel and III-6 top). A reduction in the longline Pacific cod catch in critical habitat may be due to a shift in the distribution of Pacific cod to the north (NMFS 2006).

The BS Atka mackerel fishery resulted in a steep decrease in catch in critical habitat in 1999 and maintained that low catch until 2003 and 2004 when it increased sharply (6,500 mt in 2004). Overall catch declined between 2005 and 2007, and dropped approximately 90% in 2008. Nearly all of the Atka mackerel catch in the BS has been taken from within critical habitat, primarily as bycatch in other directed fisheries. Higher catch rates in 2003 and 2004 and/or increased retention of Atka mackerel in the flatfish fishery likely corresponded with an increase in Atka mackerel biomass and a shift in this species distribution in 2003-2007 (NMFS, Sustainable Fisheries, pers. com.) (Figure III-2; bottom panel).

In the GOA, annual pollock catches have ranged between approximately 50,000 mt and 125,000 mt between 1991 and 2008 (Figure III-3; top panel), with an average of 72% taken in critical habitat (Table III-2). Since the regulatory changes implemented in 1999, the proportion taken within the 0-3 nm zone and in foraging areas remains low, while it has held to about 14% for 3-10 nm and 51% for the 10-20 nm zones.

The proportion of Pacific cod taken in GOA critical habitat dropped from 68% before 1999 to about 53% since 1999 (although this proportion trended up in 2007-2008). The overall catch amount initially declined more than 40% but has shown an increase since 2005 (Figure III-3 middle; Table III-3, bottom).

Directed fishing for Atka mackerel has been prohibited in the GOA since 1997. Most GOA Atka mackerel catch is taken in bycatch fisheries mainly in the rockfish fishery; a recently increasing catch may reflect increased abundance levels of mackerel. The 2006-2008 TACs for GOA Atka mackerel were 1,500 mt, and the 2009 TAC was set at 2,000 mt. It should be noted that the 2008 and 2009 catches exceeded (by 609 mt in 2008 and 222 mt in 2009) the TAC but were well below the ABC in those years (Lowe *et. al.* 2009, and 73 FR 10566, February 27, 2008); about one-third of the catch occurred in critical habitat (Table III-3) and was highly aggregated in the third quarter (Figure III-14).

In the AI, catch limits were placed on the pollock trawl fishery and regulations were promulgated that eliminated the directed fishery for pollock in critical habitat of the region after 1998. As a direct result, catch declined steeply and has remained at low levels between 600-2,523 mt since 2000 (Table III-3 and Figure III-4, top panel). In the AI, catch incidental to the non-pelagic Pacific cod trawl fishery continues and has perhaps increased slightly. Since 2002, about 60% of pollock catch in the Aleutian Islands occurred within critical habitat, largely within the 3-10 and 10-20 nm zones, with about twice as much taken in the outer zone (Table III-3). Under Amendment 82, commencing in 2006 the AI pollock TAC is allocated to the Aleut Corporation; the TAC level for 2010 is 19,000 mt (of which 1900 is allocated to CDQ groups). In past years under the Aleut Corp. fishery, very small amounts of pollock have been harvested (see EFP fishery data in the following). In 2006 and 2007 a portion of the annual TAC for pollock was taken in critical habitat under an EFP, and in 2008 under a Scientific Research Permit, to support a pollock abundance study (NMFS 2006b, Logerwell *et. al.* 2009) related to a potential fishery associated with a processing plant on Adak Island. In 2006, 932 mt of pollock were harvested of the 1,000 mt of pollock allocated for the EFP; in 2007, 1,122 mt were harvested out of 3,000 mt allocated, and in 2008, 8 mt were harvested (Barbeaux, pers. comm.). During the EFP, no pollock harvest occurred within 3 nm of rookeries and harvests within 3 nm of haulouts were limited to the amount necessary to verify the acoustic survey (approximately 10 mt). At this point, due to poor sample returns and unresolved issues with processing plant capabilities, it appears unlikely that a pollock fishery might develop in the AI. A Biological Opinion for the EFP determined that the activities under the EFP were not likely to jeopardize the continued existence of the western DPS of Steller sea lions or adversely modify its critical habitat (NMFS 2006b). NMFS would address any need for re-consultation should the topic of such a fishery be resurrected again in the future, especially in view of recent information documenting a continued population decline for the western DPS of Steller sea lions in the westernmost portions of the range.

Catch of Pacific cod in the Aleutian Islands declined from 2002 until 2005, and increased above 30,000 mt in 2007 and 2008. Since 2007, 3 % of the annual BSAI Pacific cod TAC is allocated to the State of Alaska in the AI region for a State GHL fishery (which generally follows Fdereal SSL protection closures in the AI). Since 1991, the proportion caught in critical habitat has averaged a nearly steady 80% with essentially no difference in catch proportion from critical habitat between the 1991-1998 and the 1999-2008 periods (Figure III-4; middle panel; Table III-3). Note that the regulatory closures of SSL critical habitat in the AI region for cod fisheries are less stringent than for pollock fisheries.

The total catch in the AI Atka mackerel fishery peaked in 1996, and has since fluctuated between approximately 44,000-65,000 mt (Figure III-4 bottom panel). The average catch since 1999 has been a fairly steady 54,000 mt. The proportion caught in critical habitat dropped from an average of about 67% in 1991-1998 to about 38% in 1999-2008 due to a series of management measures designed to spatially and temporally disperse the fishery and reduce catches within critical habitat (Table III-3, bottom).

The estimated catch of arrowtooth flounder in the BS, AI, and GOA is given in Tables III-1-III-.3. Trends in the catch of arrowtooth flounder are shown in Figure III-5. Most of the catch of this species occurred in the GOA closely followed by the BS. In general, arrowtooth catches in the GOA have increased steadily since 1999, with an average of about 45% of the catch taken in critical habitat (range 22-73%). Arrowtooth flounder biomass levels have been increasing in both the GOA and BSAI, and are at very high levels compared with historic levels; relative to available biomass, fishery quotas have been a very low proportion. Catch in the BSAI in 2009 was below 30,000 mt of an ABC of 156,000 mt. In the GOA, catch in 2009 was around 22,000 mt of an ABC of 228,405 mt. Prior to Amendment 80, most of the arrowtooth flounder harvest in the BSAI and GOA was bycatch in other fisheries, and thus most was discarded. The proportions of arrowtooth flounder catch within various zones of critical habitat by fishery are shown in Figure III-8.

4.5.2.2 Temporal Implementation of the Fisheries in the BS, AI, and GOA

One of the important issues that NMFS considered when implementing the Steller sea lion conservation measures was the need to temporally distribute fisheries to avoid locally concentrated catches that could result in localized depletions of Steller sea lion prey. New regulations, the Steller sea lion protection measures, affecting management of the Atka mackerel, pollock, and Pacific cod fisheries, went into effect in 2002. A component of these measures was the implementation of seasonal apportionments for pollock, Pacific cod, and Atka mackerel, and the use of fishery groups (or “platoons”) for the Atka mackerel fishery in statistical areas 542 and 543 (central and western Aleutian Islands). In this section NMFS will explore the changes to the fishery after implementation of these conservation measures intended to temporally distribute the fishing effort. Figures III-10 to III-15 depict the percentage of annual catch by each fishery harvested by quarter of the year.

Between 1999-2008, on average 56% of the annual catch in the BS Pacific cod trawl fishery has been taken in the first 3 months of the year (Figure III-9; top panel). Despite various changes in a complex management structure (see next two paragraphs), very little difference between temporal harvests before and after implementation of the Steller sea lion protection measures can be seen when looking at this fishery on a quarterly basis.

Prior to 2008, the first season was allocated 60% of its ITAC[19] and the second and third seasons were each allocated 20%. The trawl catcher vessels’ allocation was further allocated as 70% in the first season, 10% in the second season and 20% in the third season. The trawl C/P’s allocation was allocated 50% in the first season, 30% in the second season and 20% in the third season.

Starting in 2008, trawl gear was allocated 37.8% of the BSAI Pacific cod TAC. The trawl catcher vessels’ allocation was further allocated as 74% in the first season, 11% in the second season and 15% in the third season. The trawl C/P’s allocation was allocated 75% in the first season, 25% in the second season, and none in the third season.

The timing and amount of fishing effort in the Pacific cod pot fishery can be volatile and related to availability and value of crab fished in the Bering Sea, and the availability of Pacific cod. By regulation, the fishery occurs in two seasons: during the first season, non-trawl gear is allocated 51% of its ITAC and 49% during the second season. No seasonal harvest constraints are imposed for the Pacific cod fishery by catcher vessels less than 60 ft (LOA) using hook-and-line or pot gear. As a result the fishery typically occurs between March and April, which is why catch has peaked in either the second quarter (1998 and 1999) or the first quarter (2000-2008) (middle panel, Figure III-9). Between 2003 and 2008, about 69% of the pot fishery occurred in the first quarter. For the Pacific cod hook-and-line fishery, an average of 50%

---

[19] The ITAC is the annual TAC minus 15 percent allocated to the reserves.

of the catch was taken in the first quarter of 2002-2008 compared to 56% in 1999. Pacific cod catch in the 4th quarter increased from 17% in 1999 to 21% during 2002-2008. Very little hook-and-line fishing occurs from June 10 to August 15 because historically, halibut mortality quotas have not been allocated to the fishery until August 15.

In the GOA (Figure III-10), the Pacific cod trawl fishery catch has been variable in the first quarter, fluctuating between 35% and 83% of the annual catch (average from 1999-2008: 58%). The protection measures limit the Pacific cod harvest in the western and central GOA to 60% in the first half of the year and 40% in the second half (50 CFR 679.20(a)(11)). In 2002 about 66% was taken in the first quarter and about 17% in the second quarter, for a total of 83% (instead of the target 60%) in the first half of the year. Since 2002, there has been an increasing proportion of the catch taken in the third quarter, with second half of the years catch proportions exceeding the 40% target in 2003, 2004, and 2008. Catch proportions in 2005 and 2007 were close to the 60:40 targets (60% first half of the year and 40% the second half of the year). The proportion of the Pacific cod hook-and-line catch taken in the first quarter has gradually increased since 2003 while that taken in the second and fourth quarters has increased slightly. In general recent proportions remain heavily weighted to the first half of the year, although there have been increases (18-57%) in the proportion caught in the third quarter since 2006.

A considerable portion of the Pacific cod caught in the Aleutian Islands is taken in the trawl fishery with the catch primarily occurring in the first quarter (first quarter accounts for an average of 88% the total catch between 2000 and 2008, Figures III-11). In the AI the fish caught during the first quarter has increasingly been taken from within critical habitat zones, including some limited amounts from within the 0-3 and 3-10 nm zones, while other fisheries from 1998-2001 in the first quarter averaged 64% from critical habitat and in 2002-2008 averaged 70% from critical habitat.

The TAC for Pacific cod is currently managed for the entire BSAI. Catches from each subarea are combined to determine the overall progress towards the seasonal apportionment. There has been no discernable trend in the small pot fishery for Pacific cod in the AI (Figure III-11 middle). With some exceptions, 1st quarter catch in the longline Pacific cod fishery has generally declined with slight increases in the proportions caught in the 3rd and 4th quarters from 2001-2008.

In general, total catch of Pacific cod in each of the BS, AI, and GOA is highly aggregated in the first quarter with no change evident in the seasonal distribution of this catch from 1999-2008 (Figure III-12).

Seasonal catch of pollock in the BS, AI, and GOA is displayed in Figure III-13. Catch is seasonally apportioned in the BSAI as 40 percent in the A season and 60 percent in the B season (50 CFR 679.20(a)(5)(i)(B)(1). In the BS, catch in the first quarter had been slowly decreasing from 1998-2001 (from about 48% to 38%). Except for a small increase in first quarter catch between 2001 and 2002 (just over 40%), the catch level has stabilized in recent years, and it is now more widely dispersed across the A season (see below). Most of the catch in the second half of the year occurs in the third quarter (from July - September) with a decreasing amount being taken in the fourth quarter.

While the seasonal apportionment of pollock catch in the GOA is evenly distributed between 4 seasons in the Western and Central areas, pollock catch in the GOA has been more variable by season than in the BSAI (Figure III-13; middle panel). In 1999 the GOA catch in the first quarter of the year was about 39%, since then catch has peaked several years at or just below 60% of the annual catch, which reflects the high value of the winter roe fishery. The catch of pollock in the third and fourth quarters has fluctuated, with a trend toward increasingly larger catches occurring in the fourth quarter since 2000.

The continuing effects of fishing under cooperatives in the Bering Sea for pollock can be seen in Figure III-16. It shows the daily percentage total annual catch for 1998-2008 inside and outside of critical habitat and illustrates how the proportion caught in the A and B seasons has become more dispersed throughout the designated open fishing periods (Figure III-16, bottom right). Note that in 1999 only half of the Bering Sea fishery was operating in a cooperative that year (C/Ps only). During the A season in the three years prior to the cooperatives under the AFA, average removals peaked at more than 100,000 mt per week, corresponding to almost 11% of the annual TAC. In 1999, these absolute removals were lower due to the formation of offshore cooperatives, but shoreside harvesters had not yet formed cooperatives. After the entire Bering Sea became managed under cooperatives and the number of vessels fishing decreased, the peak weekly removal rates dropped from about 100,000 mt to about 65,117 mt (2000-2001); subsequently these have increased to 80,870 mt (2002-2005) and since stabilized (2006-2007 (80,423 mt). In 2005, the peak weekly removal was 85,358 mt; the average weekly removals were 54,059 mt for the A season, 39,488 mt for the B season. Similarly in 2007, the peak weekly removal was 72,090 mt, the average weekly removals were 50,786 mt (A season) and 34,736 mt (B season).

Previously, a peak week of 65,000 mt (e.g., 2000 and 2001) was, on average, between 4.6%-5.7% of the TAC. From 2000-2008, EBS pollock TACs were higher than 1999, with the TAC in 2004 set to 1,492,000 mt. Thus, in terms of what is being taken relative to TAC during the 2002-2005 period, the weekly maximum removals have declined to 3.6% of the TAC in the A season, 2.6% in the B season, and 5.4% overall.

Both the FMP Biological Opinion (NMFS 2000) and the 2003 Supplement examined daily catch rates in the A and B seasons and inside and outside of critical habitat. Figure III-17 updates and summarizes this information, breaking it into three categories to show how the catch goals of the 2002 management measures have resulted in an evening out of daily catch over time and reducing the magnitude of the daily catch over a specific season. Overall, progress towards reducing the magnitude of daily maximums was made between 1999 and 2000. The table at the bottom of this figure reflects a similar pattern of initial lowering of maximum daily rates followed by increases in daily rates during the 2002-2006 period. Reaching the goal of increasing the number of days where catch is within 50% of maximum dispersion has not been as successful for the A season as the B season (that is, the A season catch remains highly aggregated in time for both the overall catch and the catch within critical habitat). However, the daily B season catch rates in critical habitat peaked in 2002 and have declined steadily since, likely due to the general northward movement of the fishery out of areas with designated critical habitat.

To summarize, improvement has been made towards spatially and temporally dispersing the overall catch of pollock in the EBS although this may have weakened in recent years. Since the trawl gear closure ordered by the court in 2000, total annual harvest was maintained at nearly 1.5 million mt up until 2007. Pollock catch in critical habitat peaked in 2002 at over 800,000 and then steadily declined to 244,384 mt in 2008. The average proportion of the catch taken in critical habitat between 2002 and 2008 was 38% compared to 53% from 1991-2001. Within certain areas of critical habitat both the proportion (e.g., 10-20 nm zones) and the amount (e.g., foraging zone) of pollock removed from critical habitat increased during 2002-2005 over that harvested between 1999 and 2001 Total catch of pollock remained at near historic highs of >1,400,000 mt from 2002 -2006 as the estimated EBS pollock biomass declined 40% from a peak in 2003 (Figure III-18), which reflects a period of time between 2001 and 2006 when the TAC was appreciably smaller than the ABC.

As part of the management measures implemented in the 2002 regulations, seasonal catch regulations for Atka mackerel in the Aleutian Islands allocated 50% of the TAC to each A season (20 Jay -15 Apr) and a B season (1 September to 1 November) (Lowe *et al.* 2009). Some changes in seasonal dispersion of the catch in the Aleutian Islands have occurred since 2002 (Figure III-14) with a decreasing proportion taken in the first quarter (1998-2001 average = 54%, 2002-2008 average = 34%), a stable amount taken in the third quarter (about 50% of the annual catch), and a slightly increasing amount taken in the fourth quarter (2002-2008 average = 12.7%). The 2002 regulations set the maximum seasonal catch percentage within critical habitat in the western and central Aleutian Islands at 60%. The average proportion taken in critical habitat in the AI (2002-2008) was 37.4% (Figure III-6 and Table III-3).

4.5.2.3 Catch in Critical Habitat Zones

In the supplement to the 2001 Biological Opinion (NMFS 2003), a comparison was made between the catch under the 1999 and 2002 management regimes. Because of changes to groundfish management implemented in 2002, the expectation was that many of the fisheries would have experienced reduced nearshore amounts of catch in 2002 when compared to the amounts observed in 1999 (i.e., the fishery that NMFS determined in the FMP Biological Opinion to cause jeopardy and adverse modification). NMFS found that the performance of the management measures in reducing effort nearshore and in increased temporal/spatial dispersion was not the same in all areas or for all fisheries. For instance, in the EBS pollock fishery, there was increased temporal dispersion between 1999 and 2002 (positive performance).

NMFS has updated this comparison to provide perspective on how the catch has been distributed spatially and temporally to the 1999 situation. Figures III-6 – III-8 show the percent of the total catch by each gear type and zone from 1998-2008. Tables III-4, III-5, III-6 present this information as the change from 1999 to 2005, from 1999 to 2007, and 1999 to 2008, respectively, with the rate of change by zone displayed as a percent. When used in companion with the 2003 supplement information, these data may be useful for determining if the implemented conservation measures have been effective (Chapter 5).

In the BS, catch in all four target fisheries and all gear types was reduced or maintained at near zero levels in the 0-3 nm zone (Figures III-6 and III-8). Total catch of pollock during 2002-2006 remained at or near historical highs approaching 1.4 million mt, however the catch in critical habitat declined from over 800,000 mt to 240,000 mt in 2008 (Table III-1, top). While there was an overall decline in the proportion of pollock caught in critical habitat from 2001-2008, there has only been a slight reduction in the proportion taken within the 10-20 nm zone when comparing 1991-1998 (10.1% taken in the 10-20 nm) to 2001-2008 (9.6% from 10-20 zone) (Table III-1). Roughly a third of the pollock catch (30%) was taken out of the designated foraging zones from 2002 to 2008 (Table III-1 top), contrasting with 43% being taken from the foraging area 1991-1998. The amount of pollock taken out of the BS foraging areas ranged between 400,000-600,000 mt year between 2001-2005, and held near 300,000 mt year (until 2008) as compared to 356,077 mt taken in 1999. Data from earlier years appear to reflect the conservation strategy of closing only the 0-10 nm area, thereby allowing harvest in the 10-20 nm and foraging zones for vessels preferring to fish closer to shore. There are, however, indications from the last 2 or 3 years that the nature of this fishery may be changing with an increasing focus on both biomass and effort moving to the north (and out of critical habitat) as the available biomass declines, particularly in the B-season (June-October) (see Figure 1.3 in Ianelli *et al.* 2006). This may reflect the estimates of population age structure, which comprise relatively fewer old fish (due to poor recruitment from 2001-2005). Older fish are more common in the southern part of the EBS with younger pollock more common in the northwest region. Similar shifts in fishery distribution have occurred in the past, presumably due to similar pattern of population age structure (Ianelli *et al.* 2007).

Atka mackerel catches in the BS were at historic highs during the 2003-2007 seasons (Table III-1) and were taken almost exclusively within critical habitat (99%). Even though the total amounts removed remain lower (5-10%) relative to what is taken from the AI, more fish were removed from BS critical habitat in any of these years since 1991. When directed fishing for Atka mackerel is prohibited in the BS, there is currently a 20% maximum retention amount of Atka mackerel allowed in the other groundfish fisheries (except arrowtooth flounder which has no retention of Atka mackerel) (50 CFR part 679, Table 11). This Atka mackerel catch appears to track changes in biomass for the BS region (Lowe *et al.* 2009; Table 15.4).

There were decreases in the proportion of Pacific cod caught in recent years in the critical habitat zones in the Bering Sea (Tables III-4, III-5, and III-6). The proportion of Pacific cod taken from the foraging zone decreased from the 1991-1999 period (average 31%) to 23% over 2002-2008, while the proportion harvested in the 3-10 zone remained steady at about 3% (Table III-1,). In the BS, the Pacific cod longline harvest in critical habitat declined 53% between 1999 and 2008 as the total catch increased 7% in 2008 from the 1999 level (Table III-6). It should be noted that since 1999, the majority of Pacific cod catch in the BS has been concentrated in the first quarter of the year (average 52%).

Even though directed fishing for pollock in critical habitat was prohibited in the Aleutian Islands as a result of the 2002 Steller sea lion protection measures, some limited catch, largely within critical habitat (average over 2002-2008 is 60%) continues at low levels (Table III-3). This catch was a combination of bycatch taken in other directed fisheries for groundfish, which may occur within critical habitat, and as part of the EFP pollock fishery described above. Congress passed the Consolidated Appropriations Act of 2004 that allocated all of the Aleutian Islands pollock TAC (with 10% set aside for CDQ fisheries) to the Aleut Corporation to support economic development of Adak, Alaska. The TAC can be no more than 19,000 mt of pollock regardless of the ABC exceeding this amount (50 CFR 679.20(a)(5)(iii)). Between 2005 and 2007 less than 10% of the Aleut Corporation's pollock allocation in the Aleutian Islands was harvested (approximately 900 mt to 1,400 mt, NMFS in-season Management, pers. comm.). The proportion of pollock caught in critical habitat in the AI declined from 74% (1991-1999) to an average of 60% between 2003 and 2008, with the largest change being a drop in catch taken in the 3-10 nm zone (32% to 17%) (Table III-3).

Between 2003 and 2008, AI Pacific cod trawl catch was nearly 20% over that of 1999, with a large proportion coming out of the 10-20 nm zone and the catch from the 0-3 nm zone declining to generally under 100 mt (Table III-3). The catch of Pacific cod in the pot fishery has almost been eliminated in recent years. The Pacific cod longline catch has fluctuated in recent years but increased last year to take 12% more in critical habitat in 2008 than 1999 (Table III-6).

Total catch of AI Atka mackerel increased slightly during the 1999-2002 period, but has since stabilized around 55,000 mt year between 2003 and 2008, with a decline in the proportion of catch being made in critical habitat between 1991-1999 (average 67%) to 2000-2008 (average 36%). Between 1999 and 2008, the amount of catch removed from critical habitat has been a relatively consistent 20,000 mt, most of which (35%) was harvested in the 10-20 nm zone (Table III-3).

The proportion of pollock caught in GOA critical habitat declined slightly from an average of 75% (between 1991-1999) to an average of 68% between 2000-2008. There was a slight decrease in the proportion harvested in the 3-10 nm zone and foraging zones and an increase harvested in the 10-20 nm zone (about 51% on average during 2000-2008) (Table III-2). Comparing 1999 to 2005, a 15% decrease in the total amount of pollock caught in the GOA, coupled with a 13% decrease in the proportion of pollock caught within critical habitat, resulted in a 27% decrease in the amount of pollock taken from critical habitat in the GOA (Table III-4). Even larger decreases are noted when comparing 1999 and 2007 (Table III-5) and 1999 and 2008 (Table III-6). Harvest of pollock by trawling was virtually eliminated from the 0-3 nm zone in the GOA between 2001 and 2008 (Tables III-4, III-5, and Figure III-6). The total pollock catch decreased from 1998 to 2002, and currently the pollock biomass in the GOA is below B_{40} so the rate of fishing has been adjusted downward as required by Amendment 56.

In late 2004, revisions were made to the Steller sea lion protection measures for the GOA pollock harvest (http://209.112.168.2/frules/fr75865.pdf). These allowed for additional harvest beginning in the first part of 2005 around Puale Bay in concert with an expanded closure around Cape Douglas. The closure around Cape Douglas was much larger than the opening around Puale Bay and was designed to result in a net decrease in the area of critical habitat available for pollock fishing. Comparing the 2004 and 2005 pollock catch in the GOA, there was an increase in the total pollock caught in the GOA as well as in the amount of pollock caught within Steller sea lion critical habitat of the region (Table III-2). Subsequent to the management change, there was an 11% (2005 and 2007) and 17% (2008) increase over 1999 in the proportion caught in the 3-10 nm zone from 1999 to 2005 (Tables III-4, III-5, III-6). These recent increases contrast with decreases in proportion caught in these same critical habitat zones reported between 1999 and 2002 (see the 2003 Supplement to the 2001 Biological Opinion).

In the GOA, the average proportion of Pacific cod taken in critical habitat has declined since 1999 (1991-1999 average 68%, 2000-2008 average 53%), although the amount of both total catch and catch in critical habitat have increased since 2005 (Table III-2, Tables III-4, III-5, and III-6 and Figure III-3). For the GOA Pacific cod trawl fishery, the proportion of catch decreased within the 3-10 nm zone and generally increased in the 10-20 zone between 1999 and 2008. The proportion caught in the 10-20 nm zone by the GOA Pacific cod pot fishery in recent years is higher than in 1999 (Tables III-4 - III-6). While the average amount of all Pacific cod caught in critical habitat between 2002-2006 declined by more than 30% from that of 1999, there was a noteworthy increase in that catch in 2007 and 2008 (Table III-2).

The Atka mackerel catch in the GOA is low relative to other regions as there is no directed fishery, but incidental catch has increased steadily since 2001 (Figure III-3). The amount and the proportion of Atka mackerel caught in critical habitat also increased slowly during this period (Table III-2).

4.5.2.4 Alaska State managed Fisheries (0-3 nm)

Detailed information on fisheries in inside waters is contained in Section 4.10 of the Groundfish SEIS, as well as in Kruse *et al.* 2000 and Woodby and Hulbert 2006. This section includes a brief review of those fisheries that may affect Steller sea lions, including:

- A description of the fishery management strategy including any special measures pertaining to Steller sea lions,
- Recent changes in the spatial and temporal distribution of the fisheries, and
- A description of direct and in-direct Steller sea lion interactions.

To date, there have been few studies specifically designed to address the effects of these nearshore fisheries on Steller sea lions, so the information presented below is descriptive in nature. Soboleff (2005) analyzed State of Alaska fisheries (salmon, herring, shellfish, groundfish) fish ticket data for 1976-2002 and SSL counts by rookery (32) groupings (7). He indicated that within 50 nm of rookeries, SSL counts were both negatively and positively correlated with certain State fisheries, but few were significant and some probably spurious. This study also found negative correlation between State salmon fisheries and the SSL decline across all regions or all years, which disappeared at a regional scale, and Soboleff (2005) felt this could be plausible as salmon fisheries occur near SSL haulouts and rookeries and salmon are important SSL prey. The study concluded that few data, low power, and concentration of State fisheries outside areas where SSL declines have been most severe all may be factors that indicate a low likelihood of State-managed fisheries adversely affecting SSLs.

Significant changes in state waters fisheries since the 2001 Biological Opinion include an all-time high salmon harvest, the re-opening of several crab fisheries in the GOA, and a new Pacific cod fishery in the Aleutian Islands. This section describes recent changes in state waters including removal of greater volumes of Steller sea lion prey biomass as well as other fish and invertebrate species from nearshore areas. Because the nearshore areas may be more important for Steller sea lions than previously thought in NMFS (2000, 2001), and because some state fisheries are concentrated in time and space critical to Steller sea lions (Woodby and Hulbert 2006), this suggests that state waters fisheries may have greater effects on Steller sea lions than NMFS previously concluded (NMFS 2000, 2001).

ADF&G manages fishing activity occurring inside waters from shore to three miles seaward, herein referred to as state waters. Additionally, ADF&G oversees BSAI crab, salmon, lingcod, and some rockfish fisheries in Federal waters (EEZ – outside of three miles from shore). With the exception of State managed fisheries that have specified guideline harvest levels (GHLs) for species such as sablefish, Pacific cod, and Prince William Sound pollock, ADF&G coordinates state fishery openings and in-season adjustments with federally managed fisheries (the "parallel" fisheries). For example, when groundfish fishing is open in Federal waters, state regulations allow fishing to occur in state waters in what is referred to as the parallel fishery. The state retains regulatory jurisdiction over all fisheries within state waters.

State fisheries are managed by a highly localized system of regional offices throughout the state by species and area. Each region is responsible for issuing Guideline Harvest Limits (GHL), and providing in-season management of smaller-scale, localized fisheries. This is in contrast to the Federal fisheries, which are composed of very large management units with relatively large harvest limits. Whereas the Federal fisheries use summer and winter surveys combined with stock assessment models to assess biomass and catch limits, the state employs a variety of methods of determining catch and biomass including stock recruitment models, aerial surveys, escapement goals, and historical fishery harvest performance. Kruse *et al.* (2000) provide an overview of State managed fisheries that may interact with Steller sea lions, including historical catch, gear used, stock assessment methods, and status of the fish stocks. That information was summarized in the FMP Biological Opinion (NMFS 2000) and is not repeated here. Woodby and Hulbert (2006) expanded and updated this report to include changes between the 2000 report and the latest fisheries data available before the preparation of this document (2006). They also added information on the Pacific cod, pollock, and Atka mackerel parallel fisheries occurring inside state waters. Soboleff (2005) also evaluated State fisheries and potential effects on SSLs (summarized above).

Seasonal and temporal distributions of state waters fisheries vary widely by species, area, and gear type, and are discussed in more detail in subsequent sections. These distributions are depicted in detail in Kruse *et al.* 2000 for the year 1999, and in Woodby and Hulbert (2006) for the year 2005. Another descriptive reference is *Commercial Fisheries off Alaska* (Woody *et al.* 2005). The reader should consult these three references for a complete description of the fisheries. Only summary information is included here.

Direct interactions between State managed fisheries and Steller sea lions involve both lethal and non-lethal impacts. Lethal impacts include Steller sea lions inadvertently killed in fishing gear such as trawls, seines, and gill nets. Non-lethal effects include short-term impacts such as disturbance of Steller sea lion haulouts, vessel noise, entanglement in nets, and preclusion from foraging areas due to disturbance from active fishing vessels and gear. State managed fisheries are estimated to account for the incidental take of about 23 Steller sea lions per year (Angliss and Outlaw 2005). Recently this number has been difficult to verify due to the lack of observer coverage and the expected under-reporting of takes through a voluntary reporting program. On one hand, it might be low due to the lack of observer coverage in these fisheries, yet on the other hand this estimate is potentially biased high due to the very high estimate for a Prince William Sound gillnet fishery (Angliss and Outlaw 2005). There are no available estimates of the frequency or severity of non-lethal takes. Illegal shooting of Steller sea lions by fishermen likely still occurs, but the number of animals affected is difficult to evaluate given the lack of observer coverage on these vessels. Loughlin and York (2001) estimated the mortality level from shooting at 50 Steller sea lions per year, or more. Kruse and Huntington (2009) summarized interviews of individuals to document possible levels of intentional shooting of SSLs from 1974-1990, noting that shooting was fairly common but varied across time, region, fishery, and other factors.

Potential indirect effects of State managed fisheries include the competition for prey resources and the modification of Steller sea lion critical habitat. State fisheries remove important Steller sea lion prey species, and many fisheries are concentrated in space (usually bays or river outlets) and in time (usually spawning aggregations and salmon congregating near rivers for their return to spawning grounds in spring and summer).

The geographic range of State managed fisheries in state waters coincides almost entirely with the area designated as Steller sea lion critical habitat. To reduce interactions between Steller sea lions and State managed fisheries, in 1999 ADF&G established no fishing zones for pollock around most rookeries and a few haulouts out to 3 nm (by Emergency Order, March 17, 1999) and has closed several haulout sites seasonally in Prince William Sound out to 10 nm. Four rookeries designated as critical habitat (Agattu Island/Gillion Point, Agattu Island/Cape Sabak, Wooded Island, and Seal Rocks (Cordova)) were not protected from commercial fishing out to 3 nm by the state emergency order. Four haulouts are included in the March 17, 1999 emergency order because the entire island where a rookery was located is protected by the 3nm fishing closure. These protected haulouts are Seguam Island/Finch Point, Seguam Island/South Side, Kiska/Sobaka and Vega, and Amchitka/Cape Ivakin. The 3 nm closures and 10 nm fishing restricted areas are based upon 1999 federal regulations. Since this time, additional Steller sea lion sites have been added to the regulations at 50 CFR part 679. In 2004, ADF&G mirrored a federal change to open up several Steller sea lion haulouts in the GOA.

In an analysis of Steller sea lion diet, Sinclair and Zeppelin (2002) found that pollock, Atka mackerel, Pacific salmon, Pacific cod, and Pacific herring were consumed in relatively high frequencies by the western stock of Steller sea lions during certain times of the year (Table 3.21). Observations from biologists and fishermen indicate spatial and temporal overlap between the State managed fisheries for these species and foraging Steller sea lions (Kruse *et al.* 2000). Information on Steller sea lion foraging patterns suggest that Steller sea lions, and especially pups and juveniles, spend the majority of their time in areas within 10 nm of shore (see Section 3.1.7). Because State fisheries are concentrated in time and space in these near shore waters, there is potential for negative effects on Steller sea lion prey (critical habitat) and Steller sea lion condition. Each state waters fishery is unique in its number of participating vessels, gear used, seasonality, duration, and/or target fish species. The next four sections describe state waters groundfish fisheries, herring fisheries, salmon fisheries, and invertebrate fisheries and their potential effects on Steller sea lions.

State Groundfish Fisheries

State managed groundfish fisheries are relatively small in tonnage compared to the federally managed groundfish fisheries, and are generally confined to specific management areas. The State managed pollock fishery is limited to Prince William Sound, while Pacific cod fisheries occur in Prince William Sound, Cook Inlet, Kodiak, Chignik, and South Alaska Peninsula areas, and since 2006 in the Aleutian Islands.. For a sense of scale, in 2000 the State managed GOA pollock harvest was 1.7% of the federal pollock fishery, and the State managed Pacific cod harvest was 22.5% of the total federal ABC in the GOA. Parallel fisheries for Pacific cod, pollock, and Atka mackerel are also prosecuted in inside waters prior to the state-managed fisheries seasons, in many of the same locations. Total harvest volume in these fisheries is usually much higher. A new State waters fishery for P. cod was established in 2006 based on 3% of the Federal BSAI ABC. That amounted to about 5,000 mt annually, nearly all of which was harvested in the early years of this fishery (2006-2007). In recent years effort has been lower and not all the GHL was harvested.

In addition to Pacific cod and pollock, the state has established separate GHLs and seasons for the following fisheries in the western GOA: sablefish, lingcod, black rockfish *(Sebastes melanops)*, and blue rockfish *(S. mystinus)*. The state-managed fisheries for sablefish and Pacific cod occur within state waters, whereas the state has full management authority for lingcod and black and blue rockfish fisheries throughout the EEZ. In the central GOA, state-managed fisheries in state waters also include sablefish and all rockfish species in state waters of Prince William Sound and lower Cook Inlet.

The Alaska Board of Fisheries (BOF) created “Guiding Principles for Groundfish Fishery Regulations” (5 AAC 028.89) which stipulate that state groundfish fisheries are managed conservatively to (1) conserve groundfish resources to ensure sustained yield, (2) minimize bycatch and prevent localized depletion of stocks, (3) protect habitat and other associated fish and shellfish, (4) maintain slower harvest rates by methods and means and time and area restrictions, (5) extend the length of fishing seasons by methods and means and time and area restrictions, (6) harvest the resource in a manner that emphasizes quality and value of the product, (7) use the best available information, and (8) manage cooperatively with the North Pacific Fishery Management Council and other federal agencies associated with groundfish fisheries.

These ecosystem-based guiding principles have led to a set of conservation measures for State-managed groundfish fisheries. A number of these management measures provide, directly or indirectly, some protection to Steller sea lions. Substantial areas of state waters are closed to non-pelagic trawling (Figure 4.28). Most areas are closed year-round, and some areas are closed seasonally as in Shelikof Strait. Moreover, a portion of eastern Prince William Sound is closed to pelagic trawl gear during the pollock fishery (5 AAC 28.263) and most of eastern Prince William Sound is closed to all (non-pelagic and pelagic) trawling year-round (5 AAC 39.165). These trawl closures were established by the BOF to protect seafloor habitats, shellfish such as depressed crab populations, and non-target demersal fishes.

Under the ESA, groundfish fisheries are prohibited within 3 nm around major Steller sea lion rookeries (no-entry zones around major rookeries for all vessels; 50 CFR 223.202). The no-entry zones apply to state permitted fishing vessels as well as federal permitted fishing vessels. The rookery closures are intended primarily to avoid disturbance to rookeries during the breeding season and to maintain a no-disturbance zone year-round to protect these very important breeding sites. The loss of a breeding site to human impacts could have a substantial impact on the population.

Although the 3 nm closures were designed specifically to protect Steller sea lion rookeries, the closures have indirect effects of protecting bottom habitat which provides protection to non-target species including octopus, sculpins, flatfish, greenlings, and other forage fishes. The non-pelagic trawling ban also reduces the possibility of direct cumulative impacts from State managed fisheries on marine habitat and particularly the benthic community.

Walleye pollock

Pollock is harvested in inside waters both in a State-managed fishery in PWS, and in parallel fisheries throughout state waters. The State managed PWS pollock fishery has been declining over the past 6 years. In 2005, most of the harvest occurred in early March inside Steller sea lion critical habitat. The parallel fishery is much larger in volume than the state-managed fishery, and has increased since 2000.

The PWS fishery is based on a constant harvest rate strategy. Because reliable estimates of biomass and natural mortality are available, the PWS pollock stock falls into Tier 5 of the federal stock assessment strategy (see Section 2.4.2). The GHL is calculated as the product of the biomass estimate, instantaneous natural mortality rate (0.3) and a “safety factor” of 0.75. Biomass is estimated by bottom trawl surveys in summer and hydroacoustic surveys of spawning aggregations in winter. In 1999 the BOF directed the ADF&G to file an emergency regulation establishing a PWS pollock trawl fishery management plan to reduce potential impacts on the endangered population of Steller sea lions. The plan divides the Inside District of (PWS) into three management sections with no more than 40% of the total harvest coming from any one area (5 AAC 28.263). ADF&G manages to a target of 30% of the total harvest from any one of these areas with a 10% reserve. These spatial management measures may help reduce competition for fish between the pollock fishery and Steller sea lions. This measure was in lieu of closing two Steller sea lion haulouts that were specified to be closed under the 1998 Biological Opinion (NMFS 1998). Although pollock in the GOA are considered to be one stock, the state surveys pollock in PWS separately from NMFS surveys in the GOA. However, NMFS takes the PWS fishery into consideration when setting the GOA TAC.

The effects of the State managed pollock fishery on Steller sea lions is mitigated to some degree by existing restrictions on the fishery. The Prince William Sound outside district (including Wooded Island, Seal Rocks, Cape Hinchinbrook, and Hook Point) is closed to fishing (Figure 4.29). Since the pollock fishery occurs only in the Prince William Sound inside district, it reduces the potential for removing Steller sea lion prey in the vicinity of critical habitat sites Cape St Elias, Hook Point, Middleton Island, the Wooded Island rookery, and most of the Seal Rock and Cape Hinchinbrook sites. Pollock fishing is prohibited June 1 through November 1 within 10 nm of seven rookeries and haulouts in Prince William Sound (5 AAC 28.250). Two haulout sites within Prince William Sound, Perry Island and Point Eleanor, have no pollock fishing restrictions. The Needles, Point Elrington, and Glacier Island haulouts have no pollock harvest restrictions from November 2 through May 31. The fishery opens January 20 (concurrent with CGOA) and closes by emergency order no later than March 31, 2001. Steller sea lions using PWS inside district haulouts may experience a depletion of pollock and disruption of the prey field during part or all of the year, and the time period of the pollock fishing restriction does not provide protection during the critical winter months.

The parallel pollock fishery inside state waters in 2005 occurred in Kodiak, Chignik, South Alaska Peninsula, and the Aleutian Islands. The Kodiak fishery peaked in February/March and then again in September/October and occurred throughout all Kodiak statistical areas. The Chignik fishery had landings in January and September and was concentrated south of Chignik at Seal Cape. In the South Alaska Peninsula, most landings occurred in January and October in the Shumagins and Pavlof Bay. The Aleutian Islands fishery landings were highest in July and August, and most harvest was taken at Unalaska Island. These fishery seasons have specific start dates according to the federal pollock fisheries. Most of the A season pollock TAC is taken in January, and then harvest peaks again in early fall when the new season allocation is obtainable. Most of this catch occurs inside Steller sea lion critical habitat in the Kodiak, South Alaska Peninsula, and Aleutian Islands areas. The parallel harvest inside state waters has been between 20 and 40 times the volume of the state-managed PWS fishery in the past 6 years.

Pacific cod

In 1996, the BOF adopted Pacific cod FMPs for fisheries in PWS, Lower Cook Inlet, Chignik, Kodiak, and the South Alaska Peninsula. All five FMPs have some common elements that include: only pot or jig gear is permitted, pot vessels are limited to no more than 60 pots, jig vessels are limited to no more than five jigging machines, and exclusive area registration requirements. Vessels participating in the South Alaska Peninsula and Chignik areas are limited to no more than 58 feet in length. Catches are allocated to users as: 85% pot and 15% jig in South Alaska Peninsula and Chignik areas, 60% pot and 40% jig in PWS, and 50:50 in Kodiak and Cook Inlet areas. If target gear allocation percentages are not met by late in the season, then the unattained GHL becomes available to all gear types. State GHLs are set as a percentage of the federal TAC. State GHLs for PWS are set at 25% of the federal TAC for the eastern GOA. Similarly, up to 25% of the central GOA TAC is allocated among Chignik (up to 8.75%), Kodiak (up to 12.5%) and Cook Inlet (up to 3.75%). Finally, the state GHL for the South Alaska Peninsula fishery is set at 25% of the western GOA TAC. The fishery generally occurs in the spring following the Federal fishery, opening by regulation between 1 and 7 days after the federal fishery closes.

Pacific cod harvested in state waters in 2005 came from Steller sea lion critical habitat in the South Alaska Peninsula (most harvest occurring in March), Kodiak (February through April), and Chignik (March through May) areas primarily, with smaller harvests in PWS and Cook Inlet. The temporal distribution of catch around Kodiak was more concentrated in 2005 from February through April as compared to 1999 when there was a more substantial fall component to the harvest.

In addition to the State managed fishery, the parallel Pacific cod fishery also occurs inside state waters and mostly inside Steller sea lion critical habitat. In PWS, the 2005 parallel fishery was much more widely distributed than the state-managed fishery and most harvest was taken in April and August. A greater volume of Pacific cod was taken in the parallel fishery in Cook inlet in February and March and extends all along the outer coast from Resurrection Bay to the tip of the peninsula. The Kodiak parallel fishery was about equal in volume and spatial distribution to the state-managed fishery, but was mostly taken in January and late fall. A similar pattern emerges in the South Alaska Peninsula fisheries with most parallel harvest taken in January and February. The Chignik fisheries break this pattern in that the parallel fishery is very small in comparison to the state-managed fishery. There was no state-managed fishery for Pacific cod in the Aleutian Islands in 2005. The parallel fishery was spread along the chain and harvested most catch from February to April.

On March 15, 2006, the BOF approved the opening of a new state waters Pacific cod fishery in the Aleutian Islands west of 170W for pot, jig, longline, and non-pelagic trawl gears. This state-managed fishery opens after the parallel fishery closes. The 2005 GHL was 5807 mt, or 3% of the BSAI ABC. The fishery is temporally regulated so that no more than 70% of the GHL can be harvested before June 10, 2006, however, most of this was taken in March. The remainder of the GHL can be harvested starting June 10. Twenty-six vessels registered for the fishery, including 3 trawlers less than 60 feet, 17 larger trawlers, one large pot vessel, 5 large freezer longliners, 2 floating processors and 2 shore-based processors participated. Observer coverage and VMS are not required in this state-waters fishery, but 6 vessels chose to carry a federal observer, and 23 planned to activate VMS during the fishery.

Atka Mackerel

There is no state-managed fishery for Atka mackerel other than the parallel fishery that occurs inside state waters. The parallel Atka mackerel fishery is harvested with bottom trawl gear and has ranged between 12 and 88 mt from 2000 to 2005 (Woodby and Hulbert 2006). Because most state waters are closed to bottom trawling and Atka mackerel generally do not occur in the GOA, this fishery is largely confined to a few small locations in the Aleutian Islands, including Unalaska Island, Atka Island, and the Islands of Four Mountains. These areas are inside Steller sea lion critical habitat. Most landings occurred in June and August in 2005.

Other Groundfish

Sablefish, rockfish, and lingcod are not important in the diet of Steller sea lions, but fisheries for these species could cause indirect impacts to Steller sea lion foraging behavior through disturbance. There are no specific measures to protect Steller sea lions included in the state management plans for these species. Sablefish landings occurred inside Steller sea lion critical habitat in PWS, lower Cook Inlet, and the western Aleutian islands in 2005. Landings occurred in March through May and August in PWS, in July in Cook Inlet, and primarily May through August in the western Aleutian Islands. Most of the lingcod harvest in 2005 was taken in the Kodiak area, although catch occurred inside Steller sea lion critical habitat in Kodiak, Cook Inlet, and PWS from July through October. Similarly, most rockfish harvest occurred around Kodiak Island, but harvest occurred inside Steller sea lion critical habitat in PWS, Cook Inlet, Kodiak, Chignik, South Alaska Peninsula, and the Aleutian Islands primarily from March through August.

Harvest of Steller Sea Lion Prey Species

The amount of groundfish prey species (pollock, Pacific cod, and Atka mackerel) harvested in the parallel fisheries is presented in Table 4.7. Although the amount of fish harvested in the 3 nm area around haulouts appears low, the amount of area composed inside 3 nm of haulouts in the GOA is roughly 0.5% of the total area. Catch percentages of up to 7.4% of total (pot, Pacific cod) represent a catch rate that is two orders of magnitude higher than a theoretically dispersed fishery. Again, the type of data necessary to evaluate whether this may or may not be a problem is lacking, such as information on biomass availability on small scales. Further complicating matters, the fleet fishing within state waters during these parallel seasons are generally small unobserved vessels. Because of this, very limited information is available on these fishing activities as compared to larger boats operating in federally managed waters that have observer coverage.

State Herring Fisheries

At present, State herring fisheries that occur within Steller sea lion critical habitat include fisheries in Prince William Sound, Cook Inlet, Kodiak, Alaska Peninsula, Bristol Bay, Kuskokwim, Norton Sound, Southeast, and Port Clarence. Approximately 25 distinct fisheries for Pacific herring occur in these regions. Harvest methods are by gillnet, purse seine, and handpicking of roe from kelp. Herring are primarily caught for their roe during the sac roe harvest in the spring when they move closer to shore (and therefore Steller sea lion critical habitat) to spawn. On occasion the entire allowable harvest has been taken in less than one hour, although most sac roe fisheries occur during a series of short openings of a few hours each, spanning approximately one week. Fishing is not allowed between these short openings to allow processors time to process the catch, and for managers to locate additional herring of marketable quality.

Prior to 1999, the average annual harvest of herring for sac roe was about 48,000 mt. During the past 5 years, harvest of herring for sac roe has been stable at around 22,000 mt. due to low abundance in some areas. The major populations of herring in Alaska are at moderate levels and in relatively stable condition, with the exceptions of Prince William Sound and Cook Inlet. Since 1999, the PWS fishery has been closed due to low abundance, and in 2006 the Exxon Valdez Oil Spill Trustee Council initiated planning for a long-term herring restoration program. The lower Cook Inlet fishery has been closed since 1998 due to low abundance. Herring harvest near Kodiak has increased during the last 6 years and is distributed throughout Steller sea lion critical habitat. The fishery occurs in a concentrated time period from late April to early May.

Spawn-on-kelp fisheries harvest intertidal and subtidal macroalgae which contain freshly deposited herring eggs. Smaller amounts of herring are harvested from late July through February in herring food/bait fisheries. Herring spawn timing is temperature dependent, so that herring spawning and roe harvest timing occurs progressively later from southeast Alaska, where spawning begins in March, through the northern Bering Sea, where spawning ends in June. Herring food and bait landings in 2005 in the Alaska Peninsula area were concentrated in the Akutan district inside Steller sea lion critical habitat and occurred in late July. Smaller food and bait landings occurred in Kodiak in January, October, and December of 2005.

Harvest policies used for herring in Alaska set the maximum exploitation rate at 20% of the exploitable or mature biomass. The 20% exploitation rate is considered by ADF&G to be lower than commonly used biological reference points for species with similar life history characteristics. In some areas, such as Southeast Alaska, a formal policy exists for reducing the exploitation rate as the biomass drops to low levels. In other areas, the exploitation rate is similarly reduced, without a formal policy. In addition to exploitation rate constraints, minimum threshold biomass levels are set for most Alaskan herring fisheries. If the spawning biomass is estimated to be below the threshold level, no commercial fishing is allowed. Threshold levels are generally set at 25% of the long-term average of unfished biomass (Funk and Rowell 1995).

Most herring fisheries in Alaska are regulated by management units or regulatory stocks (i.e., geographically distinct spawning aggregations defined by regulation). Those aggregations may occupy areas as small as several miles of beach or as large as all of Prince William Sound. Herring sac roe and spawn-on-kelp fisheries are always prosecuted on individual regulatory stocks. Management of food and bait herring fisheries can be more complicated because they are conducted in the late summer, fall, and winter when herring from several regulatory stocks may be mixed together on feeding grounds distant from the spawning areas. Where possible, the BOF avoids establishing bait fisheries that harvest herring from more than one spawning population.

Interactions Between Herring Fisheries and Steller Sea Lions

Herring fisheries may affect Steller sea lions or their critical habitat when vessel activity interferes with Steller sea lion foraging, reduces prey availability, or alters long term herring biomass. Additionally, direct mortality may result when Steller sea lions are caught in nets or other fishing gear (although no direct mortalities have been observed in the herring fisheries; Angliss and Outlaw 2005). Steller sea lions are attracted to areas where herring spawn to feed on the dense aggregations of herring present during the short spawning period. Observations of Steller sea lions in Prince William Sound indicate that Steller sea lions may target herring despite the presence of much greater abundance of pollock (Thomas and Thorne 2001). These results suggest that under some conditions (e.g., when highly aggregated in shallow water), herring (or other high lipid fish) may be an important prey resource for Steller sea lions (Sigler *et al.* 2004, Womble and Sigler 2006).

Because of the variability in the timing of herring spawn, fishery managers have learned to depend on the presence of Steller sea lions to determine when spawning is imminent. Managers generally begin flying aerial surveys over potential herring spawning grounds well in advance of the expected spawning event. For several weeks prior to spawning, herring are usually present adjacent to the spawning grounds, but they occur in depths too deep to be detected from aircraft. However, the presence of Steller sea lions and cetaceans on the spawning grounds alerts fishery managers to the presence of herring and impending spawning. Fishery managers usually note the presence of Steller sea lions in their field notebooks, occasionally recording actual counts. Steller sea lions are commonly observed in the middle of these fishing areas. There are two possible hypotheses regarding these observations:

1. Steller sea lions may venture into fishing grounds because the fishery is in someway beneficial (or neutral), concentrating herring, creating confusion, and enhancing feeding opportunities for Steller sea lions.

2. Some Steller sea lions, perhaps the brave or curious ones or those that cannot afford not to forage (i.e., nutritionally limited), forage in these fishery grounds. Other Steller sea lions may avoid these fishing areas due to the intense vessel activity, nets, and other hazards (e.g.,shooting or other harassment). Steller sea lions that do choose to forage in these areas may have higher stress levels involved with avoiding vessels, gear, and dealing with noise, yet may appear to be foraging effectively but at an increased metabolic cost.

Presumably, fishing in areas that were previously unfished, yet utilized by Steller sea lions, would change the manner and success rate of foraging Steller sea lions. This could be either a positive or negative effect. Given the high caloric content of herring, the historical dependence on the species (Sinclair and Zeppelin 2002), and the large decline in herring biomass during the last century (Kruse *et al.* 2000), this fishery should be the subject of further study specifically to determine if there may be negative impacts on Steller sea lions. The important point is that although we have adequate data which displays that Steller sea lions attempt to forage during the times and places when herring fisheries occur, we have little or no information on either the net impacts to those Steller sea lions or other Steller sea lions which may avoid observation because they elect not to forage. There is no way of knowing how many Steller sea lions may be precluded from foraging in the spawning areas due to fishing activity. Steller sea lions are observed leaving the grounds within a few days after the herring have spawned. Fishery biologists make note of their departure since spawn deposition SCUBA biomass surveys do not begin, for safety reasons, until the Steller sea lions leave the area.

One example of a herring spawning event where Steller sea lion counts were quantified during aerial surveys is shown in Figure 4.30. There was no fishery at Hobart Bay in the spring of 2000 because the quota had been taken in the earlier food/bait herring fishery. However, if a fishery had occurred, managers would typically have allowed 6-12 hours of gillnet fishing about April 29. Steller sea lions were already in the area at the time of the first ADF&G aerial survey on April 19, diving on the deeply submerged herring schools, as were a number of humpback whales. Following the spawning event, large numbers of birds appeared on the beaches to feed on the herring eggs, noted in numbers of 11,000 to 20,000. Approximately 150 Steller sea lions were counted in the area. Similar descriptions of humpback whale and Steller sea lion presence on herring spawning grounds are available in field notes from other herring fishing areas.

Steller sea lions may depend on these short intervals of high prey availability to sustain them through other periods of low prey availability. Some individual Steller sea lions may be able to adapt by learning to forage among the fishing boats, but others may choose to avoid the area and may thus forego prime foraging opportunities. Since we do not observe the Steller sea lions that avoid fishing areas, we have no reliable way to estimate how many may be affected in this way, nor do we have a way to gauge the impact on those individual animals. For the Steller sea lions that remain, we have no way to gauge their foraging success among fishing vessels relative to their potential foraging success in the absence of fishing vessels. Nevertheless, based on observations of interactions between the fishery and Steller sea lions, it is reasonable to conclude that some Steller sea lions may be precluded by the fishery from foraging on spawning schools of herring. Likewise, the Steller sea lions that do forage in the vicinity of the fishery may forage less efficiently due to active competition with the fishery for the available concentrations of herring.

Hundreds of individual Steller sea lions may be affected by each of these brief fishery openings. The annual exploitation rate for herring is roughly 20% of the exploitable or mature biomass (Kruse *et al.* 2000), which is considered by the state to be conservative. This may be in relation to

> the target stock, but the question that arises is whether this is conservative from a Steller sea lion perspective? This example from Hobart Bay is merely to make the point that foraging Steller sea lions and herring fisheries operate in the same areas and times on the same resource.

State Salmon Fisheries

The State salmon fishery includes five species: Chinook, sockeye, coho, pink, and chum. These fisheries are divided into southeast, Prince William Sound, Cook Inlet, Bristol Bay, Kodiak, Chignik, Alaska Peninsula, Kuskokwim, Yukon, Norton Sound, and Kotzebue management areas. The PWS, Kodiak, Chignik, and Alaska Peninsula areas report substantial harvest inside Steller sea lion critical habitat in 2005 (Woodby and Hulbert). Salmon are taken by purse seines, gill nets, trolling, and beach seining via an extensive small boat fleet. The catch in 2000 was about 135 million fish, but Alaska's salmon landings reached an all-time high in 2005 of 221.9 million fish primarily due to high pink salmon catches, healthy salmon stocks and improving world-wide markets for wild fish. Economically, the salmon fishery is worth more than all other state fisheries combined.

Landings have increased for all salmon species except chum, and are trending towards a more temporally concentrated distribution earlier in the summer. Kodiak purse seine landings were twice as high in July and August of 2005 as compared to 1999, with more catch inside Steller sea lion critical habitat. Chignik purse seine landings were concentrated earlier in June and July. Similarly, the South Alaska Peninsula (SAP) drift gillnet landings were more temporally concentrated in June as opposed to lasting into September as in 1999. SAP purse seine catches also peaked earlier in the summer.

The fisheries are managed for minimum escapement goals, where regional ADFG biologists have determined what level of escapement seems to produce the maximum yield per year. These methods have not been standardized, and range from aerial flights to determine if the streams are "full" to fish weirs and remote sonar counters. The timing of the fisheries corresponds with the various spawning time for each run, which is highly variable and which is managed on a stream by stream basis.

State managed salmon fisheries have direct impacts on Steller sea lions through the interaction of gear. In the gillnet fishery Steller sea lions cause significant catch loss and gear damage by taking fish from nets and tearing large holes in the nets (Hoover 1988). Steller sea lions cause damage to purse seine nets when they swim inside the nets to eat salmon before the nets are closed (Hoover 1988). Prior to the mid-1990s the only quantitative study on interactions between Steller sea lions and the Alaska salmon gillnet fishery was on the Copper and Bering River deltas and the Coghill district in south central Alaska (Kruse *et al.* 2000; Matkin and Fay 1980). During the three week spring salmon season Steller sea lions damaged 1.7-4.9% of the weekly catch, and most of the damage occurred in outside waters where relatively few boats fished. Steller sea lions were infrequently seen in the Coghill district and were absent during the fall Copper River district season. Observers also monitored the Prince William Sound salmon drift gillnet (Copper River) fishery in 1990 and 1991. No mortalities were observed in 1990 and two were recorded in 1991. When these observer data are extrapolated, the mean kill rate for 1990 and 1991 is 14.5 Steller sea lions per year (Kruse *et al.* 2000). The Alaska Peninsula and Aleutian Islands salmon drift gillnet fishery was also monitored during 1990 and no Steller sea lion mortalities were observed. There were no incidental serious injuries or mortalities observed in the Cook Inlet salmon gillnet fishery in either 1999 or 2000 (NMFS unpublished data); for Bristol Bay the annual Steller sea lion mortality is thought to be 3.5 (Kruse *et al.* 2000, Ferrero *et al.* 2000).

Indirect adverse effects of State managed salmon fisheries on Steller sea lions stem from competition for seasonal aggregations of fish. Sinclair and Zeppelin (2002) found that Pacific salmon were the third most dominant fish in the diet of Steller sea lions, based on scats observed from 1990 to 1998 on summer and winter island sites across the range of the western stock of sea lions. Sinclair and Zeppelin (2002) observed that known seasonal and spatial distributions of aggregations of fish that are preyed upon by Steller sea lions parallel the highest observed frequencies of occurrence in seasonal and regional prey consumed by Steller sea lions.

The cumulative effect of early summer fisheries described above could affect Steller sea lions during an important weaning period for juveniles and leading up to the birth of pups. Due to intensive salmon fishing activity in such areas during the same times when Steller sea lions target concentrations of salmon, individual Steller sea lions may feed less efficiently or may avoid these feeding opportunities entirely. The salmon escapement goals limit the commercial harvest to the surplus above the amount needed for spawning (Kruse *et al.* 2000), but these harvest controls probably do not eliminate competition for available salmon between Steller sea lions and the fishery. However, as noted in Kruse *et al.* (2000) the abundance of salmon biomass increased dramatically during the time period that the western DPS of Steller sea lion has been in decline.

State managed salmon fisheries are open for relatively short periods, and only rarely remain open for 24 hours per day, 7 days per week (Kruse *et al.* 2000). Nevertheless, many of these fisheries take place at stream or river outlets where salmon congregate before moving upstream to spawn (Kruse *et al.* 2000). These same areas may provide important Steller sea lion foraging opportunities on high-density prey, enabling the Steller sea lions to feed efficiently and survive other periods of low prey availability.

Summary of State Fisheries

State managed fisheries represent a substantial influence on the near-shore marine ecosystem in Alaska. Both parallel and State managed groundfish fisheries occur almost entirely within Steller sea lion critical habitat (inside 3 nm). Because management of these fisheries is done on a regional basis, it is difficult to describe the overall impact of these fisheries on Steller sea lions or their critical habitat, although efforts such as Kruse *et al.* (2000) and Woodby and Hulbert (2006) are quite helpful. Most activity that occurs within state waters, including harvest and vessel traffic, occurs within Steller sea lion critical habitat.

The parallel fisheries for pollock, Pacific cod, and Atka mackerel are by far the largest groundfish fisheries within state waters by weight. The parallel fishery pollock harvest is an order of magnitude larger than the State managed fishery. Parallel Pacific cod landings are greater than State managed landings everywhere except the Chignik area. Atka mackerel is currently not harvested in a State managed fishery.

Fisheries for Pacific cod, pollock, Atka mackerel, salmon, and herring occur throughout the year. According to Woodby and Hulbert (2006), Pacific cod is harvested in nearshore waters from January through May, pollock is harvested in January and early fall, and Atka mackerel is fished in the summer. Salmon harvest was at an all-time high in 2005, and was highest in June, July and August. The herring sac-roe fisheries occur in April and May, while food and bait fisheries occur periodically throughout the year.

Fisheries for species other than pollock, Pacific cod, and Atka mackerel could potentially affect Steller sea lions due to vessel noise, disturbance, pollution, and ecosystem level effects. Fisheries for other groundfish, including sablefish, lingcod, and rockfish, occur primarily in the summer inside critical habitat. Additionally, fisheries for several invertebrate species also occur inside state waters. Crab, shrimp, scallop, and sea cucumber fisheries all occur inside critical habitat. Tanner crab fisheries have reopened and occur in January, while Dungeness crab are harvested in late summer, and Aleutian Islands golden king crab are taken in the winter. Scallops were taken July through December of 2005, and small volumes of shrimp were taken in trawl fisheries from April through September. Kodiak Island in particular has a high level of fishing activity for groundfish and invertebrates year-round (Woodby and Hulbert 2006) which could result in changes to the Steller sea lion prey field year-round.

4.5.2.5 Response of Steller Sea Lion Rookery Cluster Areas

In this section we examine catch and fishing patterns for important Steller sea lion prey species in a more detailed geographic and temporal context within the various zones comprising designated critical habitat on both an annual and seasonal basis. The analysis presents three major types of information for each RCA in this order:(a) fine-scale fishery harvest information and Total Allowable Catch (TAC) information[20] (b) available data on Steller sea lion foraging behavior; and (c) observed trends in Steller sea lion pup and non-pup counts. The catch data used in this analysis are derived from the Expanded Observer dataset. In the August 2010 draft of this Biological Opinion biomass information was also provided for each RCA. Biomass information has been revised in response to comment on the draft Biological Opinion to use the methods in the Stock Assessment and Fishery Evaluation (SAFE) reports for determining biomass by area since they comprise the best available information. Following the methods in the SAFE, biomass is reported by NMFS fishery management area. Because there is a large number of figures and tables, we have placed them into Appendix IV for ease of reference.

RCA 1

RCA 1 is the Steller sea lion Recovery Plan western Aleutian Islands sub-region and includes fishing area 543. No significant fishery take of pollock or arrowtooth flounder now occurs in this area (Table IV-1). The amount of Pacific cod taken in RCA 1 has fluctuated over time with a peak of about 13,000 mt in 2001 to a recent low of 3,152 mt in 2002. Since 2002, the amount of Pacific cod taken in RCA 1 has steadily increased. Most (90% over 1991-2008) Pacific cod taken in RCA 1 comes out of critical habitat with a 350% increase in critical habitat catch from 1999 to 2008 (Table IV-1999-2008-Area 1). The proportion of Pacific cod taken in critical habitat increased from 80% in 2002 to about 95% in 2007 and 2008 (Table IV-1). Since 1999, most (60% or greater) of the Pacific cod taken in RCA 1 has been caught in the first quarter (Figure IV-3.1). There has been a decrease in the proportion of the Pacific cod taken within 20 nm of rookeries between the 1991-1999 (pre-1999) and 2000-2008 (post-1999) periods (average of 69% and 59%, respectively); but an increase in the proportion of the catch taken within 20 nm of haulouts (46% and 64%, respectively).

Atka mackerel catches in RCA 1 have generally been stable at about 16,000 mt since a peak of 41,000 mt taken in 1996 (Table IV-1). Pre-1999 catch was predominately taken from within critical habitat (1991-1999 average 87%), since from 2000-2008 the proportion taken out of critical habitat was 41%. Since 2001, the Atka mackerel catch in RCA 1 has been concentrated in the third quarter (58% average 2001-2008). On average a declining proportion of the Atka mackerel catch has been taken within 20 nm of rookeries and haulouts in this area (rookeries: 1991-1999: 83%, 2000-2008: 28%; haulouts: 1991-1999: 38%, 2000-2008: 18%).

When comparing 1999 to 2008, there has been a 38% increase in the amount of total catch (4 primary species of pollock, Pacific cod, Atka mackerel, and arrowtooth flounder) and a 7% increase of total catch in critical habitat taken out of RCA 1 (Table IV-1999-2008-Area 1).

---

[20] The NMFS (2010a) biomass by RCA estimates are based on age 0+, 1+ or 2+ biomass for most stocks in order to quantify the prey available to sea lions (GOA and Bering Sea pollock are the exceptions). So they may not track precisely temporal patterns in TAC which is proportional to spawning biomass particularly if a large year class of subadult fish is present. Total Allowable Catch (TAC) is from the 2009 Stock Assessment and Fishery Evaluation Reports (http://www.afsc.noaa.gov/REFM/Stocks/assessments.htm).

Atka mackerel TAC in Area 543 (RCA 1) was lower in 2008 than 1999 (16,900 versus 27,000 mt; Table IV-12). Pacific cod TAC throughout the Aleutians and Bering Sea (RCAs 1-6) was also lower in 2008 than 1999 (146,837 versus 210,000 mt; Table IV-12). Total estimated pollock, Pacific cod, and Atka mackerel biomass in Area 543 in 2008 was slightly lower than 1999: 268,742 mt compared to 271,701 mt (Table IV-12). The difference was driven primarily by lower Pacific cod and pollock biomass in 2008 relative to 1999. Atka mackerel comprised 90% of the total biomass of pollock, Pacific cod and Atka mackerel in Area 543 in 2008. Atka mackerel biomass in Area 543 was approximately 7% higher in 2008 than 1999 (Table IV-12).

No Steller sea lions have been captured and instrumented with satellite tags in RCAs 1, 2, 7 and 8. However, 3 telemetered sea lions entered RCA 1 from other areas (AFSC 2010b). Foraging locations from juvenile sea lions that entered RCA 1 occurred outside of 20 nm from listed sites in a much higher proportion than Steller sea lions in RCAs 3-10 (AFSC, 2010b). In RCA 1 only one animal had 50% of their locations within 10 nm to the nearest listed haulout or rookery. Another complicating factor in the interpretation of these data is that all three were males captured and initially instrumented as 9-11 month-olds at Adak Island (RCA 4) in April or May, and all subsequently travelled west through RCAs 3, 2, and 1. Their offshore movements over the Bering Sea basin outside of critical habitat presumably correspond to foraging opportunities associated with prey-concentrating oceanographic features, such as eddies, as is observed in the Eastern and Central Aleutian Islands for animals of similar age (Fadely *et al.* 2005, Lander *et al.* 2009).

Another perspective of critical habitat use by juvenile Steller sea lions in this RCA may be achieved by grouping results from RCAs 1-4. Of the combined 28 animal-RCA individuals (some individuals occurred in multiple RCAs), 22 had 50% or more locations within 20 nm, and 19 had 75% or more of their locations associated with diving to >4 m within 20 nm of a listed site (AFSC 2010b Table 3). However because many of the diving locations by juvenile Steller sea lions in RCA 1-4 are offshore (AFSC 2010b Figures 1 and 3), distances from a listed site that encompassed 95% of an individual's diving locations were ≤10 nm for 16 of the 28 (57%), and >20 nm for 11 of the 28 (39%). In this RCA, Steller sea lions foraged primarily on Atka mackerel, cephalopods, salmon, and Pacific cod (Sinclair and Zeppelin 2002; Sinclair *et al.* 2005).

Steller sea lions foraging from RCA 1 likely also go beyond the areas discussed above and feed in waters of the western Bering Sea and eastern Kamchatka Peninsula. Branding studies have shown that Steller sea lions marked in the Kamchatka Peninsula and Commander Islands rookeries move as far as the eastern Bering Sea and eastern Aleutian Islands (Burkanov and Loughlin 2005); movement of animals from the western Aleutian Islands into Russian waters has not been documented (no branding occurs that far west) but is expected to occur. The commercial catch of walleye pollock in the western Bering Sea/Kamchatka areas averages 852,000 mt per year, with a range of between 1.2 million mt in 1999 to 633,000 mt in 2004. Atka mackerel catch averages 42,500 mt per year, with a range of between 54,351 mt in 2003 to 30,390 mt in 1999. Pacific cod, herring, and halibut are also taken by commercial catch in the western Bering Sea and Kamchatka Peninsula areas.

This RCA consists of ten trend sites, which includes four rookeries (Buldir, Agattu [2], and Attu). Steller sea lion counts at all rookeries and trend sites in RCA 1 declined from 4,920 in 1991, to 2,865 in 1998, and to 894 in 2008 (Table 2 in AFSC 2010a), a significant rate of decline at over 6% from 2000-2008 (Table 3.1c). There are no survey data for this area for 2007 and limited data (missing Buldir) for 2006, but it appears that most of the decline between 2004 and 2008 occurred in the first 2 years (-19%; loss of 230 animals) and was smaller in the last 2 years (-11%; loss of 111 animals). Pup production has also declined here with pup counts down 64 animals between 2005 and 2009. Overall, pup production has declined by 10.4% since 1997.

RCA 2

RCAs 2-5 comprise the Steller sea lion Recovery Plan central Aleutian Islands sub-region and include fishing areas 541 and 542. No significant fishery for pollock or arrowtooth flounder occurred in this area between 1991-2008 (Table IV-2). Since 1991, the catch of Pacific cod in RCA 2 has averaged 4,400mt per year (range of 230 mt-9,169 mt) with little difference between the pre-1999 (average 4,663 mt) and post-1999 (4,264 mt) periods. About 70% of the Pacific cod in RCA 2 is taken out of critical habitat with a decrease in that proportion between pre-1999 (74%) and post-1999 (65%). Since 1999, about 60% of the catch was taken within 20 nm of rookeries in the area, while about 35% was taken within 20 nm of haulouts. Seasonally, Pacific cod have primarily been taken from RCA 2 in the first quarter; some dispersion into the third quarter occurred beginning in 2006 (Figure IV-3.2).

Atka mackerel catches in RCA 2 peaked at about 40,000 mt in 1995, declined to a little over 10,000 mt in 1997 and increased steadily until 1997 (36,000 mt). The proportion of the catch taken in critical habitat declined markedly beginning in 1999 (1991-1999: 65%; 2000-2008: 41%). Since 1999, about 42% of the Atka mackerel was taken within 20 nm of rookeries in the area, while about 7% was caught within 20 nm of haulouts (Figure IV-3.2). Seasonally, about 40% of the catch of RCA 2 Atka mackerel occurs in each of the first and third quarters. When comparing 1999 to 2008, there has been a 14% increase in the total amount of total catch (4 primary species) and a 41% increase in that taken out of critical habitat of RCA 2 (Table IV-1999-2008-Area 2).

Estimated Pacific cod biomass in Areas 541 and 542 was lower in 2008 than 1999 (59,956 compared to 93,191 mt (Table IV-12). Atka mackerel TAC in NMFS Area 542 (RCAs 2 and 3) was slightly higher in 2008 than 1999 (24,300 versus 22,400 tons; Table IV-12). In contrast, Pacific cod TAC throughout the Aleutians and Bering Sea (RCAs 1-6) was lower in 2008 than 1999 (146,837 versus 210,000 tons; Table IV-12). Total estimated groundfish biomass in Area 542 (pollock, Pacific cod, and Atka mackerel) was higher in 2008 than 1999: 321,871 tons compared to 272,925 mt (Table IV-12). The difference was driven primarily by lower Pacific cod and pollock biomass. In contrast, Atka mackerel biomass in Area 542 was higher in 2008 than 1999 (268,957 mt versus 196,433 tons).

No Steller sea lions have been captured and instrumented with satellite tags in RCAs 1, 2, 7 and 8. However, 3 telemetered sea lions (all males) entered RCA 2 from other areas (AFSC 2010b). Foraging locations from juvenile sea lions that entered RCA 2 occurred outside of 20 nm from listed sites in a much higher proportion than Steller sea lions in RCAs 3-10 (AFSC 2010b). No animals in RCA 2 had more than 40% of their locations within 10 or 20 nm of a listed site. These three male 9-11 month animals were instrumented in RCA 4, and a more complete understanding of critical habitat use may be obtained from combining patterns among RCAs 1-4 (as described at Section 5.2.3). Of the combined 28 animal-RCA individuals (some individuals occurred in multiple RCAs), 22 had 50% or more locations within 20 nm, and 19 had 75% or more of their locations associated with diving to >4 m within 20 nm of a listed site (AFSC 2010b Table 3). Because many of the diving locations by juvenile Steller sea lions in RCA 1-4 are offshore (AFSC 2010b Figures 1 and 3), distances from a listed site that encompassed 95% of an individual's diving locations were ≤10 nm for 16 of the 28 (57%), and >20 nm for 11 of the 28 (39%). Steller sea lions in RCA 2 forage primarily on Atka mackerel, cephalopods, salmon, and Pacific cod (Sinclair and Zeppelin, 2002; Sinclair *et al.* 2005).

A recent analysis of opportunistic sightings of Steller sea lions (the Platforms of Opportunity, or POP database) yielded results consistent with the limited telemetry information available for the western and central Aleutian Islands. Boor (2010) derived spatially-explicit quantitative estimates of Steller sea lion at-sea habitat use by standardizing effort for the opportunistic sightings. The POP data span a 43 year period, 1958-2000, and contain 13,037 sightings representing 109,323 individual Steller sea lions. Boor (2010) also analyzed seasonal patterns of at-sea sightings. Steller sea lion encounter rates were high along the continental shelf break throughout the Gulf of Alaska and in the Bering Sea. High encounter rates also occurred in the Bering Sea's Aleutian Basin between Bower's Ridge and the Bering Sea continental shelf and in the offshore region to the south and east of Attu and Agattu Islands (Boor 2010). Offshore encounter rates were high in the breeding season (May through August) south and east of Attu and Aggattu Islands; however, there were no ship-days recorded in this region in the non-breeding season, confounding interpretation as to whether Steller sea-lions use this habitat in the non-breeding season. It is important to note that the offshore sightings south and east of Attu and Aggattu Islands in the POP database were all observed in the 1980s when the abundance of Steller sea lions was higher than present. Though, very few ship-days were recorded in the region before and after the 1980s. It is unknown as to whether or not habitat use is similar today, however NMFS assumes at-sea use inferred from telemetry and POP information summarized in AFSC (2010b) and Boor (2010) reflect the at-sea use of Steller sea lions in the respective regions as they are the best data available.

RCA 2 consists of twelve trend sites, which include four rookeries (Amchitka/Column Rock, Ayugadak, and two rookeries on Kiska). Counts of pups and non-pups dropped by approximately 4% per year from 2000 to 2008.

RCA 3

Between 20,000 mt and 40,000 mt per year of pollock were taken from RCA 3 between 1995-1997 (Table IV-3). Since 1999, the average annual removal of pollock from RCA 3 has been less than 200 mt per year.

The catch of Pacific cod in RCA 3 since 1999 has been fairly stable averaging about 1,500 mt year; most (98%) is taken inside of critical habitat (Table IV-3). Most Pacific cod in RCA 3 were caught in the first quarter, although recently (2005 on) the catch has tended to be more evenly distributed across the year (Figure IV-3.3).

On average, about 4,000 mt of Atka mackerel have been caught in RCA 3 (1999-2008). Most of the catch (99%) was taken inside critical habitat (Table IV-3). Since 2002, there has been a shift in the annual distribution of quarterly catch of Atka mackerel in RCA 3 from the first quarter (68% of the annual catch) to the third quarter (67% average 2003-2008; (Figure IV-3.3). A similar shift in the seasonality of pollock catch is evident (Figure IV-3.3). Generally, when comparing catch between 1999 and 2008 in RCA 3 (Table IV-1999-2008-Area 3), there has been a 40% decline in both the amount of total catch and the amount of catch from critical habitat.

Unlike RCAs 1 and 2, at least 75% of dive locations for Steller sea lions (n = 8) in this RCA (and most other RCAs) were within the 10 nm and 20 nm zones. Looking at all zones, just under a third of the Steller sea lions had 100% of their locations associated with diving >4 m within 10 nm (Table-- AFSC Table 3). Though more Steller sea lions were tracked in RCA 3 than in RCAs 1-2, only two were initially captured and instrumented within this RCA (AFSC 2010b) and hence another perspective of critical habitat use may be obtained from combining patterns among RCAs 1-4 (as described at Section 5.2.3). Of the combined 28 animal-RCA individuals (some individuals occurred in multiple RCAs), 22 had 50% or more locations within 20 nm, and 19 had 75% or more of their locations associated with diving to >4 m within 20 nm of a listed site (AFSC 2010b Table 3). However because many of the diving locations by juvenile Steller sea lions in RCA 1-4 are offshore (AFSC 2010b Figures 1 and 3), distances from a listed site that encompassed 95% of an individual's diving locations were ≤10 nm for 16 of the 28 (57%), and >20 nm for 11 of the 28 (39%). As with RCAs 1 and 2, Steller sea lions in RCA 3 forage primarily on Atka mackerel, cephalopods, salmon, and Pacific cod (Sinclair and Zeppelin 2002, Sinclair *et al.* 2005).

RCA 3 has twelve trend sites, of which four are rookeries (Gramp Rocks, Tag, Ulak, and Kanaga), and is part of fishing area 542 and it is part of the central Aleutian Islands complex. Non-pup counts in RCA 3 have declined by 1% per year between 2000 and 2008, while pup counts declined by 4% per year during this same time period.

RCAs 4 and 5

Prior to 1999, on average about 7,000 mt of pollock were taken in RCA 4. Since then, the average removal has been 288 mt per year (Table IV-4). The predominant fish caught in RCA 4 is Pacific cod; on average about 9,700 mt/yr are removed, with most (82%) taken within critical habitat. Since 2004, there has been a nearer-to-shore trend in fishing effort and catch for Pacific cod, with a greater proportion (50%) of the catch coming out of the 3-10 zone of critical habitat than in previous years (Figure IV-2.4). The catch has been consistently concentrated (87%) in the first quarter of the year (Figure IV-3.4). Since 1999, 42% of the catch of Pacific cod was taken within 20 nm of rookeries, and 76% was within 20 nm of haulouts (Table IV-4). Generally, there has been a 41% decline in the amount of total catch (all 4 species) and a 50% decline in the amount of catch from critical habitat within RCA 4 when comparing catch between 1999 and 2008 (Table IV-1999-2008-Area 4).

Pacific cod TAC throughout the Aleutians and Bering Sea (RCAs 1-6) was lower in 2008 than 1999 (146,837 versus 210,000 tons; Table IV-12). Total estimated groundfish biomass in Area 541 (pollock, Pacific cod, and Atka mackerel) was much higher in 2008 (422,625 mt) than 1999 (190,945) (Table IV-12). Increases in groundfish biomass in 2008 in Area 541 were due to large increases in Atka mackerel biomass and pollock biomass relative to 1999. Overall, Pacific cod biomass was lower in 2008 than 1999 in NMFS Area 541 (Table IV-12).

Catches of over 40,000 mt of pollock were taken in RCA 5 prior to 1995. Since 1995 the average annual catch has been about 400 mt year (Table IV-5). Pacific cod catches have been fairly stable at about 9,800 mt per year since 1999. Most (73%) of this Pacific cod catch has been in critical habitat, primarily (about 50%) in the 10-20 nm zone (Figure IV-2.5). The catch of Pacific cod in RCA 5 is seasonal in nature with most (87%) consistently being caught in the first quarter (Figure IV-3.5). Only 5% of the catch of Pacific cod was taken within 20 nm of rookeries, 72% was taken within 20 nm of haulouts (Table IV-5). Very little was taken in the Seguam foraging area.

Atka mackerel catch in RCA 5 averaged over 20,000 mt per year prior to 1999; between 2000 and 2006 the average annual catch declined to 6,000 mt per year, but increased in 2007-2008 to over 19,000 mt (Table IV-5). Catch of Atka mackerel in critical habitat decreased steadily in RCA 5 since 1996 to only about 1% in 2008 (1999-2008 average equals 8%). Similarly only about 3% of the catch of Atka mackerel was taken within 20 nm of rookeries, and 6% was within 20 nm of haulouts (Table IV-5). Since 1999 there has been a shift in the seasonality of catch shown by a decrease in the proportion taken in the first quarter (85% - average 1991-1999, to 26% current) and an increase in the third and fourth quarters (Figure IV-3.5). In recent years (2005-2008) the catch has been more seasonally dispersed with roughly equal amounts of mackerel caught in the 1st and 4th quarters (26%) and most of the remainder taken in the 3rd quarter (42%). There has been a 47% increase in the amount of total catch (pollock, Pacific cod, Atka mackerel, and arrowtooth flounder) and there has been a 21% increase in the amount of catch from critical habitat within RCA 5 when comparing catch between 1999 and 2008 (Table IV-1999-2008-Area 5).

Eighteen juvenile Steller sea lions were monitored with telemetry in these RCAs, more than in the RCAs farther west. Steller sea lions foraged in all zones in these RCAs; all dives >4 m were within 10 nm of a rookery or haulout site and nearly all dives >4 m were within the 20 nm zone. Apparently, juvenile Steller sea lions foraging in these islands utilize habitat within 20 nm of rookeries and haulout sites. As with the RCAs discussed above, Steller sea lions foraging in these RCAs prey on Atka mackerel, cephalopods, salmon, and Pacific cod (Sinclair and Zeppelin, 2002, Sinclair *et al.* 2005).

An unpublished study (Ortiz and Logerwell, 2010) evaluating the efficacy of trawl exclusion zones for preserving prey fields of Steller sea lions foraging on Atka mackerel found that Atka mackerel production in the Seguam Pass trawl exclusion zone (part of RCA 5) greatly exceeded the demands of Steller sea lion consumption. In contrast, the production of Atka mackerel in the Amchitka north trawl exclusion zone (RCA 2) probably could not support current or historical foraging needs of Steller sea lions. These results suggest that a "one-size-fits-all" approach to designating protection measures for Steller sea lions may not be effective, and factors such as local biomass of prey and competing predators need to be taken into consideration when designing trawl exclusion zones and other marine protected areas.

RCA 4 has thirteen trend sites of which two are rookeries (Kasatochi and Adak islands) and RCA 5 has twelve trend sites of which two are rookeries (Yunaska and Seguam islands) and includes fishing area 541. Non-pup numbers declined in RCAs 4 by 4% per year between 2000 and 2008, while pup numbers increased at 2% during this same time period. In RCA 5, there was a 2% per year increase in both non-pup and pup counts between 2000 and 2008.

RCA 6

This geographically large region is the predominant area targeted by the pollock fishery in the Bering Sea with catches between 992,601-1,494,547 mt per year (1999-2008, Table IV-6). Generally, the description of the pollock catch in the Bering Sea made in the previous section describes the nature of the catch in this specific RCA. It is relevant to note the proportion of pollock caught in RCA 6 critical habitat has on average declined from pre-1999 (46% in critical habitat) to post-1999 amounts (34% in critical habitat). Typically one-half to two-thirds of the pollock catch taken in critical habitat continues to be caught in the foraging area (SSLCA). Approximately 25% of the catch occurred in the SSLCA during 2000-2008 (down from pre-1999 level of 37%). The catch from 1991-2008 has been seasonally aggregated with roughly 40% taken in each of the 1st and 3rd quarters (Figure IV-3.6). Only about 7% of the pollock catch was taken within 20 nm of Steller sea lion rookeries or haulouts throughout the 1991-2008 period (Table IV-6). Similarly, RCA 6 is the predominant area targeted by the Pacific cod fishery in the Bering Sea, and the prior analysis generally holds for this specific area. Similar to pollock, there has been a decline in the amount of Pacific cod taken in critical habitat zones in RCA 6 since 1999 (pre-1999 – 30% in critical habitat; post-1999 was 26% (Table IV-6). Of the total catch about 16% is taken in the foraging zone of critical habitat while about 7% is caught in the 10-20 nm zone (1999-2008). Roughly 50% of the Pacific cod is taken in the 1st quarter of the year with between 10-20% being taken in the other 3 quarters (Figure IV-3-6). Only about 7% of the Pacific cod catch was taken within 20 nm of Steller sea lion rookeries and 9% near haulouts – these ratios have held throughout the 1991-2008 period (Table IV-6). Noteworthy catches of Arrowtooth flounder occur in RCA 6 (Table IV-6). Since the early 1990s there has been a general increase in ATF catch (about 10,000 mt average vs. an average of about 14,000 mt per year in recent years – and 71% increase between 1999 and 2008). More of the arrowtooth catch (45% vs. 36%) has been taken out of critical habitat since 1999. Typically, most of this catch from critical habitat has been taken in the foraging zone (27% of total). A much greater proportion of the arrowtooth catch in 2008 was made within 20 nm of rookeries (36%) and haulouts (31%) than in the 1999-2008 period (16%). Looking at the overall catch data between 1999 and 2008, there has been a nearly stable (-0.4%) amount of total catch (all 4 species) made in RCA 6; however, the amount of fish taken from critical habitat in the area declined 82,000 mt (22%) over this period (Table IV-1999-2008-6).

Pollock TAC in the Bering Sea was slightly higher in 2008 than 1999 (1,000,000 versus 992,000 tons) but is 813,000 mt for 2010. Arrowtooth flounder TAC in the Bering Sea was lower in 2008 than 1999 (75,000 versus 134,354 mt). Pacific cod TAC throughout the Aleutians and Bering Sea (RCAs 1-5 plus Bering Sea) was lower in 2008 than 1999 (146,837 versus 210,000 mt). The cod TAC for 2010 in the BSAI is 168,780 mt. Pollock and Pacific cod biomass in the Bering Sea was substantially lower in 2008 than 1999

(Table IV-12). This difference largely was driven by lower pollock biomass in 2008 compared to 1999 (3,809,000 versus 10,421,000 mt). Pacific cod biomass in the Bering Sea was 998,912 in 2008 compared to 1,422,590 mt in 1999.

As with RCAs 4 and 5, eighteen juvenile Steller sea lions were monitored in this RCA to assess foraging behavior in relation to the 10 nm and 20 nm buffer zones. Individual-based results of this analysis are consistent with that finding in that most individuals had at least 75% of their dive locations within 10 and 20 nm distances to a listed site (Table 3 in AFSC 2010b shown as Zones 6 and SSLCA). The 20 nm distance from a listed site in RCA 6 encompassed all locations associated with diving to >4 m for 15 of 18 Steller sea lions. POP data analyzed by Boor (2010) show a much broader use of the Bering Sea's Aleutian Basin between Bowers Ridge and the Bering Sea continental shelf than indicated by the available telemetry data. Telemetry and POP data align to a much greater extent in the Gulf of Alaska.

The diet in this and areas to the east shows a transition of one dominated by Atka mackerel to one of pollock. The dominant prey in order of frequency of occurrence in scat were walleye pollock, salmon, herring, Atka mackerel, Pacific cod, Irish lord, and cephalopods (Sinclair and Zeppelin 2002). Seasonal differences in prey consumed were apparent (as with most RCAs) when salmon was most prevalent in summer than winter and Irish lord and Pacific cod more prevalent in winter than summer.

RCA 6 has thirty-one trend sites, of which seven are rookeries (Ugamak, Akun, Akutan, Bogoslof, Adugak, Ogchul islands, and Sea Lion Rock), and includes all fishing areas north in the eastern Bering Sea and south in the Gulf of Alaska (part of area 610). This RCA is part of the eastern Aleutian Islands Steller sea lion complex. RCA 6 declined dramatically in the 1970s and early 1980s but has demonstrated a strong recovery since then, perhaps the result of mitigation measures imposed in the 1990s and 2000s. Between 2000 and 2008, counts of non-pups and pups increased at 4% per year and 5% per year, respectively.

RCA 7

On average about 20,000 mt per year of pollock are harvested in RCA 7 (Table IV-7). After a substantial decrease in the proportion taken in critical habitat in 2000-2003, the proportion gradually increased to a relatively stable amount (about 60%). Much of the increased pollock catch in critical habitat occurred in the 3-10 zone with some catch coming from within the 0-3 nm zone. During this period the catch has been relatively evenly distributed through the 1st, 3rd, and 4th quarters (Figure IV-3.7). In comparing pre-1999 and post 1999 catches, the average proportion of pollock taken within 20 nm of rookeries decreased from 18% to 5% of the total catch, while there was little change in the proportion taken within 20 nm of haulouts during those periods (52 vs 53%, respectively, Table IV-7). Pacific cod catch has generally averaged about 18,000 mt since the mid 1990s. The amount removed from critical habitat dropped from about 85% to 63% since 2001 with most of this change coming from reduced catches within the 3-10 nm zone (since 2003; Table IV-7). There was some initial seasonal dispersal of the fishery in 2003, but the Pacific cod trawl and pot fisheries now have returned to an aggregated seasonal catch (on average 66% of the annual catch is taken during the first quarter; Figure IV-3.7). Atka mackerel and arrowtooth flounder catches are minimal in RCA 7 (Table IV-7). Overall, there was not only a 10% decrease in the amount of total catch (all 4 species), there was a 36% decrease in the amount of catch from critical habitat within RCA 7 when comparing the catches of 1999 and 2008 (Table IV-1999-2008-Area 7).

Pacific cod biomass in NMFS Area 610 was lower in 2008 than 1999 (128,515 versus 140,108 mt) (Table IV-12). Biomass of pollock in Area 610 was lower in 2008 (150,084 mt) than 1999 (315,713) (Table IV-12). Pacific cod and pollock TAC was also lower in 2008 than 1999 in NMFS Area 610.

Twenty-four juvenile Steller sea lions were monitored with telemetry in this RCA, more than any RCA except RCA 9, though none were captured within RCA 7 and all entered from elsewhere (AFSC 2010b). Steller sea lion foraged in all zones in this RCA; all dives >4 m recorded showed dives within 10 nm of a rookery or haulout site and nearly all dives >4 m were within the 20 nm zone. Apparently juvenile Steller sea lions foraging in these islands utilize habitat within 20 nm of rookeries and haulout sites.

The highest frequency of occurrence of prey in Steller sea lion scats in this RCA was walleye pollock, salmon, and arrowtooth flounder (Sinclair *et al.* 2005). Additional prey consumed by Steller sea lions in this area include Pacific cod, salmon, Irish lord, and herring (Sinclair and Zeppelin 2002).

RCA 7 has sixteen Steller sea lion trend sites, of which five are rookeries (Pinnacle Rock, Clubbing Rocks, Atkins, Chernabura, and Jude islands), and is essentially equivalent to the Steller sea lion western GOA area; it also includes fishing area 610. Like RCA 6, Steller sea lion abundance in RCA 7 declined dramatically in the 1970s and early 1980s but has demonstrated positive growth since, perhaps due to mitigation measures adopted in the 1990s and 2000s. Between 2000 and 2008, counts of non-pups and pups increased at 5% per year and 3% per year, respectively.

RCA 8

On average in RCA 8, pollock catches have been relatively stable at around 19,000 mt per year since a substantial reduction in harvest occurred in 2000 due to a court-ordered closure of critical habitat to trawl fisheries (Table IV-8). The pollock trawl fishery is highly concentrated (84%) in the first quarter in RCA 8, with about 77% being taken from critical habitat since 1999 (Figure IV-3.8). With 2008 being the exception, most of the pollock taken in critical habitat came out of the 10-20 nm zone. Little pollock is caught within 20 nm of rookeries in RCA 8, while about 70% of the 2000-2008 catch was made within 20 nm of haulouts. Pacific cod catches have declined since 1996 in RCA 8 (with a slight uptick in 2008 (Table IV-8). Most (66%) of the catch of Pacific cod in RCA 8 tends to be taken in the first quarter, although there has been some slight seasonal dispersal with somewhat more being caught in the 4th quarter since 2004 (Figure IV-3.8). Since 1999, about 57% of the Pacific cod caught in RCA 8 comes out of critical habitat, with an increasing proportion of that coming out of the foraging zone (Table IV-8). Prior to 1999, most (77%) Pacific cod was taken from critical habitat areas, primarily from the 10-20 nm zone. The proportion of Pacific cod caught within 20 nm of Steller sea lion rookeries or haulouts in RCA 8 has dropped notably since 1999 (from 30-50% to generally 2-5% in recent years, Table IV-8). About 4,000 mt per year of arrowtooth flounder are taken in RCA 8 with an increasing proportion (73%) being removed from critical habitat since 2004 (Table IV-8). Generally, there has been a 39% decline in the amount of total catch (all 4 species) and a 45% decline in the amount of catch from critical habitat within RCA 8 when comparing catch between 1999 and 2008 (Table IV-1999-2008-Area 8).

Pollock biomass in the central GOA (NMFS Areas 620, 630 and 640) was lower in 2008 (226,771 mt) than 1999 (339,713 mt) (Table IV-12). Pollock and Pacific cod TACs were lower in 2008 than 1999 in the relevant NMFS Areas, (Table RCA-1999-2008-TAC), as reflected in the total catch statistics. Total estimated groundfish biomass (pollock, Pacific cod and Atka mackerel) was lower in 2008 than 1999 (444,219 versus 599,514 mt) (Table IV-12). This difference was driven by lower biomass of both Pacific cod and pollock. Atka mackerel accounted for less than 1% of the biomass of these three prey species in the Central GOA (NMFS Areas 620, 630, and 640).

Steller sea lion foraging locations in RCA 8 showed much greater proportions of locations outside of 20 nm to a listed site than did Steller sea lions among other RCAs (Table 3 in AFSC, 2010b). No animals in RCA 8 had more than 75% of their locations within 10 or 20 nm of a listed site. Of the four animals monitored in RCA 8, all but three had all locations associated with diving to >4 m within 20 nm of a listed site. No Steller sea lions were captured within RCA 8, but rather entered from RCA 9 (AFSC

2010b). Prey of Steller sea lions foraging in this RCA was similar to that in RCA 7 and was dominated by walleye pollock, salmon, and arrowtooth flounder with Pacific cod, salmon, Irish lord, and herring also consumed (Sinclair and Zeppelin 2002).

RCA 8 has eleven Steller sea lion trend sites, of which three are rookeries (Lighthouse Rock, Chowiet, and Chirikof), and includes fishing area 620. This area is about half of the original central GOA which also contained two large rookeries (Marmot and Sugarloaf islands) and many haulouts in the Kodiak archipelago which are now part of RCA 9. RCA 8 also declined in the 1970s and early 1980s. Between 2000 and 2008, counts of non-pups were stable, while pup counts increased at 1% per year.

RCA 9

The catch of pollock in RCA 9 has generally declined since 1991 with a fairly stable catch of about 17,600 mt per year since 2004. Most of the catch is from critical habitat with little change between pre-1999 and post-1999 proportions (91% and 89%, respectively; Table IV-9). About 70-80% of this comes out of the 10-20 nm zone. Since 2002, about 26% of the pollock taken in RCA 9 comes from within 20 nm of Steller sea lion rookeries; 92% is taken within 20 nm of haulouts. Prior to 2002 these proportions were 14% and 85%, respectively. The catch remains seasonally aggregated, although the timing has shifted, since 1999 the catch has been taken mainly within the 1st, 3rd, and 4th quarter, where prior to 1999 it was aggregated in the 1st, 2nd, and 4th quarters (Figure IV-3.9). While the catch of Pacific cod has fluctuated in RCA 9 over the 1991-2008 period, it has generally averaged about 23,000 mt per year with little change in the proportion removed from critical habitat (49% pre-1999, 45% post 1999, Table IV-2-9). Most of that portion of the catch continues to be taken from the 10-20 nm zone (Table IV-9). Since 2002, 8% of the Pacific cod has been taken within 20 nm of Steller sea lion rookeries and 43% from within 20 nm of haulouts in RCA 9. There has been only a slight increase in the seasonal dispersal of the catch of Pacific cod in RCA 9 since 1999 with a somewhat greater proportion of the catch coming in the 3rd quarter and a slight decline in the amount taken in the 1st quarter (Figure IV-3.9). Very little Atka mackerel has been taken in RCA 9. Arrowtooth flounder catches in RCA 9 have increased steadily since 1999, with a high of over 22,000 mt in 2008. The proportion of the total catch taken in critical habitat increased from 1999-2005 (77%), although it dropped in 2007 and 2008 (30%). Overall, there was a 6% decrease in the amount of total catch (all 4 species) and a 24% decrease in the amount of catch from critical habitat within RCA 9 when comparing the catches of 1999 and 2008 (Table IV-1999-2008-Area 9).

Forty-two juvenile Steller sea lions were monitored with telemetry in this RCA (AFSC 2010b). Steller sea lions foraged in all zones in this RCA; 40 of 42 dives >4 m recorded had 75% of their dives within 10 nm of a rookery or haulout site; only 9 of 42 had 100% of their dives within 10 nm. The percentages of dives within 20 nm were similar to those within 10 nm except that more (28 of 42 animals) had 100% of their dives within 20 nm. Apparently juvenile Steller sea lions foraging in this area (Kodiak Island to Kenai Peninsula) extensively utilize habitat within 20 nm of rookeries and haulout sites. Scat collections from sites in this RCA show sand lance, pollock, arrowtooth flounder, cod, salmon and herring were the dominant prey items (McKenzie and Wynne 2008).

RCA 9 has eighteen Steller sea lion trend sites of which three are rookeries (Marmot, Sugarloaf, and Ushagat islands) and includes fishing area 630. Between 2000 and 2008, counts of non-pups were stable, while pup counts increased at 1% per year.

RCA 10

Since 1999, less than 5,000 mt per year of pollock has been harvested in RCA 10. Pacific cod harvest in this area has averaged about 1,100 mt per year and virtually no Atka mackerel or arrowtooth flounder have been taken in RCA 10 (Table IV-10). Overall, there was a 60% decrease in the amount of total catch (all 4 species) and a 37% decrease in the amount of catch from critical habitat within RCA 10 when comparing the catches of 1999 and 2008 (Table IV-1999-2008-Area 10). TAC of pollock and Pacific cod in RCA 10 are the lowest in the Gulf of Alaska (RCAs 6-10; Table IV-12).

Fifteen juvenile Steller sea lions were monitored with telemetry in this RCA (AFSC 2010b). Steller sea lions foraged primarily in the shallow zones within 10 nm; 13 of the 15 Steller sea lions had 75% of their dives in this zone and none had 100% of their dives within 10 nm in this RCA. However, all of these animals had at least 75% of their dives within 20 nm, and 7 of the 15 had 100% of their dives within 20 nm of a listed site suggesting that all areas within 20 nm were used for foraging and that those that did not forage within 10 nm did so within 20 nm. Juvenile Steller sea lions foraging in this area (Kenai Peninsula to the northern tip of Southeast Alaska) extensively utilize habitat within 20 nm of rookeries and haulout sites. There are few diet studies pertaining to this region but we assume that diet is similar here to adjacent regions and that pollock, Pacific cod, arrowtooth flounder, smelt, herring, and cephalopods are the dominate prey consumed.

RCA 10 has nineteen Steller sea lion trend sites, of which four are rookeries (Seal Rocks, Wooded Island, Outer Island, and Chiswell Island), and includes fishing area 640; it is part of the eastern GOA. Between 2000 and 2008, non-pup and pup counts increased at 4% per year.

Southeast Alaska

Marine critical habitat areas for the eastern DPS of Steller sea lion are much smaller than those in the western DPS; they include a zone 3,000 ft seaward of all rookeries and major haulouts. Only in 1997 was there a mentionable catch of pollock (94 mt) in this area, otherwise scant catches of between 0-18 mt were reported (Table IV-11). Minor catches of Pacific cod have occurred in Southeast Alaska (1,367 mt average, 1991-2008); the average catch between 1999-2008 was 68 mt. The catch of arrowtooth flounder has averaged 165 mt from 1991-2008; catches since 1999 have averaged 48 mt. No catch of Atka mackerel is reported for this area.

Analysis of dive behavior by juvenile Steller sea lions in relation to 10 and 20 nm zones for Southeast Alaska has not been conducted, since extensive critical habitat zones have not been designated in the eastern DPS. Scientists with the ADFG, University of Alaska, and others have deployed a large number of telemetric instruments on juvenile Steller sea lions in this area (Raum-Suryan *et al.* 2004, *Pitcher et al.* 2005). Though not the same as distance from shore, Raum-Suryan *et al.* (2004) found that round-trip distances of pups and juveniles in Southeast Alaska were shorter than for those in the western DPS. The most common prey of 61 species identified from scat collected in Southeast Alaska were walleye pollock, Pacific herring, sand lance, salmon, and arrowtooth flounder, rock fishes, and cephalopods (Trites *et al.* 2007).

This unnumbered RCA has ten trend sites, of which three are rookeries (Forrester Island Complex, Hazy Island, and White Sisters). The area has been stable or increasing for the past twenty plus years. In 1991, 8,034 non-pups were counted, increasing to 9,855 in 2000, but showed a decline to 8,749 in 2009. Results of the non-pup survey from late June 2009 in Southeast Alaska, and in the eastern and central GOA support the hypothesis that movement of Steller sea lions into the eastern GOA, primarily from Southeast Alaska, affected sub-area and western DPS counts obtained during the early June 2008 survey. Total non-pup counts in the eastern GOA were 812 lower in the “late” 2009 survey than in the “early” 2008 survey, while they were greater by 2,642 in Southeast Alaska. An additional 404 non-pups were also counted at trend sites in the central Gulf of Alaska in 2009 compared to 2008. Using the data collected in 2009 and calculating trends in each sub-area since 2000, shows that seasonal movement from Southeast Alaska may have contributed a minimum of 570 additional Steller sea lions to the 2008 western DPS trend site counts. If these are subtracted from the 2008 western DPS total, the percent difference in non-pup counts between 2004 and 2008 is reduced from 3% to 1%. Pup production in Southeast Alaska (eastern DPS) totaled 7,462 pups in 2009, with 7,443 counted at the five rookeries (including Biali Rocks) where 5,510 were counted in 2005. The increase in production of 1,933 pups since 2005 equates to approximately 97 more pups per year at each of the Southeast Alaska rookeries.

Between 2000 and 2008, non-pup counts increased at 2% per year, while pup counts increased at 5% per year. It is interesting to note that trends in abundance based on non-pup counts between 2000 and 2008 in the Gulf of Alaska (RCAs 7-10) increased at a rate slightly larger than that observed in SE Alaska (i.e., 3% per year versus 2% per year).

4.5.2.6 Fisheries in Russian Portion of Western DPS

To give a complete overview of fishing across the range of the western DPS, we here include information gathered on Russian fisheries in the Bering Sea from 1995 to 2008. For a map of these Russian waters, see Figure Appendix VI-1, and for more details on harvests by the Russian fisheries from 1995 to 2008, see Tables in Appendix VI.

Western Bering Sea

Between 392,140 mt and 691,656 mt of pollock were taken annually from Russian waters of the Western Bering Sea between 1995 and 2008. In 2008, the latest year for which information is available, Russian fisheries harvested 554,233 mt of pollock. Harvests of Pacific cod in the same years ranged from a low of 24,445 mt in 2002 to a high of 55,726 mt in 1996. In 2008—again, the latest year for which we have information—Russian fisheries harvested 36,047 mt of Pacific cod. Finally, the Russian Atka mackerel fisheries in the Western Bering Sea took 164 mt of Atka mackerel in 1995, and those harvests have steadily increased since then, with a high of 2,991 mt of Atka mackerel taken in 2008.

Eastern Kamchatka

In the waters of the Bering Sea off Kamchatka Peninsula, between 1995 and 2008, Russian fisheries harvested amounts of pollock that ranged from a low of 10,184 mt in 1995 and a high of 66,367 mt in 1998. Harvests of pollock in this area in 2008 totaled 58,901 mt. Pacific cod harvests in the area ranged from a high of 15,000 mt in both 1995 and 1996 and have generally decreased since that time with a low of 5,331 mt in 2006, although harvests climbed again in 2008 to total 9,983 mt. The Russian Atka mackerel fisheries off Kamchatka also increased during this period, with a low of 3,111 mt in 1995 and much higher harvests each year since then, with a high of 20,866 mt in 2004 and, in 2008, taking 16,161 mt.

Northern Part of the Sea of Okhotsk

Off the western shore of the Kamchatka Peninsula, in the northern Sea of Okhotsk, the Russian pollock fishery decreased from 1,577,319 mt in 1995 to 498,406 mt in 2008. Over the same period, the Pacific cod fishery in the area increased from 4,592 mt in 1995 to 9,355 mt in 2008. Relatively few Atka mackerel were harvested from this area from 1995 to 2008, with no harvest at all recorded for some years (1995, 1996, 1998, 2003, 2007, and 2008) and a high of 51 mt taken in 2001.

Sakhalin

Russian fisheries off Sakhalin Island in the southern Sea of Okhotsk took a low harvest of 1,966 mt in 1995 and a high of 37,178 mt in the latest year for which we have information, 2008. Harvests of Pacific cod during this period ranged from a high in 1996 (1,318 mt) to a low in 2008 (299 mt), and the Russian Atka mackerel fishery off Sakhalin took a low of 2 mt in 1998 and a high of 197 mt in 2000, with no harvests recorded since 2004.

Kuril Islands

Finally, Russian fisheries off the Kuril Islands took pollock in amounts ranging from 192,131 mt in 1995 to a high of 452,767 mt in the following year, with a harvest of 206,012 in 2008. Russian fishermen harvested Pacific cod off the Kuril Islands in amounts ranging from a low of 7,964 mt in 2004 and a high of 17,193 mt in 2000, with a harvest of 13,750 in 2008. And Atka mackerel were fished off the Kuril Islands in amounts ranging from 4,334 in 1996 to 32,700 in 2008.

4.5.3 Habitat Response to Fisheries of Concern: Short Term Effects

The objective of fisheries management measures implemented in 2002 (NMFS 2001, 2003) was to mitigate potential adverse effects of fisheries on Steller sea lions and their critical habitat. 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 ($B_{20\%}$), 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 Steller 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, 2003), in which near-shore portions of critical habitat were considered more important to foraging Steller sea lions than offshore areas. However, as previously acknowledged by NMFS, most of the data used in the telemetry analysis was collected from juvenile Steller 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 was considered in the development of management measures in 2002 (NMFS 2003).

Impacts of fisheries on Steller sea lion foraging success depend on spatial, temporal, and targeted species overlap (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 Steller sea lions was 179,000 ± 36,700 mt 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 mt 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.

The amount of prey available for Steller sea lions is rarely known with confidence in the areas (and seasons) where they forage, and measures of harvest or total biomass for a larger area (i.e., total biomass in the BSAI region) may or may not be good indicators of prey availability. For example, a large catch in a small area may indicate that the prey available was substantially reduced (creating poor conditions for Steller sea lions), or it may indicate that large amounts of prey were available (good conditions). If total biomass estimates for a large region (i.e., the entire stock or some large subset of the entire stock) are used as an index of availability, then spatial and temporal patterns of distribution must be predictable or assumed constant over space. But observations of fishing distribution (Fritz 1993) and survey results indicate that the patterns of the fishery and the distribution of fish may vary considerably and, therefore, total biomass estimates may or may not be related to localized biomass estimates (i.e., Aleutian Islands pollock, see Section 4.1.4.3).

4.5.3.1 Overlap: Size of Prey

Fisheries may compete with Steller sea lions if they remove the same size of prey from the same areas. Fisheries may also reduce the spawning biomass of prey to the extent that the reproductive capacity of the fish stock is reduced and, over time, fewer fish become available for Steller sea lions or other predators.). Evaluation of the overlap is confounded by a number of factors. First, the sizes consumed by Steller sea lions are determined by the available prey and any preferential selection of prey by size. In the majority of cases, scientists do not have sufficient information to characterize the available prey and therefore can measure only what was consumed, not necessarily what was preferred (Tollit *et al.* 2004b, Zeppelin *et al.* 2004). Second, much of the information presented in the scientific literature on sizes of prey taken by Steller sea lions or fisheries has been based on the relationship between otoliths (or other hard part) size and the total length of prey. Inferences on the relative importance of prey to Steller sea lions using the occurrence in scat data is misleading, as dietary value is determined by biomass consumed and the energy content of that fish (at the time it was taken). That is, Steller sea lions may gain a great deal more nutrition (energy) from consumption of a single large prey item (in a particular season) than from the consumption of multiple small prey items and, therefore, number or occurrence is not necessarily the best indicator of dietary value and may underestimate the importance of larger, or more energy rich prey.

4.5.3.2 Overlap: Depth of Prey Species

Depth overlap between foraging Steller sea lions and fisheries may occur for any species taken by fisheries on the shelf or shelf break. Competition may be less likely for species found deeper in the water column. The extent to which competition between fisheries and Steller sea lions may be avoided through partitioning of resources by depth can be difficult to judge using the available information. Scientific studies of Steller sea lion foraging patterns are just beginning to characterize the diving depths and patterns of Steller sea lions, and they are likely capable of foraging patterns not yet understood or anticipated. Describing the overlap in depth between fisheries and Steller sea lions is further complicated by diel or seasonal vertical migrations of the fish resources for reproduction, refuge, or foraging.

4.5.3.3 Overlap: Spatial (Evidence for Localized Depletion of Important Prey)

Spatial overlap between fisheries and foraging Steller sea lions could lead to localized depletions of Steller sea lion prey. 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 troughs south of Kodiak Island.

Atka Mackerel

The potential for fisheries to reduce local abundances of fish was shown for Atka mackerel (Lowe and Fritz 1997), so a tagging study was initiated by NMFS in 1999 to estimate local abundance and fish movement. The results show large concentrations of fish at Seguam Pass (McDermott *et al.* 2005), moderate concentrations at Kiska Island and Tanaga Pass, and smaller concentrations at Amchitka Island (NMFS 2000b). 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. Tagging-based estimates of movement rates show low movement from inside to outside the trawl exclusion zone at Seguam Pass, Tanaga Pass and Kiska Island. In contrast, the movement rate from inside to outside the trawl exclusion zone at Amchitka Island was high. Local movement rates are relevant to the question of localized depletion because fish that move from inside to outside the trawl exclusion zone become vulnerable to the fishery, such that fishing outside the trawl exclusion zone can “draw down” the biomass of fish in the no-trawl area. The high biomass and low movement rates at Seguam taken together suggest that the trawl exclusion zone there is likely to be effective at maintaining local concentrations of Atka mackerel for foraging Steller sea lions. In contrast, the low biomass and high movement rate of Atka mackerel at Amchitka Island suggest that the trawl exclusion is not effective at protecting Atka mackerel for Steller sea lions. 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.

To further examine the potential for localized depletion of Atka mackerel, exploitation rates (catch/biomass) were calculated for the Atka mackerel fishery during August through October in each of the Atka mackerel tagging areas. Catch data were derived from the Norpac database and represent all Atka mackerel catches by observed commercial catcher processors in each of the specific study areas. The local exploitation rates estimated in this analysis were low for Seguam Pass, Tanaga Pass and Kiska Island (<5%) and little danger of localized depletion of prey is expected. However, higher exploitation rates at Amchitka (50%) make this area susceptible to localized depletion during the time of the fishery in the area outside the trawl exclusion zone (NMFS 2006b; Ortiz and Logerwell 2010).

Pacific Cod

There are two recent studies aimed at evaluating fisheries effects on Pacific cod in the eastern Bering Sea. One was a FIT field study conducted by NMFS (Connors *et al.* 2004, Conners and Munro 2008) and the other was an analysis of fishery and survey data presented by Fritz and Brown (2005). The FIT experiment uses a before-after, treatment-control type design to compare the seasonal rate of change in cod abundance within the Cape Sarichef no-trawl zone (near Unimak Pass) to the rate of change in the adjacent heavily-trawled area. While the cod catch rates and observed seasonal changes were variable over the three years of the study, the results of the comparison between trawled and untrawled areas were consistent. In each of the three years (2003-2005), the nonparametric statistical test indicated no difference between sites in the trawled and untrawled areas (p-values of 0.81 to 0.98). Power calculations indicate that the experiments in 2004 and 2005 would have been able to detect a reduction in the average catch of the trawled zone in the range of 20-30%. Maps of the observed catches and seasonal percentage changes show no consistent spatial pattern. Opportunistic tag release and recovery data collected as part of the FIT experiments suggests a high rate of movement through the study area. This is a compelling explanation for why no localized decreases in cod were observed, despite high levels of fishery catch (NMFS 2006b).

Fritz and Brown (2005) analyzed fishery-derived data on Pacific cod biomass collected in the eastern Bering Sea, from Unimak Pass to the outer edge of Bristol Bay. The focus of the analysis was the area north of Unimak Pass that is considered a cod spawning area. During a 1.5 month fishing period in 2001, cod abundance (as reflected by changes in fishery catch per unit effort) in the focal area north of Unimak Island declined significantly and this apparently was not related to large-scale emigration. Fishery effort outside of the cod spawning area provided no evidence that large numbers emigrated. As Fritz and Brown (2005) note, this pattern may be different depending on the pattern of migration, emigration, and fishing each year.

At first glance the two studies described above seem to contradict each other. However, the studies were conducted at different spatial scales such that they actually complement each other. The FIT study was designed to examine the potential for fishery depletion of Pacific cod at the scale of the trawl exclusion zone, tens of nautical miles. The Fritz and Brown analysis was conducted in the same area of the Bering Sea but at a larger scale, hundreds of nautical miles. The authors of the FIT study suggest that although no depletion was observed at the small-scale of the trawl exclusion zone, movement of cod through the study area could result in a larger-scale effect. During winter there may be a flow from the northwest to the southeast along the continental slope as cod aggregate prior to spawning. If net flow is occurring as cod pass near Cape Sarichef, then an intense fishery there would amount to an interruption of that flow such that the effect would be detected downstream rather than in the immediate neighborhood of the fishery. Other cod movements (possibly in response to tides, prey availability, diurnal cycles, etc.) could result in back-filling of a fishery depletion, analogous to dipping water from a bucket such that the overall depth of water in that bucket is lowered but the depth at the locale of the dipper is no lower than anywhere else in the bucket. The scale of the Fritz and Brown study may represent the larger scale at which one would expect to detect a fishery effect. Based on their results, one can hypothesize that at the scale of hundreds of nautical miles, cod movement is not sufficient to displace or disperse a fishery effect. Before-after, treatment-control type experiments at larger scales need to be conducted to test this hypothesis.

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).

Pollock

FIT conducted a walleye pollock fishery interaction study off Kodiak Island in August 2000, 2001, 2002, 2004 and 2006. The study design consisted of repeated acoustic-trawl surveys of two adjacent areas before and after the start of the commercial walleye pollock fishery. Barnabus Trough was open to fishing while Chiniak Trough was closed to fishing. Unfortunately, virtually no fishery occurred in the study area during 2000, and 2002. About 2850 mt and 1700 mt of adult pollock were harvested from the study area during the C season in 2001 and 2004, respectively. Only about 800 mt were removed in 2006. Study results from 2001 show high temporal variability in adult pollock biomass in the fished area, but not in response to fishing (Wilson *et al.* 2003). In contrast, results from 2004 show a decrease in pollock biomass in the fished area following the start of commercial fishing. No concurrent decrease in adult pollock biomass in the unfished area was observed (NMFS 2006b). Results from 2006 show a decrease in pollock in the fished trough, but a similar decrease was also observed in the unfished trough (Chris Wilson, Alaska Fisheries Science Center, pers. com.). In summary, results from two of the three study years (2001, 2006) do not provide support for a localized depletion of pollock. Further study is needed to understand the response of pollock to greater levels of fishing effort.

NMFS has also examined the potential fishery effects on pollock with analysis of acoustic data collected from commercial fishing vessels participating in the Bering Sea pollock fishery. An area identified by Barbeaux and Dorn (2003) as an intensely fished area in the southeastern Bering Sea in 2001 and 2002 was selected for an opportunistic acoustic visualization study (Barbeaux *et al.* 2005). A total of 51,200 mt of pollock were removed from the 280 km^2 study area between 20 January 2003 and 26 February 2003. Both the CPUE and the acoustic index of abundance (mean log backscatter) show a decline over the duration of the fishery. The snapshots provided by the acoustic data show a noticeable decline in density and aggregation distribution from a few large, highly dense aggregations in January to a larger number of smaller, less dense aggregations by the end of February. Note that the data show a persistent "formation" of pollock during the same year and season, but the larger scale analysis shows that the overall density of pollock still declines with time over the duration of the fishery. All changes in distribution observed from January to February cannot be attributed solely to fishery interactions, seasonal emigration from the study area could also be a factor.

Statistical and correlative analyses of fishery effort/catch with trends in local Steller sea lion populations have yielded equivocal results, some indicating a positive and some a negative relationship between catch and Steller 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.

Another important spatial consideration is the potential for an "edge effect." In the 2001 Biological Opinion, NOAA Fisheries explored the issue of the edge effect in Section 5.3.1.7. NMFS originally brought this issue to light in the 1998 Biological Opinion as a concern about the concentrated fisheries in the EBS near Sea Lion Rocks (Amak Island) and in the foraging area. The question is whether effects of fishing along the edge of a closure zone (e.g., a 10 nm closure zone) would be found on the prey field within that zone. For example, if fish are moving along the coast, entering an area around a haulout that is closed, those fish could in theory be intercepted by the fishery and therefore reduce the availability of prey within a zone in which they never fished; this concept can be compared to a downstream effect. The results of the Atka mackerel and Pacific cod FIT studies described above apply to the question of whether fishing along the edge of a closure zone could impact prey fields inside the zone. Tagging-based estimates of Atka mackerel movement rates showed low movement from inside to outside the trawl exclusion zone at Seguam Pass, Tanaga Pass and Kiska Island. In contrast, the movement rate from inside to outside the trawl exclusion zone at Amchitka Island was high (McDermott *et al.* 2005, NMFS 2006b, McDermott AFSC pers. com). Local movement rates are relevant to the question of localized depletion due to an "edge effect" because fish that move from inside to outside the trawl exclusion zone become vulnerable to the fishery, such that fishing outside the trawl exclusion zone can "draw down" the biomass of fish in the no-trawl area. The low movement rates at Seguam suggest that an edge effect is not likely to be a problem there and that the trawl exclusion zone is likely to be effective at maintaining local concentrations of Atka mackerel for foraging Steller sea lions. In contrast, the high movement rate of Atka mackerel at Amchitka Island suggests commercial fishing on the edge of the zone could impact fish inside such that the trawl exclusion is not as effective at protecting Atka mackerel for Steller sea lions. As an adjunct to the Pacific cod local depletion study, FIT scientists also conducted preliminary tagging studies of Pacific cod in 2002 and 2003. The main goal of these studies was to test feasibility of tagging for this species and work out tagging and handling procedures. Over 2,000 tags were released in April 2002 and approximately 4,000 tags were released in winter 2003 (NMFS 2006b). While tagging data from these releases provides primarily qualitative information about fish movement, the results are useful for examining the potential for an edge effect. The results show that at least a portion of the Pacific cod in the vicinity of Unimak Pass move great distances (up to 80 nautical miles) over time scales as short as a few weeks. These results are not compatible with the idea of a spatially static pool of fish that remains in one location over the season. They suggest that an appropriate spatial scale for movement of Pacific cod in this region is larger than the 10-20 nautical miles of the existing trawl closure zones. If cod move across trawl exclusion zone boundaries, as these tagging data suggest they may, then commercial fishing on the edges of the closure areas could impact the density of Pacific cod inside the closure.

4.5.3.4 Overlap: Temporal

Changes in behavior, foraging patterns, distribution, and metabolic or physiological requirements during the Steller sea lion annual cycle (Figure 3.24) are all pertinent to consideration of the potential impact of prey removal by commercial fisheries. Steller sea lions, at least adult females and juveniles, are unlike other marine mammals that store large amounts of energy (fat) to allow extended periods of fasting. Steller sea lions need more or less continuous access to food resources throughout the year. Nevertheless, the sensitivity of Steller sea lions to competition from fisheries may be higher during certain times of the year. Reproduction likely places a considerable physiological or metabolic burden on adult females throughout their annual cycle. Following birth of a pup, the female must acquire sufficient nutrients and energy to support both herself and her pup. The added demand may persist until the next reproductive season, or longer, and is exaggerated by the rigors and requirements of winter conditions. The metabolic requirements of a female that has given birth and then become pregnant again are increased further to the extent that lactation and pregnancy overlap and the female must support her young-of-the-year, the developing fetus, and herself. And again, she must do so through the winter season when metabolic requirements are likely to be increased by harsh environmental conditions.

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 3 days in winter and 2 days in summer (Trites and Porter 2002, Milette and Trites 2003). 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 2 days, suggesting that Steller sea lions do not accompany their mothers on foraging trips. Foraging trips by mothers of yearlings were longer on average than those by mothers of pups.

Weaned pups may be independent of their mothers, but may not have developed adequate foraging skills. They must learn those skills, and their ability to do so determines, at least in part, whether they will survive to reproductive maturity. This transition to nutritional independence is likely confounded by a number of seasonal factors. Seasonal changes may severely confound foraging conditions and requirements; winter months bring harsher environmental conditions (lower temperatures, rougher sea surface states) and may be accompanied by changing prey concentrations and distributions (Merrick and Loughlin, 1997). Weaned pups' lack of experience may result in greater energetic costs associated with searching for prey. Their smaller size and undeveloped foraging skills may limit the prey available to them, while at the same time, their small size results in relatively greater metabolic and growth requirements.

Other times of the year are also important for Steller sea lions. For example, the observed increases in consumption by captive animals in the fall months indicate that preparation for winter is important. Spring is also important as pregnant females will be attempting to maximize their physical condition to increase the likelihood of a large, healthy pup (which may be an important determinant of the subsequent growth and survival of that pup). Similarly, those females that have been nursing a pup for the previous year and are about to give birth may wean the first pup completely, leaving that pup to survive solely on the basis of its own foraging skills. Thus, food availability is surely important year-round, although it may be particularly important for juvenile animals and pregnant-lactating females during the winter.

Diet studies of captive Steller sea lions indicated that they adjust their intake levels seasonally, with increases in fall and early winter months (Kastelein *et al.* 1990, Rosen and Kumagai 2008), and with season being a better predictor of body condition than the quality of prey consumed (Calkins *et al.* 2005). Further, prey diversity may be lower in the winter (Sinclair and Zeppelin 2002, Trites *et al.* 2006d). These adjustments varied with age and sex of the studied animals, and the extent to which the patterns observed are reflective of foraging patterns in Steller sea lions in the BSAI or GOA regions is not known. Nonetheless, such studies support the idea that the fall and winter period is a time of high metabolic demand (Rehberg and Burns 2008, du Dot *et al.* 2008), and supports the importance of prey availability year-round.

4.5.3.5 Overlap: Compressed Fisheries

Numerous gear types have been used for fishing including jig, pot, hook-&-line, bottom trawl, and pelagic trawl gear. Also numerous vessel classes are used including everything from small skiffs, catcher boats, freezer longliners (hook-&-line), and large catcher processors. Descriptions of these fisheries are outlined in detail in the Groundfish SEIS (NMFS 2004).

Observer data were used (section 5.3.1.6 and Figures 5.1 and 5.2 in NMFS, 2001) to describe concentration in time and space for BSAI trawl, pot, and hook-&-line fisheries (Figure 4.31). In this analysis, the timing of catch was linked to the spatial and temporal concentration of fishing effort. Looking at the percentage of catch that was caught in areas with high catch rates, trawl fisheries were noted in the BSAI to have the highest proportion of their catch in cells with high catch rates. Pot gear had less of a proportion in those high catch rate cells, whereas hook-&-line gear had the highest proportions of catch in the lowest catch rate bins (Figure 4.31). These data suggest that the hook-&-line fishery is more dispersed than the trawl fishery, and may be less likely to cause localized depletions of prey. However, the critical link between fisheries removals (time, rate, location, etc.) and the effects on Steller sea lions is poorly understood and we cannot determine the relationship between these catch rates and the impacts on prey except that higher catch rates would be more likely to result in localized depletions (or prey field effects) as described by NMFS (2006b). Some published papers (Lokkeborg *et al.* 1989, Lokkeborg 1998, and Lokkeborg and Ferno 1999) have looked at the effects of gear such as hook-&-line on the distribution and abundance of fish species. Hook-&-line fisheries appear to be more dispersed in both time and space - one of the fishery components which would reduce the likelihood of resulting in adverse modification of critical habitat (NMFS 2001, 2003). The likelihood of jig gear resulting in localized depletions is extremely low, yet there are few scientific data to support this (i.e., the link between removals of fish and adverse impacts to Steller sea lions) except for extremely low catch rates.

4.5.3.6 Response of Steller Sea Lions to Habitat

Sinclair and Zeppelin (2002) is the principal publication describing Steller sea lion diet in Alaska based on scat analysis during the 1990s. In Sinclair *et al.* (2005) three more years of diet data were used to enhance the understanding of Steller sea lion diet, especially for the western DPS. They used cluster analysis that grouped Steller sea lion rookeries based on population trend and diet. Rookery clusters in the GOA based on diet included pollock, salmon, and arrowtooth flounder; clusters in the western GOA and eastern Aleutian Islands included pollock, salmon, Atka mackerel, sand lance, and herring; while those farther west clustered around a diet composed of Atka mackerel and cephalopods.

McKenzie and Wynne (2008), found that arrowtooth flounder is more prevalent in the diet of Steller sea lions than previously reported, at least in some areas. This study in the Kodiak Archipelago found that the most important species in terms of frequency of occurrence and numerical abundance were Pacific sand lance, walleye pollock, arrowtooth flounder, Pacific cod, salmon, and Pacific herring. Significant differences in diet composition were found among regions, seasons, and years, suggesting that the diet of Steller sea lions is strongly influenced by local and temporal distributions and abundances of prey. Herring dominated scat collections from the west coast of the archipelago and accounted for most (14 to 30%) of the differences in regional diet in spring and winter. Annual variation in diet was relatively low in winter, whereas spring and fall diets varied from year to year, with regional-specific shifts in dominant prey. Results from the study generally agree with diet studies conducted in the 1990s, but differ markedly in the relative importance of the major prey species, particularly the high ranking of arrowtooth flounder (frequency of occurrence at 35%) in the diet. In scats containing arrowtooth flounder, sand lance and pollock co-occurred in similar proportions (47.8 and 47.2%, respectively).

In field studies addressing Steller sea lion foraging efficiency and behavior, Sigler *et al.* (2009) found that Steller sea lions in Southeast Alaska selected intermediate-sized fish and avoided small (<10 cm) and large (>60 cm) fish, and that they moved between areas as prey became seasonally available. The number of Steller sea lions present was directly related to the amount of prey available; a standing biomass of 500 to 1700 mt of prey in a non-breeding area such as Frederick Sound, depending on species composition, can attract and sustain about 500 Steller sea lions. Their study results suggested that a diet with substantial year-round contributions from less nutritious, but abundant prey such as pollock can form part of a healthy diet as long as more nutritious prey such as herring, salmon or eulachon also are consumed. It also seems that the ability of Steller sea lions to predict locations of forage "hot spots" may be more important than the actual density of fish (Gende and Sigler, 2006). They showed that even low densities of prey were used if they were the same places every year thereby reducing Steller sea lion forage time and effort. Higher density food patches that were highly variable resulted in more Steller sea lion search effort to find and exploit them. Steller sea lions are not unusual in that they return to forage areas where they were successful during past foraging trips.

Call *et al.* (2007) reported on the relationship between foraging behavior and prey availability. Their study showed that time at-sea was significantly longer for juvenile Steller sea lions in the central and eastern Aleutian Islands and central Gulf of Alaska than for juveniles from Prince William Sound and Southeast Alaska. They concluded that if the energetic requirements of Steller sea lions are equivalent between the two areas, then this finding would suggest differences in prey abundance, distribution, or composition. Interestingly, Gelatt *et al.* (2007) report increasing occurrence of genetic haplotypes from the western DPS in Southeast Alaska (eastern DPS) and speculated that it may be the result of habitat degradation in the west. Call *et al.* (2007) further speculated that the fjord/protected coastal waters of Southeast Alaska may facilitate prey constriction and serve to limit distances Steller sea lions must travel to forage, whereas the Aleutians and Gulf of Alaska areas are more of an open-ocean environment with less prey constraint by physical features so that juveniles in those areas may have to travel greater distances or increase search times for prey.

In our earlier Biological Opinions we used data derived from satellite telemetry to describe foraging activity of Steller sea lions, especially juveniles, based on dive location, dive depth, and dive duration. For this opinion additional analyses were conducted using the same data set but the locations associated with diving to >4 m were considered by individual (AFSC, 2010b). For each individual, cumulative frequency distributions were calculated in 1 nm bins to show the proportion of locations by distance to the nearest listed Steller sea lion rookery or haulout site. Dives were also grouped based on the ten RCAs discussed above. These ten areas are analogous to our larger geographic regions (e.g., central Aleutian Islands) but numbered and with some minor changes in boundaries. This new perspective provided and indication to the extent 10 nm and 20 nm distance bins encompass potential foraging locations of juvenile Steller sea lions from the western DPS. Results were consistent with earlier findings that at least 75% of dives by juveniles were within 10 and 20 nm of a listed site. For Steller sea lions 3-10 months of age, all diving locations in summer for about two-thirds of the individuals (25 of 39 Steller sea lions) were within 10 nm and four-fifths were within 20 nm. In winter about half of the 3-10 month old individuals dove entirely within 10 nm and three-fourths were entirely within 20 nm. Steller sea lions older than 10 months of age tended to have more locations distant from a listed site, consistent with other descriptions of ontogenetic-related behavioral patterns (Loughlin *et al*. 2003, Raum-Suryan *et al.* 2004, Fadely *et al.* 2005, Pitcher *et al.* 2005, Rehberg and Burns 2008). Details on dive locations by region will be provided below in section 5.4.2. As described by AFSC (2010b) however, none of the juvenile Steller sea lions were captured in RCAs 1-2, but rather entered those areas from elsewhere. Because of the small numbers of Steller sea lions tracked in RCAs 1-3, grouping the patterns of use among individuals within RCAs 1-4 provide another perspective on critical habitat use by juvenile Steller sea lions in the western Aleutian Islands. Because there is considerable evidence of environmental and habitat variability and segregation along the Aleutian Island chain (Rodionov *et al.* 2005; also see Section 4.4.5.3) that may influence Steller sea lion abundance and behavior (Sinclair and Zeppelin 2002, Call and Loughlin 2005, Fadely *et al.* 2005, Trites *et al.* 2007, Lander *et al.* 2009), limiting the pattern combination to only RCAs 1-4 may better reflect behaviors within any of RCAs 1-4 individually than if distributions found among all RCAs were combined.

Fadely *et al.* (2005) used telemetry to study juvenile Steller sea lions in the eastern Aleutian Islands and found that most locations from the radio-equipped juveniles (5-21months of age) were within 10 nm of shore and associated with on-shelf waters less than 100 m deep. Use of off-shelf waters increased in May, as did trip durations. There were age-related and seasonal changes in juvenile trip length and diving behavior that coincided with rapid changes in oceanography and prey availability. In a different study, instrumented pups were seen to make shorter and shallower dives than juveniles and adults, but as in earlier studies, juveniles and adults dove to similar depths (Rehberg and Burns 2008). Also, subadult and adult dive activity corresponded to diurnal prey migration but pup dive behavior did not. And last, the study showed that pup dive activity reflected the pup's physiological and behavioral development where juvenile dive behavior reflected seasonal prey availability differences (Rehberg and Burns 2008).

Gregr and Trites (2008) used Steller sea lion telemetry data, as well as oceanographic variables, to model the habitat value of marine waters offshore rookery and haulout sites in the Gulf of Alaska and the Bering Sea/Aleutian Islands. They used platform of opportunity distributions of Steller sea lions to confirm their model predictions. Gregr and Trites (2008) concluded that zones within 20 nm of rookeries and haulouts were by far the most important habitat areas, particularly for juvenile and adult females, which showed very low probabilities of foraging outside 20 nm.

4.5.4 Habitat Response to Fisheries of Concern: Long Term Effects

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 while avoiding overfishing the target stock. Fishing at $F_{40\%}$ will, theoretically, reduce the average spawning stock size to 40% and total biomass to approximately 50% of pristine levels (using single-species assessments), although ecosystem modeling indicates that the single-species predictions may underestimate the status of commercially fished population relative to the unfished condition.

In NMFS (2000), all of these issues were explored and a control rule was implemented that prohibits directed fishing for pollock, Pacific cod, and Atka mackerel when biomass falls below $B_{20\%}$. Key questions remain regarding the utility of single species management, the impacts of adding an additional (and highly efficient predator) to the ecosystem, and key assumptions regarding natural mortality and fecundity (NRC 2006, Trites *et al.* 2006e).

4.5.4.1 Relationship Between Environmental Changes and Fishery Effects

Bailey (2000) suggested potential connections between climate shifts, food web effects, and pollock recruitment in the GOA, but did not address potential fishing effects. Understanding Gulf of Alaska pollock production has been difficult for stock assessment teams since pollock became an important resource to domestic fisheries during the 1980s. Despite regulation of fishing mortality since that time, production (mainly pollock recruitment) has been unpredictable, and therefore has received considerable research attention in the attempt to correct this lack of predictive capability. Scientists searching for answers have often chosen either fishing or the "natural" explanations, but integration is rare. One potential physically mediated mechanism for changes in GOA pollock population dynamics has been suggested by Bailey (2000). Notably, the biomass trajectory of GOA pollock does not correspond directly to the regime shift of 1976-77; rather biomass peaks in 1981-82 and then declines to present levels roughly equivalent with the early 1970's. This contrasts with the trajectories of several other exploited populations, notably flatfish such as arrowtooth flounder and Pacific halibut, which show a steady increase from the late 1970's through the present, suggesting a connection, though not a mechanism, for climate regime control (Clark *et al.* 1999). Bailey (2000) suggests that pollock population dynamics are nevertheless related to the 1976-77 regime shift in an indirect way. He proposes that in the 1970s, pollock recruitment was largely driven by larval mortality and that the regime shift increased plankton production in Shelikof Strait and throughout the GOA (there is no reference given for this critical phenomenon), leading to better larval survival and constantly improving pollock recruitment increasing population biomass. Then, after a 5-10 year lag the "ecosystem matured and the abundance of large predators built up," thus increasing predation on the juvenile phase in the life history and countering the effects of (presumably still) increased larval survival, reducing recruitment and ultimately pollock population biomass.

One way to assess the relative effects of fishing and environmental effects including predation is with static and dynamic food web models including as many relevant ecosystem groups as possible. One effort used simulated Aleutian Islands food web in attempt to put a relative weight/importance to the different hypotheses for causes of Steller sea lion decline (Guénette *et al*. 2006). However, this modeling exercise was performed with "in-built" assumptions; for example, killer whales were assumed prior to fitting the model to data to be able to switch prey sufficiently to become a substantial source of mortality, and bottom-up production in the Aleutians was assumed to be correlated with the PDO, despite a lack of specific process evidence. Therefore, while the Guénette *et al*. 2006 study can be described as an "exploration of the parameter space" for determining the type of ecosystem interactions which might govern Steller sea lions, it was statistically and methodologically insufficient to either confirm or strengthen particular hypotheses governing the Steller sea lion trends in the Aleutian Islands.

In particular, recent research on the Aleutians Islands (Logerwell *et al.* 2005) has emphasized the importance of spatial and longitudinal structure in testing hypotheses on factors governing fish production and therefore prey effects on Steller sea lions in the regions. Localized and broad-scale Aleutians Islands food webs (Ortiz 2007, Aydin *et al.* 2007) clearly show the Aleutians fish community to be highly dynamic and strongly spatially structured with production breaks between widely-spaced Aleutian passes, and thus less amenable to predictive modeling approaches used in Guenette *et al.* (2006) or described for the Gulf of Alaska (see below). Aydin *et al.* (2007) suggest that in the Aleutian Islands, unlike the EBS and GOA, Atka mackerel, walleye pollock, and Pacific cod production rates are tightly interlinked, consuming each other's juveniles and responding to climate variation with strong spatial structure, making current broad-scaled predictive or hypothesis-testing models and tools extremely uncertain. Two approaches are being examined to lower this predictive uncertainty. The first is a focus on local-scale modeling calibrated by local-scale field-work. The second is a detailed, age-structured construction of a three-species stock assessment model for the Aleutian Islands with food web interactions between Atka mackerel, walleye pollock, and Pacific cod (Kinzey and Punt, submitted) which will allow a more thorough exploration of hypotheses governing the dynamics of Steller sea lion prey in the Aleutian Islands.

Gaichas (2006) used an extended set of statistical fitting methods (Aydin *et al.* 2006) on a more complex dynamic ecosystem model (Aydin *et al.* 2007) to evaluate different hypotheses regarding the relative effects of fishing history, climate change, and predator prey interactions in determining biomass trajectories for important species in the Gulf of Alaska. The GOA dynamic ecosystem model is based on a food web model. The GOA food web model includes area- and time-specific production and consumption parameters based on research surveys and single species stock assessments that characterized the state of the system in the early 1990s. It also includes explicit juvenile groups for major groundfish and pinnipeds, and substantial taxonomic detail in benthos, pelagics, birds, and marine mammals. The GOA model includes 129 living groups (4 producer and 125 consumer), 5 detritus groups, and 15 fisheries. Fishery catches were reconstructed from NMFS Observer catch composition sampling data for groundfish fisheries, ADF&G catch statistics for salmon, herring, and crab fisheries, and International Pacific Halibut

Commission (IPHC) research surveys and literature values for the halibut fishery. Details of model construction and parameterization for the GOA are documented in Gaichas (2006) and Aydin *et al.* (2007)

In the dynamic ecosystem model, twelve historical time series represented the dynamics of species groups ranging from Steller sea lions through commercial groundfish to pandalid shrimp. This background information was used to develop an experimental design which compares biomass and catch trajectories predicted from the ecosystem model with the twelve species time series from the Gulf of Alaska for six hypotheses of ecosystem control (Gaichas 2006).

In general, none of the results support the idea that the historical effects of fishing are reversible (Gaichas 2006; NMFS 2006b). Removing fishing mortality from populations initialized in the early 1990s and running the model without fishing for over 100 years did not produce populations "recovered" to pre-fishing biomass. It was necessary to both estimate specific predator-prey relationships and provide some form of increased historical production, in addition to removing fishing mortality, for populations to reach historical levels.

In all modeled hypotheses, fits for several groups were consistently poor. The high historical biomass of Steller sea lions is not produced by any hypothesis. However, some potentially important relationships between juvenile Steller sea lions and pollock were suggested by one of the better-fitting models which incorporated both fishing and our best information on environmental effects on herring, POP, and pollock production (recruitment). The consistent lack of fit to the historical portion of the Steller sea lion time series in all models has been observed in other studies, but results from the present study still provide insight into ecosystem relationships and potential fishery interactions for this protected species. The Steller sea lion time series is likely to represent Steller sea lion biomass dynamics well, because the time series themselves are estimated from field sampling at the appropriate Gulfwide scale.

It is important to note that the biomass time series for juvenile Steller sea lions mirrors that for pollock, and is best explained in the model forced with pollock recruitment for the later portion of the time series from 1980-2002. The changes in juvenile biomass do not translate into changes in the adult population, however, most likely because the biomass of adults is so much larger that it absorbs these short term fluctuations. The early part of the series where biomass of juvenile and adult Steller sea lions was high is not explained by any model, suggesting that no mechanisms producing a high historical biomass of Steller sea lions were implemented under any tested forcing hypothesis. This is consistent with the findings of another modeling study specifically designed to address the Steller sea lion decline (NRC 2003). In that study, the decline of Steller sea lions in Alaska could only be explained by including increased adult mortality, apparently from undocumented culling of these predators by participants in the fishery.

The only other hypotheses supporting a realistic Steller sea lion decline involved increased mortality of small pelagic fish due to disease outbreaks, or decreased vulnerability of Steller sea lion prey as a result of the 1977 regime shift, for which no mechanism was identified (NRC 2003). The ecosim model used in the NRC hypothesis testing suffered from an unfortunate lack of precision in the inclusion of time series from both the Bering Sea and the Gulf of Alaska; for example, pollock biomass and recruitment time series from the Bering Sea were mixed with small pelagic and invertebrate time series derived from Anderson and Piatt (1999) which apply only to the GOA (and then only to the nearshore GOA as shown above). Given that Bering sea pollock biomass is an order of magnitude higher than GOA pollock biomass, and that Bering sea pollock biomass trends have been stable to increasing over the same period that GOA pollock sharply declined (Ianelli *et al.* 2005, Dorn *et al.* 2005), it seems possible that any relationship between pollock and Steller sea lions in the GOA might have been overlooked in the NRC analysis. The NRC report (2003) rightly concludes that the historical data which might support one of these population decline hypotheses over another was never collected, so the "true" explanation may never be revealed. While the present study was not designed to address Steller sea lion declines specifically, the inclusion of appropriate time series and parameters for the GOA does reveal a potential linkage between pollock recruitment and juvenile Steller sea lions. The strength of this linkage may be important for fishery and protected species management, and because it is not dependent on unobserved historical events, research efforts may evaluate it further.

The results of Gaichas (2006) demonstrate that both environmental variation and historical fishing effects (and predator prey interactions) are necessary to explain historical Gulf of Alaska ecosystem dynamics, and even then some dynamics remain unexplained. Ecosystem modeling suggests that no single hypothesis explains all biomass time series, suggesting that in the GOA, there is no single main driver of the ecosystem. Both “top-down” control by fishing and “bottom-up” environmental effects either for individual species or for the entire system are necessary to explain ecosystem dynamics. Furthermore, different groups are best explained by different control hypotheses, which in turn imply very different predator-prey relationships within the ecosystem. Gaichas (2006) concluded that fishing, environmental change, and keystone species drive regime dynamics, and must be considered together. Further, while fishing clearly has effects in the GOA, fishing effects do not damp out all other ecosystem processes. It is difficult at this point to determine whether a fished ecosystem without clear fishing dominance to be ecologically healthier relative to those with a single dominant fishing driver. Clearly, fishery management is necessary in a system with multiple drivers—and may be crucial to preventing fishing from becoming the dominant driver (NMFS 2006b).

4.5.4.2 Changes in Prey Size and Age Distribution

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.* 2006e, NMFS 2006b; see Figures 4.32 and 4.33). 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 Steller 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 Steller sea lions to obtain the equivalent amount of energy in a larger number of small fish (Calkins and Goodwin 1988, NMFS 2000, NMFS 2006b).

Recent efforts to summarize quantitative ecosystem indicators for fisheries management have identified size-based indicators, such as the community size spectrum (CSS), as an important class of indicators for tracking fishery exploitation effects on fish communities (Cury and Christensen 2005, Kruse *et al.* 2006, Hall *et al.* 2006, Greenstreet and Hall 1996, Rice & Gislason 1996, Duplisea *et al.* 1997, Greenstreet *et al.* 1999, Bianchi *et al.* 2000, Zwanenburg 2000). The CSS examines the relationship between the abundance and size of animals in a community, and has been found to explain some fishing induced changes at a system-wide level. Fishing may change the abundance of organisms of different size classes, particularly the amount of larger animals, affecting the slope of the descending limb of the size spectrum. For example, in an exploited fish assemblage, larger fish generally suffer higher fishing mortality than smaller individuals and this may be one factor causing the size distribution to become skewed toward the smaller end of the spectrum (Zwanenburg 2000), and leading to a decrease in the slope of the size relationship over time with increasing fishing pressure.

Studies in some areas, such as the Scotian shelf and the North Sea, have shown significant linear decreasing trends in CSS slopes, implying that a decreasing abundance of large fish in those systems (Bianchi *et al.* 2000). The eastern Bering Sea groundfish community, as sampled in the annual bottom trawl survey, however, does not show a significant linear trend in the CSS slopes during 1982-2006 (Boldt *et al.* 2008, Bartkiw *et al.* 2007). During the 1980s the slopes became less negative, implying an increase in the abundance of large individuals and/or a decrease in the abundance of small individuals (Boldt *et al.* 2008, Bartkiw *et al.* 2007). This trend is the opposite of what would be expected if fishing were removing the large fish. CSS slopes were variable in the 1990s and then the trend reversed during 2002-2006. Factors other than fishing, such as climate regime shifts that affect fish distribution and production, may also influence the community size spectrum.

4.5.4.3 Changes in Prey Spatial Distribution

Much of the preceding discussion on the potential for competition between the Steller sea lion and BSAI and GOA groundfish fisheries has focused on exploitative competition; that is, it examines competition that occurs when fisheries remove prey and thereby reduce prey availability to Steller sea lions. In addition to exploitative competition, fisheries may affect Steller sea lions through interactive competition. Examples of interactive competition include disruption of normal Steller sea lion foraging patterns by the presence and movements of vessels and gear in the water, abandonment of prime foraging areas by Steller sea lions because of fishing activities, and disruption of prey schools in a manner that reduces the effectiveness of Steller sea lion foraging.

The hypothesis that these types of interactive competition occur cannot be evaluated with the information currently available. The only data are from the POP database, and are not sufficient to describe the response of Steller sea lions to fishing or other vessels. For example, few observations of Steller sea lions from fishing vessels could mean that a) Steller sea lions are present and tolerant of fishing but rarely sighted, or b) that Steller sea lions are disturbed by fishing vessels and therefore abandon areas that are being fished. Incidental catch of Steller sea lions in the 1970s and 1980s indicates that at least some Steller sea lions were relatively tolerant of vessels and fishing activities. On the other hand, such interactions are relatively rare today, and it is possible there has been some selection for Steller sea lions that avoid vessels and fishing activities.

The effects of fishing on groundfish schools are not understood. Vessels fishing for Atka mackerel trawl the same locations repeatedly, as they are unable to search for schools (Atka mackerel don’t have a swim bladder and therefore are not evident on fish-finders). The number of schools affected and the effects on schooling dynamics are not known, but these factors will be important in understanding the overall impact of trawling for Atka mackerel on Steller sea lions.

Vessels trawling for other target species can use fish finders which allow them to search for and locate fish schools or aggregations of suitable densities. Trawls are repeatedly towed through fish aggregations until the size or density of the catch becomes inefficient for further trawling. When catch efficiency decreases, the search resumes for another aggregation of suitable density.

The strategies used by fishing vessels likely alter schooling dynamics and important features of target schools such as their number, density, size, and persistence (recent field studies on the effects of fishing on fish school structure are described below). If Steller sea lion foraging strategies are adapted to take advantage of prey aggregations or schools, then trawling may result not only in exploitative competition through removal of prey, but also in interactive competition through disruption of schools or aggregations and their normal dynamics. For example, the removal of a portion of a fish school by a trawl net must create at least a temporary localized depletion (i.e., a gap in the prey school). How long that gap persists and the responses of the remainder of the schooling prey to trawling are unknown. The school may aggregate again, either quickly or over time, or it may disperse. The short-term effects may be prolonged when trawling is repeated. Hypothetically, it is possible that Steller sea lions in the immediate vicinity of the trawled school are able to take advantage of the disruption to isolate and capture prey. On the other hand, Steller sea lions have probably adapted their foraging patterns to normal schooling behavior of their prey; trawling may disadvantage Steller sea lions not only by removing their potential prey within their foraging areas (exploitative competition), but also disrupting the normal schooling behavior of the prey species.

The effects of fishing on the spatial distribution of pollock schools were investigated during a multiyear field experiment conducted near Kodiak Island in the Gulf of Alaska. The study was one of a suite of FIT studies (see Section 4.5.3.3). The study design consisted of repeated acoustic-trawl surveys of two areas near Kodiak, Alaska, conducted before and after the start of commercial fishing in August. The areas chosen were two adjacent troughs: Chiniak Trough was closed to fishing and Barnabus Trough was open to fishing. The study was successfully conducted during 2001, 2004 and 2006. Virtually no commercial fishery occurred in the study area during 2000, and 2002. The large-scale geographical (horizontal) distribution of pollock before and after the start of commercial fishing was examined for the three successful years using several spatial indices, including the center of gravity (average location of fish). No significant differences were observed in the location of fish between the fishing and pre-fishing periods. Changes in the small-scale vertical distribution and school structure of pollock were also examined. No differences were detected in mean depth of pollock when pre-fishing data were compared with post-fishing data (Wilson *et al.* 2003; Wilson pers. comm.). No differences in the school shape descriptors could be attributed to the fishery in 2001 (school descriptors were school length, height, fractal dimension and density).

NMFS has also examined the potential fishery effects on pollock with analysis of acoustic data collected from commercial fishing vessels participating in the Bering Sea pollock fishery. An area identified by Barbeaux and Dorn (2003) as an intensely fished area in the southeastern Bering Sea in 2001 and 2002 was selected for an opportunistic acoustic visualization study (Barbeaux *et al.* 2005). A total of 51,200 mt of pollock were removed from the 280 km^2 study area between 20 January 2003 and 26 February 2003. The snapshots provided by the acoustic data show a noticeable decline in density and aggregation distribution from a few large, highly dense aggregations in January to a larger number of smaller, less dense aggregations by the end of February.

It is also important to note the potential cumulative effects of the federal and state fisheries on Steller sea lions. As discussed previously, pollock, Pacific cod, and Atka mackerel are very important in the diets of Steller sea lions, although they prey on a variety of other species. Since the 1970s, commercial fisheries for pollock have been focused within the foraging areas of Steller sea lions, and have sufficient fishing power to locally deplete pollock schools or disaggregate the schools (NMFS 2006b).

A predator faced with competitive pressure would normally shift its diet (if possible). Steller sea lions, however, would then have to compete with fisheries for Pacific cod, yellowfin sole, flatfish, Pacific salmon, herring, rockfish, and other species which are commercially harvested (both directly and as incidental catch). With each of these potential prey, Steller sea lions would find competitive pressure caused by a reduction of the biomass of a species, a change in its size structure, and a local reduction caused by fishing vessels in critical habitat for the Steller sea lions. Certainly, not all Steller sea lion prey species are commercially harvested.

4.5.4.4 Changes in Prey Biomass

Current groundfish harvest control rules were designed to allow for adequate prey for marine mammal piscivores in general and for Steller sea lions in particular. These subjects have been addressed at great length in the 2000 Biological Opinion (NMFS 2000), the Steller Sea Lion Protection Measures Final Supplemental Environmental Impact Statement (NMFS 2001), and the Alaska Groundfish Fisheries Final Programmatic Supplemental Environmental Impact Statement (NMFS 2004). A brief summary of some major points follows.

The FMPs each contain a pair of harvest control rules, one of which determines the fishing mortality rate corresponding to the "overfishing level" (FOFL) and the other of which determines the upper bound on the fishing mortality rate used to compute acceptable biological catch (FABC). The functional form and parameters of the harvest control rules vary according to the type of information available. In all cases, the FOFL control rule is designed (within the limits of the available data) to keep fishing mortality at or below the level corresponding to MSY, and the FABC control rule is designed to maintain a substantial buffer between ABC and OFL. In Tiers 1-3, each of the harvest control rules is defined as a function of spawning biomass. These control rules include an inflection and an intercept along the spawning biomass axis. In Tiers 1-2, the inflection occurs at B_{MSY} and the default location of the intercept occurs at 5% of B_{MSY} (the term "default location" is used here because the FMPs stipulate that the SSC can specify a different location on the basis of the best available scientific information). In Tier 3, the inflection occurs at $B_{40\%}$ (not at the B_{MSY} proxy of B35%) and the default location of this intercept is 5% of $B_{40\%}$.

In the event that a stock's spawning biomass falls below the inflection point of the control rule (Tiers 1-3), the fishing mortality rate falls linearly with spawning biomass. Because catch is roughly proportional to the product of the fishing mortality rate and biomass, this implies that *catch falls almost quadratically* with spawning biomass, meaning that catch would be constrained to a very small fraction of the catch corresponding to the inflection point of the control rule once biomass fell very far below the inflection point. Here are some examples:

Proportional reduction in biomass (from the inflection point)	Proportional reduction in catch (from the catch corresponding to the inflection point)
0.25	0.45
0.50	0.75
0.75	0.95

Because the harvest control rules in Tiers 1-3 reduce harvest dramatically in the event that the stock falls below the inflection point, much more potential prey remains available for marine mammal piscivores than would be the case if either fishing mortality or catch were held constant. Although the control rules in Tiers 4-6 lack an inflection point, it should be noted that all groundfish that are major Steller sea lion prey species are managed under Tiers 1-3.

It is also important to remember that the harvest control rules serve to constrain not only target catch but *all bycatch* as well. This can have very substantial implications for the amount of prey biomass left in the ocean. For example, suppose the following:

1. Species 1 and 2 are groundfish prey of some marine mammal piscivore.
2. In the absence of bycatch constraints, species 2 would be harvested at a rate F2.
3. In addition to supporting its own target fishery, species 1 is taken as bycatch in the target fishery for species 2 according to the equation: bycatch = β×F2×B1, where B1 is the biomass of species 1 and β is a parameter.
4. Identical yields from species 1 and 2 have equal economic value.

Under the above conditions, the target fishery for species 1 will close (i.e., species 1 will become bycatch-only) if FOFL for species 1 (FOFL1) is less than β×F2. Furthermore, if the target fishery for species 1 closes, the allowable catch of species 2 will decrease by the proportion [FOFL1/(β×F2)]–1. For example, suppose the following values:

1. F35% for species 1 = 0.25
2. β = 0.5

3. F2 = 0.2
4. Biomass of species 1 = B10%

Under these values, the allowable catch of species 2 would decrease by about 47% compared to what it would have been if the FOFL control rule for species 1 had not constrained the harvest of species 2. This means that much more potential prey remains available for marine mammal piscivores than would be the case if the harvest control rules limited target fishing mortality only.

Although the harvest control rules described above may be effective at preserving adequate prey for Steller sea lions at large spatial and temporal scales, further examination of the potential impacts of commercial fishing at localized scales is needed (see Section 4.4.5.4 Habitat Response to Fisheries of Concern: Short Term Effects).

4.5.4.5 Changes in Genetics, Reproductive Capacity, and Life History Characteristics

Fisheries generally select particular individuals (usually larger and older fish) and focus on particular locations (such as spawning or feeding grounds) such that fishing is non-random with respect to fish characteristics (or phenotypes). If there is a genetic component to differences in phenotypes between fish, then fishing will cause evolutionary change. The argument that fishing could cause phenotypic evolution is widely known in general terms. There are numerous examples of changes in life history characteristics of commercially exploited stocks over time, such as weight-at-age, length-at-age, length-at-maturation and age-at-maturation (reviewed in Law 2000). Fisheries can generate selection on life history traits by catching more fish of some ages or sizes than others. In many cases fisheries remove larger and older fish which means that early-maturing and smaller fish leave more offspring than late-maturing ones and are selected for. This has consequences for yield. For example, the current patterns of fishing are selecting a life history in Northeast Arctic cod in which fish allocate resources to reproduction rather than growth. The sustainable yield associated with this life history could be less than half the yield potentially available (Law and Grey, 1989).

It is important to note that although changes in life history characteristics of fished populations have been observed, there is uncertainty about what causes these changes. This is due in part to the fact that potential fishery effects on life history characteristics are superimposed on a backdrop of environmental change that can affect the same characteristics (such as the effect of temperature on growth). It is also unclear how fast fishery-induced evolution occurs. It is uncertain whether evolution contributes to the phenotypic changes observed in many fish stocks, or whether it is operating at a much longer time scale (Law 2000).

Fishing can impact reproductive capacity through changes in fish size. Reductions in fish size can result from long-term size-selective fishing (e.g., Zwanenburg, 2000). The decrease in the proportion of large fish might have negative impacts on reproductive capacity of the population if smaller, first-time spawners are less successful in producing viable eggs than are larger, more experienced spawners (Trippel *et al.* 1997). In addition, some temperate demersal fishes, such as cod and other gadids, are thought to rely on the longevity and size of mature individuals to bridge the gaps between years of strong recruitment (Longhurst 1999). However, this may not be an issue in Alaska. An analysis of the community size spectrum of eastern Bering Sea groundfish provides no evidence for a reduction in fish size due to size-selective fishing (Boldt *et al.* 2007).

4.5.4.6 Ecosystem Effects of Fishing

In Section 4.1 we investigated the north Pacific ecosystem, natural environmental change, and climate shifts, and explored some of the potential impacts of fisheries on those changes (Section 4.1). We also looked for climate shift signals in Steller sea lion prey recruitment and abundance. In this section we explore recent modeling results, which utilize more complex systems to assess ecosystem impacts of fishing.

This section is divided into two parts; the first is an assessment of current relationships between Gulf of Alaska groundfish predators (including Steller sea lions) and pollock; and the second part presents the results of ecosystem modeling of the direct and indirect effects of removing fishing pressure.

In this analysis we model a fishery in which the harvest strategy is similar to the Council's "F40" harvest strategy, where the maximum permissible Allowable Biological Catch (ABC) is based on an exploitation rate for commercial groundfish which is intended to reduce the equilibrium spawning stock biomass to 40% of its presumed unfished biomass (Goodman *et al.* 2002). While several of the commercially important groundfish stocks are fished with this harvest rate, not all stocks are fished at this harvest rate for a variety of reasons related to data quality, assessment, and multispecies bycatch management. Furthermore, some stocks (halibut, herring, and salmon) are not managed by the Council and may have a different harvest rate. Therefore, we elected to use exploitation rates observed in 2005 fisheries even if some of these harvest rates would not result in the removal of 40-60% of a stock's biomass relative to unfished biomass at equilibrium, rather than attempting to estimate and simulate an F40 harvest rate for every fished species.

Competitive interactions between Steller sea lions and groundfish predators in the Gulf of Alaska (based on Dorn et al. 2005 and NMFS 2006b)

In the Gulf of Alaska, the top five predators on pollock greater than 20 cm by relative importance are arrowtooth flounder, Pacific halibut, Pacific cod, Steller sea lions, and the directed pollock fishery (Figure 4.34). For pollock less than 20 cm, arrowtooth flounder represent close to 50% of total mortality. All major predators show some diet specialization, and none depend on pollock for more than 50% of their total consumption. Pacific halibut is most dependent on pollock (48%), followed by Steller sea lions (39%), then arrowtooth flounder (24% for juvenile and adult pollock combined), and lastly Pacific cod (18%). It is important to note that although arrowtooth flounder is the largest single source of mortality for both juvenile and adult pollock, arrowtooth depend less on pollock in their diets then do the other predators.

The size preference of predators for walleye pollock varied by predator species. Pacific cod and Pacific halibut fed primarily on pollock greater than 30 cm fork length; this size range is similar to that published by Sinclair and Zeppelin (2002) for Steller Sea lions. Arrowtooth flounder, on the other hand, primarily feed on fish between 10 to 30 cm long. Unlike the Bering Sea, cannibalism is a relatively minor source of mortality for pollock in the Gulf of Alaska. It is notable that the three species that focus on adult pollock as prey (cod, halibut, and Steller sea lions) all show biomass declines since the late 1980s as adult pollock biomass has declined, while arrowtooth flounder, feeding on the smaller pollock, have increased (Figure 4.35).

To better judge natural mortality, consumption was calculated for two size groups of pollock, divided at 30 cm fork length. This size break is based on finding minima between modes of pollock in predator diets. This break is different from the transition matrices used in the stock assessment; perhaps due to differences in size selection between predators and surveys. For this analysis, it is assumed that pollock less than 30 cm are ages 0 to 2 while pollock greater than or equal to 30 cm are age 3+ fish.

Consumption of age 0 to 2 pollock, per unit predator biomass (using survey biomass), varied considerably through survey years, although within a year all predators had similar consumption levels. Correlation coefficients of consumption rates were 0.98 between arrowtooth and halibut, and 0.90 for both of these species with pollock. Correlation coefficients of these three species with cod were ~0.55 for arrowtooth and halibut and ~0.20 with pollock. The majority of this predation by weight occurred on age 2 pollock.

Plotted against age 2 pollock numbers calculated from the stock assessment, consumption/biomass and total consumption by predators shows a distinct pattern. In “low” recruitment years consumption is consistently low, while in high recruitment years consumption is high, but does not increase linearly, rather consumptions seems to level out at high numbers of juvenile pollock, resembling a classic “Type II” functional response. This suggests the existence bottom-up control of juvenile consumption, in which strong year classes of pollock “overwhelm” feeding rates of predators, resulting in potentially lower juvenile mortality in good recruitment years which may amplify the recruitment. However, this result should be examined iteratively within the stock assessment, as the back-calculated numbers at age 2 assume a constant natural mortality rate. Assuming a lower mortality rate due to predator satiation would lead to lower estimates of age 2 numbers, which would make the response appear more linear.

Consumption of pollock greater than or equal to 30 cm shows a different pattern over time. A decline of consumption per unit biomass is evident for halibut and cod. Arrowtooth shows a non-significant decline; it is possible that the noise in the arrowtooth trend, mirroring the consumption of less than 30 cm fish, is due to the choice of 30 cm as an age cutoff. As a function of age 3+ assessment biomass, consumption per unit biomass and total consumption remained constant as the stock declined, and then fell off rapidly at low biomass levels in recent years. Again, this result should be approached iteratively, but it suggests increasing predation mortality on age 3+ pollock between 1990 to 2005, possibly requiring increased foraging effort from predators.

There has been a marked decline in Pacific halibut weight at age since the 1970s that Clark *et al.* (1999) attributed to the 1977 regime shift without being able to determine the specific biological mechanisms that produced the change. Possibilities suggested by Clark *et al.* (1999) include the physiological effect of an increase in temperature, intra- and interspecific competition for prey, or a change in prey quality. The two species most dependent on pollock in the early 1990s (Pacific halibut and Steller sea lion) have both shown an exceptional biological response during the post-1977 period consistent with a reduction in carrying capacity (growth for Pacific halibut, survival for Steller sea lions). In contrast, the dominant predator on pollock in the Gulf of Alaska (arrowtooth flounder) has increased steadily in abundance over the same period and shows no evidence of decline in size at age. Given that arrowtooth flounder has a range of potential prey types to select from during periods of low pollock abundance, we do not expect that arrowtooth would decline simply due to declines in pollock.

Taken together, these results suggest that recruitment remains bottom-up controlled, even under the current estimates of high predation mortality, and may lead to strong year classes. However, top-down control seems to have increased on age 3+ pollock in recent years, perhaps as predators have attempted to maintain constant pollock consumption during a period of declining abundance. Thus, increasing competition for larger prey is consistent with the parallel declines of halibut, cod, and Steller sea lions. It is possible that natural mortality on adult pollock will remain high in the ecosystem in spite of decreasing pollock abundance.

The results presented above are taken from Gulfwide weighted averages of consumption; Steller sea lions and the fishing fleet are central place foragers, making foraging trips from specific locations (ports in the case of the fishing fleet, and rookeries or haulouts for Steller sea lions). Foraging bouts (or trawl sets) begin at the surface, and foragers attack their prey from the top down. For such species, directed and local changes in fishing may have a disproportionate effect compared to these results (Dorn *et al.* 2005, NMFS 2006b).

In contrast, predation by groundfish is not as constrained geographically, and captures are likely to occur when the predator swims upwards from the bottom. Changes in the vertical distribution of pollock may tend to favor one mode of foraging over another. For example, if pollock move deeper in the water column due to surface warming, foraging groundfish might obtain an advantage over surface foragers. Alternatively, pollock may respond adaptively to predation risks from groundfish or surface foragers by changing its position in the water column.

Potential effects of stopping all fishing using a dynamic food-web model

To examine the relative role of pollock natural versus fishing mortality within the GOA ecosystem, a set of simulations were run using an extended dynamic food-web model based in part on Ecopath with Ecosim (Christensen *et al.* 2005), but modified to include more specified and accurate age-structured dynamics for key species such as pollock, cod, and arrowtooth flounder (Aydin *et al*. 2006; Aydin *et al.* 2007). Two approaches were taken to the modeling. First, sensitivity analyses were performed on the model to determine the relative importance of direct and indirect effects within the model. Secondly, projections were made under several ecological and fishing scenarios to attempt to examine the effects of fisheries removals.

Following the method outlined in Aydin *et al.* (2003), the sensitivity analysis indicated that the largest effects of declining adult pollock survival would be declines in halibut and Steller sea lion biomass. Declines in juvenile survival would have a range of population effects, including populations of halibut and Steller sea lion, but also releasing a range of competitors for zooplankton including rockfish and shrimp. The pollock trawl itself has a lesser effect throughout the ecosystem (fishing mortality is small in proportion to predation mortality for pollock); the strongest modeled effects are not on competitors for prey but on incidentally caught species, with the strongest effects being on sharks.

To address this question for the Gulf of Alaska ecosystem, NMFS performed a simulation analysis using the Gulf of Alaska dynamic ecosystem model parameterized with information from the early 1990s (Gaichas 2006, Aydin *et al.* 2007). To determine what potential food web or cumulative effects of fishing might impact Steller sea lion prey or the ecosystem as a whole, we compared predicted biomass trajectories for key species between two fishing scenarios: a “status quo” scenario where 2005 exploitation rates in all fisheries were continued for 100 years, and a “no fishing” scenario where all fishing was stopped for 100 years. The simulation includes an assessment of uncertainty, and considers alternative future production regimes for both pollock and primary production. The difference between the results from the “no fishing” scenario and the “status quo” scenario under each potential future production regime is intended to provide insight into broad potential food web or cumulative effects of fishing in the Gulf of Alaska ecosystem. In particular, we focus on results for pollock and Steller sea lions, as well as halibut, cod, and arrowtooth flounder. The groundfish species presented have been identified as potentially important structural components of the GOA ecosystem (Gaichas 2006). However, results are available for all groups in the ecosystem.

The GOA dynamic ecosystem model is fully described in Gaichas (2006) and Aydin *et al.* (2007). The initial food web model was built with the Ecopath algorithms (Christensen *et al.* 2005). For the key groundfish species under analysis here, fully age structured population dynamics were implemented based on the life history “stanza” implementation of Ecosim (Walters and Martell 2004), with modified maturity accounting used to better match Gulf of Alaska groundfish. For most other species, the biomass dynamics equations described in Walters *et al.* (1997) were used to make forward projections. For this analysis, we included all fisheries operating in the Gulf of Alaska, including groundfish fleets, the halibut fishery, the herring and salmon fisheries, and the subsistence fishery which takes a small number of Steller sea lions each year. For more details on methods see NMFS (2006b).

NMFS attempted to discern the potential food web or cumulative effects of fishing the ecosystem as a whole by stopping fishing, allowing the ecosystem model to re-equilibrate with no fishing, and comparing equilibrium no fishing biomass levels with biomass at status quo fishing in ecosystem model simulations. Overall, the clear effects of fishing on the ecosystem are apparent only for top predators which are directly exploited, either as target species or as bycatch. It is less clear how fishing affects species with high predation mortality, and unfished species via prey interactions.

Under the no fishing scenario, while pollock increase between 5-10% for the first few years of the simulation (2006-2009), this is followed by a decrease back to the level of the status quo scenario; ceasing fishing within the ecosystem model has little or no effect on pollock standing stock in 50% of the modeled ecosystems (NMFS 2006b). This is due to the increase in predators as fishing ceases, the predators of pollock are able to absorb increases in biomass through increased predation. It is important to note that, as these confidence regions represent a range of whole ecosystems, there are ecosystems between the 50% and 95% intervals in which ceasing fishing causes pollock to increase (and decrease) substantially. This highlights the fact that, for some species, the single species F40 reference point does not mean that the cessation of fishing will result in a 40-60% increase in the standing stock of biomass; it is possible that this will simply redirect prey from fisheries to other predators.

There were two cases in which ceasing fishing has a predictable effect similar to single-species projections. Ceasing fishing on Pacific halibut and Pacific cod results in a 40-60% increase in those species. In both cases, as these are top predators within the ecosystem, there is little or no buildup of predators above them. For Steller sea lions, the model predicts that the cessation of fishing would cause Steller sea lions to increase in biomass (50% confidence intervals is between 2-10% increase). It is important to note that this effect, similar to that for pollock, is also dampened over time.

The effect of ceasing fishing (on all species) on arrowtooth flounder results in a definite downward trend for arrowtooth. This counterintuitive result is a reflection of indirect effects. Arrowtooth are lightly fished compared to their competitors such as cod and halibut; stopping fishing for all species greatly shifts the competitive advantage to those other species, thereby out-competing arrowtooth for prey.

The examination of the trophic relations reveals a great deal of uncertainty, as shown by the uncertainty of the projections (NMFS 2006b). However, some of the counterintuitive results can be seen against the background of overall uncertainty; namely, the limited increase of pollock under no fishing scenarios, reflects that many of the stock assessment reference points (e.g., F40) are reflections of historical conditions and not necessarily predictive of future ecosystem states.

An additional, important note from these results is that, on the scale of an entire large marine ecosystem, fish predators, with faster life-histories and greater area coverage, have a greater capacity to respond to prey releases than do central place foragers such as Steller sea lions. The results shown here represent fishing policies applied on the ecosystem scale; it is quite possible that spatially and temporally targeted fishing reductions (e.g., near haulouts) would direct more of the prey released towards Steller sea lions (NMFS 2006b).

Potential effects of stopping only pollock fishing using the dynamic food-web model

The reaction of the pollock stock to a reduction or stoppage in fishing depends, in part, on how important predation is relative to fishing in causing pollock mortality at present (Figure 4.36). Using different modeling assumptions suggests a range of possible changes in pollock biomass if pollock fishing is reduced or stopped relative to continuing with status quo fishing for other species over the same projection period. Single species bars (blue) report differences between year 2018 projections for the no fishing (scenario 5) and author's recommended F (scenario 1 or 2) from 2005 stock assessments for each species, except for arrowtooth flounder which reported only a five year projection to 2010. Predation bars report differences between ecosystem model runs from the mid-1990s to the end of a 20 year period for two scenarios: no pollock fishery but other fisheries continue ("no pollock F", pink bars), and no fisheries for any species including herring, salmon, and all groundfish ("no F", burgundy bars).

The difference between the single species outcome for pollock and the ecosystem model outcomes has to do with the relative importance of fishing and predation mortality assumed in each model, and whether the fishery for pollock stops or all fishing stops. In the single species stock assessment model for pollock, natural mortality is assumed to be higher than fishing mortality ($M = 0.3$, $F = 0.19$ in 2005; Dorn *et al.* 2005). Therefore, in the single species assessment model, fishing mortality represents nearly 39% of total mortality for pollock. When this source of mortality is removed in the model, the pollock stock is predicted to increase by over 100% by 2018 relative to applying the author's recommended F over that same time period (which varies between 0.19 and 0.23, Dorn *et al.* 2005). In this model, it is assumed that fishing on pollock stops, which strictly may be interpreted as an end to all fishing which might catch pollock unintentionally. More commonly, the assumption is that the target fishery for pollock would stop, which in this case accounts for over 99% of all fishing mortality on pollock (2005 NMFS AKRO Catch Accounting System data, Terry Hiatt AFSC personal communication). Therefore, clarifying whether pollock fishing is reduced or stopped by stopping the pollock fishery or all fisheries is irrelevant in the single species case. We note that for most other groundfish, achieving $F=0$ implies shutting down more than just the fishery targeting that fish.

The outcome of stopping only the targeted pollock fishery in the ecosystem model results in a considerably lower predicted increase in pollock biomass than the single species assessment model predicts. After accounting for diets and consumption of predators in the ecosystem, we can confirm that the assumption that natural mortality exceeds fishing mortality is correct; however, the ratio of fishing to natural mortality is estimated to be quite different in the ecosystem model. The vast majority of adult pollock mortality is caused by predation; further, the vast majority of the predation mortality is caused by three groundfish predators: arrowtooth flounder (33% of total mortality), halibut (23%), and cod (16%; Figure 4.34). The pollock trawl fishery causes only 6.6% of adult pollock mortality, which is similar in magnitude to that caused by sablefish, Steller sea lions (adults and juveniles combined), and by pollock cannibalism. When the fishery on pollock is removed in the ecosystem model, just under 7% of mortality is removed, rather than nearly 39% as in the single species model. Therefore, the ecosystem model predicts that the pollock stock would increase by about 10% if the pollock fishery were to stop (but all other fisheries continued).

When all fishing is stopped in the entire ecosystem (burgundy bars in Figure 4.36), pollock biomass is actually predicted to decrease relative to status quo fishing. This apparently

> counterintuitive pattern is explained by the indirect effects of fishing on the system: a complete lack of fishing in the ecosystem increases predator biomass and therefore the predation mortality experienced by pollock. Because pollock are apparently predicted to experience more of an increase in predation mortality than the decrease in fishing mortality when all fishing is stopped, overall pollock mortality increases and their biomass decreases in a Gulfwide no fishing scenario.

Initial modeling results give similar results for the Bering Sea for pollock; both age-structured multispecies models (MSFOR) and ecosystem dynamics models (ECOSIM) give similar results, suggesting that pollock would only increase 10 to 20% with cessation of fishing, while single-species models produce increases of up to 60%. Part of this difference may be due to recruitment methodology; the single-species stock assessment model draws projected recruitment from density-independent past recruitment rather than assuming a stock-recruitment relationship.

4.5.5 Indirect Effects on Water Quality

After fish are harvested from the ocean, they are usually processed before they are delivered to markets. Seafood processing covers a range of activities that can be as simple as removing viscera and storing whole fish on ice, it can require cutting fish into fillets or steaks, or it can involve more processing to form products like surimi or fish meal. Seafood processing generates waste that consist of highly biodegradable constituents such as tissue solids, oil and grease, along with fluids from viscera, heads, bones, and other discarded materials. The major constituents that are not highly degradable are crab and shrimp shells. These materials are usually ground up before being discharged from seafood processing facilities. However, most fishing operations send offal and other discarded materials to a meal plant where tissues and reduced to a saleable fish meal. This greatly reduces discharge of processing wastes by both shoreside plants and offshore factory harvesting or mothership vessels.

The adverse effects of discarding this material tend to be highly local and usually depend on flushing rates and dispersal regimes of the receiving waters. When discharges exceed the dispersion and biodegradation rates of the receiving waters, they can build up, increase the biochemical oxygen demand of the receiving waters, and can produce noxious smells. Waste generated by seafood processing can cause receiving waters to become anoxic, can elevate ammonia levels, can smother benthic organisms, and attract scavengers such as gulls or rodents, which may cause public health problems (Patten and Patten 1979). This scenario is generally limited to bays and inlets along the coast where seafood processing operations discharge into protected, often low turnover, embayments. Offal discharges offshore may provide nutrients utilized through the water column and in the benthic environment.

In the 1970s, fish and shellfish waste discharged from mobile and shore-based processors at Kodiak, Dutch Harbor, and Akutan polluted coastal waters around those communities. In 1971, about 3.3×10^4 mt of waste was discharged at Kodiak (Jarvela 1986). In 1976, about 2.1×10^4 mt of waste was discharged at Dutch Harbor. In 1983, the shore-based Trident Seafoods plant at Akutan released between 9 and 11×10^4 mt of codfish and crab wastes into Akutan Harbor before the plant was destroyed by fire. Sonar surveys of Akutan Harbor identified a waste pile that was about 7 m thick and 200 m in diameter.

Section 303(d)(1)(C) of the Clean Water Act and the EPA's implementing regulations (40 CFR 130) require the establishment of a Total Maximum Daily Load (TMDL) to achieve state water quality standards when a body is limited by water quality. A TMDL identifies the degree of pollution control needed to maintain compliance with standards using an appropriate margin of safety. The focus of the TMDL is reduction of pollutant inputs to a level (or load) that fully supports the designated uses of a given waterbody. In 1997, the Alaska Department of Environmental Conservation (AKDEC) identified Udagak Bay (Beaver Inlet on Unalaska Island in the Aleutian Islands) and King Cove lagoon in King

Cove (on the Alaska Peninsula in the Aleutians East Borough) as being water quality-limited for seafood wastes. TMDLs were established for both facilities in 1998.

For Udagak Bay, AKDEC concluded that the Northern Victor Partnership facility P/V Northern Victor produced seafood processing wastes (from Pacific cod, Pacific halibut, herring, walleye pollock, salmon, and a variety of other fish) that created a waste pile deposit of solid residues measuring at least 2.4 acres in area and 7 feet thick on the seafloor. AKDEC concluded that the waste pile exceeded Alaska's water quality standards for residues. For King Cove, the AKDEC concluded that the Peter Pan Seafoods facility created a waste pile covering 11 acres of seafloor to an average depth of 3 feet.

In 2004, the list of impaired waters that was prepared by the AKDEC included only 1 site which was impacted by seafood wastes; Popof Strait in the East Aleutians Borough[21]. Additional water bodies in Cold Bay, Dutch Harbor, Kodiak, etc., have been impaired by logging operations, military materiel, or fuel storage. The effects of these facilities appear to be localized and would not be expected to adversely affect threatened or endangered species under NMFS's jurisdiction.

4.6 Response of Other Pinnipeds to Environmental Change, Prey Depletion, or Direct Takes

The growth of marine mammal populations, as for all vertebrates, is fundamentally governed as a bottom-up process by prey availability. However, other processes such as intraspecific social dynamics, environmental disturbances, or top-down control through predation can supersede or interact with that bottom-up control and result in complex population responses (Boyd and Murray 2001, Sinclair and Krebs 2002, Frid *et al.* 2006). A substantial amount of literature associates physiological, behavioral, or population level responses of pinnipeds with reductions in prey availability in both inferential and direct studies. In nearly all of these studies, individual or population responses were associated with reductions in prey availability due to stochastic environmental conditions or intraspecific competition, though prey abundance can also be reduced by interspecific competition, long-term climatic influences, or commercial fisheries. Management concerns of competition for prey with commercial fisheries have recently been expressed for California sea lions (*Zalophus californianus*) breeding in the Central Gulf of California (Szteren *et al.* 2006), New Zealand sea lions (*Phocarctos hookeri*)(Chilvers *et al.* 2005), and Australian sea lions (*Neophoca cinerea*, Campbell *et al.* 2006, Fowler *et al.* 2006).

For a comparison of responses to reduced prey availability that might be observed in Steller sea lion populations, it is appropriate to limit review to studies of other otariid (sea lion and fur seal) populations. All otariids utilize a forage-cycle maternal strategy (Schulz and Bowen 2004) that relies upon energy obtained during lactation to provision pups, an income breeding strategy that contrasts with the capital breeding strategy of phocids (Boyd 2000). This strategy is energetically costly (Costa 1993), but provides for maximization of energy transfer to pups when local prey availability is high (Pitcher *et al.* 1998). Conversely, responses to decreased prey availability are poor maternal and/or pup condition. The type and magnitude of response depends on the timing, duration and magnitude of prey depletion in relation to the period of gestation and lactation. Longer-term life-history consequences appear to develop over sustained periods of low food availability.

Undernutrition of reproductive females at the time of implantation results in unsuccessful or delayed implantation (observed in Antarctic fur seals *Arctocephalus gazella*, Lunn and Boyd 1993a, b; Boyd 2000). Because energy requirements increase throughout gestation the effects of undernutrition during that period can greatly affect subsequent birth rates. Poor maternal condition during gestation is associated with decreased birth rates due to increased abortions in several species (South American sea

---

[21] http://www.dec.state.ak.us

lion (*Otaria flavescens*), Soto *et al.* 2004); Antarctic fur seal, Duck 1990, Lunn and Boyd 1993b; South African fur seal (*Arctocephalus pusillus*), Guinet *et al.* 1998; and South American fur seal (*Arctocephalus australis*), Lima and Paez 1995). Low food availability in late pregnancy is also associated with smaller subsequent birth masses (Antarctic fur seals, Boyd and McCann 1989; Lunn and Boyd 1993b; Lunn *et al.* 1994).

Depending upon the species, pups are either wholly dependent upon maternal provisioning until weaned (fur seals), or may supplement energy needs with independent foraging during mid to late lactation (some sea lions). Thus local prey abundance and distribution strongly influences maternal foraging trip duration and attendance patterns (Boness and Bowen 1996) and has consequences for the ability of lactating females to provision pups. As a consequence of decreased prey availability, maternal foraging trips may increase in duration, become more variable, or otherwise show changes in diving behavior indicative of increased foraging effort (California sea lion, Costa *et al.* 1991; Juan Fernandez fur seal, Francis *et al.* 1998; Antarctic fur seal, Boyd *et al.* 1994, McCafferty *et al.* 1998, Lea *et al.* 2006; South American sea lion, Soto *et al.* 2006). Extended foraging trips have an associated decrease in time spent onshore for pup attendance (South American sea lion, Soto *et al.* 2006; California sea lion, Heath *et al.* 1991), increasing the duration of pup fasting periods.

Changes in maternal attendance and provisioning efficiency subsequently have effects on pup growth rates and weaning mass, which decrease in response to declining prey availability (South African fur seal, Guinet *et al.* 1998; Subantarctic fur seal, Chambellant *et al.* 2003; Antarctic fur seal, Boyd and Murray 2001, Boyd *et al.* 1994; California sea lion, Boness *et al.* 1991; New Zealand fur seal (*Arctocephalus forsteri*), Bradshaw *et al.* 2000). Increased pup mortality is also associated with decreased prey availability (Antarctic fur seal, Boyd *et al.* 1994; Lunn *et al.* 1994) and is particularly acute during extremely low food availability years associated with strong El NiZo conditions (South American sea lion, Soto *et al.* 2004, 2006; Galapagos fur seal, Trillmich and Limberger 1985; Galapagos sea lion, Trillmich and Limberger 1985), during which there is also increased mortality of the youngest age classes (South American sea lions, Soto *et al.* 2004; Galapagos fur seal, Trillmich and Limberger 1985).

Reductions in prey availability during a breeding season may have consequences that extend into subsequent years. There may by a trade-off between pregnancy and lactation (South African fur seal, Guinet *et al.* 1998), and reduced prey availability during a breeding season delayed birth dates in the subsequent year (South American sea lion, Soto *et al.* 2004; Antarctic fur seal, Lunn and Boyd 1993a). There is also evidence that care of a yearling during an extended lactation period reduces natality or survival of subsequent pups (Galapagos fur seal, Trillmich 1986; Australian sea lion, Higgins and Gass 1993).

Otariids clearly can be affected by reductions in prey availability throughout the breeding cycle, and acute prey depletion circumstances (as occurs during some El Niño events among temperate and sub-tropical dwelling species) can have catastrophic impacts extending beyond the season of depletion. However, otariid life-history strategies have evolved in association with intra- and inter-annual variations in prey availability. When low food availability becomes chronic however, the combined impacts clearly regulate populations. For example, in association with long-term decreased prey availability likely associated with density dependency, Subantarctic fur seals exhibited slower maturation rates, lower age-specific reproductive rates, and lower older-age class female survival, resulting in an overall limitation of the number of weaned pups produced per a female lifetime (Dabin *et al.* 2004).

4.7 Response of Steller Sea Lions and Critical Habitat to the Environmental Baseline

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); in addition, there may have been density-dependent responses at lower population levels.

We have only a limited or qualitative understanding of how multiple factors interact to create an overall cumulative effect on Steller sea lion populations. 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 Steller sea lions can be delineated with certainty (NRC 1996, NMFS, 2001, NRC 2003). Yet, a number of theories attempting to explain the decline in Steller sea lions and apparent changes in the structure of North Pacific ecosystems since the 1970s have been developed, and these involve direct (e.g., top-down) and indirect (e.g., bottom-up) 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). 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.

4.7.1 Summary of the Likely Current Environmental Baseline Stressors

In the sections above, NMFS discussed the various factors which may affect Steller sea lion health and population numbers. The intent of this section is to determine which of those factors represent a continuing impediment to Steller sea lion survival and recovery or the maintenance of critical habitat to provide for recovery. A summary table is provided in Table 4.8. In that table NMFS contrasts what we knew in 2000 (FMP Biological Opinion) and what we have learned since then. In some cases NMFS' view of the past decline has changed, or our view of whether a stressor is currently acting or not also may have changed. This section synthesizes information made available through research since 2000.

4.7.1.1 Environmental Change

The potential impact of environmental variability, through a reduction in the biomass and quality of Steller sea lion prey species, has 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. Adult females and juveniles are likely the most vulnerable age-classes to environmental change.

The change in North Pacific fish community structure stemming from the regime shift in 1976-77 may have been substantial enough to alter the quality and availability of prey for Steller sea lions, resulting in changes in the success of pollock and other gadids, which may have resulted in part to nutritional stress of juvenile Steller 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 Steller sea lion vital rates. Interestingly, high energy salmon populations in the North Pacific increased dramatically from the late 1970s to present allowing large commercial, sport, and subsistence harvests annually over the past three decades. 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 Steller sea lions during the 1900s. If it were within the normal range and Steller 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 (NMFS 2008a).

It is likely 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 Steller sea lions. Gadids have been and will continue to be a principal component of the diet of Steller 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 Steller sea lion diets between regimes were unremarkable. Thus the potential impact of environmental variability on recovery in the near term is minimal.

Fishing could have exacerbated the regime shift related impacts through relatively high local harvest rates of Steller sea lion prey species, increasing their foraging costs. These changes would decrease carrying capacity, yet their impact on Steller sea lions should decrease as the population declined. However, a threat to recovery will persist until the environment, and associated fish distributions and populations, change again to favor Steller sea lions, increasing their carrying capacity and subsequently Steller sea lion survivorship and birth rates. It is likely that environmental change, coupled with fishery impacts, affected Steller sea lion at the population level during the decline and is currently a stressor to consider in Chapter 5.

4.7.1.2 Indirect Fisheries Effects

The potential impact of competition with fisheries, through a reduction in the biomass and quality of Steller sea lion prey species, is a highly debated topic among the scientific community. The primary issue of contention is whether fisheries reduce Steller sea lion prey biomass and quality at both the local and regional spatial scales that may lead to a reduction in Steller sea lion survival and reproduction, and if sustained, possibly regional reductions in carrying capacity. The effect of fisheries on the distribution, abundance, and age structure of the Steller sea lion prey field, at the spatial scale of foraging Steller sea lions and over short and long temporal scales, is largely unknown. As noted above, it is likely that no one factor accounts for the dynamic trends in Steller sea lion abundance in the western population, and that factors responsible for the period of steep decline (e.g., 1980s), slow decline(e.g., 1990s) and slow recovery (e.g., 2000s) differ.

Fisheries are likely to lower Steller sea lion carrying capacity. A decreased carrying capacity could result from the combined effects of seasonally compressed fishing in Steller sea lion foraging areas, the long term impacts of exploitation of Steller 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 Steller sea lion prey field. Population declines could have been driven in part by reductions in the quality and quantity of available Steller 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 Steller 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 it seems reasonable to conclude that commercial fisheries for primary prey species of Steller sea lions has lead to a reduction in the carrying capacity of the environment, it is much less clear how commercial fisheries are currently affecting the recovery of the western DPS. Further, it seems reasonable to conclude that at least in some areas (e.g., southeastern Alaska), commercial fisheries have not adversely impacted the recovery of the eastern DPS of Steller sea lions over the last 20 years. Nonetheless, regarding the indirect effects of commercial fisheries on the western DPS, it seems reasonable to conclude that such activities may be a continuing stressor.

4.7.1.3 Direct Human Effects

Commercial fisheries can directly affect Steller sea lions in the BSAI, and GOA by capturing, injuring, or killing them in fishing gear or in collisions with fishing vessels, and if fishermen kill them intentionally. These impacts were described in detail above in Sections 4.3.3 (incidental take in commercial fisheries), 4.3.4 (intentional and illegal killing), and in 4.3.7 (disturbance). In general, the current level of direct impact to Steller sea lions is relatively small (see summary in Section 4.3.10). However, it is likely that historical direct impacts influenced the rapid decline rate observed in the 1980s, but by the mid-1990s was no longer an important factor in the decline and lack of recovery. Vital rate analyses confirm the reduction in direct mortality (Holmes and York 2003, Holmes *et al.* submitted). Thus, this is unlikely to be a continuing stressor.

4.7.1.4 Predation

Some hypothesize that predation by killer whales is causing continued decline or is a significant contributor to the lack of recovery of Steller sea lions (Section 4.3.2). One study of killer predation in the eastern Gulf of Alaska found that transient killer whales are having only a minor effect on the survival of Steller sea lions in that area (Maniscalco *et al.* 2007). As summarized in NMFS (2008), Steller sea lions (and other pinnipeds) have likely always been prey of mammal eating killer whales and killer whale predation has been a component of that natural mortality. Additionally, if the population of killer whales is assumed to have been the same size historically as it is now and if predation events were approximately constant, that level of predation would be a greater proportion of mortality on the western population of Steller sea lion, and in the areas of greatest reduction (e.g., western Aleution sub-region) in particular. Durban *et al.* (2010) recently reported that transient killer whales from the central GOA to the central Aleutians number approximately 345 and noted the extremely high potential level of predation pressure that comes from this high a number of killer whales.

Life-history changes in the western DPS of Steller sea lion through time argue against the hypothesis that killer whale predation alone was responsible for the decline. Density dependent responses seen in the western DPS 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 Steller 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, Fay and Punt 2006, Holmes *et al.* 2007) in those areas where adequate data exist to estimate trends in rates of survival and reproduction. The observed changes in age and growth and the pattern of increasing survival in some regions of the range of the western population of Steller sea lion argue against killer whale predation as a primary cause of the observed steep decline in the 1980s and slow decline in the 1990s. While possible, it seems unlikely that predation pressure would have led to the observed decrease in natality in certain sub-regions of the range. But in some subareas, the extremely low number of Steller sea lions and the high numbers of killer whales seasonally present in these waters suggest killer whales may limit the degree to which this SSL subpopulation can increase appreciably (see Durban *et al.* 2010). Springer *et al.* (2003) argues killer whales could have caused large declines in marine mammal populations in the North Pacific. Although rebutted by many, Springer *et al.* (2008) hold their hypothesis as a possible factor influencing the decline of the Steller sea lion and its lack of recovery in some subareas. 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. Recent information on juvenile Steller sea lion survival from branding (Figure 3.13) also argues against killer whale predation as a significant impediment to recovery throughout a part of the range. Yet LHX tagging work on transient juvenile SSLs in the Kenai Fjords/Prince William Sound in the eastern GOA indicate killer whale predation may be the single largest cause of mortality. Rates of Steller sea lion juvenile survival in the 2000s in the eastern Aleutian Islands and the central Gulf of Alaska are similar to those estimated for the pre-decline 1970s population in the central Gulf of Alaska (Holmes *et al.* 2007), suggesting that direct mortality is not currently inhibiting Steller sea lion recovery. And so the debate continues, with competing yet logical representations of the influence transient killer whales may have on Steller sea lions and other marine mammals.

4.7.1.5 Inter-specific Competition

Piscivorous fish, other marine mammals, and some birds 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, the following annual differences have been reported: size and distribution of young-of-the-year, levels of cannibalism by adult pollock (Livingston 1991, Wespestad *et al.* 2000). Differential rates of fishing within the groundfish community may have also indirectly contributed to an increasing biomass of arrowtooth flounder, 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.

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 (in terms of biomass) by cetaceans approaches or exceeds removals by commercial fisheries in some areas of the world's oceans (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). 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). As populations of piscivorous cetaceans recover, this potential competitive interaction would be expected to increase. Thus, it is possible that inter-specific competition with other predators has influenced the demographics of Steller sea lions and may further in the future.

4.7.1.6 Disease, Parasites and Contaminants

Adult females and pups are likely the age-classes most vulnerable to disease and parasitism. Available serologic evidence does not support the possibility that a disease epidemic occurred during the Steller sea lion decline of the late 1970s and 1980s; however, due to sampling limitations the possibility can not be excluded completely. Although Steller sea lions have recently been exposed to several endemic disease agents that could potentially impede recovery in the future, 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 effect on Steller 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 Steller 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 Steller sea lion populations and to what extent this may have on reproductive capacity. Preliminary results indicate higher levels of stress hormones (i.e., haptoglobin) in the eastern DPS, where population densities are higher and high prevalence of hookworm parasites have been found (Rea *et al.* 2010).

Adult females and pups are likely the age-classes most vulnerable to toxic substances. Steller sea lions have shown relatively low levels of toxic substances as well as heavy metals and mercury, 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. Steller 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. Of recent concern is the potential for methyl-mercury impacts, but little is known about the individual-level response of a Steller sea lion to mercury loads. Holmes *et al.* (2008) studied mercury levels in tissues of eastern DPS and western DPS Steller sea lion pups; they reported that western SSLs had statistically higher mercury levels in kidney and liver tissues and lead in liver tissues than animals from the eastern DPS. Finally, pups in the western portion of the SSL range appear to have higher mercury and PCB levels than the eastern portion of their range (Castellini *et al.* 2009). Thus, overall, there is still some concern that toxic substances may have indirect impacts on individual vital rates, including reproductive potential.

4.7.2 Synthesis of the Likely Responses of Steller Sea Lions and their Habitat to Direct and Indirect Stressors

As listed above, several factors act as direct or top-down sources of Steller sea lion mortality; i.e., commercial harvest, intentional shooting, entanglements or incidental catch by fishing gear, disturbance, and predation. Direct sources of mortality were significant contributors to the Steller sea lion population declines observed prior to the 1990s, when there were relatively large reductions in juvenile survival rates, and smaller reductions for adults (Pascual and Adkison 1994, York 1994, Holmes and York 2003, Fay 2004). Since 1990, rates of mortality from harvests, shooting, entanglement, and incidental catch have been substantially reduced and likely have contributed to a rebound in both juvenile and adult survival rates (Holmes and York 2003, Fay 2004, Holmes *et al.* 2007). Subsistence harvests of Steller sea lions continue but have declined substantially and are not thought to be an important factor in the dynamics of this DPS.

As previously described, some hypothesize that predation by killer whales has the potential to be a significant additional top-down source of mortality (Williams *et al.* 2004, NRC 2003). 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 hypothesis, however, 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.* 2006c, DeMaster *et al.* 2006, Wade *et al.* 2009). Springer *et al.* (2008) provided responses to these critiques, and maintain their hypothesis is valid and probable. Analyses presented by Holmes and York (2003) is contradictory to top-down stressors in the region of Kodiak Island where killer whales are known to specialize on Steller sea lions, yet adult and juvenile survival rates are high. Although the NRC (2003) concluded that killer whale predation and top-down impacts were the likely driver for the decline of Steller sea lions, some current information contradicts that hypothesis, other data affirm it, and some data suggest that bottom-up factors at least in some areas may currently be more important. That is not to say that killer whale predation or shooting were not important in the past and may not be important in some areas currently. Historical data do not allow us to adequately evaluate the potential impacts of these various factors (NRC 2003, NMFS 2006a); thus in this Biological Opinion we have focused our concerns on current stressors.

Evidence that indirect or 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, Calkins *et al.* 1998), possible depressed late-term pregnancy rates (Pitcher *et al.* 1998), significantly reduced pregnancy rates for lactating females (Pitcher *et al.* 1998), and a decline in per capita natality of female Steller sea lions at some rookeries (Holmes and York 2003, Fay 2004, Winship and Trites 2006, Holmes *et al.* 2007). These responses by Steller sea lions are opposite to those predicted by direct, 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. These bottom-up factor(s) appeared to be affecting Steller sea lions as early as the 1960s and 1970s (see Section 3.1.14), at about the same time that large numbers of Steller sea lions were also killed directly (especially in the late 1970s and 1980s). The combination of reduced population abundance and poor body condition indices is consistent with a substantial reduction in carrying capacity (Gerrodette and DeMaster 1990, Calkins *et al.* 1998).

The changes in vital rates (see above) may have been a function of nutritional stress resulting from a combination of reduced prey availability and quality (Trites *et al.* 2006a). Two stressors were likely to have affected the prey field for Steller sea lions: (1) climate induced changes in the species composition, distribution or nutritional quality of Steller sea lion prey (see review by Trites and Donnelly 2003 and Trites *et al.* 2006a), and (2) fishery-induced changes in localized or overall prey abundance and quality (Braham *et al.* 1980, NMFS 1998a, 2000). Both climate change and fisheries induced changes in prey communities likely 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), and Boyd (2010) recently hypothesizes that the Steller sea lion throughout its range may be at carrying capacity. Yet what may have been unusual about the decline in Steller sea lions observed through 2000 is the introduction of large-scale commercial fisheries on Steller 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 (NMFS 1998, 2000, 2001). Overall, 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.* 2007) despite climate shifts to potentially more favorable environmental conditions that may have occurred in 1989 and 1998 (Hare and Mantua 2000, Bond *et al.* 2003).

In addition, changes in the overall energy density of the prey field due to both climate shifts and long term fisheries impacts, may have reduced the efficiency of Steller sea lions and affected their ability to obtain adequate energy to maintain body condition and full reproductive potential. In our review of climate and regime shifts, gadids were not necessarily affected across the range of Steller sea lions by the 1977 shift. Although it appears that EBS pollock did benefit from this change, GOA pollock and Atka mackerel likely were unaffected or affected in different ways that are still not clear. Results by Hennen (2006) correlate Steller sea lion declines with fisheries around rookeries in the 1980s, and find no correlation between fisheries and Steller sea lion dynamics in the 1990s after conservation measures were enacted around rookeries and shooting was prohibited[22] (Hennen 2006, Dillingham *et al.* 2006).

Both direct and indirect stressors can affect Steller sea lion population growth and vital rates. In addition, both types of stressors can operate simultaneously and at various levels. Steller sea lions have been affected by climate and 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 (Walker *et al.* 1999, Dixon 1986). 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 Steller sea lion population itself. Steller sea lions persisted in the North Pacific despite the adverse impact of these stressors, and they did so without an apparent loss of genetic diversity which would indicate that the population had gone through a "genetic bottleneck" (NMFS unpublished data). 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, human-related kills, and apparently maintained, on average, a relatively large population size (i.e., above the point that would have resulted in an obvious genetic bottleneck). This is not to say that the population did not go through historical changes in population size or distribution as reported by Nelson (1887) or similar changes for seabirds (Causey *et al.* 2005), but that it appears unlikely that rapid and large population increases and decreases were common for Steller sea lions.

In the last several decades, several stressors have developed as a result of human influence such as contaminants, incidental take, shooting, fisheries, and, potentially, global climate change[23] (NRC 2003, NMFS 2006). The absolute impact of each stressor on survival and reproduction during the Steller sea lion population decline are unknown. Yet, based on several PVAs, some argue that there is a significant probability that either a portion of the range of the western DPS of Steller sea lion may be extirpated (Winship and Trites 2006) or that the entire western DPS will go extinct in the next 100 years (York *et al.* 1996, Gerber and VanBlaricom 2001, NMFS 2008a; Appendix A); others argue that changes have been and will be far less dramatic (Boyd 2010). The eastern DPS is likely to continue to increase and appears to be large, healthy, and, based on Goodman (NMFS 2008a; Appendix A), is not in danger of extinction or likely to become endangered. Therefore, a working model of factors that may be contributing to the lack of a robust recovery of the western DPS of Steller sea lions, as well as the significant declines in abundance in the western, and in the western portion of the central Aleutian Island sub-areas, is as follows:

1) No one factor can explain the overall and local patterns in trends in abundance (Loughlin and York 2000, NRC 2003, NMFS 2008a, Atkinson *et al.* 2008).

2) While considerable scientific evidence is inconsistent with the nutritional stress hypothesis as one of the primary factors adversely impacting the recovery of this DPS, information on the pattern of decline in the reproductive rate and size at age of this population relative to the eastern DPS since the mid-1970s is consistent with the nutritional stress hypothesis. It should be noted that all of the available data that are inconsistent with the nutritional stress hypothesis are from areas east of the central Aleutian Islands. Therefore, nutritional stress cannot be dismissed as an important factor in understanding the dynamics of this population range wide.

---

[22] Numerous sea lion conservation measures were implemented throughout the 1990s, see Sections 4.4.1 and 4.5 for a thorough historical review.

[23] Global climate change is a highly debated theory, both on the mechanisms and results. See NMFS (2006a) for a short discussion of the topic. In this opinion we recognize the possibility of global climate change and the potential influence on sea lions and changing habitat and range. The southern contraction of the range of the eastern DPS may be in response to warming (see NMFS 2006a for this discussion).

i) Acute nutritional stress does not appear to be an important mechanism in understanding the dynamics of this population (Trites and Donnelly 2003, Trites *et al.* 2006a, NMFS 2008a, Atkinson *et al.* 2008).

ii) Chronic nutritional stress, if it is occurring, is a reasonable mechanism related to the lack of a robust population growth rate in this DPS.

iii) Environmental forcing undoubtedly changes the prey field for SSL over time. In some cases, these changes could be beneficial (i.e., increase the carrying capacity) and in other cases, these changes could decrease the carrying capacity. Carrying capacity may differ markedly from one subarea to another; new studies suggest oceanographic conditions in some subareas may differ sufficiently to provide less favorable habitat for producing SSL prey (Lander *et al.* 2010). The information needed to ascertain which environmental conditions improve the prey field and which environmental conditions degrade the prey field is not available.

iv) It is possible that commercial fisheries have adversely impacted, and in the future could continue to adversely impact, the prey field of Steller sea lions, which could contribute to the conditions that support a poor prey field (i.e., chronic nutritional stress). Recent studies, however, show very inconclusive relationships between fishery removals of prey and SSL sub-population growth (e.g., Trites *et al.* 2010). Thus, it is likely that these conditions vary geographically and temporally within the range of the western DPS of Steller sea lion (NMFS 2001, NMFS 2003).

3) Information necessary to completely dismiss the hypothesis that contamination or disease are important factors in understanding the dynamics of this population is not available (NRC 2003, NMFS 2008a, Atkinson *et al.* 2008). According to NMFS (2008) disease is less likely to be a significant factor than are contaminants.

4) While the information necessary to confirm the hypothesis that human caused-mortality was one of the primary drivers of the steep decline in abundance in the 1980s does not exist, there is a general consensus among experts that this is or could be the case (NRC 2003, Hennen 2006, NMFS 2008a, Atkinson *et al.* 2008, Kruse and Huntington 2009).

5) Predation by killer whales is likely to be an important factor in understanding the dynamics of Steller sea lions in some of the sub-areas (Horning and Mellish 2009, 2010a, 2010b), and in particular those sub-areas that have relatively small numbers of SSL (NRC 2003, Guinette *et al.* 2010, Durban *et al.* 2010). Nonetheless, there is compelling evidence to seriously question the hypothesis that killer whale predation was the primary factor driving the overall decline of this DPS in the 1980s and 1990s (see Springer *et al.*, 2003, Trites *et al.* 2006, Springer *et al.* 2008, Wade *et al.* 2008).

6) Fishing has occurred in the action area for decades, starting in the 1960s with very large catches in thriving foreign groundfish fisheries throughout the BSAI and GOA (Ketchen, 1968, Buck 1973), on the order of catch levels in recent decades. Thus, the action area has been a fished ecosystem before, during, and after the SSL decline, and continues in this state today while the overall western DPS begins an apparent rebound. Uncoupling commercial fisheries in the action area from the multi-faceted stressors likely acting on the western DPS is not possible to attain with much clarity. However, in Chapter 5, fishing patterns are examined in light of trends in SSL subpopulation growth rates and other vital rates to assess potential effects of the groundfish fisheries on the persistence and recovery of the western DPS of Steller sea lions.

4.8 Human Impacts Affecting Humpback Whale Status

4.8.1 Effects of Historic Whaling

The worldwide population of humpback whales was thought to have been in excess of 125,000 animals prior to commercial whaling (NMFS 1991). Approximately 15,000 animals were believed to have been present in the North Pacific prior to 1905, and intensive commercial whaling during the 20th century may have reduced this population to as few as 1,000 before it was placed under international protection by the International Whaling Commission (IWC) in 1965. This estimate likely underestimates the actual kill as a result of the under-reporting of Soviet catches (Yablokov 1994).

Whaling was considered the primary threat to the worldwide populations of humpback whales when the species was placed under the protection of the IWC. At present, commercial whaling is not considered a significant threat to this species, although some illegal Japanese whaling continues to occur. It is not known currently how many individuals are killed on an annual basis in these commercial harvesting operations.

4.8.2 Direct Effects of Commercial Fisheries on Humpback Whales

Humpback whales are killed incidentally in federal groundfish and longline fisheries and State managed commercial salmon fisheries. The primary source for data on incidental mortalities of humpback whales in commercial fisheries is from the North Pacific Groundfish Observer Program database (NMFS GOP) and Alaska Marine Mammal Observer Program. The incidental mortalities included here are summarized from Angliss and Allen 2009.

4.8.2.1 Central North Pacific Population

Until 2004, there were four different federally-regulated commercial fisheries in Alaska that occurred within the range of the central North Pacific humpback whale stock that were monitored for incidental mortality by fishery observers: BSAI groundfish trawl, GOA groundfish trawl, longline, and pot fisheries. Average annual mortality from the observed fisheries was 1.5 (CV = 0.47) humpback whales from this population. As of 2004, changes in fishery definitions in the List of Fisheries have resulted in separating these four fisheries into 17 fisheries (69 FR 70094, 2 December 2004). This change does not represent a change in fishing effort, but provides managers with better information on the component of each fishery that is responsible for the incidental serious injury or mortality of marine mammal stocks in Alaska. Between 2002 and 2006, there were incidental serious injuries and mortalities of Central North Pacific humpback whales in the Bering Sea/Aleutian Islands sablefish pot fishery (Table 4.9). Estimates of marine mammal serious injury/mortality in observed fisheries are provided in Perez (2006) and Perez (unpubl. ms.).

Reports of entangled humpback whales found swimming, floating, or stranded with fishing gear attached occur in both Alaskan and Hawaiian waters. During the 5-year period from 2001 to 2005, there were 54 reports of human-related mortalities or serious injuries. Of these, there were 40 incidents which involved commercial fishing gear, and 15 of those incidents involved serious injuries or mortalities. These estimates should be considered a minimum. No observers have been assigned to several fisheries that are known to interact with this stock, making the estimated mortality rate unreliable. Further, due to limited Canadian observer program data, mortality incidental to Canadian commercial fisheries (i.e., those similar to U.S. fisheries known to interact with humpback whales) is uncertain. Though interactions are thought to be minimal, data regarding the level of humpback whale mortality related to commercial fisheries in northern British Columbia are not available, again indicating that the estimated mortality incidental to commercial fisheries is underestimated for this stock (Angliss and Allen 2009).

4.8.2.2 Western North Pacific Population

Within the range of the western North Pacific humpback whale population, there were six different federally-regulated commercial fisheries in Alaska that were monitored for incidental mortality by fishery observers until 2004. At that time, changes in fishery definitions in the List of Fisheries have resulted in separating these six fisheries into 22 fisheries (69 FR 70094, 2 December 2004). Estimates of marine mammal serious injury/mortality in each of these observed fisheries are provided in Perez (2006) and Perez (unpubl. ms.). Between 2002 and 2006, there were incidental serious injuries and mortalities of Western North Pacific humpback whales in the Bering Sea/Aleutian Islands sablefish pot fishery (Table 4.9). Average annual mortality from observed fisheries was 0.20 humpbacks from this stock (Table 4.9). Note, however, that the stock identification is uncertain and the mortality may have involved a whale from the Central North Pacific stock of humpback whales. Thus, this mortality is assigned to both the Central and Western stocks.

The estimated annual mortality rate incidental to U. S. commercial fisheries is 0.2 whales per year from this stock based on 0.2 from observed fisheries. However, this estimate is considered a minimum because there are no data concerning fishery-related mortalities in Japanese, Russian, or international waters. In addition, there is a small probability that fishery interactions discussed in the assessment for the Central North Pacific stock may have involved animals from this stock because the only known matches to feeding areas come from areas typically used by the Central North Pacific stock.

No reports of entangled humpback whales from the western North Pacific population were reported through fishery observer programs between 2001 and 2005; however, observing effort in western Alaska is low (Angliss and Allen 2009). However, one humpback whale in western Alaska was reported to the NMFS Alaska Stranding Program during this time frame (2002) as entangled in black cod gear.

The western population of humpback whales may also be impacted by takes in other countries' fisheries. Brownell *et al.* (2000) compiled records of bycatch of humpback whales in Japanese and Korean commercial fisheries between 1993 and 2000 and found that, during 1995-99, six humpback whales were taken as bycatch. In addition, two strandings were reported by the Japanese or Korean during this period. Analysis of four samples from meat found in Japanese markets indicated that humpback whales are being sold (Angliss and Allen 2009). Where these animals are taken, however, and whether they are taken intentionally or as bycatch, is unknown.

4.8.3 Entanglements

In recent years, an increasing number of entangled humpback whales have been reported to NMFS Alaska Region stranding program. One hundred eighteen humpback whales were reported (ninety six confirmed) entangled in Alaska from 1997-2009; the majority of these involved southeast Alaska humpbacks (NMFS Alaska Region Stranding Data 2010)(Table 4.10). In 2005, twenty-one entangled humpback whales were reported to the NMFS Alaska stranding program, sixteen of those were confirmed. Nine of these were reported in southeast Alaska, and seven in southcentral Alaska in the Kodiak, Homer, and Seward regions. There were nineteen entanglements reported (sixteen of those confirmed) in 2006, nine reported entanglements (eight confirmed) in 2007, seven confirmed reports in 2008, and nine entanglements reported (five confirmed) in 2009. For some of these reports, it remains unclear whether they represent distinct events or re-sights of the same entangled animal.

For many of these reports, it is not possible to identify the gear involved in the entanglement to a specific fishery. This is based on a general lack of data in reports received, the difficulty in accurately describing gear at a distance, and the fact that most entanglements are not re-sighted for follow-up analysis. The majority of gear reported in Alaska humpback whale entanglements involves crab, shrimp, and unidentified pot gear (Figure 4.37, Table 4.10). Some of the unidentified pot gear incidents may involve groundfish fisheries, such as pot cod. In addition, several Alaska humpback whale entanglements reported to the stranding network have involved longline gear and could be attributed to the sablefish groundfish fishery under analysis in this opinion.

To understand more about the prevalence of these entanglement incidents, a study in 2003 and 2004 documented entanglement scarring in the humpback population in northern southeast Alaska. Using methodology developed in the Gulf of Maine to investigate scarring in Atlantic large whales, Neilson *et al.* (2005) photographed the caudal peduncle of individual humpbacks as they dove and examined them for scars indicative of previous entanglement. Their results indicate that, based on caudal peduncle scarring, 71% (95% CI: 62%-78%) of the humpback whales in northern southeast Alaska have been entangled at least once. The study also found that eight percent of the whales photographed in Icy Strait/Glacier Bay acquired new entanglement scars between the two years that they were sampled. Calves were less likely to have entanglement scars than older whales, and there was no significant difference in scarring percentages between males and females. Overall, the percentage of whales with entanglement scars in northern southeast Alaska is comparable to Gulf of Maine humpback whales (48%-65% entanglement percentage). Based on similar scarring investigations carried out in Hawaii, 14% of the humpbacks there appear to have been entangled (Robbins and Mattila 2004).

For entanglements that do not result in immediate or discernable mortality, it is difficult to determine the extent of impact to the animal. Most entangled whales reported to the Marine Mammal Stranding Network in Alaska are not re-sighted. Without further information, it is unclear which types of entanglements are ultimately life-threatening. Data such as that collected by Neilson *et al.* (2005), however, leads to the conclusion that many humpback whales survive their entanglements. Some, it would appear, survive multiple entanglement incidents (NMFS 2006d).

In recent years, the NMFS Alaska Region has partnered with NOAA's Hawaiian Humpback Whale National Marine Sanctuary to develop a response program for entangled large whales in Alaska. Numerous teams throughout coastal areas of the state have been trained in first response activities, and workshops have been held with fishermen regarding measures that can be taken to prevent entanglements. Caches of disentanglement gear have been placed with trained responders in various communities throughout Alaska, including Petersburg, Juneau, Sitka, Tenakee, Gustavus, Seward, Homer and Kodiak. Educational forums have been held for members of the public to an effort to increase accurate reporting, and encourage mariners to stand-by entangled animals until a response team can be mobilized.

At the time of this opinion, the number of entanglements that might result in serious injury or mortality for humpback whales is not known to be at a level to have population level effects for the species. While a number of humpback whales have been reported entangled in fishing gear in Alaska in recent years, it is difficult to quantify the impact relative to a specific fishery and to the whales themselves because of insufficient information obtained from these events.

The number of confirmed reports of entangled Central North Pacific humpback whales in Hawaiian waters has increased in recent years (Table 4.11). Many of the whales reported entangled in Hawaiian waters most likely brought the gear with them from higher latitude feeding grounds. While the whales are not typically at risk from drowning or immediate death, they are at increased risk of starvation, infection, physical trauma from the gear, and ship strikes as a result of the entanglement. Since 2002, the Hawaiian Islands Humpback Whale National Marine Sanctuary and NMFS have worked together to improve outreach by advertising response capabilities and creating an emergency hotline number (NMFS 2006d).

4.8.4 Impacts from Ship Strikes

A compilation of available records of vessel collisions with large whales from 1975 to 2002 indicated that humpback whales are one of the most frequently hit cetacean species worldwide (Jensen and Silber 2003). Of the database containing 292 records, humpback whales (44 records) were the second most commonly hit species after fin whales (75 records). In Alaska, opportunistic reports of vessel collisions with humpback whales since 1986 have shown an average of one to two humpback whales struck per year. This is a minimum estimate, as not all whales struck are reported and not all whales struck are identified to species or cause of mortality. The fate of struck animals is also not always determined unless the whale dies immediately upon impact or is discovered as a carcass on the bow of a ship and it can be determined that the strike was the cause of death.

Humpback whale distribution overlaps significantly with the transit routes of large commercial vessels that ply the waters off Alaska. The larger vessels are cruise ships, large tug and barge transport vessels, and oil transport tankers. Cruise ships frequent the inside waters of southeast Alaska, passing through areas used by humpback whales for feeding, such as Glacier Bay National Park and Preserve, Point Adolphus and, adjacent to the action area, the waters of Lynn Canal en route to Skagway and Haines. Tug and barge transport follows much of the traffic pattern of the cruise ships, as they frequent the same coastal communities. Oil transport tankers are generally operating farther offshore where there are presumably fewer concentrations of humpback whales, except for transit through Prince William Sound. Collisions in Alaska can occur throughout the region, peaking during the summer season.

Generally, there is a direct relationship between the occurrence of a whale strike and the speed of the vessel involved in the collision. Most collisions that have killed or severely injured whales involved vessels greater than 80 meters in length traveling at speeds in excess of 13 knots (Laist *et al.* 2001). In Jensen and Silber (2003), vessel speed at the time of strike was reported for 58 (19.8%) of the 292 cases. Operating speeds of vessels that struck various species of large whale ranged from 2 to 51 knots with an average speed of 18.1 knots. The average speed resulting in injury or mortality to the whale was 18.6 knots. In Alaska, records show that vessels have struck humpback whales at a range of speeds, from a skiff traveling at 29 knots to vessels drifting or idling (NMFS unpublished data, see Table 3.65). These records indicate that two incidents with associated vessel speeds resulted in death to the animal. One of these fatal collisions occurred in southeast Alaska with a cruise ship traveling at 19 knots; the other, with a container ship traveling at 12-19 knots reported from Anchorage, Alaska.

4.8.4.1 Humpback Whale Ship Strikes in Alaska

Although there is no official reporting system for ship strikes, numerous incidents of vessel collisions have been documented in Alaska. Fifty-eight reports from 1997 to 2009 representing confirmed, unconfirmed and suspected ship strikes with humpback whales have been reported to the Alaska Stranding Network (Table 4.12). This is a minimum estimate, as not all whales struck are reported and not all whales struck are identified to species or cause of mortality. The fate of struck animals is also not always determined unless the whale dies immediately upon impact or is discovered as a carcass on the bow of a ship and it can be determined that the strike was the cause of death.

Records of vessel collisions with large whales in Alaska indicate that strikes have involved cruise ships, recreational cruisers, whale watching catamarans, fishing vessels, and skiffs. Vessel lengths associated with these records ranged from approximately 20 feet to over 250 feet, indicating that all types and sizes of watercraft pose a threat of collision for whales (Jensen and Silber 2003). Cruise ships are of particular concern, as they operate at considerably high speeds and frequent the inside waters of southeast Alaska with routes passing through areas of humpback whale abundance such as Glacier Bay National Park and

Preserve, Point Adolphus and Lynn Canal. In addition to large ships, which are most likely to cause significant injury or death to humpback whales, smaller tour, charter and private vessels also significantly overlap with inshore humpback whale distribution in Alaska waters and also have the potential to cause disturbance, serious injury, and possibly mortality.

Some incidents of vessel collisions with humpback whales are known to result in mortality. Between 2001 and 2009, confirmed reports of vessel collisions with humpback whales indicated an average of five humpback whales struck per year in Alaska; between 2005 and 2009, two humpback deaths were attributed to ship strikes. In 2001, a dead and pregnant humpback whale was discovered in Glacier Bay, and a necropsy determined the whale likely had been killed by blunt trauma, possibly from a large vessel collision. In 2003, a humpback whale was necropsied that had been first seen at Pt. Manby, Yakutat Bay. The results of that necropsy indicated that the whale had been killed by blunt trauma as a result of large vessel collision. In 2004, a humpback whale calf in Glacier Bay was necropsied on Strawberry Island. Severe dislocation of six ribs caused massive bleeding and tissue damage; blunt trauma indicated injury consistent with vessel collision. A second incident in 2004 involved a humpback (nursing calf) necropsied on the south end of Douglas Island outside of Juneau. Results of this necropsy showed a severe scapular fracture and again indicated likely collision with a vessel based on blunt trauma to the animal. Incidents in 2005 and 2007 also indicated cases where humpback whales likely died due to the impact from a vessel based on necropsy findings.

Vessel collisions with humpback whales remains a significant management concern, given the increasing abundance of humpback whales foraging in Alaska, as well as the growing presence of marine-based tourism in Alaska's coastal waters. Based on these factors, injury and mortality of humpback whales as a result of vessel strike may likely continue into the future (NMFS 2006a).

4.8.4.2 Humpback Whale Ship Strikes in Hawaii

Central North Pacific humpback whales, especially calves and juveniles, are highly vulnerable to ship strikes and other interactions with non-fishing vessels while in Hawaii. Younger whales spend more time at the surface, are less visible and closer to shore (Herman *et al.* 1980; Mobley, Jr. *et al.* 1999), thereby making them more susceptible to collisions. There appears to be an increased frequency at which collisions with humpback whales and vessels are occurring in Hawaiian waters (Table 4.11), especially in the shallow waters (less than 100 fathoms) of the four-island region of Maui county and Penguin Banks, the preferred habitat by the whales wintering in Hawaii (Lammers *et al.* 2003). Three types of collisions reports were documented: collisions with little/no forewarning; collisions resulting from effort to avoid whales; circumstantial collisions not reported but evidence of trauma known. The majority of the collisions are with boats from 19-80 ft in length, including both slow and fast moving vessels. Also, the highest incidents of collisions were documented from the island of Maui, and the lowest number documented was from the island of Kaua`i.

The increasing rate of whale and vessel collisions may have a number of contributing factors, the most important of which may be that the population of humpback whales in Hawaii is increasing (Lammers *et al.* 2003). In addition, there is a corresponding rise in the number of vessels in the preferred habitat for humpback whales, a direct result of the growing popularity of eco-tourism in Maui and the surrounding areas. Efforts to reduce these interactions include improved technological research into mapping models and radar and sonar detection systems, state regulations prohibiting parasailing and personal watercrafts in Maui waters during whale season, and a NOAA hot line to report humpback whale interactions.

4.8.5 Impact from Disturbance

The whalewatch industry in Alaska has developed over the last decade and is primarily directed toward humpback whales and killer whales. As the population of humpback whales has increased, the whalewatch industry in Alaska likewise has grown along with this increasing presence. The industry has experienced particular growth in the last five years due to the popularity of Alaska as a tourist destination and an increasing public desire to view marine mammals in the wild. Vessel class types engaged in whalewatching include kayaks, zodiac inflatables, jet boats, catamarans, yachts, mid-size expedition vessels, and large cruise ships. Sport-fishing charters and recreational boaters (both locals and visitors) also widely participate in whalewatching activities.

Numerous incidents of vessel interactions with humpback whales have been documented in Alaska. There have been several incidents that included close approaches and possible harassment by vessels of different vessel classes including kayaks, cruise ships and catamarans. In 2007, fines were levied for two incidents with humpback whales which occurred in Alaska, one involving a cruise ship collision, the other harassment by a whalewatch jet boat. At present, it is difficult to measure the extent of potential disturbance impacts to whale populations from this industry.

To minimize the potential for harassment and the possibility of collision, NMFS implemented regulations on July 2, 2001 that imposed vessel restrictions on approaching humpback whales closer than 100 yards in Alaska. Operating at a “slow, safe speed” when near humpback whales was also required. The National Park Service has implemented even greater minimum approach distances in Glacier Bay National Park (1/4 mile in all Park waters) for humpback whales, which likely reduces the whales’ underwater noise exposure and potential for behavioral disturbance. In addition, the Park has passed vessel management measures that allow speed restrictions of 13 knots to be imposed by Park management on an as-warranted basis in the bay.

In addition to the approach regulations, marine mammal viewing guidelines strive to mitigate disturbance impacts. NOAA Fisheries guidelines include recommendations to limit time spent with individuals, avoid trapping or encircling animals, avoid sudden changes in speed, and minimize the use of outdoor PA systems in the vicinity of whales.

4.8.6 Subsistence Harvest

Although the harvest of humpback whales is not restricted for the indigenous people of Alaska, humpback whales are not harvested by Alaska Natives.

4.9 Human Impacts Affecting Sperm Whale Status

4.9.1 Commercial Whaling

The population of sperm whales in the Pacific was likely well below pre-whaling levels before modern whaling became especially intense in the late 1940s (Reeves and Whitehead 1997). A total of 258,000 sperm whales were reported to have been taken by commercial whalers operating in the North Pacific between 1947 and 1987 (C. Allison, pers. comm., International Whaling Commission, United Kingdom, in NMFS 2006a). This value underestimates the actual kill in the North Pacific as a result of under-reporting by U.S.S.R. pelagic whaling operations, which are estimated to have under-reported catches during 1949-71 by 60% (Brownell *et al.* 1998). In addition, new information suggests that Japanese land based whaling operations also under-reported sperm whale catches during the post-World War II era (Kasuya 1999). The Japanese officially stopped catching sperm whales in the North Pacific in 1988

(Reeves and Whitehead 1997), but as part of their 2005 research program, they planned to take 10 sperm whale in the western North Pacific (details at the International Whaling Commission website at http://www.iwcoffice.org/conservation/permits.htm). Since 2000, Japan has harvested an average of 6 sperm whales per year in the western North Pacific. It is not known if the sperm whales taken in the western North Pacific may also occur in Alaskan waters (NMFS 2006a). The implications of this action for the status and trend of sperm whales are uncertain.

4.9.2 Direct Effects of Commercial Fisheries on Sperm Whales

The vulnerability of sperm whales to incidental capture in fishing gear, especially gillnets set in deep water for pelagic fish (e.g., sharks, billfish, tuna) and bottom-set longline gear, is well documented (Di Natale and Notarbartolo di Sciara 1994, Haase and Félix 1994, Félix *et al.* 1997, Hill *et al.* 1999, Straley *et al.* 2005, Warner *et al.* 2005). In U.S. Pacific waters, drift gillnet operations incidentally killed or seriously injured an average of 9 sperm whales per year from 1991-95 (Barlow *et al.* 1997).

In 2004, the definitions of commercial fisheries in Alaska were changed to reflect target species: these new definitions have resulted in the identification of 22 observed fisheries in the Gulf of Alaska and Bering Sea that use trawl, longline, or pot gear (69 FR 70094, 2 December 2004). Of these, there is at least one fishery that has incurred incidental serious injuries or mortalities of sperm whales, the Gulf of Alaska sablefish longline fishery. Between 2002 and 2006, there were three observed serious injuries of sperm whales in the Gulf of Alaska this fishery (Table 4.13). Each animal was designated as seriously injured because it became caught in the gear, and was released alive with trailing gear. However, since 2001 there have been no observed mortalities in federally observed Alaska fisheries and therefore the minimum estimated annual mortality rate incidental to U.S. commercial fisheries is zero (Angliss and Allen 2009).

4.9.3 Entanglements

During 1997, the first entanglement of a sperm whale in Alaska’s longline fishery was recorded, although the animal was not seriously injured (Hill and DeMaster 1998). The available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions, although the nature and extent of interactions between sperm whales and longline gear is not yet clear.

Sperm whales may break through or carry away fishing gear. Whales carrying gear may die at a later time due to trailing fishing gear, become debilitated or seriously injured, or have normal functions impaired, but with no evidence of the incident recorded. Sperm whales may also become entangled while attempting to depredate fish off fishing gear (NMFS 2006b).

4.9.4 Depredation

Sperm whales have learned to depredate deep-water sablefish longlines in the waters of the eastern North Pacific (Hill *et al.* 1999, Straley *et al.* 2005). Reports of depredation were first noted in 1978 and have steadily increased in frequency and severity, with a notable increase since the late 1990s. Based on information documented from 1999-2003 (observer data), one sperm whale was observed with trailing gear from the Gulf of Alaska sablefish longline fishery. Estimated mean annual mortality for the North Pacific Stock (Alaska) is 0.45 (CV=0.75) (Angliss and Outlaw 2005). During annual longline surveys conducted between 1989-2003, 38 of the surveyed stations recorded sperm whale predation on catch; all predation events occurred in the Gulf of Alaska, none in the Bering Sea (Angliss and Allen 2009).

Male sperm whales are known to be attracted to groundfish fishing activities. In the GOA, sperm whales have been observed feeding off demersal longline gear targeting halibut and sablefish since the mid-

1970's (NMFS 2006b, Straley *et al.* 2005). A quota system implemented in 1995 reduced overall fishing effort but expanded the season to eight months, which resulted in greater depredation rates. However, no sperm whales were reported to have been seriously injured in these fishing activities (NMFS 2006a).

Approximately 90 male sperm whales are believed to participate in this activity at present. The whales appear to have become more attracted to these vessels in recent years as reliable and easy sources of food. The interaction with commercial longline gear may have an adverse impact on individual whales due to the potential for entanglement; however, researchers have observed that sperm whales predating on longline gear appear to be able to avoid becoming entangled (Jan Straley, UAF, personal communication, March 13, 2006, in NMFS 2006a). No mortalities have been observed.

Investigations conducted in 2003-2004 showed that sperm whales were present near the fishing vessels one-third of the time and 65% of the hauls had evidence of depredation. Based on their proximity to vessels, 40 photo-identified individuals were believed to participate in depredation activity. Mark-recapture estimated a total of 127 sperm whales present in the study area at large. Generally, between one and seven highly vocal animals were observed near a fishing vessel hauling catch. Sperm whales were seen less frequently during the soak and set portion of the fishing activity. When no vessels were nearby, whales were seen diving along the shelf edge, indicating they were also engaging in normal foraging behavior (Straley *et al.* 2005). Research in the eastern Gulf of Alaska is ongoing to develop deterrents to predation by sperm whales on sablefish longlines (Straley *et al.* 2005) which may reduce potential for entanglement.

In 1997 and 2000, one sperm whale in each year was observed entangled and seriously injured by longline gear in statistical area 640 in the eastern portion of the GOA (Perez 2003). The fishery incidental take for sperm whale is based on observing these single animals, and therefore, the estimated take of sperm whale is very small. Based on the 2004 stock assessment, estimated annual rate of human-caused mortality and serious injury appears minimal for the sperm whale stock (Angliss and Lodge 2004).

Sperm whale depredation in longline fishing operations off southeast Alaska has been increasing since 1995, with at least 40 individual sperm whales documented in these encounters (Tiemann *et al.* 2005, Thode *et al.* 2005). In 2002, the Southeast Alaska Sperm Whale Avoidance Project (SEASWAP) began studying this behavior, and in 2004 began using passive acoustics to monitor sperm whale behavior around longlines. Tracks of whale activity were created from hydrophone data; typical foraging depth was found to be 200-400 m, 50 m or more from the ocean bottom. SEASWAP findings based on tracking sperm whale acoustic activity indicate that when fishing vessels are absent, sperm whales are foraging at mid-water depth (~250-500 m) with dive durations similar to those observed elsewhere. However, animals around fishing vessels had much shorter dive cycles and appeared to remain in shallower water (~50 m). Characteristics of their acoustics also changed in the presence of fishing vessels. Evidence indicates that acoustic cues from longline vessels hauling their catch attract sperm whales whenever the animals are within 10 nm of the activity (Thode *et al.* 2005).

In collaboration with fishermen, studies have been conducted in the Sitka area involving genetics, acoustics and fishing behaviors to gain insight into what may attract sperm whales to longlining activity (Sigler *et al.* 2003, Straley *et al.* 2005). Preliminary analyses found that during a typical encounter when sperm whales are present during the haul, about 3%-6% of the catch was estimated to be removed, but sometimes over 50% of the catch has been lost by individual fishermen. As the frequency of depredation events increases, there are growing concerns about the potential for sperm whale entanglements and the prospect of growing economic losses.

4.9.5 Sperm Whale Ship Strikes

Sperm whales spend long periods (typically up to 10 minutes; Jacquet *et al.* 1998) “rafting” at the surface between deep dives. This could make them exceptionally vulnerable to ship strikes. Berzin (1972) noted that there were “many” reports of sperm whales of different age classes being struck by vessels, including passenger ships and tug boats. There were also instances in which sperm whales approached vessels too closely and were cut by the propellers (NMFS 2006b). Two whales described as “possibly sperm whales” are known to have died in U.S. waters in 1990 after being struck by vessels (Barlow *et al.* 1997).

Jensen and Silber’s (2003) large whale ship strike database, which details global reports of vessel collisions with whales, contains 17 sperm whale ship strike records. Sperm whales represent the sixth most commonly reported species in the compilation of records, behind fin, humpback, gray and minke whales. Vessel collisions with sperm whales are recorded from the east coast of the United States and Canada, the Marquesas, the Canary Islands, the Caribbean Sea, the coast of Italy, California, and Alaska. The Alaskan vessel collision incident occurred in 1997, 60 miles southwest of Middleton Island outside of Prince William Sound. A fishing vessel was underway at six knots when the whale surfaced and was hit on the tail. According to the captain, the animal appeared unharmed and resumed feeding.

Although little recent documentation is available from the eastern North Pacific, this lack of evidence should not lead to the assumption that no mortality or injury from collisions with vessels occurs. Carcasses that do not drift ashore may go unreported, and those that do strand may show no obvious signs of having been struck by a ship.

4.9.6 Impacts from Disturbance

Concern about the effects of whale-watching vessels prompted field studies of sperm whale responses to boat approaches in New Zealand (Gordon *et al.* 1992). The results suggest that sperm whales adjusted their diving and acoustic behavior to the whale-watching boats, but also that with frequent exposure, whales become increasingly tolerant of the vessels’ presence. Sperm whales are not often seen from whale-watching vessels in Alaska (either because the vessels are not located in areas where sperm whales are typically found, or the vessels are disruptive and the sperm whales avoid them) and the potential for disturbance to sperm whales by such vessels is probably low (NMFS 2006b).

4.9.7 Strandings

Thirty-one sperm whale incidents were reported to NMFS Alaska marine mammal stranding program between 1976-2009 (Table 4.14). The cause of death for most of these cases is unknown.

4.9.8 Subsistence Harvest

Sperm whales have never been reported to be taken by subsistence hunters (Rice 1989).

4.10 Human Impacts Affecting Fin Whale Status

4.10.1 Fishery Entanglements[24]

Entanglement in fishing gear can pose a threat to fin whales. About 73 balaenopterids were killed per year in the southern California offshore drift gillnet fishery during the 1980s (Heyning and Lewis 1990). Some

---

[24] Source: NMFS 2010

of these may have been fin whales and some of them sei (*B. borealis*) whales. According to Barlow *et al.* (1997), fin whales in particular may also be taken in the drift gillnet fisheries for sharks and swordfish along the Pacific coast of Baja California, Mexico. Some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net; however, fishermen report that large whales usually swim through nets without entangling and with very little damage to the nets. However, there have been no reports of entangled fin whales to date in the EEZ offshore of Alaska.

4.10.2 Non-fishery Vessel Interactions in Alaska

Laist *et al.* (2001) compiled information available worldwide regarding documented collisions between ships and large whales (baleen whales and sperm whale) and found that fin whales were struck most frequently. In some areas studied, one-third of all fin whale strandings appeared to involve ship strikes. Fin whales are occasionally injured or killed by ship strikes off the east coast (Waring *et al.* 1997) and west coast of the United States. At least one, and probably more, fin whales were killed by collisions with ships off California in the early 1990s (Barlow *et al.* 1997). More recently, three fin whales were documented as killed due to ship strikes off California; one in 1997 and two between the period 2000-2005 (Carretta *et al.* 2009; California Marine Mammal Stranding Network Database 2009). Four fin whales were struck off the Northwest coast of the United States; three were identified in Washington and one was identified in Oregon (S. Norman, NMFS, pers. comm., 2006). Additional mortality from ship strikes probably goes unreported because the whales do not strand, or if they do, they do not always have obvious signs of trauma (Carretta *et al.* 2009).

The NMFS Alaska Region stranding data have twelve reports of fin whale mortalities dating back to 1981; for most of these cases, cause of death is unknown. In 2006, a fin whale was discovered on the bulbous bow of a cruise ship arriving in the port of Seward; a necropsy revealing fractured ribs and hemorrhage indicated that ship strike was the likely cause of death. The death of a fin whale necropsied on Kodiak Island in 2010 was attributed to ship strike based on skull fracturing and bruising. A non-lethal strike was reported in 2000, when a U.S. Coast Guard cutter traveling at 17 knots struck a fin whale in Uyak Bay, Kodiak; the animal exhibited no signs of injury or distress during immediate post-contact observations. Based on the one mortality between the 2002-2006 five year period, the minimum mean annual mortality/serious injury from ship strikes is 0.20 fin whale/yr in Alaska (Angliss and Allen 2009).

4.10.3 Subsistence Harvest

Subsistence hunters in Alaska and Russia have not been reported to take fin whales from this stock.

4.10.4 Other Serious Injury or Mortality

As early as the mid-seventeenth century, the Japanese were capturing fin, blue, and other large whales using a fairly primitive open-water netting technique (Tønnessen and Johnsen 1982). After blue whales were depleted in most areas, the smaller fin whale became the focus of whaling operations and more than 700,000 fin whales were landed in the twentieth century (NMFS 2006a). Between 1925 and 1975, 47,645 fin whales were reported killed throughout the North Pacific (International Whaling Commission, BIWS catch data, February 2003 version, unpublished), although newly revealed information about illegal Soviet catches indicates that the Soviets over-reported catches of about 1,200 fin whales, presumably to hide catches of other protected species (Doroshenko 2000). There are no reports of direct human-related injuries or mortalities to fin whales in Alaska waters included in the Alaska Region stranding database for 2001-2005 (NMFS unpublished data).

4.10.5 Incidental Mortality & Serious Injury in Commercial Fisheries

4.10.5.1 Impacts of Fisheries in Alaska

Until 2004, there were six different federally-regulated commercial fisheries in Alaska that occurred within the range of the Northeastern Pacific fin whale stock that were monitored for incidental mortality by fishery observers. As of 2004, changes in fishery definitions in the List of Fisheries have resulted in separating these six fisheries into 22 fisheries (69 FR 70094, 2 December 2004). This change does not represent a change in fishing effort, but provides managers with better information on the component of each fishery that is responsible for the incidental serious injury or mortality of marine mammal stocks in Alaska.

Prior to 1999, there were no observed or reported mortalities of fin whales incidental to commercial fishing operations within the range of the Northeastern Pacific fin whale stock. However, in 1999, one fin whale was killed incidental to the Gulf of Alaska pollock trawl fishery. This take occurred in federal waters of statistical area 620, southwest of Kodiak Island (Perez 2003). Between 2002 and 2006, there was one observed incidental mortality of a fin whale in the Bering Sea/Aleutian Island pollock trawl fishery, resulting in a mean annual mortality/serious injury of 0.23 fin whale/yr incidental to commercial fisheries in Alaska for this 5 yr period (Angliss and Allen 2009).

4.10.5.2 Impacts of Fisheries outside of the US EEZ

No information is available addressing incidental catch in fisheries in Russia or Japan for NEP fin whales.

5 EFFECTS OF THE ACTION

The action addressed in this Biological Opinion is ongoing, and has been the subject of several preceding opinions (NMFS 2000, 2001, 2003). The actions are the (a) authorization of groundfish fisheries in the BSAI under the FMP for Groundfish of the BSAI, (2) authorization of groundfish fisheries in the GOA under the FMP for Groundfish of the GOA, and (3) authorization of parallel fisheries within State-managed waters. The deconstruction of the action in Chapter 2 yields four main groups of interrelated activities, which comprise the proposed action:

- Fisheries management policy
- Exploitation strategy
- Annual fisheries assessment
- Implementation of the fisheries

The action has generally been described and previously assessed as consisting of two sub-parts; a programmatic-level action by which the overarching rules for the fishery are established and the global operational factors that provide for the broad guidelines by which the fisheries operate (i.e., fisheries management policy, exploitation strategy and annual fisheries assessment), and the project-level action involving the implementation of the fisheries.

The groundfish fisheries off Alaska are continuing actions. Most of these actions were implemented from about 1999 to 2002. As such, resulting biological consequences are largely described in the Environmental Baseline. When NMFS developed the Environmental Baseline (Chapter 4), we assessed the base condition of the listed species and the populations those species comprise. Thus, this exposure and response analysis will rely extensively on effects described in the Environmental Baseline (see summary in Section 4.7).

In this chapter, we review effects of that pre-existing regime, extend them into the future, and assess effects likely to result from the fisheries. The Effects of the Action analyses (this chapter) consist of (a) continuing the pre-existing response regime (given that some of the natural subsidies and stressors are likely to change without any action on our part); then (b) identifying the probable responses of the species that are likely to be exposed to that regime over some period of time; (c) identifying the probable consequences of those responses on the fitness of those individuals; (d) identifying the probable consequences of any changes in fitness on the viability of the populations those individuals represent; and (d) identifying the probable consequences of changes in population viability on the viability of the "species" those populations comprise. Because the action is continuing, no new stressors relating to the groundfish fisheries will be applied in this chapter. However, continuing to expose individual organisms to stressors can result in consequences that we have not detected previously.

In this Biological Opinion, NMFS will determine whether (a) the species can be expected to survive with an adequate potential for recovery (e.g., trending toward recovery) when the effects of the action are added to the effects of the environmental baseline and any cumulative effects, and (b) whether affected designated critical habitat is likely to remain functional (or retain the ability to become functional) to serve the intended conservation role for the species in the near and long term when the effects of the action are added to the environmental baseline and any cumulative effects. Through an exposure and response analysis (Figures 4.24 and 4.25) we will determine population level impacts that inform determinations in Chapter 7 as to whether the response of the species and the risk to its viability (or not) constitutes jeopardy and adverse modification of critical habitat (i.e., risk assessment). NMFS completed a revised recovery plan for Steller sea lions (NMFS 2008a), which provides additional information on historical stressors and also provides demographic and threats criteria, which will be used to assess the risks to the species and its critical habitat.

There are two points of reference available when we consider data, information, or other evidence to support our analyses: (1) we can analyze the information available and subsequently conclude that an action has an effect, when in fact it does not (false positive), or (2) we can analyze the information available and subsequently conclude that an action does not have an effect, when if fact it had (false negative). In statistics, these two points of reference are called errors: the first point of reference is designed to avoid what is called Type I error while the latter is designed to avoid what is called Type II error. Although analyses that minimize either type of error are statistically valid, most biologists and ecologists still focus on minimizing the risk of concluding that there was an effect when, in fact, there was no effect (Type I error) and tend not to emphasize Type II error. However, because of the responsibility mandated by the ESA to protect and promote the recovery of ESA listed species, NMFS has made an effort to manage the risk of making a Type II error. We consider this approach to be consistent with the precautionary approach and purposes of the ESA and similar direction from the U.S. Congress and the courts.

Jeopardy and adverse modification analyses must look into the future to identify the effects of activities conducted today on the future of threatened and endangered species. Some human activities have delayed effects. This could be because a species population takes time to respond to an effect, because the population only responds when effects accumulate, or a combination of these two. If we do not look far enough into the future, our analyses will not detect a population's response to human activities, and we are more likely to falsely conclude there was no effect when, in fact, an effect occurred. If we look too far into the future, our analyses can mask short-term collapses in a population and, again, we increase our likelihood of falsely concluding there was no effect when, in fact, an effect occurred.

5.1 Steller Sea Lion: Western DPS and Critical Habitat

The endangered western DPS of Steller sea lion has declined by almost 90% throughout its range over the past 30 years, reaching its smallest size in 2000; however, since that time the population appears to be increasing in abundance. Prior to the 1990s, the primary causes of the decline may have been commercial harvests, entanglement of juvenile Steller sea lions in commercial fishing gear, and intentional shooting by fishermen. However, since 1991 the effects of the latter two factors have been nearly eliminated.

At present in the scientific community, there is no clear leading hypothesis to explain the decline of the western population of Steller sea lions in the 1990s (DeMaster *et al.* 2001, Atkinson *et al.* 2008) nor its lack of a robust population growth rate since 2000 (NMFS 2008a). Chronic nutritional stress resulting from an altered prey field due to commercial fishing and/or natural environmental variability, killer whale predation, and toxic substances are all considered to be continued threats to the recovery potential for the western population of Steller sea lions (NMFS 2008a). The age groups originally thought most likely to be affected during the steep declines of the 1980s were juveniles and, to a lesser extent, adult females (Merrick *et al.* 1987, Pitcher *et al.* 1998, Rosen *et al.* 2000a, Alaska Sea Grant 1993); data collected in the last decade emphasize potential risks to adult females due to indications of a potential decline in their reproductive performance (NMFS 2008a).

There is general scientific agreement that the decline of the western population of Steller sea lions in the 1980s resulted primarily from declines in the survival of juvenile Steller sea lions. Evidence for this comes primarily from the central GOA, but trends in population counts along the range are consistent with this general conclusion. At the same time, however, smaller reductions in adult survival also occurred, and a three-decade long decline in natality in the central GOA began. It is uncertain as to whether these changes had a nutritional cause (Merrick *et al.* 1987, Pitcher 1998, Rosen *et al.* 2000a, Alaska Sea Grant 1993, DeMaster *et al.* 2001), and whether fishery-induced changes in the forage base of Steller sea lions contributed to and continue to contribute to dietary or nutritional deficits (DeMaster *et al.* 2001; NMFS 2008a).

Under normal circumstances, the life history of Steller sea lions would protect them from short-term declines in the reproductive success of adult females or the survival of juvenile Steller sea lions. Steller sea lions are long-lived species with overlapping generations, a life-history strategy that protects them from short-term, environmental fluctuations. Their life history strategy would protect sea lion populations from variable survival and mortality rates caused by short-term phenomena like ENSO events. However, this life-history strategy may not protect Steller sea lions from chronic nutritional stress or changes in predation pressure that continue for two or three decades.

Recent data on population trends (2000 – 2008) indicate that the western DPS as a whole is growing at a positive rate (i.e., non-pup trend in abundance is +1.4% per year with a P value of 0.21). Of the seven sub-regions identified in the Recovery Plan (NMFS 2008a), four are increasing in abundance, with two increasing at a statistically significant rate (note: type 1 error value set at 0.10). However, the declines in pup production and the continued large declines of non-pups in the western Aleutian sub-region, declines in pups and non-pups in the central Aleutian sub-region, and to a lesser extent trends in non-pups and pups in the central Gulf of Alaska appear to be inhibiting robust recovery in the western DPS.

Evidence provided in the Environmental Baseline indicates that predation, disease, contaminants, and direct human effects including direct fisheries effects (e.g., mortality due to catch in gear, shooting by fishermen, collisions with vessels), are unlikely to affect the ability of the western DPS or eastern DPS to survive or to recover (Section 4.3, 4.4, and synthesis in 4.7). Thus, those limited direct fisheries effects are incorporated here by reference and will be considered to continue as described under this continuing action.

5.1.1 Effects of Fisheries Management Policy

The MSA, 16 USC 1801 et seq., is the primary domestic legislation governing management of the nation's marine fisheries. The MSA requires FMPs to be consistent with a number of provisions, including ten national standards, with which all FMPs must conform and which guide fishery management. Besides the MSA, U.S. fisheries management must be consistent with the requirements of other laws including the MMPA, the ESA, and several other federal laws.

Under the MSA, the Council is authorized to prepare and submit to the Secretary of Commerce for approval, disapproval, or partial approval, an FMP and any necessary amendments for each fishery under its authority that require conservation and management. The Council conducts public hearings to provide interested persons an opportunity to be heard in the development of FMPs and amendments, and reviews and revises, as appropriate, the assessments and specifications with respect to the OY from each fishery (16 USC. 1852(h)).

The Council's policy is to proactively apply judicious and responsible fisheries management practices, based on sound scientific research and analysis, to ensure the sustainability of fishery resources, to prevent unregulated fishing, and to protect associated ecosystems for the benefit of current users and future generations. For the past 30 years, the Council's management policy for Alaska fisheries has incorporated forward-looking conservation measures that address differing levels of uncertainty. This management policy has in recent years been labelled the precautionary approach. Recognizing that potential changes in productivity may be caused by fluctuations in natural oceanographic conditions, fisheries, and other non-fishing activities, the Council intends to continue to take appropriate measures to ensure the continued sustainability of the managed species. It will carry out this objective by considering reasonable, adaptive management measures, as described in the MSA and in conformance with the National Standards, the ESA, the NEPA, and other applicable law. This management policy takes into account the National Academy of Science's recommendations on Sustainable Fisheries Policy.

One explicit consideration of ecosystem-based fishery management should be biomass tradeoffs. The sum of single species MSY is greater than MSY for the system, and it is impossible to simultaneously maximize yield for all species (Brown *et al.* 1976, May *et al.* 1979).

Single-species fishery harvest policies (e.g., MSY) allow for the dietary needs of other components of the ecosystem through the natural mortality parameter, M. Natural mortality is the estimated biomass of the fished species that would die naturally each year, largely through predation by other species in the ecosystem, and is, in most cases, a fixed proportion of the fished species biomass in any given year. As such, the actual biomass estimated to be consumed by predators does not consider the size of the predator populations nor their dietary needs, but is assumed to be a constant fraction of the available fished species biomass. Single-species fishing strategies, by design, attempt to reduce the average standing stock biomass of the fished species to an intermediate size to maximize productivity. As a consequence, the biomass assumed to be consumed by other predators also declines, regardless of the population trends or the dietary needs of the predators themselves. Single-species harvest policies assume that other predators will compensate by switching to other "forage" species (whether they are also subject to fishing or not) in the ecosystem to ensure their nutritional needs. Single-species management policies applied to multiple "forage" species at the same time, then, are likely to compound this potential deficit.

Ecosystem assessments evaluate the state of the environment, including monitoring climate–ocean indices and indicator species to detect ecosystem changes. Ecosystem-based fisheries management reflects the incorporation of ecosystem assessments into single species assessments when making management decisions and explicitly accounts for ecosystem processes when formulating management actions. The new US Ocean Policy issued by President Obama on July 19, 2010 contains the following as a priority objective: "adopt ecosystem-based management as a foundational principle for the comprehensive management of the ocean, our coasts, and the Great Lakes." [25] In addition, the Pew Commission report and the Oceans Commission report point to the need for changes in the organization of fisheries and oceans management to institutionalize ecosystem considerations in policy making (Pew 2003; U.S. Commission on Ocean Policy 2004). The Oceans Commission, for example, points to the need to develop new management boundaries corresponding to large marine ecosystems, and to align decision-making with these boundaries (U.S. Commission on Ocean Policy 2004).

Ecosystem-based fisheries management may still encompass traditional management tools, such as TACs, but these tools will likely yield different quantitative results. To integrate such factors into fisheries management, NMFS and the Council may develop policies that explicitly specify decision rules and actions to be taken in response to preliminary indications that a regime shift has occurred. These decision rules could be included in long-range policies and plans. Management actions should consider the life history of the species of interest and can encompass varying response times, depending on the species' lifespan and rate of production. Stock assessment advice may explicitly indicate the likely consequences of alternate harvest strategies to stock viability under various recruitment assumptions.

Management strategy evaluations (MSEs) can help in this process. MSEs use simulation models of a fishery to test the success of different management strategies under different sets of fishery conditions, such as shifts in ecosystem regimes. The AFSC is actively involved in conducting MSEs for several groundfish fisheries, including for several flatfish species in the Bering Sea, and for pollock in the GOA.

---

[25] President Obama's US Ocean Policy: http://www.whitehouse.gov/administration/eop/ceq/initiatives/oceans/

The 2009 SAFE reports include an appendix on ecosystem considerations that addresses climate and physical environmental trends, ecosystems trends, and fishing and fisheries trends. Information on trends by trophic level is included for the consideration of the stock assessment authors as the individual sections are written for the managed fish stocks. Assessments for BSAI and GOA pollock, BSAI and GOA Pacific cod, GOA arrowtooth flounder, and Aleutian Islands Atka mackerel include sections on ecosystem consideration, specific to each of these stocks.

Since the publication of the Oceans Commission report, the President has established a cabinet-level Committee on Ocean Policy by executive order. The Committee explores ways to structure government to implement ecosystem-based ocean management (Evans and Wilson 2005).

Congress reauthorized the MSA in December 2006 to address ecosystem-based management. NMFS and the Council are continuing to develop their ecosystem management measures for the fisheries in the EEZ off Alaska. Previous measures include those that prevent target fisheries on forage species such as capelin and eulachon that are important to other ecosystem components such as marine mammals and seabirds. The Council has created a committee to inform the Council of ecosystem developments and to assist in formulating positions with respect to ecosystem-based management. The Aleutian Islands Fishery Ecosystem Plan was completed in 2007 and is currently being reviewed by the Council for updates. The Council and the State of Alaska have created a staff-level interagency Aleutian Islands Ecosystem Team to support this effort.

The Council has taken the lead in the establishment of the interagency Alaska Marine Ecosystem Forum to improve interagency coordination and communication on marine ecosystem issues. The SSC has begun to hold annual ecosystem scientific workshops at the February Council meetings.

In addition to these efforts to explore how to develop its ecosystem management efforts, the Council and NMFS continue to initiate efforts to take account of ecosystem impacts of fishing activity. For example, the Council recommended Amendment 89 to the BSAI groundfish FMP which provides protection to areas of bottom habitat that may have been impacted by nonpelagic trawling. This action was in response to concerns of the ecosystem impacts from shifts in fishing activity to the north. Ecosystem protection is supported by an extensive program of research into ecosystem components and the integrated functioning of ecosystems, carried out at the AFSC and in cooperation with other research organizations including the North Pacific Research Board. EFPs currently support investigation of new management approaches for the control of salmon bycatch in the BSAI, and research into salmon and halibut excluder devices.

The Council reconstituted its Ecosystem Committee to discuss ecosystem initiatives and advise the Council on: (1) defining ecosystem-based management; (2) identifying the structure and Council role in potential regional ecosystem councils; (3) assessing the implications of NOAA strategic planning; (4) drafting guidelines for ecosystem-based approaches to management; (5) drafting MSA requirements relative to ecosystem-based management; and (6) coordinating with NOAA and other initiatives regarding ecosystem-based management. As of December 2009, the committee has addressed and provided recommendations regarding the following issues: Marine Protected Areas nomination process, the Ocean Policy Task Force and Marine Spatial Planning, the Northern Bering Sea Research Area research planning, the EFH five-year review, and Habitat Areas of Particular Concern criteria.

5.1.2 Effects of the Exploitation Strategy

As NMFS stated in the 2000 and 2001 Biological Opinions, fishing significantly reduces the spawning stock biomass from an “unfished” level to a “fished” level (Section 4.5.4). Effects of reducing the spawning stock biomass by fishing may occur directly on listed species or critical habitat, or indirectly through changes in the ecosystem, including target species, non target species, habitat, and the ecosystem at large. Fisheries can also have indirect biological effects that occur when fisheries remove large numbers of target species and non-target species (bycatch) from a marine ecosystem. These removals can change the composition of the fish community with associated effects on the distribution and abundance of prey organisms. Fishery removals of biomass can also compete with other consumers that depend on target organisms for food. These biological effects are generally termed cascade effects and competition.

5.1.2.1 Global Control Rules

The global control rule is one aspect from the programmatic level of the current FMPs which describes the exploitation strategy and resulted from recent Biological Opinions to reduce fishing mortality when the spawning biomass is estimated to be less than 40% of the projected unfished biomass. This rule established that if a biological assessment of stock condition for pollock, Pacific cod, or Atka mackerel projects that the spawning biomass will be equal to or below 20 percent of the projected unfished spawning biomass during a fishing year, the directed fishery for the relevant species is prohibited. When the global control rule was established in the early 2000s NMFS did not anticipate that fish species' biomass would reach the B_{20} level. In recent years, the downward trend in the Bering Sea pollock biomass has approached the B_{20} threshold, currently at $B_{27\%}$.

5.1.2.2 Effects of the Harvest Strategy on Steller Sea Lions

Various analyses have been conducted to evaluate the effects of fisheries on Steller sea lions (e.g., Loughlin and Merrick 1988; Ferrero and Fritz 1994; Hennen 2006) and recently another paper has been completed that examines the effects of the Atka mackerel fishery (Trites *et al.* 2010). The AFSC also has conducted various Fishery Interaction Team studies (described previously), and additional analyses were conducted for this Biological Opinion (AFSC 2010a). Opinion varies in the literature and among experts in pinniped foraging ecology, fishery management, and other fields as to indirect linkages between fishing activities and Steller sea lions. Some have found significant correlations of fishery removals of prey resources and Steller sea lion abundance, for example, but these effects are often localized or suffer from potential sources of error, including particularly the intangible effect of fluctuations in ocean carrying capacity, climate change, and other confounding impacts that are difficult to measure (levels of disease or predation). Hennen (2006) found correlations between fishery harvest amount and Steller sea lion counts in the 1970s through the 1990s, concluding that more conservative harvest regulations may have been a cause of an upward shift in sea lion trends, but also concluded other factors may have influenced these trends such as a change in ocean conditions. Recently, conservation measures implemented after the 2001 Biological Opinion was released (effective in 2002) coincide with continued and more rapid improvement in Steller sea lion counts in many subareas across the range of the western DPS.

5.1.3 Effects of the Annual Fisheries Assessment

The Annual Fisheries Assessment cycle is described in Section 2.4, and consists primarily of resource surveys, stock assessments, and setting the catch specifications. Resource surveys yield estimates of fish stock biomass, age structure and distribution. Stock assessments integrate data from multiple sources to estimate time series of biomass and recruitment to individual fish populations that are then used to project the populations several years into the future. Setting of the catch specifications (e.g., ABC) primarily involves applying the harvest control rule for the species (based on its tier assignment), which in its simplest form is a harvest rate times the projected stock biomass for the following year. Most resource surveys yield estimates of biomass, age and distribution of multiple species of fish at the same time, while the other two parts of the annual fisheries assessment cycle are done on a single-species basis.

The MSA and resulting regulations (50 CFR § 600.310 (f)(3)(iii)) require that relevant ecological factors be considered in the setting of optimum yield for a fishery, including "…predator-prey or competitive interactions, and dependence of marine mammals and birds or endangered species on a stock of fish." Because the fish stocks themselves are assessed using primarily single-species methodologies, the review of the assessments and catch specifications by the Council's groundfish Plan Teams, SSC and the Council itself provide the only opportunity for relevant ecological factors to be explicitly considered in the annual fisheries assessment cycle. Currently, most fish stock assessments include some of the information presented in the Ecosystem Considerations chapter regarding the ecological role of the species being assessed, but this information is not formally incorporated into the process of setting catch specifications. Some of the stock assessment authors include in their draft SAFE chapters some of the available information on the ecological role of the species being assessed, including estimates of its consumption by other ecosystem consumers and its role in consuming other ecosystem components. For instance, some target species are also nodal species within their ecosystems, and as such have many trophic connections with other species. Nodal fish species (e.g., pollock, Atka mackerel, Pacific cod) are forage fish for trophic levels above them, including the Steller sea lion. This is a "relevant ecological factor" that must "be considered in the setting of optimum yield", since it pertains directly to the "dependence of …[an] endangered species on a stock of fish." To some extent, these relationships are considered early in the overall stock assessment process by the authors of the individual species chapters in the annual draft SAFE documents.

5.1.4 Effects of the Implementation of the Fisheries

In Chapter 4, NMFS walked through a detailed exposure analysis (Figure 4.24) showing the overlap of the effects of the fisheries and the prey resources of Steller sea lions (Section 4.5.2) and discussed the short-term and long-term responses (Sections 4.5.3 and 4.5.4). In this section we focus on the effects of the prosecution of the fisheries on Steller sea lions and their critical habitat under the continuation of the proposed action. In the 2001 Biological Opinion, NMFS used a three-step inquiry to determine whether the proposed action would cause jeopardy to Steller sea lions. This three-step inquiry included: (1) identify the probable direct and indirect effects of the proposed action on the action area (exposure analysis), (2) determine whether reductions in Steller sea lion reproduction, numbers, or distribution would reasonably be expected, (response analysis) and (3) determine if any reductions in Steller sea lion reproduction, numbers, or distribution from the proposed action could be expected to appreciably reduce the species' likelihood of both surviving and recovering in the wild (jeopardy and adverse modification analysis). We used this three-step inquiry in the present analysis to determine the direct and indirect effects of the proposed action on Steller sea lions and their critical habitat.

5.1.4.1 Fishery Harvest Rate Analysis

In the August 2, 2010 draft Biological Opinion, NMFS reported biomass by RCA according to methods in AFSC (2010a). During the special Council meeting convened in August 2010, the SSC noted that the available data, particularly for patchily distributed Atka mackerel abundance, do not support apportionment at the scale of the RCAs. The SSC recommended that methods and scale used by the fishery stock assessment authors to determine and apportion biomass be used in the Biological Opinion as they comprise the best data available. In this section we present fish biomass data following the protocols used in the stock assessment process to estimate biomass of Atka mackerel, Pacific cod, and pollock in the Aleutian Islands and Gulf of Alaska. Biomass estimates by NMFS management area from 1999-2009 are provided in Tables 5.1, 5.2, 5.4, and 5.5. The annual stock assessment protocols, including a description of NMFS' surveys are described in section 2.4.

Catch data by NMFS management areas are provided in Tables 5.2 and 5.5. The data source for these catch estimates is the NMFS AKR catch accounting database, which is the official record of total catch for the groundfish fisheries. Tables 5.3 and 5.6 show the harvest rate (catch divided by the estimated available biomass) for each NMFS management area from 1999-2009 in the Aleutian Islands and Gulf of

Alaska, respectively. Data provided in this section replace fish biomass estimates and harvest rates by RCA contained in the August 2, 2010 draft Biological Opinion.

In the Aleutian Islands, harvest rates were lowest for pollock (ranging from 0-6%) and highest for Pacific cod (ranging from 8-64%) (Table 5.3). In Area 543, the western Aleutian Islands, the average harvest rate of pollock, Pacific cod and Atka mackerel from 1999-2009 was 2%, 26%, and 7%, respectively. In Area 542, the average harvest rate of pollock, Pacific cod and Atka mackerel from 1999-2009 was 1%, 22%, and 11%, respectively. In Area 541, the average harvest rate of pollock, Pacific cod and Atka mackerel from 1999-2009 was 1%, 45%, and 8%, respectively. Section 4.5.2.5 describes changes in biomass in 2008 relative to 1999 for each fishery management area. Relative to 1999, pollock biomass has decreased in Areas 543 and 542 and increased in Area 541; Pacific cod biomass has decreased in all fishery management areas in the Aleutian Islands; and Atka mackerel biomass has remained relatively constant in Area 543, increased in 542, and increased substantially in Area 541 (Table 5.2).

Harvest rates from 1999-2009 were calculated for Pacific cod and pollock in the Gulf of Alaska (Table 5.6). In NMFS Area 610, pollock and Pacific cod harvest rates averaged 11% and 16%. The maximum harvest rate over the period analyzed was approximately 20% for these two species. In Areas 620-640 the combined harvest rate from 1999-2009 averaged 20% for pollock and 16% for Pacific cod (Table 5.6). Relative to 1999, pollock biomass has decreased substantially in Area 610 and less dramatically in Areas 620-640 (Table 5.5). Pacific cod biomass was slightly lower in 2008 than in 1999; biomass in 2009 exceeded 1999 levels.

Table 5.7 shows the projected female spawning stock biomass (SSB) for Aleutian Islands, Bering Sea and GOA groundfish stocks relative to B_{100} (from 2009 SAFE reports). The only stocks projected to be $\geq B_{40}$ in 2010 and 2011 were Aleutian Island Atka mackerel and Gulf of Alaska Pacific cod (Table 5.7).

The spatial relationships between Steller sea lion Recovery Plan sub-regions, NMFS Fishery Management Areas, and RCAs are shown in Table 5.8. Data on Steller sea lion trends, diet, total fishery harvest, and harvest by Steller sea lion critical habitat zones are shown in the highest spatial resolution available for each data category. Harvest rates inside Steller sea lion critical habitat are shown for each NMFS fishery management area. For example, in NMFS Area 543 the fishery harvested 7% of the available Atka mackerel biomass in 2008; of that 7%, 65% was harvested in the 10-20 nm zone of Steller sea lion critical habitat (Table 5.8). The highest harvest rates in 2008 were for Pacific cod from 3-10 nm of critical habitat in RCAs 1-4. A high proportion of the Atka mackerel harvest occurred in 10-20 nm of Steller sea lion critical habitat in RCAs 1-3 and from 3-10 nm in RCA 4; however the overall harvest rates for Atka mackerel ranged from only 6% to 8% in the corresponding fishery management areas. One caution in interpreting these results is that the scale at which the data are available to assess fishery biomass and harvest may not correspond with the scale of removals important for foraging sea lions. However, the data in Table 5.8 comprise the best available.

5.1.4.2 Zonal Analysis: Fishery Overlap with Steller Sea Lions

A more detailed assessment of catch within critical habitat zones can be found in Section 4.5.2.3; here we present a brief summary and analysis. In the Supplement to the 2001 Biological Opinion (issued 2003), a comparison was made between the catch under the 1999 and 2002 management regimes. Because of changes to groundfish management implemented in 2002, the expectation was that many of the fisheries would have experienced reduced nearshore amounts of catch in 2002 when compared to the amounts observed in 1999 (i.e., the fishery that NMFS determined in the FMP Biological Opinion to cause jeopardy and adverse modification). NMFS found that the performance of the management measures in reducing effort nearshore and in increased temporal/spatial dispersion was not the same in all areas or for all fisheries (NMFS 2003). When used in companion with the 2003 supplement information (NMFS

2003), these data which look at comparisons between recent years and 1999 are useful for determining if the implemented conservation measures have been effective in dispersing catch in time and space and reducing catch in nearshore areas important to foraging Steller sea lions.

In the Bering Sea, catch in all four target fisheries by all gear types was reduced or maintained at near zero levels in the 0-3 nm zone (Figures III-6 and III-8). Total catch of pollock during 2002-2006 remained at or near historical highs approaching 1.4 million mt; however the catch in critical habitat declined from over 800,000 mt to 240,000 mt in 2008 (Table III-1, top). While there was an overall decline in the proportion of pollock caught in critical habitat from 2001-2008, there has only been a slight reduction in the proportion taken within the 10-20 nm zone when comparing 1991-1998 (10.1% taken in the 10-20 nm) to 2001-2008 (9.6% from 10-20 zone) (Table III-1). Roughly a third of the pollock catch (30%) was taken out of the designated foraging zones from 2002 to 2008 (Table III-1 top), contrasting with 43% being taken from the foraging area 1991-1998. The amount of pollock taken out of the Bering Sea foraging areas ranged between 400,000-600,000 mt year between 2001-2005, and held near 300,000 mt per year (until 2008) as compared to 356,077 mt taken in 1999. Data from earlier years appear to reflect the conservation strategy of closing only the 0-10 nm area, thereby allowing harvest in the 10-20 nm and foraging zones for vessels preferring to fish closer to shore. There are, however, indications from the last two or three years that the nature of this fishery may be changing with both biomass and effort moving to the north.

Atka mackerel catches in the Bering Sea were at historic highs during the 2003-2007 seasons and were taken almost exclusively within critical habitat (99%). Even though the total amounts removed remain lower (5-10%) relative to what is taken from the Aleutian Islands, more fish were removed from Bering Sea critical habitat in any of these years since 1991.

There were decreases in the proportion of Pacific cod caught in recent years in the critical habitat zones in the Bering Sea. The proportion of Pacific cod taken from the foraging zone decreased from the 1991-1999 period (average 31%) to 23% over 2002-2008, while the proportion harvested in the 3-10 zone remained steady at about 3% (Table III-1). In the Bering Sea, the Pacific cod longline harvest in critical habitat declined 53% between 1999 and 2008 as the total catch increased 7% in 2008 from the 1999 level (Table III-6). It should be noted that since 1999, the majority of Pacific cod catch in the Bering Sea has been concentrated in the first quarter of the year (average 52%).

Some limited catch of pollock in critical habitat in the Aleutian Islands continues at low levels (about 60%; Table III-3). This catch was a combination of bycatch taken in other directed fisheries for groundfish, which may occur within critical habitat and as part of the EFP described above. The proportion of pollock caught in critical habitat in the Aleutian Islands declined from 74% (1991-1999) to an average of 60% between 2003 and 2008, with the largest change being a drop in catch taken in the 3-10 nm zone (32% to 17%) (Table III-3). The annual quota of pollock in the Aleutian Islands is wholly allocated to the Aleut Corporation and harvest levels in recent years have been very small.

Between 2003 and 2008, AI Pacific cod trawl catch was nearly 20% over that of 1999, with a large proportion coming out of the 10-20 nm zone and the catch from the 0-3 nm zone declining to generally under 100 mt (Table III-3). The catch of Pacific cod in the pot fishery has almost been eliminated in recent years. The Pacific cod longline catch has fluctuated in recent years but increased last year to take 12% more in critical habitat in 2008 than 1999 (Table III-6).

Total catch of Aleutian Islands Atka mackerel increased slightly during the 1999-2002 period, but has since stabilized around 55,000 mt year between 2003 and 2008, with a decline in the proportion of catch being made in critical habitat between 1991-1999 (average 67%) to 2000-2008 (average 36%). Between

1999 and 2008, the amount of catch removed from critical habitat has been a relatively consistent 20,000 mt, most of which about 35% was harvested in the 10-20 nm zone (Table III-3).

The proportion of pollock caught in GOA critical habitat declined slightly from an average of 75% (between 1991-1999) to an average of 68% between 2000-2008. There was a slight decrease in the proportion harvested in the 3-10 nm zone and foraging zones and an increase harvested in the 10-20 nm zone (about 51% on average during 2000-2008) (Table III-2). Comparing 1999 to 2005, a 15% decrease in the total amount of pollock caught in the GOA, coupled with a 13% decrease in the proportion of pollock caught within critical habitat, resulted in a 27% decrease in the amount of pollock taken from critical habitat in the GOA (Table III-4). Even larger decreases are noted when comparing 1999 and 2007 (Table III-5) and 1999 and 2008 (Table III-6). Harvest of pollock by trawling was virtually eliminated from the 0-3 nm zone in the GOA between 2001 and 2008 (Tables III-4, III-5, and Figure III-6).

In late 2004, revisions were made to the Steller sea lion protection measures allowing for additional harvest beginning in the first part of 2005 around Puale Bay in concert with an expanded closure around Cape Douglas. The closure around Cape Douglas was much larger than the opening around Puale Bay and was designed to result in a net decrease in the area of critical habitat available for pollock fishing. Comparing the 2004 and 2005 pollock catch in the GOA, there was an increase in the total pollock caught in the GOA as well as in the amount of pollock caught within Steller sea lion critical habitat of the region (Table III-2). Subsequent to the management change, there was an 11% (2005 and 2007) and 17% (2008) increase over 1999 in the proportion caught in the 3-10 nm zone from 1999 to 2005 (Tables III-4, III-5, III-6).

In the GOA, the proportion of Pacific cod taken in critical habitat has declined since 1999 (1991-1999 average 68%, 2000-2008 average 53%), although the amount of both total catch and catch in critical habitat have increased since 2005 (Table III-2, Tables III-4, III-5, and III-6 and Figure III-3). For the GOA Pacific cod trawl fishery, the proportion of catch decreased within the 3-10 nm zone and generally increased in the 10-20 nm zone between 1999 and 2008. The proportion caught in the 10-20 nm zone by the GOA Pacific cod pot fishery in recent years is higher than in 1999 (Tables III-4 and III-6). While the average amount of all Pacific cod caught in critical habitat between 2002-2006 declined by more than 30% from that of 1999, there was a noteworthy increase in that catch in 2007 and 2008 (Table III-2).

5.1.5 Response of Steller Sea Lions

The potential response of Steller sea lions to the proposed action is provided in a schematic in Figure 4.25 and is dependent on the Steller sea lion life-cycle (Figure 3.24).

5.1.5.1 Physiological Response of Individuals

Numerous individual hypotheses to explain the Steller sea lion steep decline in the 1980s, slow decline in the 1990s, and lack of a robust recovery in the 2000s are concerned with both proximate and ultimate factors that range in scale from ecosystem changes to individual Steller sea lion physiology (NMFS, 2008). To make things more complex, many of these hypotheses are inter-related (Guenette *et al.* 2006, Atkinson *et al.* 2008). One hypothesis that has received considerable attention has become known as the nutritional stress hypothesis (Chapters 3 and 4). In its simplest form, the nutritional stress hypothesis suggests that the decline in Steller sea lions is linked to inadequate nutrition induced by changes in the quality or quantity of available prey. Much of the research to validate this hypothesis has been conducted in a captive setting, but a coherent summary of all the studies, linked with relevant field research, had been lacking until recently. Rosen (2009) summarized the data from the majority of dietary studies conducted with pinnipeds in the laboratory that can be used to evaluate the validity of the nutritional stress hypothesis. Rosen (2009) found that:

1. Overall, there is strong evidence for biologically meaningful differences in the nutritional quality of major prey species. Steller sea lions can partly compensate for low-quality prey by increasing their food consumption.

2. There appear to be no detrimental effects of low-lipid prey on Steller sea lion growth or body composition when Steller sea lions can consume sufficient quantities of all types of prey. However, the ability to increase consumption is physiologically limited, particularly in young animals. It is more difficult to maintain energy intake on a diet of low-quality prey than on a normal diet.

3. Under conditions of inadequate food intake (either due to decreased prey availability or quality, or increased energy requirements) the overall impacts of nutritional stress are complex, and are dependent upon season, prey quality, age and the duration and intensity of the nutritional stress event.

Rosen (2009) commented that while Steller sea lions possess the ability to alter food intake in response to prey quality and availability, sufficient intake may be limited by ecological factors (prey availability or distribution) or physiological factors (increased energy demands or digestive capacity). Based largely on studies of animals in captive facilities, younger animals have increased energy requirements and more limited digestive capacities relative to adults and are less able to increase food intake sufficiently when consuming low quality prey to obtain sufficient nutritional intake. Adults are less limited by digestive capacity, but their ability to make seasonally appropriate adjustments to their energy budget (which minimizes the bio-energetic consequences of food restriction) is impeded when consuming low-quality prey. Additionally, it appears that, although severe food restriction in the winter may induce greater rates of mass loss and physiological stress than in the summer, Steller sea lions have greater difficulty recovering from more moderate restrictions in the summer months.

Physiological stress and mass loss can result in reduced body size, reduced energy stores, alteration of foraging behavior, altered reproductive rates, and other survival parameters. The effects on reproduction could include delayed ovulation, failure to implant a fetus, abortion, under-developed or under-weight pups at birth, or any other disruption of the reproductive process. Additional research is needed to address these information gaps.

5.1.5.2 Response of Populations

In this analysis we integrate the response of individuals and RCAs to the response of regional populations within the western DPS. In the western DPS, the overall trend (2004 - 2008) in the non-pup population in Alaska was stable or declining slightly. This follows a four-year period of population increase (at approximately 3% per year) between 2000 and 2004 which has been the only increasing period observed since trend information began to be collected in the 1970s.

Regional Trends

NMFS conducted an additional survey in 2009 (DeMaster 2009) designed to count pups and to clarify possible movement of non-pup Steller sea lions from the eastern DPS into the range of the western DPS. The concern was that the timing of the 2008 survey might have altered the 2008 population estimate. Results of the non-pup survey from late June 2009 in southeast Alaska and in the eastern and central GOA support the hypothesis that a very small number of Steller sea lions (about 570) moved into the eastern GOA, primarily from southeast Alaska. Total non-pup counts in the eastern Gulf of Alaska were

812 animals lower in the "late" 2009 survey than the "early" 2008 survey, while numbers of animals was greater by 2,642 in southeast Alaska.

An additional, 404 non-pups were also counted at trend sites in the central GOA in 2009 compared to 2008. Using the data collected in 2009 and calculating trends in each sub-area since 2000, it is likely that seasonal movement from southeast Alaska may have contributed a minimum of 570 additional Steller sea lions to the 2008 western DPS trend site counts.

The survey of Steller sea lion pups in 2009 (DeMaster 2009) resulted in 10,792 pups within the range of the western DPS in Alaska on rookeries and major haul-outs. However, survey sites in the western Aleutian Islands or Pribilof Islands were not surveyed in 2009, two regions with declining pup production. To estimate total western DPS pup production in 2009, counts obtained in 2005 and 2008 at five rookeries and one haul-out in these two regions were included to obtain a total of 11,120, an increase of 1,170 from the 9,950 pups counted in 2005. Pup production at major rookeries (N=31) increased by 921 pups between 2005 and 2009 (+10%), or by approximately seven pups per rookery per year.

By region, rookery pup production declined in the western (minus 64 pups) and central Aleutian Islands (minus 120 pups), but increased in the eastern Aleutian Islands (+378 pups), and in the western (+355 pups), central (+169 pups), and eastern (+203 pups) GOA between 2005 and 2009. Pup production on all major western DPS rookeries increased at a statistically non-significant rate of 0.6% per year between 1998 and 2009, and 1.7% between 2001/02 and 2009.

Between 2001/2002 and 2009, rookery pup production declined 43% in the western and 7% in the central Aleutian Islands, but increased 47% in the eastern Aleutian Islands, and 23%, 6%, and 57% in the western, central, and eastern GOA, respectively, for an overall western DPS change of +14%. This is equivalent to an increase of approximately five pups per rookery per year from 2001/02 through 2009. While, these observed increases are encouraging for some sub-regions, it is not consistent across the range.

Vital Rates

Steller sea lion survival was calculated based on pups marked at rookeries in the GOA and compared to animals marked in southeast Alaska (Pendleton *et al.* 2006). Their analysis showed that juvenile survival in the central GOA was lower in 1988-1991 than in the 1970s and lower than southeast Alaska in the 1990s. They also found similar survival for adult females in the 1970s to 1990s but lower than in Southeast Alaska. The finding of female survival is different than that of Holmes *et al.* (2007) perhaps because of low re-sight probability for the Marmot Island cohorts through much of the 1990s (the sub-rookeries at Marmot Island where pups were marked have all but disappeared).

As previously described in Chapter 3, Holmes and York (2003) and Holmes *et al.* (2007) modeled the adult female population in the central GOA for the period 1976 – 2004. Their model indicated that birth rate in the central GOA steadily declined from 1976 to 2004. Over the same period, survivorship first dropped severely in the early 1980s, when the population collapsed, and then survivorship steadily recovered. The best model fit indicated that in 2004, the birth rate in the central GOA was 36% lower than in the 1970s, while adult and juvenile survivorship were close to or slightly above 1970s levels.

As noted in previous chapters, survival information (e.g., cumulative survival to age 3 for female Steller sea lions branded as pups on rookeries) is only available from RCAs 6 and 9-11 between 2000-2005 with resights through 2008 (NMML unpublished; Pendleton *et al.* 2006). There are no survival data from RCAs 1-5 and 7-8.

Likewise, direct information on natality (e.g., probability of a mature female producing a pup in a given year or average age of sexual maturity) is only available from RCAs 8, 9, 10 and 11.

As noted in the previous paragraph, information on vital rates is unavailable for the western Aleutian Islands sub-region. This is the only sub-region where there is a statistically significant decline in abundance since 2000. Available information on natality based on pup and non-pup counts indicates that the productivity of animals in this sub-region is less than it is in other sub-regions (e.g., 0.29 versus 0.37 – 0.42: see Table 5.8. A cursory review of the relationship among survivorship, natality, and trends in abundance indicates the following (assuming a maximum age of 30 years):

Adult survival	0.87	0.95	0.95
Recruitment (to ASM)	0.33	0.45	0.45
Age of sexual maturity (ASM)	6	6	6
Natality (female pups in year t+1 per adult female in year t)	0.15	0.15	0.30
Trend in Abundance	-7.0%/yr	-0.5%/yr	4%/yr

That is, given the first four parameters (and assuming a maximum longevity of 30 years), the resulting trend in abundance can be calculated. These results suggests that juvenile or adult survival of animals in the western Aleutian Islands sub-region, given the estimate of natality is approximately correct, may also be lower than what has been estimated for sub-regions to the east where information on vital rates is available. Another contrast to the above combinations of vital rates and estimates of trends in abundance can be developed by investigating the trend in abundance associated with the vital rates in Table 3.8. Here, vital rates change smoothly with age. The maximum annual survivorship value is 0.93 at age 3 (with a recruitment value to age 6 of 0.44), while the maximum natality value is 0.3285. In this scenario, the population is stable (i.e., the trend in abundance is 0.0%/yr).

Clearly, direct estimates of age specific rates of natality and survival are needed to better understand and inform discussions of measures that might be developed to mitigate the observed decline in abundance in the western Aleutian Islands sub-region, where the most recent estimate of the trend in abundance is -7.2%/yr. Nonetheless, it appears reasonable that the observed decline in abundance may be attributed to factors that lead to lower rates of reproduction and survival.

5.1.6 Fishery Effects on Critical Habitat

Critical habitat for the western DPS includes a 20 nm buffer around all major haulouts and rookeries, as well as associated terrestrial, air, and aquatic zones, and three large offshore foraging areas, including the Shelikof Strait area, the Bogoslof area, and the Seguam Pass area (50 CFR 226.202).

NMFS has implemented a complex suite of fishery management measures designed to minimize competition between fishing and the endangered population of Steller sea lions in critical habitat areas and other areas recognized as important to Steller sea lions. Those management measures are the Steller sea lion protection measures which are described in detail in the April 2006 BA for this action (Appendix B and maps in Appendix A of the BA).

In prior sections of this chapter (and in the Environmental Baseline), NMFS discussed the indirect effect of prey removal on Steller sea lions. When considering fishery effects on critical habitat those same fishery actions can be considered direct impacts as they remove prey species from designated critical habitat. These fishery removals can change the composition of the fish community with associated effects on the distribution and abundance of prey organisms (Section 4.5.4). The basis of the “destruction or adverse modification" analysis is to evaluate whether the proposed action results in negative changes in the function and role of the critical habitat in the conservation of the species. Thus, an adverse effect occurs when fishery removals of Steller sea lion prey in designated critical habitat inhibits the ability of that habitat to remain functional (or retain the ability to become functional) to serve the intended conservation role for the species (western DPS) in the near and long-term.

5.1.6.1 Essential Features (Primary Constituent Elements) of Marine Critical Habitat

In general, the physical and biological features of critical habitat essential to the conservation of Steller sea lions are those items that support successful foraging, rest, refuge, and reproduction. The August 27, 1993 final rule to designate critical habitat for the Steller sea lion (58 FR 45269) describes essential aquatic (foraging areas) and terrestrial features (rookeries and haulouts) of critical habitat and the rationale behind the regulatory definition of critical habitat.

Prey resources are the most essential feature of marine critical habitat for Steller sea lions (58 FR 45269). Marine areas may be used for a variety of other reasons (e.g., social interaction, rafting or resting), but foraging is the most important Steller sea lion activity that occurs when the animals are at sea. A discussion of Steller sea lion foraging patterns and prey use has been discussed in Sections 3.1.6-3.1.10. While many of the important physical and biological elements of Steller sea lion critical habitat can be identified, most of those features (particularly biological features) cannot be described in a complete and quantitative manner.

The status of critical habitat is best described as the status of the important prey resources contained within those areas. Pollock, salmon, Atka mackerel, Pacific cod, sand lance, cephalopods (squid and octopus), arrowtooth flounder, sculpins, herring, capelin, and other forage fish are important prey items found in Steller sea lion critical habitat. Due to the dynamic nature of aquatic ecosystems and fish, NMFS was unable to describe the specific attributes of prey within critical habitat at the time of listing. Thus, prey resources were described in general, and are constantly re-assessed to determine their conservation value to Steller sea lions. These fishery resources are evaluated annually from a fishery stock assessment perspective and that description is contained in the annual SAFE reports produced by the Council and NMFS, and to some extent the ecosystem relationships among some of these Steller sea lion prey items that also are targets for commercial fisheries (what preys upon them, what they prey upon) are considered in the individual stock assessment chapters of the SAFE report.

5.1.6.2 Status of Important Prey Resources

Understanding the status and trends of fish species known to be prey items essential to Steller sea lions is a crucial aspect in understanding the quality of critical habitat and, in turn, potential impacts to Steller sea lion populations. Although this consultation addresses the entire federal groundfish fishery, as well as parallel fisheries, the following four fish species are considered to be some of the principal prey items for Steller sea lions in the western DPS: Atka mackerel, walleye pollock, Pacific cod, and arrowtooth flounder. This determination results from species found to make up at least 10% of the Steller sea lion diet (Sinclair and Zeppelin 2002, McKenzie and Wynne 2008). Seasonal changes in diet preferences of Steller sea lions may confound our ability to confirm which prey species are most important given seasonal and regional differences found in diet studies. For example, in the Aleutian Islands, based on scat studies conducted in the 1990s (Sinclair and Zeppelin 2002), Atka mackerel may occur in 96% of Steller sea lion scats in winter and 55% in summer, while in the western GOA, sand lance and salmon are most prevalent (65% and 57%, respectively) in summer diets, then in winter pollock and P. cod predominate (93% and 31%, respectively). Thus, the overlap of fishery harvests with Steller sea lion habitat must also consider the importance of fishery target species to both regional and seasonal diets.

Chapter 3 includes a brief summary of prey status for Atka mackerel, walleye Pollock, Pacific cod and arrowtooth flounder from the 2009 SAFE reports, including historical ABC and TAC data where available (Tables 3.24-3.31). Further information on these species can be accessed by referencing the SAFE reports in their entirety (NPFMC 2009). Projected female spawning stock biomass (SSB) for Aleutian Islands, Bering Sea and GOA groundfish stocks relative to B_{100} (from 2009 SAFE reports) is provided in Table 5.7.

5.1.6.3 Value of Critical Habitat for Steller Sea Lions

In the Environmental Baseline (Chapter 4), we used the term environmental carrying capacity (the relationship between the distribution and abundance of prey and the number of predators an area could support at a particular time) to represent the value of critical habitat for Steller sea lions. Even without the presence of humans, other species compete with Steller sea lions for food in their designated critical habitat. Adult pollock, arrowtooth flounder, Pacific cod, northern fur seals, spotted seals, harbor seals, and numerous species of seabirds compete for small pollock in the action area; harbor seals compete with Steller sea lions for larger pollock; orcas, humpback whales, gulls, and pinnipeds compete with Steller sea lions for species like herring and capelin; and there are similar competitive interactions for species like salmon, rockfish, and sablefish.

One approach used in previous Biological Opinions was to compare the amount of forage available to a given predator in a given area with the amount of prey consumed by that predator in a time period of one year (referred to as Biomass Ratios). However, NMFS has found difficulty in interpreting this metric because the forage available to SSL based on all species (and not just pollock, cod and Atka mackerel-Ianelli *et al.* 2010b) indicates that the there was on average (2002 to 2006) over 700,000 mt of forage in RCA 1 for SSLs. This is comprised of over 10 species, and includes Pacific ocean perch (232,000 mt), Atka mackerel (244,000 mt), northern rockfish (127,000 mts) and cod (18,000 mt). Given the current estimate of consumption by SSLs in RCA1 is 17.8 kt/yr (Fadely *et al.* 2010), the resulting forage ratio is 40.6. Given the long-standing decline in abundance of SSL in RCA 1, it is clear that a high forage ratio alone is not sufficient for understanding trends in abundance. Similarly, Ortiz and Logerwell (2010) reported Atka mackerel production alone was sufficient to meet SSL consumption requirements at Seguam, Tanaga and Kiska trawl exclusion zones, yet these are areas where the numbers of sea lions in the last decade have declined.

Long-used rookery sites were likely selected by Steller sea lions for a variety of reasons, including substrate and terrain, protection from land-based and marine predators, protection from harsh wave or surf conditions, and local availability of prey. Successful reproduction for the species depends on the availability of rookery sites where animals can aggregate for sufficiently long periods of time to give birth, mate, and raise their young until the young are able to survive at sea. As the reproductive period requires at least several months, food supplies in the vicinity of the rookeries must be sufficient to meet the energetic needs of animals involved in reproduction (adult females and males). Once the reproductive season and the need for social aggregation is over, and pups have gained sufficient competence at sea, then animals (including mothers with pups) may disperse to other haulout sites. Throughout the remainder of the year, the local availability of prey remains a crucial factor in determining their movements and distribution. Mature females with dependent pups are still likely to be constrained in their foraging distribution relative to mature females without a dependent pup. All pups are susceptible because they have limited energy reserves compared to adult animals. Pups less than two years of age are likely relatively poor foragers that may be susceptible to reductions in prey availability. Pups are likely dependent on nearshore prey resources while they make the difficult transition to independent foraging. Juveniles, older but still immature, must continue to develop their foraging skills over time. Like other, older animals, they may range more widely, but their distribution and haulout patterns are determined, in large part, by the availability of prey.

Gregr and Trites (2008) modeled adult female foraging zones around Steller sea lion sites to examine the currently-designated critical habitat for the western DPS. They used telemetry, oceanographic, and POP data sets to develop and validate a model that would predict the most important foraging areas for adult females based on known foraging distances from sites during winter and summer. They determined that Steller sea lions rarely forage beyond 20 nm, and they noted that juvenile and adult female Steller sea lions particularly forage relatively close to rookeries and haulouts.

The foraging success of Steller sea lions, whether based on rookeries or haulouts, is determined by their ability to balance the energetic gains from foraging with the energetic costs of daily activities, including the act of foraging itself (i.e., energy balance). If the prey resources around rookeries and haulouts are inadequate for their needs, then they are forced to increase the time and energy expended to find sufficient prey. As a result, they are more likely to fail in securing the resources necessary for growth, and reproduction, and, if severe enough, survival.

Thus, the value of critical habitat for Steller sea lions is determined in large part by the abundance and distribution of prey species, as well as the species composition and the season of the year. To a large extent, the abundance of prey over time determines the number of predators the habitat can support; as the abundance increased, the area would be able to support more predators, as the abundance decreased, the area would be able to support fewer predators. Similarly, the distribution of prey species will determine whether prey are available to foraging Steller sea lions and will determine whether they can forage successfully. Factors that would determine an area's value to predators like Steller sea lions include the distance of prey from shore, the depth of prey in the water column, the distribution and abundance of prey, and the dispersal of prey over time and space.

5.1.6.4 Removal of Prey from Critical Habitat

Fishery removals of large numbers of target and non-target (bycatch) species have direct and indirect biological effects on marine ecosystems and specifically designated critical habitat (Section 4.5 for a detailed description of fishery removals and the response of fish). First is the immediate removal of fish during the implementation of the fishery and the reduction in local prey abundance for Steller sea lions. These removals, or localized depletions, were first described for the Atka mackerel fishery in the Aleutian Islands (Lowe and Fritz 1997) and further studied by the FIT program at the AFSC in the 2000s (McDermott *et al.* 2005, Conners and Munro 2008) and by Battaile and Quinn (2006) for pollock in some areas of the eastern Bering Sea. The Atka mackerel studies have shown that spatially concentrated fisheries in critical habitat can result in local harvest rates (e.g., south of Amchitka) that far exceed the target rate set for the fish species in the entire management area (e.g., for Atka mackerel in area 542).

In some areas, management measures to mitigate these fishery impacts (e.g., Atka mackerel fishery exclusion zones around rookeries) may not be equally effective in all areas due to differences in the spatial distribution of essential prey habitat inside and outside of these zones (Ortiz and Logerwell 2010) - for example, Atka mackerel habitat around pinnacles in bathymetrically diverse regions that extend offshore beyond 20 nm, and therefore outside of critical habitat (Figure 5.2). Exclusion zones appear to be more effective in isolating fishery effects in regions open to the fishery where zone and habitat boundaries are similar (e.g., Seguam Pass). In these areas, aggregations of fish inside the exclusion zone have low exchange rates with aggregations of fish outside at least over the course of short-term fisheries (i.e., one or two weeks). Exclusion zones are less effective in those areas where zone boundaries cut across habitat where fish would be expected to move freely (e.g., Pacific cod; Figure 5.3), thus allowing fisheries outside to negatively influence prey populations thought to be protected inside the zone (e.g., Amchitka). FIT and other studies of trawl exclusion zone effectiveness around Cape Sarichef (Unimak Island) have shown similar results with respect to the Pacific cod trawl fishery on the southeastern Bering Sea shelf

(Fritz and Brown 2005, Conners and Munro 2008). And Ortiz and Logerwell (2010) found that other competitors to fisheries for the same prey (here, Atka mackerel) may reduce target species, confounding effects of trawl exclusion zones established to "protect" these prey items. Thus, fishery exclusion zones within parts of critical habitat should take into account the spatial heterogeneity of important prey habitat, and other factors such as predators on Steller sea lion prey items.

Regulations enacted in 2002 to mitigate Atka mackerel fishery impacts in the Aleutian Islands are illustrative of the potential effects of fisheries on critical habitat. Directed fishing for Atka mackerel is prohibited within Steller sea lion critical habitat in the Aleutian Islands management area east of 178° W. This includes all of RCAs 4 and 5, and with respect to fishery management areas, all of area 541 and part of area 542. By contrast, up to 60% of the TAC for Atka mackerel can be taken from Steller sea lion critical habitat areas west of 178°W in the Aleutian Islands; this includes all of RCAs 1-3, and part of area 542 and all of area 543. The boundary at 178°W does not coincide with any major biogeographic boundary in the Aleutian Islands, and is located at approximately Tanaga Pass which is about 60 miles east of Amchitka Pass in the Aleutian Islands. Steller sea lion populations in areas where directed fishing for Atka mackerel in critical habitat is prohibited (RCAs 4 and 5) have generally been stable or increasing slightly since 2000. However, populations in areas where fishing inside critical habitat is permitted (RCAs 1-3) have continued to decline. This correlation is consistent with the hypothesis that Atka mackerel fishing within critical habitat areas west of 178°W may have contributed to the continued decline of Steller sea lion numbers in RCAs 1-3. As noted later in this document, we believe this management history can be used in an adaptive management sense to better explore the efficacy of conservation measures implemented to promote the recovery of the western DPS of Steller sea lions.

In contrast to the fishing example above, there has never been any significant directed fishing for Atka mackerel in RCA 4. Yet, shortly after the ESA listing regulations were implemented to prohibit shooting and reduce bycatch of Steller sea lions in the early 1990s, Steller sea lion numbers in this area began to increase. This is the only RCA that experienced increases in Steller sea lion numbers in the 1990s, and it is also the only RCA that had never experienced substantial fishery removals of important prey items inside critical habitat. Thus, once direct substantial mortality stressors were curtailed, Steller sea lion numbers began to increase in RCA 4 relatively quickly (correlative evidence). Based on this analysis, it is reasonable to conclude that prey availability RCA 4 within critical habitat was sufficient to contribute to steady Steller sea lion population increases. However, recent increases in the harvest rate of Pacific cod in RCA 4 may have reduced the forage value of the habitat in this RCA to a point that Steller sea lion numbers stopped increasing (Table 5.8).

In critical habitat areas (RCAs 1-3) where fishing has occurred at varying levels since the inception of the Atka mackerel fishery (since 2002 up to 60% of the TAC in critical habitat), Steller sea lion numbers continued to decline. No change in population trajectory was observed after the enactment in the early 1990s of prohibitions on direct mortality. Thus, it is likely that prey availability in critical habitat in RCAs 1-3 has not been sufficient to sustain a sufficient forage value of habitat in this RCA to a point that Steller sea lion numbers can stabilize and recover.

No directed Atka mackerel fishing has been permitted within critical habitat in RCA 5 since 2002. In addition, 20 nm trawl exclusion zones around rookeries on Seguam and Agligadak Islands were established in the early 1990s when the Bogoslof area (519) was closed to directed pollock fishing. This action taken in the 1990s reduced trawl fishery activity in large portions of critical habitat in RCA 5 earlier than in other RCAs. Atka mackerel harvest rates overall in RCA 5 have also been relatively low in the 2000s (<10%; AFSC 2010a) compared to RCAs 1-3. Meanwhile a fishery for Pacific cod has persisted in the region with about 70% taken in critical habitat. Steller sea lion abundance in RCA 5 stabilized in the mid-late 1990s and has increased through 2008. Based on these data, we conclude that the conservation value of critical habitat in RCA 5 was sufficient to prevent further declines, but may not have been sufficient to support recovery.

In RCA 6, measures taken in 2002 reduced the proportion of catch taken from critical habitat from 44% to 33% for pollock and from 30% to 25% for Pacific cod. Most of the critical habitat removals came from the foraging zone outside of 20 nm of rookeries and haulouts. The total removals of prey species from critical habitat also declined 24% between 1999 and 2008. However, about half of the annual cod catch and 40% of the annual pollock catch is taken in the first quarter of the year, largely within critical habitat. Steller sea lion non-pup counts in RCA 6 were largely stable through much of the 1990s following dramatic declines in the 1980s. The cessation of the steep decline in the 1980s and the change to a slow decline in the 1990s is most likely due to measures taken to reduce direct mortality when the species was listed in 1990. Populations did not begin to increase until approximately 2000; non-pup counts increased 4% per year between 2000 and 2008, while pup production has increased at approximately 5% per year over this period. This is considered a robust rate of increase by NMFS (i.e., here we refer to a robust rate of recovery as an annual rate of increase in excess of 3% per year). Catches in critical habitat of pollock and cod within RCA 6 through the 1990s were higher than those of the 2000s. The observed association of lower catches in critical habitat in the 2000s with a robust recovery of Steller sea lion abundance in RCA 6 indicates that recently enacted management measures in this region may have improved prey availability for Steller sea lions in this region. This in combination with the high estimated rates of juvenile survival in the 2000s from RCA 6 (Ugamak; Figure 3.13) suggest that these measures may have contributed to the improvements in Steller sea lion population trends observed in RCA 6.

RCA 6 and RCA 7 are the regions that have experienced the greatest increase in Steller sea lions since 2000 and are largely responsible for the increases observed in the western DPS as a whole. Populations in RCA 6 have also declined proportionally more than any other area (tied with the central GOA at negative 91%) since their historic highs in the early 1960s. As a consequence, density-dependent increases in per capita prey availability may also have contributed to the larger improvements in Steller sea lion population performance observed in RCA 6 than other areas.

In RCA 7, total groundfish prey removals from critical habitat dropped 36% between 1999 and 2008, with significant declines in the pollock catch within 20 nm of rookeries and in the cod catch within 3-10 nm of all rookeries and haulouts. However, there also continue to be significant catches of both species in the first quarter of the year (66% of the annual catch of both), largely within critical habitat. Steller sea lion populations in RCA 7 have responded similarly to those in RCA 6 since the early 1990s (stable in the 1990s, increasing since 2000). Based on these data, we conclude that prey availability in critical habitat in RCA 7 is sufficient to begin the recovery process given the increases observed here since 2000.

In RCAs 8-9, Steller sea lion declines were abated in approximately 2000; since then, counts of non-pups and pups have been relatively stable, with a 6% increase in pup counts and a 1% increase in non-pup counts between 2000 and 2008. Thus, Steller sea lions in this RCA continued to decline and apparently did not stabilize until the additional conservation measures to reduce fishery-Steller sea lion interactions were taken in the early 2000s. As a consequence, the total amount of pollock and Pacific cod taken in RCAs 8 and 9 declined beginning in the early 2000s. However, the proportions of annual catch of both pollock and Pacific cod taken from critical habitat in RCAs 8 and 9 have not changed significantly since the early 1990s, and remain relatively high compared with other zones with improving Steller sea lion populations (RCAs 6, 7 and 10). In addition, most of the pollock in both RCAs 8 and 9 is caught inside critical habitat in winter. Also since the early 2000s, the catch of arrowtooth flounder has generally increased in these areas with most of the catch being taken (and increasingly so) from within critical habitat. Based on these data, we conclude that prey availability in critical habitats in RCAs 8 and 9 may be sufficient to provide for a slow recovery, but may not be sufficient to provide for a robust rate of recovery.

In RCA 10, Steller sea lion populations declined steeply through the 1990s even after direct mortality was reduced, much like in RCAs 8 and 9. Since 2000, there have been significant increases in both non-pup and pup counts in RCA 10, which are more similar to RCA 11 (southeast Alaska) than to the rest of the western DPS (particularly in the ratio of pups to non-pups on rookeries). Pollock and cod fisheries in RCA 10 are much smaller than those of other RCAs in the GOA (7-9), except for RCA 11, and have considerably more inter-annual variability. Critical habitat catches of pollock between 1991 and 2008 ranged from a low of 254 mt to a high of 13,672 mt. Pollock harvest rates declined from an annual average of 28% in the late 1990s to only 7% in the 2000s, while cod harvest rates dropped from 31% to 12% in the same period. Most pollock catch in RCA 10 occurs in winter, but annual winter harvest rates have declined from an average of 12% in the 2000s from 32% in the late 1990s (AFSC 2010a). Primary prey resources for Steller sea lions in RCA 10 are salmon, sand lance, herring, and capelin, with minor amounts of pollock, P. cod, and arrowtooth flounder. Based on these data, we conclude that prey availability is sufficient in critical habitat in RCA 10 to support a robust rate of recovery.

5.1.7 Response of Critical Habitat

As summarized in Section 4.6.2 the changes in Steller sea lion vital rates and trends in population growth may have been a function of nutritional stress resulting from a combination of reduced prey availability and quality, as well as other factors, such as predation. Two stressors were likely to have affected the prey field for Steller sea lions: (1) regime shift/environmental variability induced changes in the species composition, distribution, or nutritional quality of Steller sea lion prey, and (2) fishery-induced changes in localized or overall prey abundance and quality. Changes in prey communities resulting from regime shifts or environmental variability likely have affected the condition of Steller sea lions since they first began to inhabit the North Pacific Ocean, while changes that were fisheries-induced have occurred only in the last 40 years, but the relative importance of each is a matter of considerable debate.

5.1.7.1 Changes in Prey Density

As summarized in the 2001 Biological Opinion,[26] there are significant concerns about the assumptions inherent in an analysis of fisheries effects on critical habitat. These include: (a) Steller sea lions eat more than just the principal prey species identified, (b) biomass is not the only factor that affects successful foraging, and (c) Steller sea lions forage beyond critical habitat boundaries.

NMFS analyzed changes in distribution of important Steller sea lion prey (pollock and Pacific cod) in Section 4.1.4.4 and found that inter-annual spatial variability was high for both pollock and Pacific cod, but with Pacific cod having a somewhat broader but lower density overall than pollock (Figures 4.8-4.13). Pollock summer bottom-trawl concentrations within Steller sea lion critical habitat area show considerable variability with some years having relatively low densities (e.g., 1982, 1988, 1991, 1997-1999, 2006) and other years having high concentrations in critical habitat (Figures 4.8-4.10). Since 2000, moderate densities of pollock have been consistently present in critical habitat (with the exception of 2006). For Pacific cod, the relative density in critical habitat was higher in the 1980s compared to later years (Figures 4.11-4.13).

The summer NMFS bottom-trawl survey data were also evaluated for CPUE patterns within and outside of Steller sea lion critical habitat areas. Mean values of CPUE were computed inside critical habitat and compared to the mean CPUE outside of this region over time. Figure 4.15 shows a high degree of inter-annual variability of the relative CPUE inside Steller sea lion critical habitat compared to outside, especially for pollock (top panel). The bottom panel of this figure is the same data but aggregated into 5-

---

[26] See the 2001 Biological Opinion for an expanded discussion of these three considerations.

year blocks. This shows that there appears to be a downward trend in Pacific cod CPUE within the critical habitat relative to outside. Pollock, on the other hand, appears to show a stable to increasing trend in relative CPUE within critical habitat.

5.1.7.2 Steller Sea Lion Energetic Requirements (Daily)

Numerous bioenergetic experiments and models have been carried out for Steller sea lions. Winship *et al.* (2002) and Winship and Trites (2003) developed a bioenergetics model that predicted that Steller sea lions required 45–60% more food per day in early spring (March) than after the breeding season in late summer (August) because of seasonal changes in the energy density of the diets (along with seasonal changes in energy requirements). Overall energy requirements were lowest in the summer breeding season (June to August) and highest in the winter (December to February) and spring (March to May) mainly due to changes in activity budgets. Predicted relative daily food requirements were highest for young animals and decreased with age. The predicted mean daily food requirement of pregnant females was only marginally greater than the predicted mean daily food requirement of non-pregnant females of the same age. However, the model suggested that the mean daily food requirement of females nursing pups was about 70% greater than females of the same age without pups. Per capita food requirements differed by as much as 24% between regions of Alaska depending on the relative amounts of low energy-density prey (e.g., gadids) versus high-energy density prey (e.g., forage fish and salmon) consumed. Estimated food biomass requirements were highest in regions where Steller sea lions consumed higher proportions of low energy-density prey. Rosen (2009) concluded from captive feeding experiments that adult female Steller sea lions compensate by consuming more biomass during periods when low energy-density prey dominate the diet. Thus, in the wild, low-energy density prey must be readily available to adult female Steller sea lions during these periods such that they can consume more biomass and not become nutritionally compromised.

5.1.7.3 Local Prey Requirements

In contrast to broadly-integrative estimates of forage ratios used in previous Biological Opinions, recent field studies have calculated a type of forage ratio based on focal-area surveys of prey biomass relative to local Steller sea lion abundance. Sigler *et al.* (2004) inferred the seasonal importance of pre-spawning aggregations of eulachon in Berners Bay, southeast Alaska, to numbers of Steller sea lions hauled out nearby by comparing how many daily Steller sea lion energy requirements (daily ration) could be met by the prey biomass available, assuming a daily energy requirement of 79,464 kJ/day/Steller sea lion (calculated from Winship *et al.* 2002). Dividing the number of daily energy rations available by the number of sea lions present produced a forage ratio comparable to those estimated by comparing consumption requirements with biomass available. The forage ratio was 10-16 for Steller sea lions in Berners Bay feeding on eulachon for a few weeks in April-May. Thus, the prey available would meet the daily energy requirements for 10-16 times more Steller sea lions than were present in the area (assuming they could eat every last fish). Womble and Sigler (2006) compared numbers of Steller sea lions at Benjamin Island with availability of herring and pollock during 2001-2004. Seasonal numbers of Steller sea lions at Benjamin Island corresponded with peak herring availability, which represented up to 9,000 times more daily requirements than Steller sea lions present (and about 2,800 times in two other years). Sufficient pollock were always available to meet the daily energy requirements of the sea lions in the study area, and depending on the year of study pollock abundance would support 44 times more daily requirements than needed by Steller sea lions present.

Though not calculating a forage ratio in their study, Adams *et al.* (2008) estimated the August mid-water biomass available to Steller sea lions within 10 nm of the Chiswell Island rookery in 2003 and present enough information in their paper to derive a ball-park estimate. Adams *et al.* (2008) found the available prey dominated by herring and pollock, and present biomass availability on a per square nm basis.

Assuming a breeding population of 90 Steller sea lions on the rookery (Adams *et al.* 2008), an area of 314 square nm within 10 nm (ignoring a small amount of land mass), assuming prey energy densities based on Logerwell and Shaufler (2005), and the daily energy requirement of Sigler *et al.* (2004), in August of 2003 the available prey would support the daily needs of 107 more Steller sea lions than were likely present. This represents a multiplier of approximately 2 (i.e., comparing energy requirements of sea lions to energy available).

In the broadest focal-area study of prey availability, Sigler *et al.* (2009) estimated that 500-1,700 mt of prey within 10 km of shore in Frederick Sound attracted and sustained about 500 Steller sea lions during the non-breeding season. This amount represented 126 times more forage than required to meet the daily needs of 500 Steller sea lions. They also found that Steller sea lion numerical response to prey availability was non-linear in that more energy per Steller sea lion was necessary to attract greater numbers of Steller sea lions. Sigler *et al.* (2009; their Figure 7) showed a significant relationship between Steller sea lion count and nutritional energy available for a broad range of prey availability. Using the Sigler *et al.* (2004) daily ration there was 71-750 times more forage available than necessary to meet the daily needs of sea lions counted within the study area (their Figure 7). The number of Steller sea lions supportable by the energy available increased at a faster rate than did increases in observed Steller sea lions.

5.1.7.4 Regional Prey Requirements

There are several estimates of annual Steller sea lion consumption requirements and estimates of prey availability over broad spatial scales of varying scope. A very broad spatial and temporal forage ratio can be taken from Hobbs and Fowler (2008) who estimate a mean pollock consumption rate for 20 fish, bird, and mammal species to be 1.3% of the annual pollock production in the Bering Sea and North Pacific Ocean, where production was assumed to be 30% of the standing stock. This results in a ratio of 250 times more pollock biomass than is consumed by these species annually.

Appendix 3 of the FMP Biological Opinion (NMFS 2000) presented forage ratio estimates based on estimated Steller sea lion consumption to groundfish biomass availability, narrowed to availability of pollock, Atka mackerel and Pacific cod within Steller sea lion critical habitat. A multiplier of 54 (1 to 54) was calculated for the estimated 43,000 western DPS Steller sea lions in the BSAI/GOA. This multiplier used a consumption model (Winship 2000), a 1999 western DPS population abundance estimate, and groundfish biomass estimates in critical habitat. NMFS (2000) then estimated a forage ratio for a historic level of 200,000 Steller sea lions in an unfished environment based on extrapolation of the above model to be 1:21, which NMFS (2000) conservatively changed to a multiplier of 22. A second annual multiplier estimate of 46 for an unfished environment was based on Fowler (1999) and Perez and McAlister (1993). NMFS (2001) updated the Winship-based estimated multiplier for groundfish available in critical habitat to 55, using a changed groundfish biomass estimate.

NMFS (2001) noted the spatial scales of BS/AI and GOA are much larger than the geographic scale important to foraging by Steller sea lions, and both NMFS (2000) and NMFS (2001) explored forage ratios on slightly finer scales. As noted above, NMFS (2000) presented forage ratios by month for western DPS Steller sea lions using monthly biomass estimates of pollock, Pacific cod and Atka mackerel in critical habitat, giving an overall average multiplier of 113, ranging between 68 and 192. NMFS (2001, 2003) explored in more detail the underlying assumptions and subsequently estimated foraging ratios using biomass estimates of all groundfish by fishery areas of eastern Bering Sea, Aleutian Islands and GOA on an annual basis, and presented estimates of 446 times more forage available in eastern Bering Sea (this includes the eastern Aleutian Island Steller sea lion metapopulation area), contrasting with 11 times more in the Aleutian Islands (west of 178°W) and 17 times more in the GOA. It should be noted that Agency scientists discovered in the process of analyzing available information the forage ratios reported in NMFS (2001, 2003) were based on all groundfish species and both critical habitat and non-critical habitat in a given area, and not as stated for biomasses of pollock, cod and Atka mackerel in critical habitat.

Recently, Ortiz and Logerwell (2010) estimated biomass of Atka mackerel in the central Aleutian Islands area relative to their predators (such as arrowtooth flounder) and Steller sea lions, also predators. The objective of this study was to evaluate production of prey versus consumption of prey *vis-à-vis* size and location of trawl exclusion zones established under the premise of protecting prey resources for Steller sea lions. They found that production of Atka mackerel at most study sties was equal to or greater than estimated annual consumption for both Steller sea lions and other predators of Atka mackerel, including fisheries, but at their Amchitka North site estimated consumption exceeded production. They determined that estimates of consumption of Atka mackerel by other groundfish and Steller sea lions can either exceed estimated local production, or be well below it, depending on geographic location.

Regarding a determination of a threshold value for “sufficient” in evaluating the significance of the ratio of forage biomass available for annual consumption, it seems clear that a ratio of 1.0 is the mathematical limit for such a threshold. That is, a ratio of 1.0 implies that the predators are somehow just able to survive a given year on available production and then the forage biomass is reset and consumption can continue. This scenario seems highly unlikely from an ecological perspective. However, NMFS believes that the quantitative tools are not available at this time to reliably determine a threshold for this value that would necessarily be consistent with sufficient forage biomass to promote and achieve recovery of the western DPS of SSL, as defined under the ESA. Values of between 20 and 50 based on the work described in NMFS (2000) and Fowler (1999) in the preceding paragraphs of this section seem more realistic as a threshold for sufficient forage availability.

NMFS has updated the forage ratio estimates for SSL (i.e., forage biomass available divided by annual consumption) in the table below from those reported in NMFS 2001 and 2003 (see Fadely *et al.* 2010). Forage biomass was based on survey estimates of all groundfish, as reported in Fadely *et al.* 2010. Forage ratios are provided by region (e.g., Aleutian Islands, EBS, GOA) and by time period (data from surveys between 1997 and 2002 are referred to as “past” and data from surveys between 2004 and 2010 are referred to as “current”). Forage ratios are also reported for biomass in the entire area and biomass only from Critical Habitat (CH).

	Areas 543, 542, 541	Eastern Bering Sea	Gulf of Alaska
Past (all area)	19	322	26
Current (all area)	26	201	24
Past (CH only)	5	14	4
Current (CH only)	8	4	3

As noted elsewhere in this Biological Opinion, forage ratios are very difficult to interpret relative to the adequacy of forage to provide for the recovery of Steller sea lions in waters off Alaska. Part of this difficulty is related to the coarseness of reliable estimates of forage biomass relative to the feeding ecology and movements of Steller sea lions. Part of the difficulty is also related to uncertainty in describing the seasonal movement patterns of sea lions, as well as their dependency on forage that occurs inside and outside of critical habitat. Finally, it should be noted at this time NMFS has no reliable way to ascertain the uncertainty associated with a forage ratio estimate for a given area and for a given time period.

Nonetheless, some interpretation of these data is possible. First, it appears that the forage ratio in the Bering Sea is likely to be larger than the ratio in the other two regions (which appear to be similar in value). This difference is more than an order of magnitude and could possibly be a factor in explaining the increasing abundance of sea lions in the eastern Bering Sea. In addition, the forage ratio in the Bering Sea seems to have declined over the last decade. This would be consistent with our understanding that groundfish biomass has declined, while sea lion numbers have increased, over the last decade in the Bering Sea area. However, to the contrary, forage ratios in the GOA and Aleutian Islands, while similar in value, are associated with trends in sea lion abundance that appear to differ between these two areas. Further, forage ratios in the Aleutian Islands are in the range of 20 to 50 that should theoretically support sea lion recovery, yet sea lion numbers in this area appears to be in decline.

Finally, it appears that forage availability in critical habitat is an order of magnitude less than for the entire area. It is certainly not surprising that the estimated forage ratio would increase when comparing a relatively small area with a larger area. However, these data indicate that forage outside of critical habitat, at least in certain times of the year, is likely to be important to the successful foraging of sea lions.

Fadely *et al.* (2010) also provide estimates of forage ratios for SSL by fishery management area in the Aleutians based on biomass of all groundfish and biomass of pollock, Pacific cod, and Atka mackerel:

	All groundfish (kt)	Consumption (kt/yr)	Ratio	Pollock, cod, Atka mackerel (kt)	Ratio
543	721.7	17.8	40.6	464.5	26.1
542	702.3	53	13.2	285.3	5.4
541	900.0	59.5	15.1	267.2	4.5

As was the case for forage ratios at the marine ecosystem scale, forage ratios by fishery management regions are not correlated with SSL trends in abundance aggregated at this same scale. That is, annualized trends in non-pup counts from fishery management areas 543, 542, and 541 was -6.83%/yr, -2.33%/yr, and -0.39%/yr. This finding was the case whether one considered only pollock, cod, and Atka mackerel biomass or the biomass of all groundfish species. As was the case for the larger spatial scale, the amount of forage biomass in critical habitat alone in areas 543, 542, and 541 appears to be insufficient by itself to support efficient foraging (i.e., relative to forage ratios of 20-50; see above).

5.1.7.5 Summary of the Response of Critical Habitat

When estimating Steller sea lion forage ratios for studies associating surveys of prey availability with Steller sea lion abundance, an assumption must be made to relate the amount of support the local Steller sea lion population derives from that resource. That is, since both Steller sea lions and prey are dynamic in distribution, over what duration does the standing biomass support the associated Steller sea lion population? As an index similar to the forage ratio, Sigler *et al.* (2004) estimated the daily energy requirements for sea lions present in the study area, and calculated the number of these daily rations that could be supported by the measured standing biomass. There is a direct and strong association between available biomass and Steller sea lion abundance because the series of studies in Southeast Alaska clearly showed numerical responses of Steller sea lions to varying amounts of biomass availability (Sigler *et al.* 2004, Womble and Sigler 2006, Sigler *et al.* 2009). These forage ratios indicate the amount of biomass that attract and support Steller sea lions, and can perhaps be considered as a “snap-shot” view in contrast to the broadly-integrative forage ratio estimates calculated from estimates of annual or monthly Steller sea lion energy requirements relative to estimates of groundfish biomass within all or regions of critical habitat (NMFS 2000, 2001).

Fishery removals can change the composition of the fish community with associated effects on the distribution and abundance of prey organisms. Fisheries may also affect Steller sea lions through interactive competition. Examples of interactive competition include disruption of normal Steller sea lion foraging patterns by the presence and movements of vessels and gear in the water, abandonment of prime foraging areas by Steller sea lions because of fishing activities, and disruption of prey schools in a manner that reduces the effectiveness of sea lion foraging. The hypothesis that these types of interactive competition occur cannot be discounted, but also cannot be quantitatively evaluated with the information currently available and will not be addressed further in this section (see Section 4.5.3).

5.1.7.6 Testing Relationships between Fishing and Response of Prey in Critical Habitat

In this Biological Opinion we have shown that fisheries cannot be excluded as a factor that affects Steller sea lion population dynamics and their habitat on a number of levels. However, it should be noted that several analyses failed to show statistically significant impacts of commercial fisheries on the western DPS of Steller sea lion. Nonetheless, short-term effects are described in Section 4.5.3 and suggest a potential interaction between the immediate effects of fishing and Steller sea lions life history and population dynamics. Long-term effects are described in Section 4.5.4, and we suggest how the exploitation strategy over time could alter the distribution of prey, size structure, biomass, and underlying productivity of prey species. In addition, in Section 4.5.3.3 we explored the evidence that even though fisheries may attempt at the global scale to "fish proportional to biomass" that those relationships often break down at the local level due to spatial heterogeneity of prey habitat, biomass, or due to the effectiveness of fisheries themselves in removing prey in local areas to levels below those predicted under the global fishing rate. Further, in Section 4.5.4.3 we discuss the potential operational effects of fishing on the local aggregations of prey. We acknowledge that the elusive cause-effect connection between the catch of fish in "Boat A" and response of "Steller sea lion B" will likely never be made. Some experiments have attempted to observe Steller sea lion movements and the offshore fisheries, but synthesis of these complicated experiments has been problematic. Because Steller sea lions are a long lived species that freely move across artificially created human boundaries (management regions), utilize a variety of prey, and respond to changes in their environment differently depending on the season and age/sex group considered, a series of relatively modest local effects is likely to be integrated across years and subpopulations. These circumstances would be consistent with a chronic rather than an acute response relationship. However, there is evidence that acute responses to nutritional stress may have occurred in the 1980s. We have considered evidence that after the implementation of the conservation measures in the early 2000s the Agency is not required to establish a statistically significant cause-and-effect relationship under the ESA. Rather as noted above, the ESA requires the Agency to conclude that a given action is not likely to jeopardize the continued existence of a DPS or adversely modify its critical habitat.

The use of catch amounts or the proportion of catch inside critical habitat is an incomplete measure of fishing impacts on Steller sea lion prey availability. For example, if biomass inside critical habitat is proportional to stock biomass, an increased catch or proportion of catch in critical habitat would imply an increased amount of prey available to Steller sea lions. In some cases, fishery responses (e.g., reduction in catch in critical habitat, increases in percent taken inside critical habitat) are a direct response to changes in the distribution of prey within a given management area and are not indicative of a change in fishery regulations. Clearly, we assume that fishermen will go to locations that maximize benefits while reducing costs, within the regulatory framework that constrains their actions. Thus, increases or decreases in any one fishery metric do not necessarily correlate with a subsequent negative impact on critical habitat or on Steller sea lions; it simply is a description of what occurred using the best available data.

The harvest rate analysis (Section 5.1.4) assesses potential impacts of fishing on Steller sea lion prey availability at the NMFS management area and multi-year scale by incorporating abundance of prey over large regions. However, fine-scale analysis of fishery operations and Steller sea lions at a local scale are precluded by the spatial and temporal resolution of the fishery abundance data used to determine harvest rates.

Earlier in this section we described Steller sea lion energetic requirements highlighting critical life-history characteristics, specifically the sensitivity of adult females and juveniles to reductions in prey quality and availability. We then explored local prey requirements and found that local, small-scale movements of Steller sea lions are related to the temporal and spatial distribution of prey; the presence and absence of Steller sea lions could be explained by the observed changes in prey densities. We then walked through a more detailed description of forage ratios and their relationship to Steller sea lion population dynamics. Trends in Steller sea lion abundance as estimated from changes over time in pup production and numbers of non-pups integrate all of these factors, plus the indirect factors described above including oceanographic conditions. No one stressor or response can be relied upon as a cause-effect relationship; however when the impact of multiple stressors is integrated, their combined effect on Steller sea lion response becomes more apparent.

The synthesis of these discussions will occur in Chapter 7. The effects of the action are summarized below:

- Fisheries have short-term effects on the prey field for Steller sea lions both within and outside of critical habitat including reduction in local prey amounts from what would have existed without fishing, and thus fisheries may have disproportionate impacts on a local scale relative to the global harvest rate due to localized depletions and spatial heterogeneity of prey habitat
- Fisheries have long-term effects on the prey field for Steller sea lions both within and outside of critical habitat due to changes in prey size, distribution, and productivity
- Oceanographic conditions vary across the Steller sea lion range and may exacerbate or ameliorate the implementation of the fisheries in the short-term
- Fishery impacts are likely confounded by regionally specific factors, such as predation and contaminants.
- At this time with available data, it is not possible to demonstrate a statistically significant relationship between commercial fisheries on pollock, cod, Atka mackerel and arrowtooth flounder and the productivity of Steller sea lions in the western DPS. However, it is also not possible with the available data to conclude that commercial fisheries are not having a significant impact on the recovery of the western DPS of the Steller sea lion.

5.2 Steller Sea Lion: Eastern DPS and Critical Habitat

The eastern DPS range extends west from 144°W longitude along Alaska's southern coast, and south to California. In 2008, the population of Steller sea lions in the eastern DPS was estimated at 63,000 animals. In contrast to the decline and lack of a robust recovery documented in the western DPS, the eastern DPS increased at over 3% per year between the late 1970s and 2002, more than doubling in abundance in southeast Alaska, British Columbia, and Oregon. Currently, this population is estimated to be at its highest level in recent history. Since 1996, pup production in southeast Alaska has increased at a rate of 5% per year, while non-pup numbers are increasing at 3.7% per year. Surveys of pups and non-pups conducted in British Columbia in 2006 and in southeast Alaska, California and Oregon in 2009 indicate that the overall eastern DPS trend described by Pitcher *et al.* (2007) continued through 2009, and is particularly strong in the northern portion of the eastern DPS in southeast Alaska and British Columbia.

5.2.1 Effects of the Fisheries on the Eastern DPS of Steller Sea Lion

Steller sea lion populations in southeast Alaska increased at 3% per year through 2002 and likely at greater rates than that between 2002 and 2009 (DeMaster 2009). NMFS recognizes that there may be a small number of western DPS animals present in southeast Alaska, but that this may constitute an extremely small number of animals relative to the current population. Thus the majority of the increase in the eastern DPS is due to production by animals born in the eastern DPS, and only a small fraction due to production by animals born in the west that gave birth in the east. There are no fisheries for the principal groundfish prey species within critical habitat in southeast Alaska, and only very small fisheries for those species in the range of the eastern DPS, even though large, seasonal commercial salmon fisheries harvest the principal prey item (salmon – Sinclair and Zeppelin 2002) of Steller sea lions in Southeast Alaska. Given the overall sustained increase in the eastern DPS for at least 30 years and the migration of some animals from the western DPS to the eastern DPS (with evidence of pupping of those females), it is likely that habitat conditions in the eastern DPS provide for adequate survival and the ability to recover based on long-term demographics.

5.2.2 Response of Steller Sea Lions

Pup production in Southeast Alaska (eastern DPS) totaled 7,462 pups in 2009, with 7,443 counted at the five major rookeries where 5,510 were counted in 2005. The increase of 1,933 in rookery pup production since 2005 equates to approximately 97 more pups per year at each of the southeast Alaska rookeries. Pup production in southeast Alaska increased at the rates of 5.0% per year since 1996 and 3.6% per year since the late 1970s. Between 2001/2002 and 2009, rookery pup production increased 50% (from 4969 to 7443) in southeast Alaska, which is equivalent to an increase of approximately 62 pups per rookery per year. These trends indicate that recovery in the eastern DPS is steady and is continuing.

The eastern DPS is likely to continue to increase and appears to be large, healthy, and based on Goodman (PVA in NMFS 2006a) has a very small likelihood of becoming endangered.

5.2.3 Effects of the Fisheries on Critical Habitat

Critical habitat within the eastern DPS of Steller sea lion consists of a 3,000 ft zone seaward of the three designated rookeries, a number of haulouts, and some identified terrestrial areas. There is no directed fishery for the principal groundfish prey species within critical habitat in southeast Alaska. In addition, fishing with trawl gear in Southeast Alaska has been prohibited since 1995. There is very little removal of groundfish prey species of importance to Steller sea lions in this nearshore area designated as critical habitat in the eastern DPS.

5.2.4 Response of Critical Habitat

Given that there is very little removal of groundfish prey species of interest to Steller sea lions in this nearshore area designated as critical habitat in the eastern DPS, it is reasonable to conclude that there is no substantial effect on critical habitat within the eastern DPS. Based on the effects of the proposed action, it is likely that designated critical habitat within the eastern DPS is likely to remain functional (or retain the ability to become functional) to serve the intended conservation role for the species (eastern DPS) in both the near and long-term.

5.3 Humpback Whales

5.3.1 Prey Resources

Humpback whale prey species are not targeted or taken in significant amounts by the fishery actions evaluated in this opinion: (1) the groundfish fisheries for the Bering Sea and Aleutian Island Management Area, (2) the groundfish fisheries for the GOA, and (3) all State of Alaska parallel groundfish fisheries. Because humpback whales forage primarily upon small schooling fish (herring, eulachon, capelin, sand lance) and zooplankton (krill), it is unlikely that groundfish fisheries would compete for prey resources.

5.3.2 Fishery Interaction and Entanglement

Gear entanglements may debilitate, seriously injure, or kill humpback whales. Between 2002 and 2006, there were incidental serious injuries and mortalities of central North Pacific humpback whales in the BSAI sablefish pot fishery (Angliss and Allen 2009). Numerous entanglements have also been documented by the Alaska Marine Mammal Stranding Network in the GOA and BSAI regions (see Environmental Baseline). Given that it is often difficult to confirm the gear type in reported entanglements based upon a lack of information, it is unclear to what extent entanglements reported to the stranding network in Alaska involve groundfish fishing gear.

Over the last 30 years, over 100 humpback whale entanglements have been reported to the NMFS Alaska stranding program. Most of these incidents have been reported from the inside waters of southeast Alaska, although the locations where the animals have become entangled are rarely determined. Regions in southcentral Alaska, notably Kodiak, Homer and Seward, also consistently report humpback whale entanglements. It is unclear which types of entanglements are ultimately life-threatening, though recent scarring research in northern southeast Alaska has shown that many humpback whales survive their entanglements.

About half (51%) of the gear reported in Alaska humpback whale entanglements between 1997-2009 involves crab, shrimp, black cod and unidentified pot gear (see Environmental Baseline). Approximately 3% of the entanglements during this time were reported to the stranding network as longline gear; several of these events involved longline gear attributable to the sablefish fishery. Some of the unidentified pot gear incidents (22%) may also involve groundfish fisheries.

Overall, the number of entanglements that that might result from interactions with groundfish fisheries appears to be low in contrast to other gear types. For such events that do occur with individual whales, the extent of entanglement from groundfish fisheries is not expected to have negative consequences for humpback whales in the North Pacific.

5.3.3 Ship Strike

It is not known to what extent the increased vessel traffic in the action area will result in humpback whale injury or mortality due to ship strikes. Vessels engaged in groundfish fisheries will likely result in increased disturbances to whales and pose a higher risk of collision than those posed by baseline conditions. The risk of vessel collision is likely to be higher during the summer months when the population of humpback whales in Alaska is at its peak. Throughout the remainder of the year, the chance of collision is likely to be low given the limited usage of the action area by humpback whales.

The incidence of ship strikes and/or serious injury from vessels involved in the groundfish fisheries are likely to be negligible, as fishing vessels usually operate at slow speeds and often spend their time in the pelagic environment rather than inside waters where humpbacks tend to forage. Although a heightened risk of collision may impact individual humpback whales using the action area, it is unlikely to have population level consequences for humpback whales in the North Pacific.

5.3.4 Disturbance

Humpback whales may be disturbed by noise from fishing vessel engines. Research has suggested that noise may cause humpback whales to avoid or leave feeding or nursery areas (Jurasz and Jurasz 1979). Other research has suggested that humpback whales may become habituated to vessel traffic and its associated noise. Still other researchers suggest that humpback whales may become more vulnerable to vessel strikes once they habituate to vessel traffic (Swingle *et al.* 1993, Wiley *et al.* 1995). While measurable startle responses might diminish with time, this does not necessarily indicate that a negative impact has not occurred. Vessels could still cause physiological stress impacts or could disrupt prey aggregations forcing whales to spend a greater amount of time and energy foraging (NMFS 2005).

A number of studies have been conducted in areas with seasonally high numbers of humpback whales to assess short-term impacts of vessel activity (e.g. Peterson 2001). Studies of vessel impact to marine mammals have most often looked at short-term effects (e.g., measuring disturbance or avoidance behaviors) rather than longer-term or cumulative effects of repeated exposure to numerous vessels over time (e.g., decreased survivability or reproductive effects such as increased birthing intervals which are directly related to productivity). Immediate responses to vessel presence, such as avoidance behavior or changes in dive patterns, can be measured more easily; longer-term effects can often be difficult to define and to measure. Typical measures of a whale's reaction to the presence of a vessel have been visible changes in behavior, such as avoidance reactions or displacement; increased fluke or flipper activity; changes to blow intervals, dive patterns, swimming orientation and speed. These reactions are measurable and can be assumed to have a certain energetic cost. However, animals could also incur an energetic cost through behaviors that are not necessarily measurable (i.e., physiological stress responses such as increased heart rate or pathological conditions) (NMFS 2005).

Behavioral alterations in the presence of fishing vessels could result in an individually incurred energetic cost, causing whales to leave the action area if sufficiently disturbed. Displacement may adversely affect individual animals by requiring additional energy investment to forage elsewhere, and thus may translate into the reduced fitness of an individual. However, the effect of such displacement on individual humpback whales, if it were to occur, would not compromise the recovery or survival of the species.

In many cases, groundfish fishing vessels target different areas than those where humpback whales display high foraging site fidelity (e.g., Frederick Sound, Icy Strait, Lynn Canal, Kachemak Bay). In addition, these vessels are not targeting humpbacks in the manner that whale-watching vessels do by remaining in their vicinity for extended periods of time. What disturbance individual animals may experience by passing fishing vessels is not expected to be of a magnitude to impact a significant portion of the population in the action area.

5.3.5 Effects of the Action on Critical Habitat

There is no listed critical habitat for North Pacific humpback whales.

5.4 Sperm Whales

5.4.1 Prey Resources

Sperm whales in the GOA feed primarily on squid and fish and may depend on some of the same species that are harvested in the groundfish fisheries. Sperm whales have demonstrated their preference for sablefish through their predation on sablefish longline sets (NMFS 2006). They are also known to feed on Atka mackerel, Pacific cod, and pollock, all species targeted by the fisheries reviewed in this opinion. This results in the potential for competition between sperm whales foraging for prey species and groundfish fisheries in the GOA; however, the extent of this impact is currently not well understood. Although some sperm whales have learned depredation behavior associated with sablefish fisheries and consistently engage in this activity, it appears that their motivation is a reliable and easy source of prey rather than a lack of prey available in the natural environment. The potential does exist for a reduction in prey biomass to occur in the future, which could threaten sperm whales' ability to successfully forage.

Currently, however, there is no evidence that sperm whale diet is compromised by the groundfish fisheries in Alaska.

5.4.2 Fishery Interaction and Entanglement

Sperm whale depredation in longline fishing operations in Alaska has been increasing throughout the last decade. Most recorded episodes of predation on catch have occurred in the GOA rather than in the BSAI region; thus, this is where most exposure is likely to occur resulting in fisheries interactions and entanglements.

The nature and extent of interactions between sperm whales and longline gear is not yet clear. However, in 1997 and 2000, one sperm whale in each year was observed entangled and seriously injured by longline gear in the eastern portion of the GOA. Beyond these two events, available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions. Researchers have observed that sperm whales predating on longline gear appear to be able to avoid becoming entangled. No mortalities have been observed, though it is possible that whales may break through or carry off trailing gear and become debilitated, injured, or die as a result, with no observation of the event. Future potential for entanglement may be reduced by deterrent research in the eastern GOA to reduce predation by sperm whales on sablefish longlines.

In general, the incidence of sperm whale entanglement in Alaska appears to be low. For those cases in which current groundfish operations may adversely affect individual whales, effects would not be expected to rise to a level which would negatively impact Alaska sperm whale population viability.

5.4.3 Ship Strike

Ship strikes from fishing vessels are possible due to the overlap of sperm whale habitat with vessels involved in the Alaska groundfish fisheries. The incidence of ship strikes and/or serious injury from ship strikes involved in the groundfish fisheries are likely to be minimal, as such fishing vessels usually operate at slow speeds and often are forewarned of sperm whale presence in their vicinity due to depredation behaviors. Although sperm whale rafting behavior at the surface between deep dives could make them vulnerable to ship strikes, few reports exist to provide evidence for this. In general, fewer world-wide reports exist of vessel collisions with sperm whales than for many other large whale species, including humpback, fin, right, gray and minke whales (Jensen and Silber 2003). Further, there are very few reports of sperm whale ship strikes in Alaska; NMFS Alaska Region stranding records contain only one fishing vessel incident from 1997 which occurred outside Prince William Sound (see Environmental Baseline).

Certainly, some carcasses may go unreported, while cause of death for others may never be identified to implicate ship strike. Between 1976 and 2009, 31 sperm whale incidents were reported to NMFS Alaska marine mammal stranding program. The cause of death for most of these cases is unknown; it is possible that some could have been due to ship strike. Despite these considerations, what injury and mortality may occur from vessel collisions is not expected to result in an adverse population level effect for sperm whales in Alaska.

5.4.4 Disturbance

Sperm whale responses to vessel approaches have indicated that the animals adjust their diving and acoustic behavior to boats. With frequent exposure, whales have also become increasingly tolerant of vessels' presence. In Alaska, sperm whales appear to be much attuned to acoustic cues from longline vessels hauling their catch, particularly when they are within 10 nm of the activity. Given that many individual sperm whales are attracted to the sound of groundfish vessel engines and gear hauling catch, it would appear that they do not interpret such noise as disturbance. On the contrary, it is possible that engine noise from fishing vessels represents a source of disturbance that could otherwise displace the whales or alter their behavior, but the prey rewards available through depredation may outweigh this response and cause animals to remain near vessels or habituate to them. However, in light of the fact that depredation behavior in Alaska is only known to involve male sperm whales, any theoretical population impact that could result would be lessened by not affecting females in the population.

5.4.5 Effects of the Action on Critical Habitat

There is no listed critical habitat for the Alaska population of sperm whales.

5.5 Fin Whales

5.5.1 Prey Resources

Fin whales feed primarily on zooplankton and fish and may depend on some of the same species harvested in the groundfish fisheries (e.g., pollock). This results in the potential for competition between fin whales foraging for prey species and groundfish fisheries in the GOA and BSAI; however, the extent of this impact is currently not well understood. The potential does exist for a reduction in prey biomass to occur in the future, which could threaten fin whales' ability to successfully forage. Currently, however, there is no evidence that fin whale diet is compromised by the groundfish fisheries in Alaska.

5.5.2 Fishery Interaction and Entanglement

Prior to 1999, there were no observed or reported mortalities of fin whales incidental to commercial fishing operations within the range of the Northeastern Pacific fin whale stock. However, in 1999, one fin whale was killed incidental to the Gulf of Alaska pollock trawl fishery. This take occurred in federal waters of statistical area 620, southwest of Kodiak Island (Perez 2003). Between 2002 and 2006, there was one observed incidental mortality of a fin whale in the Bering Sea/Aleutian Island pollock trawl fishery, resulting in a mean annual mortality/serious injury of 0.23 fin whale/yr incidental to commercial fisheries in Alaska for this 5 yr period (Angliss and Allen 2009).

5.5.3 Ship Strike

Ship strikes from fishing vessels are possible due to the overlap of fin whale habitat with vessels involved in the Alaska groundfish fisheries. The incidence of ship strikes and/or serious injury from ship strikes involved in the groundfish fisheries are likely to be minimal, as fishing vessels usually operate at low speeds. Laist *et al.* (2001), Jensen and Silber (2003), Vanderlaan and Taggart (2007), and Van Waerebeek and Leaper (2008) compiled information available worldwide regarding documented collisions between ships and large whales. From 1993–2002, one fin whale was struck but appeared uninjured in Alaska (Jensen and Silber 2003).

The possible impact of ship strikes on recovery of fin whale populations is not well understood. Because many ship strikes go unreported or undetected for various reasons and the offshore distribution of fin whales may make collisions with them less detectable than with other species. Ship strikes were classified as a medium threat in terms of severity with a high level of uncertainty in the Fin Whale Recovery Plan (NMFS 2010). However, no fin whale ship strikes have been documented incidental to the Alaska groundfish fisheries. Certainly, some carcasses go unreported, though what injury and mortality may occur from vessel collisions is not expected to result in an adverse population level effect for fin whales in Alaska. The minimum mean annual mortality/serious injury from ship strikes for all vessel types in

Alaska is 0.20 fin whale/yr. The mortality/serious injury attributed to strikes by Alaska groundfish vessels is assumed to be lower and could be zero.

5.5.4 Disturbance

Short-term changes in fin whale behavior have been documented when approached by a vessel. Documented responses to the presence of vessels include decreasing dive times, surface times, and number of blows per surfacing, and abandoning feeding (Stone *et al.* 1992; Jahoda *et al.* 2003).The low-frequency sounds used by fin whales for communication and (possibly) in courtship displays (Watkins 1981) could be masked or interrupted by ship noise. The Final Fin Whale Recovery Plan classified vessel disturbance as a low severity level with a medium level of uncertainty (NMFS 2010). Effects of vessel disturbance from vessels involved in the groundfish fisheries are not expected to be of a magnitude to impact a significant portion of the population of Northeast Pacific fin whales.

6 CUMULATIVE EFFECTS

6.1 Overview

Cumulative effects include the effects of future state, tribal, local or private actions, not involving federal activities, that are reasonably certain to occur in the action area considered in this Biological Opinion (50 CFR 402.02). Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the Act. Past and present impacts of non-Federal actions are part of the environmental baseline discussed in Section 4 of this Biological Opinion. Cumulative effects that reduce the capacity of listed species in the action area to meet their biological requirements increase the risk to the viability of the species, and consequently increase the risk that the proposed action on the species or its habitat will result in jeopardy (NMFS 1999). The cumulative effects of future state, tribal, local, and private actions on Steller sea lions and their critical habitat and other affected species, including both lethal and nonlethal direct and indirect effects, are considered below.

The 2008 Revised Recovery Plan for Steller sea lions (NMFS 2008a) provides the most comprehensive recent review of various threats to the recovery of the eastern and western DPSs of Steller sea lion. Thus, it serves as a source document for discussion and evaluation of human and natural factors that have, may be currently, and may in the future affect the population status and health of Steller sea lions in both population segments.

Other summaries or syntheses of factors that may affect recovery of these populations include the Final Programmatic EIS (NMFS 2007b) and the Summary Document (NMFS 2009b), the 2008 Alaska Marine Mammal Stock Assessments (Angliss and Allen 2009), Atkinson *et al.* (2008), MMS' Final EIS on Cook Inlet Oil and Gas Lease Sales 191 and 199 (MMS 2003) and chapters in Trites *et al.* (2006). Loughlin and York (2001) provided a detailed accounting of the various sources of Steller sea lion mortality, including anthropogenic sources and predation.

For the western DPS of Steller sea lion, the Recovery Plan threats assessment concluded that the following threats pose a relatively minor threat to recovery currently: (1) Alaska Native subsistence harvest, (2) illegal shooting, (3) entanglement in marine debris, (4) disease, and (5) disturbance from vessel traffic and scientific research. The threats assessment also concluded that "…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)." These threats are presented in Table IV-1 of the Steller Sea Lion Recovery Plan.

The availability of new information since the publication of the Recovery Plan in early 2008 has increased our uncertainty about disease, entanglement, Alaska Native subsistence hunting (in the central Aleutians), and pollution for the western DPS. Since the release of the Recovery Plan, phocine distemper virus has been confirmed (Goldstein *et al.* 2009) to now infect sea otters in parts of the North Pacific within the range of the Steller sea lion. This increases our concern and uncertainty about the potential risk of disease to the recovery of the western DPS. New unpublished data has emerged from studies by ADF&G that indicate current published estimates of entanglement in, and ingestion of, marine debris may be underestimates of actual levels. Thus, there is increased uncertainty about this threat. Lastly, the level of subsistence take at Atka in the central Aleutians is high relative to the abundance of Steller sea lions in the area and overall non-pups counts at Atka were down sharply in the 2008 count. Thus, we are concerned that subsistence take in this local area could be impeding local recovery and contributing to the downward trend in the central Aleutians as a whole. We also note that since the writing of the Recovery Plan, plans for State of Alaska oil and gas leasing and lease sales in areas north of the Alaska Peninsula have occurred. This increases our uncertainty about the threat posed by pollution.

The threats assessment in the 2008 Recovery Plan identified no threats to continued recovery for the eastern DPS of Steller sea lion. In the Recovery Plan NMFS (2008a) concluded that: "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." Since the Recovery Plan was finalized additional information has become available from the ADF&G that indicates that the incidence of entanglement of Steller sea lions in the eastern DPS may occur with greater frequency than had been documented at the time of finalization of the Recovery Plan. Additionally, a disease (phocine distemper virus) that is likely novel to the North Pacific has now been detected in another species of marine mammal (sea otters) in multiple portions of the range in which individuals from this Steller sea lion population are known to feed and haul out. Hence, there is new information that increases our uncertainty about potential threats to this population, but no information indicates that recovery is being impeded by these factors.

In the Environmental Baseline section of this opinion (Section 4), past and present anthropogenic activities and natural processes within the action area affecting Steller sea lions and other affected species and Steller sea lion critical habitat are described in detail. The table in Section 4.3.10 shows estimates of mortality from human activities since 1958. This table estimates a total of 1,900 Steller sea lions killed from human activities between 2000 and 2004, a substantial decline from the 1990s level of takes from subsistence harvests, shootings, and incidental catch and entanglement.

Many factors that have affected Steller sea lions within the action area in the past are likely to continue to affect them in the future (pollution, disease, predation, competition for prey, etc.). However, some factors thought to have contributed to the decline of Steller sea lions have been mitigated so that the level of effects is substantially reduced (e.g., intentional, non-subsistence-related, shooting) or eliminated in some cases (e.g., commercial harvests). Given available information, we assume that for most factors the current level, not the historic level, of effect is that which we anticipate in the foreseeable future.

6.2 Subsistence Harvest

We provided information about past and current levels of subsistence harvest of Steller sea lions by Alaska Natives in the environmental baseline section. Harvest of Steller sea lions by Alaska Natives results in direct lethal takes and we expect subsistence harvest to continue into the foreseeable future. As noted in the Baseline Section, data on the subsistence harvest of Steller sea lions is collected by both the ADF&G working with the Alaska Native Harbor Seal Commission (ANHSC) and the Ecosystem Conservation Office (ECO) of the Aleut Community of St. Paul. ADF&G and the Alaska Native Harbor Seal Commission have recently been collaborating to document subsistence harvest through the use of a retrospective survey undertaken in a large number of villages in coastal Alaska. As noted above, documented annual take of Steller sea lions by subsistence hunters between 1998 and 2007 ranged from 171-218, with 217 animals reported taken in 2007 (Wolfe *et al.* 2009). Confidence intervals around these estimates are large (e.g., in 2007, the 95% confidence interval was 147-324).

The vast majority of the reported takes come from just a few locations located within the range of the western DPS, including the Pribilofs, the central Aleutians, and Prince William Sound. Patterns over the past years have been variable with levels of take increasing in some areas (e.g.,Tatiklek) and decreasing in others. Thus, it is hard to predict future patterns. In the western DPS, Angliss and Allen (2008) reported a mean annual take of Steller sea lions by subsistence hunters of 135. This mean level of take represents a large proportion of the potential biological removal calculated for the western DPS of Steller sea lions (e.g., PBR was 234 in 2008) (Angliss and Allen 2009).

The reported take levels associated with subsistence harvest in the eastern DPS are low (e.g., 6.1 with an upper estimate of 9.5 in 2007) and, as concluded in the Recovery Plan, should not impede recovery of that DPS.

Levels of harvest in a few locations in the western DPS (e.g., Atka, where the total take estimate for 2007 was 54, with an upper range estimate of 87.2) could contribute substantially to the already downward trend in the local area and contribute to the overall downward trend in the subarea. Non-pup counts at North Cape on Atka fell drastically in the last four years (2000: 76; 2002: 224; 2004:383; 2006: 279; 2007: 140; 2008:34) and have risen slightly at Cape Korovin (2000:12; 2002:1; 2004: 4; 2006: 0; 2007: 30; 2008:39). As we concluded in the 2007 and 2009 Biological Opinions on the effects of Research on Steller Sea Lions and Northern Fur Seals, the overall future impact of the subsistence harvest on the western population will be determined by the number of animals taken, their sex and age class, and the location where they are taken. As with other sources of mortality, the significance of subsistence harvesting may increase, especially in certain areas, such as the western or central Aleutian Islands, if Steller sea lion abundance continues to decline. The future subsistence harvest may contribute to subarea-wide or localized declines of Steller sea lions and/or impede recovery if the harvest is concentrated geographically.

6.3 Entanglement in Marine Debris

As noted above, and in the Baseline section, in the Recovery Plan NMFS (2008a) concluded that the threat to Steller sea lions from entanglement was low. Levels of entanglement captured in the stranding database are low. However, recently published (Raum-Suryan *et al.* 2009) information collected during resighting trips by ADF&G indicate that only a small percentage of the entanglements, at least in Southeast Alaska, are reflected in the currently available entanglement estimates. During an eight year (2000-2007) study in Southeast Alaska and northern BC, researchers quantified Steller sea lion entanglement by sex and age class of the entangled animal, debris type, entanglement incidence (entanglement incidence was 0.26%, SD = 0.0064, n = 69 sites), and estimated population level effects. They observed 386 Steller sea lions of all age classes that were either entangled or that had ingested fishing gear. Most of the entangling debris that could be identified was fishery based (e.g., packing bands, rubber bands, net, rope, line). About half of the Steller sea lions had some kind of neck entanglement whereas about half of the animals had ingested either commercial or sport lures. Over half (54%) of the neck entanglements observed in which the entangling material could be identified were plastic packing bands which Raum-Suryan (2009) identify as being used to secure cardboard bait boxes. Rubber bands (which the authors state is used to secure rain gear and is used in commercial crab fisheries) were another common (30%) visible material causing neck entanglement in this study. Other identified materials causing neck entanglements were: net (7%), rope (7%), and monofilament line (2%). Animals were also observed that had ingested fishing gear such as salmon fishery flashers (lures: 80%), longline gear (12%), hook and line (4%), spinners/spoons (2%), and bait hooks (2%).

These entanglements, especially the neck entanglements, pose a lethal threat to affected animals. Raum-Suryan *et al.* (2009) stated that, "Neck entanglements are especially lethal to animals that become entangled at a young age. As a sea lion grows, the entangling material tightens, eventually strangling the animal. Lesions from netting or packing bands are often infected and associated with necrotic tissue and if the infection surpasses the ability of the lymph system to control it, the lungs will often become infected, often leading to mortality" (Angliss and DeMaster 1998). In addition, microbes that enter the blood stream can cause secondary infections in the heart (e.g., heart valves), brain, or other vital organs (Angliss and DeMaster 1998). These entangling materials injure and kill not only Steller sea lion but many other marine species worldwide (Laist 1997).

These observations indicate that current estimates of entanglement, including entanglements that are known to have caused death and those that almost certainly caused death or serious injury, may be serious underestimates. However, there also may be strong spatial and/or temporal differences in the frequency of entanglement. Results from Raum-Suryan differ greatly from those of a study of the frequency of Steller sea lion entanglements in the Aleutian Islands in 1985 in which only 0.07% of the counted adult population (11 Steller sea lions) had evidence of entanglement with debris (Loughlin *et al.* 1986). In that study no Steller sea lions were observed to be entangled with packing bands. Identified materials in which Steller sea lions were entangled included trawl net or twine. However, in a recent study in the Pribilof Islands, Zavadil *et al.* (2007) reported a high incidence of plastic packing bands in the debris observed on and removed from Northern fur seals.

Based on this new information, we acknowledge increased uncertainty about the future threat posed by entanglements. At present, we do not have information sufficient to adequately document recent and current entanglement frequency in most parts of the range or to indicate whether or not current rates of Steller sea lion entanglement are likely to increase or decrease in the foreseeable future.

6.4 State managed Commercial Fisheries

Fisheries that occur in state waters and are counted toward the federal TAC were analyzed as part of the action in this opinion (refer to Chapter 2 – Description of the Action). Additional ADF&G groundfish fisheries in state waters and in the EEZ that are not managed under parallel regulations and are not included in a NMFS FMP include lingcod, black and blue rockfish, state waters sablefish, Prince William Sound pollock, and the state waters Pacific cod fishery, which is managed under a GHL and not included in the federal TAC. These fisheries are not considered part of the Federal action in this opinion, but their continued prosecution is analyzed here as a cumulative effect.

Seasonal and temporal distributions of state waters fisheries vary widely by species, area, and gear type, are discussed in more detail in subsequent sections. These distributions are depicted in detail in Kruse *et al.* (2000) for the year 1999, and in Woodby and Hulbert (2006) for the year 2005. Another descriptive reference is Woodby *et al.* 2005. Soboleff (2005) evaluated State fisheries for salmon, herring, shellfish, and groundfish based on fish ticket data for 1976-2002 and SSL counts by rookery groupings. Matthiopoulos *et al.* (2008) modeled fishery effects on Steller sea lions, providing some perspectives on not only groundfish but also eulachon and herring fisheries. The reader should consult these first three references for a complete description of these fisheries; the latter two provide some additional perspectives on State fishery effects on Steller sea lions. Only summary information is included here.

The geographic range of State managed fisheries in state waters coincides almost entirely with the area designated as Steller sea lion critical habitat (Figure 2.13). To reduce interactions between Steller sea lions and State managed fisheries, in 1999 ADF&G established no fishing zones for pollock around most rookeries and a few haulouts out to 3 nm (by Emergency Order, March 17, 1999) and has closed several haulout sites seasonally in Prince William Sound out to 10 nm. Four rookeries designated as critical habitat (Agattu Island/Gillion Point, Agattu Island/Cape Sabak, Wooded Island, and Seal Rocks [Cordova]) were not protected from commercial fishing out to 3 nm by the state emergency order. Four haulouts are included in the March 17, 1999 emergency order because the entire island where a rookery was located is protected by the 3 nm fishing closure. These protected haulouts are Seguam Island/Finch Point, Seguam Island/South Side, Kiska/Sobaka, and Vega, and Amchitka/Cape Ivakin. The 3 nm closures and 10 nm fishing restricted areas are based upon 1999 federal regulations. Since this time, additional Steller sea lion sites have been added to the regulations at 50 CFR Part 679.

State managed groundfish fisheries are managed regionally and prosecuted according to an ADF&G Emergency Order (Appendix 1). Recent harvest is presented in Table 2.27. The State managed pollock fishery is limited to Prince William Sound, while Pacific cod fisheries occur in Prince William Sound, Cook Inlet, Kodiak, Chignik, South Alaska Peninsula, and Aleutian Islands areas.

In addition to Pacific cod and pollock, the state has established separate GHLs and seasons for the following fisheries in the western GOA: sablefish, lingcod, black rockfish (*Sebastes melanops*), and blue rockfish (*S. mystinus*). The state-managed fisheries for sablefish and Pacific cod occur within state waters, whereas the state has full management authority for lingcod and black and blue rockfish fisheries throughout the EEZ. In the Central GOA, state-managed fisheries in state waters also include sablefish and all rockfish species in state waters of Prince William Sound and lower Cook Inlet.

The Alaska Board of Fisheries created "Guiding Principles for Groundfish Fishery Regulations" (5 AAC 028.89) which stipulate that state groundfish fisheries are managed conservatively to (1) conserve groundfish resources to ensure sustained yield, (2) minimize bycatch and prevent localized depletion of stocks, (3) protect habitat and other associated fish and shellfish, (4) maintain slower harvest rates by methods and means and time and area restrictions, (5) extend the length of fishing seasons by methods and means and time and area restrictions, (6) harvest the resource in a manner that emphasizes quality and value of the product, (7) use the best available information, and (8) manage cooperatively with the Council and other federal agencies associated with groundfish fisheries.

These ecosystem-based guiding principles have led to a set of conservation measures for state-managed groundfish fisheries. A number of these management measures provide, directly or indirectly, some protection to Steller sea lions. Groundfish fisheries are excluded within 3 nm around Steller sea lion rookeries (through federal regulation) and some haulouts on a seasonal basis. Regulations at 50 CFR 223.202 prohibit entry of any vessel within 3 nm of ESA listed rookeries. These no-entry regulations apply to state permitted fishing vessels as well as federal permitted fishing vessels. These closures are intended to minimize disturbance of land-based animals and to maintain unaltered supplies of prey resources in the nearshore waters around rookeries that are critical to juveniles, pregnant females, and females with pups.

Another conservation measure is the closure of most state waters in the GOA and Bristol Bay to non-pelagic trawling (Figure 2.13). Most areas are closed year-round, and some areas are closed seasonally as in Shelikof Strait. Moreover, a portion of eastern Prince William Sound is closed to pelagic trawl gear during the pollock fishery (5 AAC 28.263) and most of eastern Prince William Sound is closed to all (non-pelagic and pelagic) trawling year-round (5 AAC 39.165). These trawl closures were established by the Alaska Board of Fisheries to protect seafloor habitats, shellfish such as depressed crab populations, and non-target demersal fishes. Although only the 3 nm closures around most rookeries were designed specifically for Steller sea lions, the trawl area closures protect bottom habitats within Steller sea lion critical habitat, and they afford protection to non-target species that are part of the Steller sea lion diet in various amounts, including octopus, sculpins, flatfish, greenlings, and other forage fishes which are associated with bottom habitats. The non-pelagic trawling ban also reduces the possibility of direct cumulative impacts from State managed fisheries on marine habitat and particularly the benthic community.

State managed groundfish fisheries are likely to reduce the abundance and/or alter the distribution of several Steller sea lion key prey species, including walleye pollock and Pacific cod for several reasons. Groundfish fisheries may cause dense schools of prey to scatter (depending upon the gear type used), affecting the foraging behavior of marine mammals and seabirds that target aggregated prey (Brock and Riffenburgh 1960, Dayton *et al.* 1995). Repeatedly causing fish schools to scatter reduces their density and may decrease the value of foraging areas to Steller sea lions. As a result, individual Steller sea lions may feed less efficiently and would have to expend more time and energy to consume the same number of fish. On the other hand, research by Lokkeborg *et al.* (1989) and Lokkeborg (1998) indicate that some gears such as hook-and-line and pot gear may attract fish. At larger spatial scales, reductions of biomass due to fishing may exacerbate the effects of small-scale depletions, leaving fewer spawning-aged fish to replenish areas where fishing has occurred.

The Prince William Sound pollock fishery is based on a constant harvest rate strategy. Because reliable estimates of biomass and natural mortality are available, the Prince William Sound pollock stock falls into Tier 5 of the federal stock assessment strategy (Section 2.4.2.3). The GHL is calculated as the product of the biomass estimate, instantaneous natural mortality rate (0.3) and a “safety factor” of 0.75. Biomass is estimated by bottom trawl surveys in summer and hydroacoustic surveys of spawning aggregations in winter. In 1999 the Alaska Board of Fisheries directed the ADF&G to file an emergency regulation establishing a Prince William Sound pollock trawl fishery management plan to reduce potential impacts on the endangered population of Steller sea lions. The plan divides the Inside District of (Prince William Sound) into three management sections with no more than 40% of the total harvest coming from any one area (5 AAC 28.263). ADF&G manages to a target of 30% of the total harvest from any one of these areas with a 10% reserve. These spatial management measures may help reduce competition for fish between the pollock fishery and Steller sea lions. This measure was in lieu of closing two Steller sea lion haulouts that were specified to be closed under the 1998 Biological Opinion (NMFS 1998). Although pollock in the GOA are considered to be one stock, the state surveys pollock in Prince William Sound separately from NMFS surveys in the GOA. However, NMFS takes the Prince William Sound fishery into consideration when setting the GOA TAC and the Prince William Sound pollock GHL is subtracted from the ABC before setting TAC. In 2008, the fishery began on January 20, and was estimated to harvest near 1,400 mt of pollock. The fishery closes at the end of March.

The effects of the State managed pollock fishery on Steller sea lions is mitigated to some degree by existing restrictions on the fishery. The Prince William Sound outside district (including Wooded Island, Seal Rocks, Cape Hinchinbrook, and Hook Point) is closed to fishing (Figure 2.14). Since the pollock fishery occurs only in the Prince William Sound inside district, it reduces the potential for removing Steller sea lion prey in the vicinity of critical habitat sites Cape St Elias, Hook Point, Middleton Island, the Wooded Island rookery, and most of the Seal Rock and Cape Hinchinbrook sites. Pollock fishing is prohibited June 1 through November 1 within 10 nm of seven rookeries and haulouts in Prince William Sound (5 AAC 28.250). Two haulout sites within Prince William Sound, Perry Island and Point Eleanor, have no pollock fishing restrictions. The Needles, Point Elrington, and Glacier Island haulouts have no pollock harvest restrictions from November 2 through May 31. The fishery opens January 20 (concurrent with CGOA) and closes by emergency order no later than March 31. Steller sea lions using Prince William Sound inside district haulouts may experience a depletion of pollock and disruption of the prey field during part or all of the year, and the time period of the pollock fishing restriction does not provide protection during the critical winter months.

In 1996, the Alaska Board of Fisheries adopted Pacific cod FMPs for fisheries in Prince William Sound, LCI, Chignik, Kodiak, and the South Alaska Peninsula. All five FMPs have some common elements that include: catch is allocated to specific gear types on a percentage basis, pot vessels are limited to no more than 60 pots, jig vessels are limited to no more than five jigging machines, and exclusive area registration requirements. Vessels participating in the South Alaska Peninsula and Chignik areas are limited to no more than 58 feet in length. Catches are allocated to users as: 85% pot and 15% jig in South Alaska Peninsula and Chignik areas, 60% pot and 40% jig in Prince William Sound, and 50:50 in Kodiak and Cook Inlet areas. If target gear allocation percentages are not met by late in the season, then the unattained GHL becomes available to all gear types. State GHLs are set as a percentage of the Federal ABC. State GHLs for Prince William Sound are set at 25% of the Federal ABC for the eastern GOA. Similarly, up to 25% of the central GOA ABC is allocated among Chignik (up to 8.75%), Kodiak (up to 12.5%), and Cook Inlet (up to 3.75%). Finally, the state GHL for the South Alaska Peninsula fishery is set at 25% of the western GOA ABC. The fishery generally occurs in the winter and spring following the federal fishery, the state Pacific cod fishery opens by regulation between one and seven days after the federal fishery closes.

The State managed Pacific cod fisheries also include management measures that may help to reduce interactions with Steller sea lions by dispersing effort spatially. The eastern section of the Prince William Sound outside district is closed to Pacific cod fishing where several Steller sea lion haulouts and rookeries are located.

Pacific cod harvested in state waters in 2008 came from Steller sea lion critical habitat in the South Alaska Peninsula, Kodiak, and BSAI areas primarily, with smaller harvests in Prince William Sound, Chignik, and Cook Inlet.

On March 15, 2006, the Alaska Board of Fisheries approved the opening of a new state waters Pacific cod fishery in the Aleutian Islands west of 170°W for pot, jig, longline, and non-pelagic trawl gears. This state-managed fishery opens after the parallel trawl catcher vessel fishery closes. The 2008 GHL was 5,280 mt, or 3% of the BSAI ABC. The fishery is temporally regulated so that no more than 70% of the GHL can be harvested before June 10, however, most of this is taken in March. The remainder of the GHL can be harvested starting June 10. Twenty-six vessels registered for the fishery, including trawlers, pot vessels, and freezer longliners. Two floating processors and two shore-based processors participated. Observer coverage and VMS are not required in this state-waters fishery, but six vessels chose to carry a federal observer, and 23 planned to activate VMS during the fishery. Steller sea lion rookery closures are enforced.

Sablefish, rockfish, and lingcod are not important in the diet of Steller sea lions, but fisheries for these species could cause indirect impacts to Steller sea lion foraging behavior through disturbance. There are no specific measures to protect Steller sea lions are included in the state management plans for these species. Sablefish landings occurred inside Steller sea lion critical habitat in Prince William Sound, lower Cook Inlet, and the Aleutian islands in 2008. Landings occurred in March through May and August in Prince William Sound, in July in Cook Inlet, and primarily May – August in the western Aleutian Islands. Most of the lingcod harvest in 2005 was taken in the Kodiak area, although catch occurred inside Steller sea lion critical habitat in Kodiak, Cook Inlet, and Prince William Sound from July through October. Similarly, most rockfish harvest occurred around Kodiak, but harvest occurred inside Steller sea lion critical habitat in Prince William Sound, Cook Inlet, Kodiak, Chignik, South Alaska Peninsula, and the Aleutian islands primarily from March through August.

The amount of groundfish harvested in the state fisheries is presented in Table 2.27. Although the amount of fish harvested in the 3 nm area around haulouts appears low, when compared to the actual area in the GOA, it may not be that clear. The amount of area composed inside 3 nm of haulouts in the GOA is roughly 0.5% of the total area, with catch percentages up to 7.4% (pot, Pacific cod), this represents two orders of magnitude higher catch rate than a theoretically dispersed fishery. Again, the type of data necessary to evaluate whether this may or may not be a problem is lacking, such as information on biomass availability on small scales. Further complicating matters, the fleet fishing within state waters during these parallel seasons are generally small unobserved vessels. Because of this, very limited information is available on these fishing activities as compared to larger boats operating in federally managed waters.

Soboleff (2005) evaluated State fisheries relative to Steller sea lion trends throughout the range of the western DPS. This study determined that, within 50 nm of rookeries, SSL counts were both negatively and positively correlated with certain State fisheries, but few were significant and some probably spurious. Soboleff (2005) found negative correlation between State salmon fisheries and the SSL decline across all regions or all years, which disappeared at a regional scale, and he felt this could be plausible as salmon fisheries occur near SSL haulouts and rookeries and salmon are important SSL prey. The study concluded that few data, low power, and concentration of State fisheries outside areas where SSL declines have been most severe all may be factors that indicate a low likelihood of State-managed fisheries adversely affecting SSLs.

6.4.1 Herring Fisheries

At present, state herring fisheries that occur within Steller sea lion critical habitat include fisheries in Prince William Sound, Cook Inlet, Kodiak, Alaska Peninsula, Bristol Bay, Kuskokwim, Norton Sound, Southeast, and Port Clarence. Approximately 25 distinct fisheries for Pacific herring occur in these regions. Harvest methods are by gillnet, purse seine, and handpicking of roe from kelp. Herring are primarily caught for their roe during the sac roe harvest in the spring when they move closer to shore (including portions of Steller sea lion critical habitat) to spawn. On occasion, the entire allowable harvest has been taken in less than one hour, although most sac roe fisheries occur during a series of short openings of a few hours each, spanning approximately one week. Fishing is not allowed between these short openings to allow processors time to process the catch, and for managers to locate additional herring of marketable quality.

Prior to 1999, the average annual harvest of herring for sac roe was about 48,000 t. Harvest for the past five years has been between 30,000-40,000 t. The major populations of herring in Alaska are at moderate levels and in relatively stable condition, with the exception of Prince William Sound and Cook Inlet. The Prince William Sound fishery has been closed since 1999 due to low abundance, and in 2006 the Exxon Valdez Oil Spill Trustee Council initiated planning of a long-term herring restoration program. The lower Cook Inlet fishery has been closed since 1998 due to low abundance. Herring harvest is concentrated temporally from late April to early May due to the nature of the fishery. Herring food and bait landings in the Alaska Peninsula area are concentrated in the Akutan district inside Steller sea lion critical habitat and occur in late July. Smaller food and bait landings occur in some years around Kodiak.

Spawn-on-kelp fisheries harvest intertidal and subtidal macroalgae containing freshly deposited herring eggs. Smaller amounts of herring are harvested from late July through February in herring food/bait fisheries. Herring spawn timing is temperature dependent, so that herring spawning and roe harvest timing occurs progressively later from southeast Alaska, where spawning begins in March, through the northern Bering Sea, where spawning ends in June.

Harvest policies used for herring in Alaska set the maximum exploitation rate at 20% of the exploitable or mature biomass. The 20% exploitation rate is considered to be lower than commonly used biological reference points for species with similar life history characteristics. In some areas, such as Southeast Alaska, a formal policy exists for reducing the exploitation rate as the biomass drops to low levels. In other areas, the exploitation rate is similarly reduced, without a formal policy. In addition to exploitation rate constraints, minimum threshold biomass levels are set for most Alaskan herring fisheries. If the spawning biomass is estimated to be below the threshold level, no commercial fishing is allowed. Threshold levels are generally set at 25% of the long-term average of unfished biomass (Funk and Rowell 1995).

Most herring fisheries in Alaska are regulated by management units or regulatory stocks (i.e., geographically distinct spawning aggregations defined by regulation). Those aggregations may occupy areas as small as several miles of beach or as large as all of Prince William Sound. Herring sac roe and spawn-on-kelp fisheries are always prosecuted on individual regulatory stocks. Management of food and bait herring fisheries can be more complicated because they are conducted in the late summer, fall, and winter when herring from several regulatory stocks may be mixed together on feeding grounds distant from the spawning areas. Where possible, the Alaska Board of Fisheries avoids establishing bait fisheries that harvest herring from more than one spawning population.

Adverse impacts may accrue to Steller sea lions from herring fisheries when vessel activity interferes with Steller sea lion foraging. Additionally, direct mortality may result when Steller sea lions are caught in nets or other fishing gear (although no direct mortalities have been observed in the herring fisheries; Ferrero *et al.* 2000). Steller sea lions are attracted to areas where herring spawn, and they feed on the dense aggregations of herring present during the short spawning period. Nighttime observations of Steller sea lions in Prince William Sound using infrared scanning technology and acoustic surveillance of their prey revealed that Steller sea lions fed exclusively on herring, despite the presence of much greater abundances of pollock (Thomas and Thorne 2001). These results suggest that under some conditions (e.g., when highly aggregated in shallow water), herring may be a preferred prey resource for Steller sea lions. Rosen and Trites (2000) found that Steller sea lions on a pollock-only diet showed progressive metabolic depression while losing body mass. The authors attributed these responses to the lower gross energy content of pollock versus herring, the higher energetic cost of digesting pollock, and the increased energy loss from digesting a larger quantity of fish needed to compensate for the lower energy content of pollock. The Steller sea lions would have had to consume 35% to 80% more pollock than herring to maintain similar net energy intakes (Rosen and Trites 2000). Thus, one could speculate that when herring are available in high enough densities, Steller sea lions may prefer to feed on herring due to its higher energy content. However, field data to either support or refute this speculation are lacking.

Because the time when herring spawn is somewhat variable, fishery managers have learned to depend on the presence of Steller sea lions to determine when herring spawning is imminent. Managers generally begin flying aerial surveys over potential herring spawning grounds well in advance of the expected spawning event. For several weeks prior to spawning, herring are usually present adjacent to the spawning grounds, but they occur in depths too deep to be detected from aircraft. However, the presence of Steller sea lions and cetaceans on the spawning grounds alerts fishery managers to the presence of herring and impending spawning. Fishery managers usually note the presence of Steller sea lions in their field notebooks, occasionally recording actual counts. Steller sea lions are commonly observed in the middle of these fishing areas. There are two possible hypotheses regarding these observations:

1. Steller sea lions may venture into fishing grounds because the fishery is in some way either beneficial (or neutral), concentrating herring, creating confusion, and enhancing feeding opportunities for Steller sea lions.

2. Some Steller sea lions, perhaps the brave or curious ones, or those that cannot afford not to forage (i.e., nutritionally limited), forage in these fishery grounds. Other Steller sea lions, those that are not observed (and would not be due to the type of observations) avoid these fishing grounds due to the intense vessel activity, nets, and other hazards. Additionally, some Steller sea lions that do forage, may have higher stress levels involved with avoiding vessels, gear, and dealing with noise, yet may appear to be foraging effectively.

There is insufficient information to determine which hypothesis is more or less likely. Presumably, fishing in areas that were previously unfished, yet utilized by Steller sea lions, would change the manner and success rate of foraging Steller sea lions. This could be either a positive or negative effect. Given the high caloric content of herring, the historical dependence on the species (Sinclair and Zeppelin 2002), and the large decline in herring biomass during the last century (Kruse *et al.* 2000), this fishery should be the subject of further study specifically to determine if there may be negative impacts on Steller sea lions. The important point is that although we have adequate data which displays that Steller sea lions attempt to forage during the times and places when herring fisheries occur, we have little or no information on either the net impacts to those Steller sea lions or other Steller sea lions which may avoid observation because they elect not to forage. There is no way of knowing how many Steller sea lions may be precluded from foraging in the spawning areas due to fishing activity. Steller sea lions are observed leaving the grounds within a few days after the herring have spawned. Fishery biologists make note of their departure since spawn deposition SCUBA biomass surveys do not begin, for safety reasons, until the Steller sea lions leave the area.

One example of a herring spawning event where Steller sea lion counts were quantified during aerial surveys was at Hobart Bay. There was no fishery in the spring of 2000 because the quota had been taken in the earlier food/bait herring fishery. However, if a fishery had occurred, managers would typically have allowed 6-12 hours of gillnet fishing about April 29. Steller sea lions were already in the area at the time of the first ADF&G aerial survey on April 19, diving on the deeply submerged herring schools, as were a number of humpback whales. Following the spawning event, large numbers of birds appeared on the beaches to feed on the herring eggs, noted in numbers of 11,000 to 20,000. Approximately 150 Steller sea lions were counted in the area. Similar descriptions of humpback whale and Steller sea lion presence on herring spawning grounds are available in field notes from other herring fishing areas.

Steller sea lions may depend on these short intervals of high prey availability to sustain them through other periods of low prey availability. Some individual Steller sea lions may be able to adapt by learning to forage among the fishing boats, but others may choose to avoid the area and may thus forego prime foraging opportunities. Since we do not observe the Steller sea lions that avoid fishing areas, we have no reliable way to estimate how many may be affected in this way, nor do we have a way to gauge the impact on those individual animals. For the Steller sea lions that remain, we have no way to gauge their foraging success among fishing vessels relative to their potential foraging success in the absence of fishing vessels. Nevertheless, based on observations of interactions between the fishery and Steller sea lions, it is reasonable to conclude that some Steller sea lions may be precluded by the fishery from foraging on spawning schools of herring. Likewise, the Steller sea lions that do forage in the vicinity of the fishery may forage less efficiently due to active competition with the fishery for the available concentrations of herring.

Hundreds of individual Steller sea lions may be affected by each of these brief fishery openings. The annual exploitation rate for herring is roughly 20% of the exploitable or mature biomass (Kruse *et al.* 2000), which is considered by the state to be conservative. This may be in relation to the target stock, but the question that arises is whether this is conservative from a Steller sea lion perspective? This example from Hobart Bay is merely to make the point that foraging Steller sea lions and herring fisheries operate in the same areas and times on the same resource. However, it should also be noted that in the four most eastern sub-regions, where the state herring fisheries are prosecuted, Steller sea lions are increasing in abundance (albeit at different rates).

6.4.2 Salmon Fisheries

The state salmon fishery includes five species: Chinook, sockeye, coho, pink, and chum. These fisheries are divided into southeast, Prince William Sound, Cook Inlet, Bristol Bay, Kodiak, Chignik, Alaska Peninsula, Kuskokwim, Yukon, Norton Sound, and Kotzebue management areas. The Prince William Sound, Kodiak, Chignik, and Alaska Peninsula areas report substantial harvest inside Steller sea lion critical habitat in 2005 (Woodby *et al.* 2005). Salmon are taken by purse seines, gill nets, trolling, and beach seining via an extensive small boat fleet.

The fisheries are managed for minimum escapement goals, where regional ADF&G biologists have determined what level of escapement seems to produce the maximum yield per year. These methods have not been standardized, and range from aerial flights to determine if the streams are “full” to fish weirs and remote sonar counters. The timing of the fisheries corresponds with the various spawning time for each run, which is highly variable and which is managed on a stream by stream basis. Economically, the salmon fishery is worth more than all other state fisheries combined.

State managed salmon fisheries have direct impacts on Steller sea lions as well. In the gillnet fishery Steller sea lions cause significant catch loss and gear damage by taking fish from nets and tearing large holes in the nets (Hoover 1988). Steller sea lions cause damage to purse seine nets when they swim inside the nets to eat salmon before the nets are closed (Hoover 1988). Prior to the mid-1990s the only quantitative study on interactions between Steller sea lions and the Alaska salmon gillnet fishery was on the Copper and Bering River deltas and the Coghill district in south central Alaska (Kruse *et al.* 2000; Matkin and Fay 1980). During the three week spring salmon season Steller sea lions damaged 1.7-4.9% of the weekly catch, and most of the damage occurred in outside waters where relatively few boats fished. Steller sea lions were infrequently seen in the Coghill district and were absent during the fall Copper River district season. Observers also monitored the Prince William Sound salmon drift gillnet (Copper River) fishery in 1990 and 1991. No mortalities were observed in 1990 and two were recorded in 1991. When these observer data are extrapolated, the mean kill rate for 1990 and 1991 is 14.5 Steller sea lions per year (Kruse *et al.* 2000). The Alaska Peninsula and Aleutian Islands salmon drift gillnet fishery was also monitored during 1990 and no Steller sea lion mortalities were observed. There were no incidental serious injuries or mortalities observed in the Cook Inlet salmon gillnet fishery in either 1999 or 2000 (NMFS unpublished data); for Bristol Bay the annual Steller sea lion mortality is thought to be 3.5 (Kruse *et al.* 2000, Ferrero *et al.* 2000).

Indirect adverse effects of State managed salmon fisheries on Steller sea lions stem from competition for seasonal aggregations of fish. State managed salmon fisheries are open for relatively short periods, and only rarely remain open for 24 hours per day, seven days per week (Kruse *et al.* 2000). Nevertheless, many of these fisheries take place at stream or river outlets where salmon congregate before moving upstream to spawn (Kruse *et al.* 2000). These same areas may provide important Steller sea lion foraging opportunities on high density prey, enabling the Steller sea lions to feed efficiently and survive other periods of low prey availability. As discussed above, salmon are a common prey resource for Steller sea lions. Sinclair and Zeppelin (2002) found that Pacific salmon were the third most dominant fish in the diet of Steller sea lions, based on scats observed from 1990-1998 on summer and winter island sites across the range of the western stock of Steller sea lions. Sinclair and Zeppelin (2002) observed that known seasonal and spatial distributions of aggregations of fish that are preyed upon by Steller sea lions parallel the highest observed frequencies of occurrence in seasonal and regional prey consumed by Steller sea lions. Due to intensive salmon fishing activity in such areas during the same times when Steller sea lions target concentrations of salmon, individual Steller sea lions may feed less efficiently or may avoid these feeding opportunities entirely. ADF&G’s identified salmon escapement levels limit the harvest to the amount that is surplus to that needed for spawning (Kruse *et al.* 2000), but these harvest controls probably do not eliminate competition for available salmon between Steller sea lions and the fishery. However, as noted in Kruse *et al.* (2000) the abundance of salmon biomass increased dramatically during the time period that the western stock of Steller sea lions has been in decline. Further study and consideration is necessary to determine what affects salmon fisheries have on the availability of prey for Steller sea lions.

6.4.3 Invertebrate Fisheries

ADF&G manages fisheries for several species of crab, shrimp, scallops, and sea cucumbers inside Steller sea lion critical habitat. Invertebrates are not important in the diet of Steller sea lions, but the fisheries could cause indirect impacts to Steller sea lions by influencing their foraging behavior due to disturbance. No specific measures to protect Steller sea lions are included in the state management plans for these species.

Since 1999, the tanner crabs stocks have recovered enough to re-open fisheries in Kodiak, Chignik, south Alaska peninsula, eastern Aleutian Islands, and the EBS management areas. Thus, new effort and therefore increased vessel traffic is now occurring in the south Alaska peninsula, Kodiak, and Chignik areas of Steller sea lion critical habitat primarily in January. In 2005, Dungeness crabs were harvested inside Steller sea lion critical habitat around the east side of Kodiak and the South Alaska Peninsula. Harvests are highest in late summer peaking in July, August, and September.

Crab rationalization in 2005 for king and tanner crab stocks in the BSAI provides for a prolonged harvest season, given that fishermen can fish their quota when they desire. The 2005/06 harvest of golden king crab in the Aleutian Islands Steller sea lion critical habitat areas started in the eastern Aleutians in August, peaking in September and October, and then shifted to the western Aleutians where harvest stayed lower through March.

Other invertebrates harvested in State managed fisheries inside Steller sea lion critical habitat include shrimp, scallops, and sea cucumbers. These species, like the crab species discussed above, are not important components of the Steller sea lion diet, but fisheries for these species could cause indirect impacts to Steller sea lion foraging behavior through disturbance caused by vessel traffic.

6.4.4 Summary of Effects of State Waters Non-parallel Fisheries

With regard to direct effects, State managed fisheries are likely to continue to account for an annual mortality of approximately 30 Steller sea lions, based on current levels of direct mortality (Angliss and Outlaw 2005), although it should be recognized that the data used to estimate direct mortality are almost twenty years old and are based on a relatively small sample. There are no available estimates of the frequency or severity of nonlethal takes in State managed fisheries, but presumably nonlethal takes will continue at current levels.

Regarding indirect effects, NMFS concludes based on available information that State managed fisheries for pollock, Pacific cod, herring, and salmon are likely to continue to compete for fish with foraging Steller sea lions. Given the importance of near shore habitats to Steller sea lions, this competition for fish may have consequential effects. Specifically, these interactions may contribute to nutritional stress for Steller sea lions, and may reduce the value of the marine portions of designated Steller sea lion critical habitat. State managed fisheries will likely continue to reduce the availability of prey within these marine foraging areas and may alter the distribution of certain prey resources in ways that reduce the foraging effectiveness of Steller sea lions. More data on the foraging habits of Steller sea lions from research in key geographic areas could aid our understanding of where and when these effects might be most important.

6.4.5 Effects of State Fisheries on Listed Whale Species

State-managed fisheries outside the scope of this opinion clearly have a direct impact on humpback whales. Over 100 entanglement incidents have been reported to the NMFS Alaska stranding program over the last 30 years, many involving pot gear and/or gill net gear from fisheries in inside waters in southeast

Alaska, and areas around Kodiak, Homer, and Seward. Of the events involving pot gear from 1997-2009, 28% involved crab and shrimp gear, while 22% involved unidentified pot gear. In recent years, these events appear to be increasing, though higher numbers could be a result of greater awareness of the problem or, in a few cases, re-sighted animals. For many of these incidents, when disentanglement is not possible or the animal is not re-sighted, the ultimate fate of the animal remains unknown. State managed fisheries represent an additional source of anthropogenic impact beyond those posed by the proposed action through entanglements to the central North Pacific population of humpback whale, the western North Pacific population of humpback whale, and the Alaska population of sperm whale.
In summary, NMFS expects the existing state-managed fisheries and their direct and indirect effects on Steller sea lions and affected species to continue into the foreseeable future.

6.5 Sport, Subsistence, and Tribal Fisheries

Meeting public demand for recreational fishing opportunities in Alaska, while at the same time maintaining and protecting fishery resources, has become a significant challenge for ADF&G (Howe *et al.*1996). Increasing tourism and continued population growth lead to increased pressure on existing sport fisheries and development of new fisheries. At the core of sport fisheries management is the ADF&G onsite creel surveys. ADF&G staff survey fishermen as they return to the docks, requesting information on catch and time fished, as well as collecting biological samples, fish tags, and other information. Additionally, ADF&G conducts surveys through the mail requesting further information from fishermen on the annual harvest. This information is compiled and published in annual sport fishery reports (Howe *et al.* 1996). Of the 469,436 anglers who fished in Alaska in 1995, about 51% were Alaska residents and 49% were nonresidents, resulting in about 3 million angler-days fished. This effort resulted in 2,909,979 fish harvested which included 1,299,945 razor clams (*Siliqua patula*) and 52,905 smelt and capelin (Osmeridae). Of the remaining 1,657,129 harvested fish, 55% were salmon, 20% were halibut, 7% were rainbow trout, 5% were rockfish, 4% were Dolly Varden and Arctic char, 3% were grayling, and 1% were landlocked salmon. Also harvested, at much lower amounts, were lingcod, whitefish, steelhead, and sheefish.

Since 1985, the number of anglers fishing in Alaska has increased 35%, about 3% per year. Trends in annual catch rates are most affected by fluctuations in salmon abundance. Abundance of species such as halibut and rockfish has been more consistent over the last 20 years (Howe *et al.* 1996). For perspective, the sport fishery harvests about 1% (4,000 mt) of the annual Alaska total fish harvests, while the commercial fisheries accounted for 97% (900,000 mt) of the annual harvest in 1998. Sport fishery harvests would be expected to continue in relatively low amounts in the future. It is likely that increased levels of tourism will also increase the amount of fish taken for sport. However, this additional harvest would likely result in a comparatively small amount of fish taken. The nature of most of the fisheries is slow removal rates and dispersed catch. The most concentrated catches are in the salmon fisheries; however, many of these (such as the Kenai fisheries) take place upriver outside of foraging areas for Steller sea lions. For these reasons, future State managed sport fisheries will not contribute measurably to the total cumulative effects of state, tribal, local, and private actions on Steller sea lions.

Subsistence hunting and fishing are important to the economies of many families and rural communities in Alaska, and subsistence uses are central to the customs and traditions of many Alaska Native groups, including the Aleut, Athabaskan, Alutiiq, Haida, Inupiat, Tlingit, Tsimshian, and Yup'ik. NMFS expects that traditional uses of natural resources will continue. About 20% of Alaska's population participates in subsistence harvests (124,367 people in 270 communities in 1998). Most of the harvest is composed of fish (about 60% by weight). For perspective, the subsistence fishery harvests about 2% (8,000 mt) of the annual Alaska total fish harvest, while commercial fisheries accounted for 97% (900,000 mt) of the annual harvest in 1998. Consequently, although subsistence harvests are likely to continue into the future, and possibly grow if population increases, the amount taken for consumptive uses will remain very small compared to the commercial catch of fishery resources (ADF&G 1998 “Subsistence in Alaska: 1998 Update”) and will not contribute measurably to the total cumulative effects of state, tribal, local, and private actions on Steller sea lions.

As we stated in the 2007 and 2009 Biological Opinions on research permitting, we expect that sport and tribal/subsistence fisheries have an incremental effect on listed Steller sea lions relative to that in commercial fisheries. Alaska’s sport fishery harvests about 1% (4,000 mt) and subsistence fishery harvests 2% (8,000 mt) of the annual State of Alaska total fish harvests, while the commercial fisheries accounted for 97% (900,000 mt) of the annual harvest in 1998. Impacts are likely limited to minor removals of the potential foraging base, but in such small volumes that we expect only incremental adverse effects, if any. Effects due to lost gear and potential entanglements and ingestion are documented but we have uncertainty about the magnitude of the current levels (see discussion of entanglements above). Steller sea lions can also be disturbed in key areas by the sport fishermen and be attracted to sport-fishery related cleaning areas in harbors. Sport and subsistence fisheries are expected to continue into the foreseeable future throughout the action area and may increase in the future as tourism and population increases.

NMFS expects the existing state-managed subsistence and recreational fisheries and their direct and indirect effects on Steller sea lions and other affected species to continue into the foreseeable future.

6.6 State Oil and Gas Leasing

Oil and gas leasing on state lands in areas near Steller sea lion habitat and in Alaska state waters is likely to occur in the future. Such leasing and exploration has occurred for a long period of time. However, given changes in energy prices and increasing demand, it is also likely that more development and production will also occur.

At present, a number of active fields produce oil in Cook Inlet, all of which is processed at the refinery at Nikiski on the Kenai Peninsula. Estimated oil reserves in Cook Inlet are 72 million barrels of oil. Currently there are additional lease sales planned for the next five years (State of Alaska Oil and Gas Leasing Program [available at www.dog.dnr.state.ak.us]). In areas where Steller sea lions could be affected, oil and gas lease sales are currently scheduled for Cook Inlet (annual sales 2009-2013). In Cook Inlet, the state proposes to offer leases throughout the inlet from the northernmost areas south to approximately Anchor Point on the southeast and areas outside of Cook Inlet that would fall within the action area. There are active leases along the Kenai Peninsula coast as far south as approximately Ninilchik.

6.7 Vessel and Aircraft Activity

Our conclusions about potential cumulative effects from vessel and aircraft activity are similar to those stated in previous Biological Opinions. Disturbance from vessel and aircraft traffic has variable effects on Steller sea lions ranging from no reaction at all to temporary departure from haulouts and rookeries, trampling of smaller animals by large ones, injury, and even abandonment of haulouts and rookeries (e.g., Johnson *et al.* 1989; Calkins and Pitcher 1982; Thorsteinson and Lensink 1962; Kenyon 1962). These effects stem primarily from noise emanating from cruise ships, ferries, small boats, and aircraft. The consequences of such disturbance to the overall Steller sea lion population are difficult to measure in part because most instances of such disturbance are not documented or studied. Disturbance may have contributed to or exacerbated the decline of Steller sea lions, although it likely has not been a major factor in the decline. NMFS has taken steps to reduce disturbance around rookeries by the placement of 3 nm no-entry zones. NMFS expects disturbance from vessels and aircraft to continue in the future at levels comparable to or, more likely greater than, the present.

6.8 Population Growth

In general, as human communities increase in size and number, habitat alterations and environmental impacts also increase. Native plants and animals become displaced by the construction of housing, roads, commercial facilities, and other infrastructure such as facilities for waste disposal. Thus, throughout the action area, the potential foreseeable impact from human population growth rate varies greatly. Alaska has the lowest population density of all of the states in the U.S. and many areas of the state adjacent to Steller sea lion habitat (e.g., the western Aleutians) do not have many permanent year-round settlements. Other coastal areas adjacent to Steller sea lion habitat, such as parts of southern and central California, are densely populated.

In general, human population density is greater in the range of the eastern DPS than in the western DPS. Although Alaska's population has increased by almost 50% in the past 20 years, most of that increase has occurred in Anchorage and Fairbanks. Outside of Anchorage, the largest human populations occur on the Kenai Peninsula, the Island of Kodiak, Fairbanks, Juneau, Bethel, and in the Valdez - Cordova region. Except for Anchorage, few of the cities, towns, and villages would be considered urbanized.

As we noted in the 2007 and 2009 Research Permitting Biological Opinions, within Alaska, rural populations may increase or decrease based on their ability to exploit resources such as fisheries and secure necessities to live in these remote areas. Many rural villages have experienced population declines, mostly in the Aleutians. To bolster these communities, the state has begun to develop local fisheries. For example, the state has implemented a local Adak Pacific cod fishery where vessels fishing under the state GHLwould be excluded by size in order to allow the local small boat fleet to harvest in that area. This effectively shifts management control from the Federal Government, concentrates catch inside state waters (0-3 miles), while providing economic opportunities in the form of fisheries to specific coastal communities. This method of economic development may put pressure on fishery managers in the future to provide for near-shore fisheries, leading to potential conflicts with federal measures to limit direct and indirect impacts to critical habitat for Steller sea lions and to the Steller sea lions. NMFS has also recently become aware of plans by the Bureau of Land Management (BLM) to transfer large amounts of federal land in the Aleutians into ownership by Alaska Natives under provisions of Alaska Native Claims Settlement Act (ANCSA). While this is a future Federal action, we discuss it here because it is not clear that the BLM will consult under Section 7 on this action. If this land transfer occurs, land in the Aleutians (currently primarily on USFWS refuge land) including land adjacent to and possibly within Steller sea lion critical habitat will go out of federal ownership into private ownership. Some of the areas are being subdivided, but the future intended uses of these areas are not clear, and, hence, it is not possible to predict impacts.

The risk of interactions between people and listed species increases as human use of areas within and adjacent to Steller sea lion habitat increases. Steller sea lions can be affected by nearby human settlements in many ways including increases in: disturbance and potential loss of suitable habitat (e.g., due to increasing ship. boat and air traffic; nearby roads); pollution (due to run-off, sewage and gray-water discharge, industrial discharge, shipping accidents; in some areas, increased oil and gas spills offshore; disease; coastal noise and displacement due to construction; and potentially increased harvests (in Alaska). Steller sea lions also sometimes utilize human structures and/or begin to interact with humans in harbors where they may haul out and/or pursue accessible fish waste. This type of interaction can result in risks to Steller sea lions. In general, as the size of human communities increases, there is an accompanying increase in habitat alterations and impacts on landscapes and biota. As areas are modified for the construction of housing, roads, commercial facilities, and other infrastructure, native plants and animals are displaced and waste disposal needs increase.

Steller sea lions typically haul out on offshore (and occasionally shore-based, rocks and islands; however, in some areas they have adopted man-made structures (e.g., jetties) as haulout habitat. The south jetty at the mouth of the Columbia River is such a location and is routinely used by several hundred Steller sea lions in addition to similar numbers of California Steller sea lions. Steller sea lions also occasionally forage in estuaries and the mouths of rivers along the west coast. Many of these estuary areas are also developed as marine terminals for shipping and boat moorage. The lower river reaches and estuaries are kept in navigable condition by maintenance dredging. Noise from dredge operations may cause temporary behavioral avoidance by Steller sea lions in the vicinity of the activity.

The human population of Alaska will most likely continue to increase, especially in urban areas. As noted above, the increasing population will likely result in increases in vessel activity, subsistence uses of natural resources and sport fishing. The further development of commercial fisheries is not as dependent on Alaska population as these fisheries are mostly fully developed and are mostly operated by entities outside of the state (L. Queirolo, NMFS Alaska Region Regional Economist, personal communication February 5, 2010). Any support for future commercial fisheries development is expected to come from outside the state. Rural populations may increase or decrease precisely on their ability to exploit resources such as fisheries and to secure necessities to live in these remote areas. To bolster rural communities that have experienced population declines, such as in the Aleutians, the state has developed local fisheries.

6.9 Climate Change, other Environmental Variability, and Ocean Acidification

We discuss these factors in greater detail in the baseline section. In this section, as in the recent status review of the spotted seal under the ESA (NMFS 2009b; 74 FR 53683), we have attempted to assess the threats to DPSs of Steller sea lions and their designated critical habitat “to the extent such threats can be forecast into the future, keeping in mind that there is greater uncertainty the farther out the analysis extends.” Following the approach, and incorporating much of the summary of expected environmental changes discussed in that review, we have projected the consequences of the key threat of climate change through both 2050 and the end of the 21st Century, though under widely varying assumptions.
It is clear from recent reports (e.g., IPCC 2007a) that global warming is likely to continue for some time regardless of international public policy related to decreases in greenhouse gas emissions. In the spotted seal status review, NMFS (2009c) summarized that: “…the scientific consensus projections are for continued and perhaps accelerated warming and sea ice decline in the foreseeable future. A second major concern, related by the common driver of C_{O2} emissions, is the modification of habitat by ocean acidification, which may alter prey populations and other important aspects of the marine ecosystem. However, the magnitude and rate of change in both key parameters (warming and ocean acidification) is expected to be influenced by such emissions. Thus, we have uncertainty about how rapidly change will occur, what the upper limit of warming in the North Pacific will be, what specific effects this warming will have on basic oceanographic processes (such as currents, areas of upwelling, and primary production), and Steller sea lion prey distribution and abundance). Hansen *et al.* (2008) have pointed out that current climate forcing may trigger “tipping points” in which climate changes could happen rapidly and the climate system could shift to a qualitatively different state. Such a tipping point may have already been reached in the Arctic (e.g., Lindsay and Zhang 2005).

Loss of multi-year ice likely will not be a concern in sub-Arctic seas, such as the Bering Sea and the GOA, as only annual ice is formed in these areas. However, annual ice thickness, ice coverage, and ice extent southward could be affected by climate warming with uncertain effects on Steller sea lion prey. Some believe that seasonal ice in the Bering Sea is expected to continue forming annually during the winter for the foreseeable future with large interannual variability in both duration and extent. However, NMFS (2009) predicted that it was likely that there would be more frequent years in which ice coverage is reduced.

Regarding potential global warming environmental effects in the western part of the western DPS (i.e., the Sea of Okhotsk), some believe that because of model deficiencies and the small size of the regions compared to the spatial resolution of the models, sea ice models do not satisfactorily represent effects on a regional scale. As a result, inferences about future ice conditions in these areas were drawn indirectly from projections of air or sea surface temperatures, and thus contain greater uncertainty.
The U.S. Global Change Research Program's 2009 Report "Climate Change Impacts in the United States" (USGCRP 2009) pointed out that changes in the timing of plankton blooms are altering the benthic ecosystems in the Bering Sea.

Steller sea lion distribution may shift northward if key prey species move northward. Such a northward shift of prey could have strong negative consequences for Steller sea lions if females, young pups, and juveniles cannot find sufficient food near rookeries and haulouts. Prey species may be simultaneously affected by global warming, natural climate variability, ocean acidification, and fishing removals with uncertain, and potentially overall negative results. As noted in the Baseline section, with regards to ocean acidification only, the Interacademy Panel on International Issues (IAP 2009) stated that:

> "The high CO2 waters in polar and upwelling regions such as the eastern Pacific and Bering Sea for example, will experience low pH more rapidly than other regions…The ocean chemistry changes projected will exceed the range of natural variability, which is likely to be too rapid for many species to adapt to. Many coastal animals and groups of phytoplankton and zooplankton may be directly affected with implications for fish, marine mammals and the other groups that depend on them for food…The impacts of these changes on oceanic ecosystems…cannot yet be estimated accurately but they are potentially large…Although some species may benefit, most are adapted to current conditions and the impacts on ocean biological diversity and ecosystem functioning will likely be severe."

Hester *et al.* (2008) concluded that:

> "The waters in the upper ocean are now undergoing an extraordinary transition in their fundamental chemical state and at a rate not seen on Earth for millions of years, and the effects are being felt not only in biological impacts but also on basic geophysical properties including ocean acoustics."

As with spotted seals and many other species within these ecosystems, Steller sea lion reproduction and survival could be adversely affected if food webs dependent on calcifying organisms are disrupted.
Sea level rise threatens terrestrial components of Steller sea lion critical habitat. However, detailed modeling studies are not available to support analysis of how many sites and which sites might be affected first. In some areas, it is likely that Steller sea lions can move higher up the beach. However, the topography in other sites will prevent this. Alternate remote sites are not available in many areas of the range, either because many alternate sites are now occupied by humans (such as in many parts of the range of the eastern DPS) or because the unique characteristics of Steller sea lion terrestrial critical habitat are not easily met. In some areas, isostatic rebound from the retreat of glaciers may exceed the rate of ocean rise so that Steller sea lion rookeries and haulouts may not be covered by rising ocean levels (Larson *et al.* 2005) (e. g. southeast Alaska).

Overall, we anticipate continuing, probably accelerating, and probably interacting effects from global warming, environmental variability, and ocean acidification on Steller sea lions and their prey. However, data are not sufficient to be able to predict how these factors will affect Steller sea lions in a given area over the foreseeable future. Some authors have concluded that C_{O2} related changes in climate, such as sea level rise and temperature increases, are largely irreversible for 1,000 years after emissions cease (Archer and Brovkin 2009, Solomon *et al.* 2009). Richardson *et al.* (2009) concluded that increases in ocean acidification will persist for hundreds of thousands to millions of years.

Effects due to ocean acidification will almost certainly be negative. There are no currently known regulatory mechanisms that effectively address reductions in sea ice habitat or ocean acidification at this time.

6.10 Toxic Substances

The threat to Steller sea lions and their critical habitat posed by toxic substances is likely to be similar to, or to rise, from baseline levels. A rise is possible due to projections of worldwide population growth and the release of toxic substances, especially in the developing world, but also in areas of the U.S., Canada, and Russia adjacent to Steller sea lion habitat. Plans to develop oil and gas reserves in the southeastern Bering Sea increase the risk to Steller sea lions from an oil spill. If this activity occurs, shipping and vessel traffic in Steller sea lion habitat will increase, thereby raising both levels of disturbance and risk of the release of toxic substances over baseline levels. At present, data are insufficient to know whether our previous conclusions (NMFS 2008) about the threat of contaminants need to be modified and whether it is likely levels of key contaminants are increasing.

In the Russian portion of Steller sea lion habitat, cumulative effects from offshore oil and gas development are expected to increase in the foreseeable future due to increased exploration and development in Russian waters. Large oil and natural gas extraction projects both on and offshore Sakhalin Island have been ongoing since 1994. In 1999, development in the Sea of Okhotsk resulted in an oil spill which released about 3.5 mt of oil (NMFS 2009).

6.11 Disease and Parasitism

NMFS (2008a) concluded that the threat posed by disease and parasitism to the recovery of the western DPS of Steller sea lions was low. As noted in the Environmental Baseline, we have an increased level of concern about the threat to Steller sea lions due to the threat of infectious disease based on documentation by Goldstein *et al.* (2009) of the presence of phocine distemper virus in sea otters in areas of the Aleutians, the Kodiak Archipelago, and Kachemak Bay. Goldstein *et al.* (2009) concluded that:

> “These results demonstrate that PDV has been introduced to the North Pacific Ocean since 2000. All Pacific marine mammal species are now at risk for phocine distemper–induced population decreases…Viral nucleic acid in nasal swabs from free-ranging, live-captured otters confirms viral shedding. Therefore, otters are capable of transmitting PDV to conspecifics and other species. Because the PDV fragment isolated from Alaskan otters is identical to that of the 2002 Atlantic isolate, this virus was likely transmitted to the North Pacific Ocean after the 2002 European epidemic, although it is remotely possible that it may have originated in the North Pacific Ocean during 2000–2002.”

Goldstein *et al.* (2009) speculated that the entry of this virus into North America may have occurred due to global warming. They wrote:

> “The decrease in sea ice during the 14 years between these epidemics may have affected movement of Arctic seal populations…This reduction was even more pronounced in 2004 and

> 2005, years in which PDV was confirmed to have infected sea otters…Ice coverage is at its lowest level during August and September…In 1988 and 2002, the PDV epidemic had reached gray and harbor seal populations in the North Sea and Norwegian Sea by August. This sea ice reduction may have altered seal haul-out and migration patterns, resulting in contact between Atlantic, Arctic, and Pacific Ocean species that was not possible in 1988 and the few years afterwards."

If this is indeed the case, the entry of this virus into the North Pacific may be an effect of global warming. Regardless, the virus poses a threat to Steller sea lion populations to which, available evidence indicates the virus is novel. Potential impacts are high. In the Atlantic, this virus has resulted in the deaths of tens of thousands of harbor seals during multiple epidemics.

The recent occurrence of adult and young Steller sea lions hauling out on the Russian mainland in the city of Petropavlovsk, Kamchatsky raises an issue of special concern. These sea lions are exposed to harassment from packs of feral dogs which may well be vectors of canine diseases that could be transmitted to sea lions hauling out at this location and subsequently be transmitted to Steller sea lions on rookeries throughout the Russian Far East and the U.S. Aleutian Islands (U.S./Russia Mammal Working Group 2009). Currently, this is the only site within the range of the western Steller sea lion where dogs and sea lions are in regular contact. Similar situations with other species of pinnipeds have previously led to significant mortality of pinniped species. Diseases of particular concern are the morbillivirus, canine distemper and phocine distemper as well as bacterial diseases such as leptospirosis.

The Petropavlovsk-Kamchatsky Steller sea lion hauling ground is also located at the point of entry into the sea for the outfall of the urban sewage treatment plant. In that sewage there are undoubtedly bacteria of anthropogenic origin much of which can be expected to show patterns of antimicrobial resistance. These antibiotic resistant bacteria are frequently highly virulent and pathogenic. They may be incorporated into the local fish population which form the prey base of the sea lions, are subsequently integrated into the enteric bacterial flora of the sea lions, and can cause health challenges to the exposed individuals. There is also the possibility that these human pathogens can then be transported by sea lions hauling out near Petropavlovsk to rookeries throughout the Russian Far East as well as the U.S. Aleutian Islands (U.S./Russia Mammal Working Group 2009).

6.12 Illegal Shooting of Steller Sea Lions

Loughlin and York (2001) speculated that the mortality level from illegal shooting of Steller sea lions is at least 50 animals per year. NMFS has worked closely with the participants in the North Pacific Fishery Management Council to eliminate this source of mortality in recent years. At present, the data necessary to properly evaluate this source of mortality are not available.

7 SYNTHESIS AND CONCLUSIONS

The purposes of the ESA are, "*… to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved, to provide a program for the conservation of such endangered species and threatened species, and to take such steps as may be appropriate to achieve the purposes of the treaties and conventions set forth in subsection (a) of this section*." To help achieve these purposes, the ESA requires that, "*Each Federal agency shall, in consultation with and with the assistance of the Secretary, ensure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered species or threatened species or result in the destruction or adverse modification of habitat…*"

Consistent with these purposes and requirements of the ESA, NMFS must make determinations and take actions to conserve wildlife species from the risk of extinction. These determinations and actions are also guided by implementing regulations and judicial precedent. The following rationale provides a basis for NMFS' determinations and actions as required by law.

7.1 Legal and Policy Framework

This Biological Opinion is informed by nearly 15 years of consultations to apply the ESA's jeopardy and critical habitat standards to the fisheries of concern here.

This section describes the analytical approach we use to evaluate the effects of the proposed action on listed species under NMFS jurisdiction. The approach is intended to ensure that NMFS comports with the requirements of statute and regulations when conducting and presenting the analysis. This includes the use of the best scientific and commercial data available relating to the status of the species and critical habitat and the effects of the proposed action. The following discussion of our analytical approach is organized into several sub-sections describing the legal framework provided by the ESA, case law, and policy guidance related to Section 7 consultations.

7.1.1 Jeopardy Standard

The "jeopardy" standard has been further interpreted in regulation (50 CFR 402.02) as a requirement that Federal agencies ensure that their actions are not likely to result in *appreciable reductions in the likelihood of both the survival and recovery of the species in the wild by reducing its numbers, reproduction, or distribution*. It is important to note that the purpose of the analysis is to determine whether or not appreciable reductions are reasonably expected, but not to precisely quantify the amount of those reductions. As a result, our assessment often focuses on whether a reduction is expected or not, but not on detailed analyses designed to quantify the absolute amount of reduction or the resulting population characteristics (abundance, for example) that could occur as a result of proposed action implementation.

In part, NMFS considers a listed species' probability or risk of extinction with the likelihood of both the survival and recovery of the species in the wild for purposes of conducting jeopardy analyses under section 7(a)(2) of the ESA. A designation of a high risk of extinction indicates that the species faces significant risks from internal and external processes that can drive a species to extinction. The status assessment considers and diagnoses both the internal and external processes affecting a species' extinction risk.

The parameters of productivity, abundance, and population spatial structure are important to consider because they are predictors of extinction risk, the parameters reflect general biological and ecological processes that are critical to the survival and recovery of the listed species, and these parameters are consistent with the "reproduction, numbers, or distribution" criteria found within the regulatory definition of jeopardy (50 CFR 402.02). Because there are demographic recovery criteria (NMFS 2008a), NMFS will use those criteria to assess whether the species can be expected to survive with an adequate potential for recovery (e.g., trending toward recovery) under the effects of the action, the effects of the environmental baseline, and any cumulative effects.

7.1.2 Destruction or Adverse Modification of Critical Habitat Standard

For critical habitat, NMFS does not rely on the regulatory definition of "destruction or adverse modification" of critical habitat at 50 CFR 402.02. Instead, we have relied upon the statutory provisions of the ESA to complete the analysis with respect to critical habitat. NMFS will evaluate "destruction or adverse modification" of critical habitat by determining if the action reduces the value of critical habitat for the conservation of the species. Thus, NMFS must determine whether affected designated critical habitat is likely to remain functional (or retain the ability to become functional) to serve the intended conservation role for the species both in the near and long term under the effects of the action, environmental baseline and any cumulative effects.

7.1.3 Additional Requirements

Additional requirements on the analysis of the effects of an action are described in regulation (50 CFR 402) and our conclusions related to "jeopardy" and "destruction or adverse modification" generally require an expansive evaluation of the direct and indirect consequences of the proposed action, related actions, and the overall context of the impacts to the species and habitat from past, present, and future actions as well as the condition of the affected species and critical habitat.[27]

Recent court cases have reinforced the requirements provided in Section 7 regulations that NMFS must evaluate the effects of a proposed action within the context of the current condition of the species and critical habitat, including other factors affecting the survival and recovery of the species and the functions and value of critical habitat. In addition, the Courts have directed that our risk assessments consider the effects of climate change on the species and critical habitat and our prediction of the impacts of a proposed action.

In *National Wildlife Federation v. National Marine Fisheries Service*, 481 F.3d 1224 (9th Cir. 2007), the Ninth Circuit reviewed the agency's use of its jeopardy standard at 50 CFR 402.02 in a Biological Opinion and found that the agency is required "to consider both recovery and survival impacts." Id. at 1237. This is consistent with the Ninth Circuit's view on the application of the adverse modification regulation in *Gifford Pinchot Task Force v. U.S. Fish and Wildlife Service*, 378 F.3d 1059 (9th Cir. 2004).

National Wildlife Federation made it clear that it would be unreasonable for the agency to consider only the likelihood of survival when applying the jeopardy standard and ignore the likelihood of recovery. The court emphasized that an interpretation of a law that gives no significance to a portion of that law is disfavored**.** ~~Id.~~ Since *National Wildlife Federation*, NMFS considers the likelihood of recovery, as well as the likelihood of survival, when it applies the jeopardy standard to a listed species.

In *Gifford Pinchot*, the Ninth Circuit reviewed the USFWS's use of the adverse modification standard at 50 CFR 402.02. The service had issued several Biological Opinions in which the service determined that destruction or adverse modification of critical habitat would not occur unless the action appreciably diminishes the value of the critical habitat for the species survival, even if the action appreciably

---

[27] For example, see the definitions of "cumulative effects," "effects of the action," and the requirements of 50 CFR 402.14(g)

diminishes the value of the critical habitat for the species recovery. This interpretation of the regulatory standard was challenged. The Ninth Circuit found that Service's interpretation was inconsistent with the Endangered Species Act. The court found that sufficient critical habitat for the survival of the species was not enough because "Congress said that 'destruction or adverse modification' could occur when sufficient critical habitat is lost so as to threaten a species' recovery." After the Ninth Circuit's decision in *Gifford Pinchot,* NMFS issued a memorandum on November 7, 2005, describing how the agency should apply the destruction and adverse modification standard. Thus, NMFS must determine whether affected designated critical habitat is likely to remain functional (or retain the ability to become functional) to serve the intended conservation role for the species both in the near and long term under the effects of the action, environmental baseline and any cumulative effects (NMFS 2005).

7.2 Evidence Available for the Analysis

To conduct these analyses, NMFS considered many lines of evidence, using the best scientific and commercial data available found in published and unpublished sources that would help us make the required determinations. Detailed background information on the status of the species and critical habitat has been published in a number of documents including the Steller Sea Lion Recovery Plan (NMFS 2008a)(see below); the draft Supplemental EIS for this action (Section 3.1.1); the draft Supplemental EIS for the FMPs (NMFS 2000); the marine mammal stock assessments (NMFS 2009); recent Biological Opinions (NMFS 2000, 2001, 2003); Loughlin and Tagart (2006) and the numerous papers described in this document. Despite the unprecedented amount of research in the 2000s and our review of the published and unpublished information on Steller sea lions, our knowledge of the biology and ecology of Steller sea lions, including their life history, population dynamics, and their response to environmental change, is still imperfect. Numerous reports based on the information available for the 2000 FMP Biological Opinion (NMFS 2000) noted the limitations of the available information to make educated, scientifically sound determinations (SSC 2001 [Review of the FMP Biological Opinion], Bowen *et al.* 2001 [Review of the November 30, 2000 Biological Opinion], ASSLRT 2001, DeMaster *et al.* 2001 [Summary of the Is It Food? II Workshop]). However, since those reports, literally hundreds of papers on Steller sea lion ecology, marine ecology, and fisheries have been published that relate directly to our analysis (many included in Loughlin and Taggart 2006). Much insight has been gained on predation, changes in Steller sea lion abundance, reproductive success~~,~~ and other vital rates, as well as their likelihood of extinction (NMFS 2008a). Yet, as in most natural systems, it is extremely difficult to make causal relationships. Thus, most of the analyses are based on making correlations between likely stressors and the likely effects. In order to support these analyses, NMFS will look at these potential mechanisms from different perspectives to see if our conclusions hold up to rigorous evaluation.

In previous Biological Opinions and conservation actions, NMFS utilized four types of management measures to reduce the likelihood that fisheries were competing with Steller sea lions or adversely modifying its critical habitat:

1. Fishery restrictions: areas and periods that were restricted or closed for the pollock, Pacific cod, and Atka mackerel fisheries (i.e., rookery and haulout closures relevant to both critical habitat and to RPA measures),
2. Temporal distribution of TAC (disperse catch throughout the year),
3. Spatial distribution of TAC (fish according to biomass distribution),
4. Global control rule: assessment mechanism to reduce TACs at a faster rate than status quo when biomass falls below the target biomass level.

In this Biological Opinion, NMFS has provided additional evidence (Chapters 4 and 5) and new analyses, which indicate that fisheries, as authorized under the BSAI and GOA FMPs, have a likelihood of negatively impacting the ability of the western DPS of Steller sea lions to survive and recover in the wild. These impacts, however, are regional in scale and NMFS will describe below the rationale for focusing on these areas to ensure future recovery of Steller sea lions throughout the range of the western DPS. In this chapter NMFS will assess available information to make the necessary determinations under the ESA. To do this, NMFS has taken a fine-scale look at the exposure and response of Steller sea lions (i.e., RCA analysis), which was not conducted in previous Biological Opinions. We also have examined new research studies that address the broader, regional requirements of foraging Steller sea lions. NMFS also considers additional information on the performance of Steller sea lions, environmental conditions, and stressors including fisheries, in a subregional context. This subregional evaluation of Steller sea lion vital rates and overall performance since the last Biological Opinion was issued provides a closer look at the status of many factors influencing ~~a~~ the decision on whether the current action is likely to jeopardize the western DPS or adversely modify its designated critical habitat. Below, we synthesize the available information on Steller sea lions and their habitat by sub-region, including pup and nonpup trends throughout their range at all trend sites, prey dynamics and prey preferences, natality information, harvests of the prey field within and outside critical habitat zones, harvest as a component of overall biomass, sea lion consumption of prey as a component of overall available forage biomass, and several metrics of biomass to consumption, and fishery harvest, ratios. It is these types of evidence that NMFS will use to determine the impacts of fishery policy contained within the FMPs and the fisheries that are implemented based on those FMPs.

This Biological Opinion uses recommendations for agency action found in the Revised Steller Sea Lion Recovery Plan (NMFS 2008a). The Revised Recovery Plan replaced the 1992 Recovery Plan, which was developed before the threatened Steller sea lion population was divided into an eastern and a western distinct population segment (DPS) and the western DPS was determined to be endangered. The replacement of the 1992 Recovery Plan by the Revised Recovery Plan responded to a conservation recommendation in the 2000 FMP Biological Opinion (NMFS 2000), which found that the 1992 Recovery Plan was out of date and needed replacement. NMFS assembled a new Steller Sea Lion Recovery Team (Recovery Team) in 2001 to assist NMFS in replacing the 1992 Recovery Plan.

The first draft of the revised plan was written by the Recovery 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 representatives from conservation organizations. In March 2006, the Recovery Team submitted a draft of the Revised Recovery Plan to NMFS, at which time it became an agency document. The Revised Recovery Plan then went through peer and public review and NMFS completed the Revised Recovery Plan on February 29, 2008.

The Recovery Team believed that it was plausible that the conservation measures implemented since 1990 positively affected 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 sub-regions, and as the Revised Recovery Plan was being developed, incomplete data from 2006 and 2007 indicated the population overall was either stable or declining slightly. It was not known whether the slowdown in decline, the period of increase, and the stability or near stability was a result of management actions, natural changes in the ecosystem, or other factors. In developing the Revised Recovery Plan, it became evident to the Recovery Team that the eastern DPS had been recovering for nearly 30 years, increasing at about 3% per year throughout much of its range. 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 provided a possible recovery scenario for the western DPS. Population trends in the eastern DPS indicated that Steller sea lions in the North Pacific could 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. However, population trends since 2004 have not shown that rate of growth. The population growth rate for the western DPS is 1.4% for 2000-2008.

A major factor in our inability to determine the extinction risk of Steller 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 Steller sea lion population to demonstrate that threats are reduced is to increase over an extended time period.

The Recovery Team believed that it was important to consider sub-population declines in recovery and therefore established specific sub-regions in the geographical range of the western DPS. These sub-regions were designed to ensure that a viable population remained throughout the geographical range of the western DPS for recovery purposes. These sub-regions were not included as part of the PVA due to a lack of data on exchange rates between sub-populations. The PVA developed for the Revised Recovery Plan 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 did not capture the more complex sub-area population changes observed in the 1980s and 1990s. However, the Recovery Team strongly believed that all parts of the range must remain occupied to ensure recovery. The western DPS of Steller sea lion does not migrate, but breeds at a number of widely distributed rookeries that serve to maintain populations throughout its range. Reproductive isolation of populations has occurred in the past (hence the eastern and western DPSs), but it is not clear to what extent the western DPS could withstand further fragmentation of breeding populations if a portion of the range were extirpated. 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 sub-regions or more) could indicate that extinction risk may still be high and that further research would be needed to understand the threats and would indicate a lack of recovery for the western DPS as a whole. NMFS, in finalizing the Revised Recovery Plan, believed that it was important to maintain viable sub-populations within the western DPS and not rely solely on the core of the range to provide for increasing population numbers over the short-term.

For recovery, NMFS expects to see natality increase 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 potential of Steller sea lions. Therefore, the choice of a time period in the recovery criteria in the Recovery Plan (i.e., 15 years for downlisting and 30 years for delisting) was influenced not only by the need to be confident that abundance has increased, 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 Steller 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.

Another factor to consider is that the ability of the Steller 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). This factor was not included in the PVA, and thus time to recovery may be longer (or different) than expected, especially if all animals in a given subarea are extirpated.

When considering if demographic recovery criteria are being met (i.e., Steller sea lions are recovering), NMFS will use available information on the population ecology and vital rates to ensure that they support the trends observed in the western DPS as a whole and in each sub-area. Available information on pup counts, production (natality), 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-region trend rates (NMFS 2008a). To avoid adverse modification of critical habitat, NMFS must show that the fisheries are not impeding the ability of Steller sea lions to "recover" and that they are "recovering." To make this evaluation, NMFS constructed the criteria in the Revised Recovery Plan to facilitate the evaluation of whether the western DPS is "recovering" by employing demographic criteria that use trend rates over long time periods. This establishes both short-term and long-term benchmarks of trend rates and population needs (i.e., prey resources).

The recovery criteria from the 2008 Revised Recovery Plan are:

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

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 cannot be declining significantly. The population trend in any sub-region cannot have declined by more than 50%. The 7 sub-regions are the same as listed in a-g above.

Demographic results (and modeling [DeMaster 2009, Johnson 2010, Ianelli 2010]), indicate that the western DPS in the U.S. increased at an average rate of about 3% per year from 2000 to 2004 and has been stable overall from 2004 to 2008, for an average of about 1.4% per year between 2000 and 2008 (Tables 3.1b and 3.1c). DeMaster (2009), Johnson (2010), and Ianelli (2010) found the 2000 to 2008 trend to be uncertain and not statistically significant ($P>0.1$), thus failing recovery criterion #1 at the DPS level for downlisting.

Looking at the sub-region trends considered in criterion #2 in the delisting criteria, the number of Steller sea lion non-pups declined at a statistically significant rate between 2000 and 2008 in only one sub-region (i.e., the western AI). This sub-region is also close to reaching a 50% overall decline that would violate delisting criterion #2, which does not allow for delisting when one subarea declines by more than 50%. The western AI decreased by 45% in non-pup counts between 2000 and 2008. In the central AI, non-pup counts declined, but not at a statistically significant rate (P>0.1), between 2000 and 2008. The change in the number of non-pups in the central Aleutian Islands sub-area from 2000-2008 was -11% or an annual rate of change of -1.5%. The Russia/Asia sub-region has been increasing since the early 1990s (due entirely to increases in abundance of the Asian component), and is currently estimated to be increasing at a statistically significant rate of 4.3% per year. Thus, the current sub-region trajectories satisfy criterion #2, because no two adjacent sub-areas are declining at a statistically significant rate.

It is important to recognize that in 2001 and 2003 NMFS did not have any demographic recovery criteria against which to assess population performance. Thus, NMFS looked at underlying health indices and the overall trend of the western DPS. And in 2003, NMFS did not have new information on natality indices or the detailed information we now have on the underlying vital rates of the population. Also, NMFS did not directly consider the Russia/Asia component of the western DPS in 2003. The recovery criteria in the Revised Recovery Plan provided NMFS with guidance on determining whether the western DPS is recovering or not when it conducts its section 7 consultations. While the western DPS in U.S. waters appears to have responded to the fishery management measures in the 2000s much as predicted, some of those predictions for sub-regions were continued declines (e.g., western AI). Continued population declines in some regions along with lingering signs of nutritional stress (e.g., relatively low levels of pup to non-pups in some sub-regions) in others indicates that the western DPS still has an uncertain future regarding full recovery.

7.3 Risk Analysis

The definition of recovery (i.e., conservation) under the ESA does not require a species to be restored to its historic range and abundance. It also does not require the restoration of a species to all of the remaining suitable habitat. What it does require is the use of all methods and procedures that are necessary to bring any endangered or threatened species to the point at which the measures provided by the ESA are no longer necessary.

The phrase “throughout all or a significant portion of its range” (SPOIR) is used in the definitions of endangered and threatened. Neither “significant” nor “range” are defined in the ESA or implementing regulations. Hence, the ESA provides NMFS with discretion, based on the best scientific and commercial data available, to develop recovery goals and implement recovery plans designed to conserve and recover species.

Given that the ESA is intended to avoid species extinction, NMFS avoids the pitfalls of a purely quantitative approach by 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 having multiple self-sustaining populations that are genetically and demographically viable. Specific to the western DPS of Steller sea lion, we have assumed that the currently observed robust (and significant) rates of recovery in the Russian/Asian, eastern Aleutian, western GOA, and eastern GOA sub-regions will continue. This represents a very large portion of the existing range of the western DPS and therefore creates a strong position on which to diminish further the risk of extinction to this DPS.

NMFS believes that it is important to consider trends in specific sub-regions that constitute the entire distribution of the western SSL DPS, as outlined in the Revised Recovery Plan for the western DPS. The PVA developed in the Recovery Plan was based upon a single population and did not consider sub-population/meta-population dynamics for the western DPS, which some believe could be an important influence on persistence (e.g., Winship and Trites 2006). Others have followed this meta-population approach, and have conducted PVAs that that combined the dynamics of the western DPS and the eastern DPS and demonstrated at that level of aggregation that Steller sea lions are likely not at risk of extinction (e.g., Boyd 2010). However, concerning the results of Boyd (2010), NMFS cannot rely on this finding because Steller sea lions are recognized as two distinct populations under the ESA.

In the Revised Recovery Plan NMFS did not consider results from PVAs that captured the more complex sub-area population structure because of a lack of movement data between putative sub-areas. However, the Recovery Team believed that because all parts of the range are currently occupied, it would be wise to maintain those historic 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. It is plausible that significant declines in the western Aleutian Islands sub-region could indicate that the extinction risk for the western DPS may still be too high unless stressors affecting the population in sub-regions are mitigated. Additional research will be important to better understand the threats and risks to each sub-region and the DPS overall. NMFS believes it is important to maintain viable sub-populations within the western DPS and not rely solely on the core of the range to provide for increasing population numbers over the short-term. Thus, by protecting those sub-regions in decline, the western portion of the range is expected to stabilize and the overall rate of recovery for this DPS should move toward the 3% per year standard defined in the Revised Recovery Plan.

7.4 Integrating the Effects

The preceding discussions describe the various quantitative and qualitative models, decision frameworks, and ecological foundations for the analysis presented in this Biological Opinion. Many of the methods described focus the analysis on directed fishing for particular species. Key to the overall assessment, however, is an integration of the effects of these fisheries with each other and with the baseline set of stressors to which the species and critical habitat are also exposed. In addition, the final steps of the analysis require a consideration of the effects of the action within the context of the reference (or without action) condition of the species and critical habitat. Here, we integrate effects of the ongoing action (Chapter 5) from the time of the most recent opinion (FMP-level in 2000, project-level in 2003) and add these to the baseline (Chapter 4) and cumulative effects (Chapter 6) to assess whether it is reasonable to expect that the proposed action is not likely to: (1) result in appreciable reductions in the likelihood of both survival and recovery of threatened and endangered species in the wild by reducing their numbers, reproduction, or distribution, or (2) reduce the value of designated critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and critical habitat (Chapter 3).

This chapter is organized by species or DPS such that we first integrate and synthesize the effects to survival and recovery, and follow with the effects on designated critical habitat for each of the species under consideration in this Biological Opinion: western DPS of Steller sea lion and its critical habitat, eastern DPS of Steller sea lion and its critical habitat, humpback whales, and sperm whales.

7.4.1 Steller Sea Lion: Western DPS

7.4.1.1 Summary of Responses to the Environmental Baseline, Effects of the Action, and Cumulative Effects

Our review of the Environmental Baseline was extensive, evaluating direct and indirect effects on Steller sea lions and their habitat. Table 4.8 provides an overall summary of the major stressors, which were identified in the FMP Biological Opinion (2000), and what we believe is currently influencing Steller sea lion population vital rates and population trajectories. Tables 5.1 to 5.7 provide an overview of fisheries catch and biomass information. Table 5.8 provides a summary by area of SSL trends in abundance, diet, catch and biomass (both in critical habitat and for the entire area).

Our review of the condition of the western DPS of Steller sea lion (Chapter 3) within the action area found that the overall western DPS decline was likely due to the cumulative effect of multiple factors, and that the marked change in the rate of the decline since 1990 suggests that the factors that contributed to the more rapid declines may not be the most significant stressors now operating (Section 3.1.14). Direct sources of mortality likely were significant contributors to the Steller sea lion population declines observed prior to the 1990s (Section 4.3). Since 1990, rates of mortality from these sources such as harvests, shooting, entanglement, and incidental catch have been substantially reduced. The reduction in the rate of mortality from these sources has likely contributed to a rebound in both juvenile and adult survival rates (Section 3.1.4). Subsistence harvests of Steller sea lions continue but have declined substantially and are unlikely to be a substantial factor.

Another factor that has the potential to be a significant top-down source of mortality is killer whale predation. In Section 4.2.3 we walk through the competing hypotheses regarding the likely effects of killer whales and provide a summary in Section 4.7.1.4. Generally, available evidence from survival rates, isotope analyses, field observations, and observations on the distribution and abundance of prey types place the level of predation by killer whales within what we would expect for natural mortality of Steller sea lions. However, we note that in some areas (e.g., central Aleutians) where sea lion numbers are relatively low, the effects of killer whale predation could be amplified, although the data to evaluate this hypothesis are not available.

Evidence that indirect or bottom-up factors (see Section 4.7.2) may have contributed to the decline observed from the mid-1970s through the late 1990s include reductions in size at age, possible depressed late-term pregnancy rates, significantly reduced pregnancy rates for lactating females, and a decline in per capita natality of female sea lions at some rookeries. These responses by Steller sea lions are opposite to those predicted by direct, top-down, factors. Typically, body condition, growth rates, and natality should increase or remain the same when population abundance is reduced by direct, top-down factors. Bottom-up factors appeared to be affecting Steller sea lions as early as the 1960s and 1970s (see Section 3.1.14), at about the same time that large numbers of Steller sea lions were also killed directly (especially in the late 1970s and 1980s).

One possible explanation for reduced population abundance and poor body condition indices is a reduction in carrying capacity. A reduction in the carrying capacity of an ecosystem for a species could be associated with bottom-up factors. The carrying capacity of the North Pacific for Steller sea lions likely fluctuates in response to changes in the environment. Maschner *et al.* (2010) reported from the archeological record in the western GOA and southern Bering Sea regions that Steller sea lions likely have gone through at least three major declines in the last 1000 years and that these collapses are closely tied to reported regime shifts in the North Pacific ecosystem. Maschner *et al.* (2010) also note that the archeological record from this part of Alaska suggests some response of the Steller sea lion population from Aleut harvest for skin-covered kayaks in the period 1400-1700 AD. Both regional and localized fisheries removals of prey could have exacerbated natural changes in carrying capacity, possibly in non-linear and unpredictable ways. Changes in the overall energy density of the prey field due to both climate shifts and long-term fisheries impacts (e.g., exploitation strategy) may have reduced the efficiency of Steller sea lions and affected their ability to obtain adequate energy to maintain body condition and full reproductive potential.

In our review of climate and regime shifts, gadids were not necessarily affected across the range of Steller sea lions by the 1977 regime shift. Although it appears that EBS pollock did benefit from this change, GOA pollock and Atka mackerel appear to have been unaffected or affected in different ways that are still not clear. Also, the Steller sea lion population may have been increasing during the warm climate regime of the 1940s and 1950s, a period that was likely rich in gadids, but may have been affected by nutritional stress as early as the 1960s and 1970s, before the 1977 regime shift (Trites and Larkin 1992). Results by Hennen (2006) correlate sea lion declines with fisheries around rookeries in the 1980s. However, Hennen found no correlation between fisheries and sea lion dynamics in the 1990s after conservation measures were enacted around rookeries and shooting was prohibited.

To summarize the Environmental Baseline impacts, climate and regime shifts, diseases, parasites, and predation have affected Steller sea lions throughout their 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 Steller sea lion population itself. Steller sea lions persisted in the North Pacific despite the adverse impact of these stressors. Therefore, for tens of thousands of years prior to the 1970s, Steller sea lions adapted to and accommodated fluctuations in their carrying capacity due to natural variability, disease and parasitism, killer whale predation, and apparently maintained, on average, a relatively large population. This is not to say that the population did not go through historical changes in size or distribution as reported by Nelson (1887) and Maschner *et al.* (2010) or similar changes for seabirds (Causey *et al.* 2005). The western portion of the range of Steller sea lions was probably at a relatively large population size at the beginning of the sharp declines in the 1980s, and may have been increasing prior to that decline.

The cumulative effects of future state, tribal, local, and private actions on Steller sea lions and their critical habitat have been presented in Chapter 6. These cumulative effects include State fisheries, subsistence harvest, illegal shooting, entanglement in marine debris, disease, and disturbance from vessel traffic. The reported take level associated with subsistence harvest in the western DPS is 206 kills per year on average between 2003 and 2007 (Angliss and Allen 2009). As concluded in the Recovery Plan (NMFS 2008a), this level of removal should not impede recovery of the western DPS. However, levels of Steller sea lion harvest in a few locations in the western DPS (e.g., Atka, where the total take estimate for 2007 was 54, with an upper range estimate of 87.2) could contribute to the already downward trend in the local area and contribute to the overall downward trend in the sub-region.

State-managed fisheries include salmon fisheries that occur in nearshore and offshore areas. Herring fisheries are also managed by the State. Both of these target high-energy forage species that may be important components (at least seasonally) of the diet of Steller sea lions. Additional cumulative effects include state groundfish fisheries in state waters and in the EEZ that are not managed under parallel regulations and are not included in an FMP. This includes lingcod, black rockfish, blue rockfish, state waters sablefish, and the state waters Pacific cod fishery. The State of Alaska employs various management measures that indirectly provide some measure of protection to Steller sea lions, and all waters within 3 nm of Steller sea lion terrestrial sites in the western DPS are closed to vessel entry, including vessels fishing under State programs. These state fisheries as currently prosecuted would present similar issues and concerns as those associated with the proposed actions, although for some fisheries, such as pollock, at reduced levels. However, the effects of state cod fisheries in certain regions may be of more concern. For example, a new state cod fishery opened in 2006, which is prosecuted solely in near-shore waters (i.e., 0-3 nm from shore). The harvest level of this state fishery is similar in magnitude to the Federal fishery in this area. The amount to which state fisheries may add to the cumulative effects to the western DPS remains unknown, but could be significant in that they take place within near-shore areas and often target highly-concentrated, high value prey species for Steller sea lions.

As discussed previously, indirect or bottom-up factors may have contributed to the decline observed from the mid-1970s through the late 1990s. These bottom-up factor(s) appeared to be affecting Steller sea lions as early as the 1960s and 1970s, at about the same time that large numbers of Steller sea lions were also killed directly (especially in the late 1970s and 1980s). The combination of reduced population abundance and poor body condition indices is consistent with a substantial reduction in carrying capacity (Section 3.1.14.1). Changes in vital rates may have been a function of acute direct mortality factors, as well as chronic nutritional stress resulting from a combination of reduced prey availability and quality. Two stressors were likely to have affected the prey field for Steller sea lions: (1) climate induced changes in the species composition, distribution, or nutritional quality of Steller sea lion prey and (2) fishery-induced changes in localized or overall prey abundance and quality. While the carrying capacity of the North Pacific for Steller sea lions likely fluctuates in response to changes in the environment, fisheries removals of prey likely have exacerbated natural changes in carrying capacity, especially in some sub-regions (Sections 5.1.5 and 5.1.6).

Overall, the western DPS of Steller sea lion has had a statistically significant improvement in the rate of change from the 1990s to the 2000s. In this recent decade, four of the seven sub-regions have positive rates of growth, although only two are statistically significant. Only one sub-region is declining at a statistically significant rate (i.e., western Aleutian Islands) in this last decade. Over the last eight years, the numbers of sea lions in this sub-region have declined by over 40%. If it were not for this one sub-region, it could be argued that the western DPS of Steller sea lions were moving toward recovery, as (1) overall the population is increasing and moving toward the number of animals required for downlisting, (2) no two juxtaposed sub-regions are in significant decline, and (3) no one sub-region has a decline in abundance of over 50%. However, because of the current decline in the western Aleutians, as well as the slow decline observed in the central Aleutian sub-region, the recovery of this DPS is not meeting the criteria in the Revised Recovery Plan (NMFS 2008a). If population growth trends in the western and central Aleutians continue at current rates, Steller sea lions may be extirpated from this portion of their range in the foreseeable future.

In an analysis of smaller geographic areas, RCAs, we find that the abundance of Steller sea lions is in significant decline in the western Aleutian sub-area (RCA 1). In the western portion of the central Aleutian sub-area (RCAs 2 and 3), the sea lion population appears to be declining, but the rate of decline since 2000 is not significant. In the eastern portion of the central Aleutian sub-area (RCAs 4 and 5), non-pup counts are decreasing in RCA 4 and increasing in RCA 5. However, pup production is increasing in both areas (Table 5.8 and Figure 5.1). In the eastern Aleutian sub-area (RCA 6 and part of RCA 7), sea lion numbers are increasing at a robust rate (i.e., 3% or over per year). In the western GOA sub-area, sea lion numbers are increasing at a robust rate (i.e., 4.3% per year). Further, in the eastern portion of this sub-area (RCA 8), the rate of recovery is not robust (between 0 and 3% per year). In the central GOA sub-area (part of RCA 8 and part of RCA 9), sea lion numbers are stable. Finally, in the eastern GOA the rate of increase is both robust and statistically significant. However, in the western portion of the sub-area (RCA 9), the rate of recovery is not robust, while in the eastern portion of the sub-area, the rate of recovery is robust.

The western Aleutian Islands and the western portion of the central Aleutian Islands (RCAs 1-3) area representative of ecosystems characterized by a narrow continental shelf. Throughout much of the Aleutian Islands, Steller sea lions rely primarily on two key prey species: Atka mackerel and Pacific cod. Telemetry results indicated that juveniles foraged intensively inside of critical habitat, although Steller sea lions have been observed ranging widely offshore, perhaps foraging within eddy systems to the west and north of the Aleutians. Since 1999, fisheries have removed a consistently high proportion of the total catch within critical habitat. During this same time Steller sea lions numbers continued to decline. While some regionally limited fishery changes have been implemented to reduce seasonally aggregated catch within critical habitat or to improve dispersal of catch (e.g., seasonal/spatial quotas, platoons), in general these measures have not been associated with any significant change in Steller sea lion trends. In the far west (RCAs 1-3), a combination of factors has possibly affected Steller sea lions: (a) total removals of Atka mackerel and Pacific cod with a high proportion of catch inside critical habitat, (b) spatial heterogeneity of Steller sea lion foraging habitat (highest habitat patchiness, fewest prey options) and (c) killer whale predation.

RCAs 4 and 5 have marine habitats that are similar to those in RCAs 1-3 due to a narrow shelf, relatively small groundfish biomass, and a low diversity Steller sea lion diet. There has been no fishing for the primary groundfish prey species (Atka mackerel) within critical habitat in RCA 5 since 2002, nor any directed Atka mackerel fishery ever in RCA 4. In RCA 4 Steller sea lion non-pup numbers show a decline between 2000 and 2008, while pup production has increased during this time period. In RCA 5, sea lion numbers are increasing, but not robustly. The results of a high proportion of Pacific cod catch within critical habitat in winter in an intermediate Steller sea lion foraging environment in RCAs 4-5 has possibly contributed to chronic nutritional stress that has adversely affected survival and reproduction to the extent that the population is not recovering at a robust rate.

The central Bering Sea and GOA region (RCAs 6-7) represents an ecosystem characterized by a broad continental shelf, the largest biomass of important Steller sea lion groundfish prey species (pollock and cod), and a relatively diverse Steller sea lion diet. Here, Steller sea lion numbers are increasing at a robust rate. Telemetry data from RCA 6 shows that juveniles forage primarily within critical habitat areas including the extensive SSLCA. In RCA 7 juveniles utilize critical habitat areas intensively, but they also move offshore into eddy systems. Since 1999, fishery measures have significantly reduced catch in critical habitat and have seasonally dispersed catch as well, particularly in RCA 6. Fishery management measures in this highly productive region appear to be sufficient to allow for recovery of these RCAs.

RCAs 8 and 9 are characterized by a continental shelf and groundfish prey biomass of intermediate magnitudes compared to Areas 1-5 (smaller) and Areas 6-7 (larger). The diet of Steller sea lions is relatively diverse in these areas, and the chief groundfish prey species are pollock, salmon, Pacific cod and arrowtooth flounder. A high proportion of the total catch of pollock and Pacific cod is caught in winter and within critical habitat. Steller sea lion numbers have stabilized over the last 20 years, but have shown only slight increases in the 2000s in these RCAs, suggesting that fishery measures may have provided for limited recovery.

RCA 10 is the area that contains the easternmost region of the western DPS. It contains Prince William Sound, the eastern GOA, and the Kenai Coast. Steller sea lion numbers appear to be recovering in this region at a robust rate and are meeting the demographic criteria contained within the Recovery Plan. The composition of Steller sea lion prey differs here from that in RCAs 1-9, shifting to a prey complex more similar to that found in southeast Alaska. Telemetry data indicates that most juveniles forage within critical habitat zones within PWS and the offshore islands. These areas are closed to trawling for pollock and Pacific cod. No changes in fishery management measures appear necessary within RCA 10.

In the Russian/Asian portion of the western DPS (see NMFS 2007 for a detailed description of the Russian component of the western DPS), 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 Steller sea lions at major rookeries and haul-outs in the Kuril Islands alone declined 74% (Merrick *et al.* 1990). By 1990, the total Russian population had declined by approximately 50% to about 13,000 (the populations most proximate to the U.S. portion of the western DPS). Between the early 1990s and 2008, the Russia/Asian population (including pups) increased at a statistically significant and robust rate to about 25,000 overall (Burkanov and Loughlin 2005, V. Burkanov personal communication, J. VerHoof, personal communication).

7.4.2 Assess Risk to the Western DPS

7.4.2.1 Threats to Survival and Recovery

Currently there is no evidence strongly indicating that infectious disease, acute toxicity, or pollutants are responsible for the lack of a robust recovery in Steller sea lion abundance in the western DPS, nor are they thought to have an effect large enough to adversely affect critical habitat. Likewise, given the current rate of juvenile survival in Steller sea lions from the eastern Aleutians through the eastern Gulf of Alaska, predator-prey dynamics between killer whales and Steller sea lions are unlikely by themselves to prevent Steller sea lion survival and recovery (see Section 4.7.2), although significant impacts are possible.

Steller sea lions collected in the GOA during the mid-1980s showed evidence of reproductive failure and reduced rates of body growth consistent with nutritional limitation. Comparatively low birth rates for females from the western DPS from mid-1970s to mid-1980s coupled with elevated embryonic and fetal mortality appear to have contributed to decreased reproductive performance during the period of early decline. Age-structured models fit to pup and non-pup counts suggest that this decline in reproductive success continued through the 1990s and into the 2000s (Holmes *et al.* 2007), but there is no comparable direct evidence that acute nutritional stress was responsible for the continued decline of the western DPS during the 1990s. The continued drop in the reproductive rate in the 1990s and relative low rate increasing abundance in the 2000s, may have been due *inter alia* to chronic poor nutrition. 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 4-6 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 affected reproductive performance of adult females, but not survival of juveniles. There have not been any adult female Steller sea lions captured and handled since the late 1990s, due to the focus on juveniles from 2000 to 2006, and the termination of permits necessary to handle adult females since 2006. Thus, we must examine a variety of data sources that are not nearly as direct as the data collected in the 1970s and 1980s to evaluate the potential for adverse effects due to chronic nutritional stress.

Evidence from the central GOA suggests that decreased juvenile survival is largely responsible for the steep decline of Steller sea lions in at least this portion of the western DPS in the 1980s. However, in the 1990s juvenile survival began to recover while lower reproductive success contributed to the slower declines of that period. A lack of a robust recovery between 2000 and 2008 was possibly produced by rising adult and juvenile survival rates, and perhaps declining natality rates. This same process may be affecting RCAs 1-3. Models with population structure resulted in extirpated regions while those built as one population resulted in high likelihoods of extinction for the western DPS. However, while loss of subpopulation connectedness was not considered in the PVA reported in the Revised Recovery Plan and although some subpopulations persisted in the models, NMFS believes that management at the sub-regional level is important to appropriately reduce the risk of extinction overall and to meet the guidelines for downlisting and delisting. Regional extirpation is a threat identified in the Revised Recovery Plan with measures in the demographic criteria to avoid this result (i.e., the requirement to avoid no two adjacent subareas from statistically significant rates of decline over the recovery period).

The western DPS of Steller sea lion shows significant improvement in the numbers of pups produced in the core of its former range, the eastern Aleutian Islands and western GOA, as well as the eastern GOA (Prince William Sound) and the Russian/Asian sub-region. These improvements could be related to the imposition of fishery measures in the early 2000s. However, pup production continues to decline precipitously in the western and part of the central Aleutian Islands (RCAs 1 and 2). The 178°W longitude boundary (near Tanaga Island) within the central Aleutian Islands separates rookeries declining in abundance to the west from rookeries increasing to the east (RCAs 4-10, with a few exceptions). Reduced natality is considered to be a primary driver in the current lack of a robust recovery in the western DPS.

For the western DPS to be downlisted from endangered to threatened, the Revised Recovery Plan requires a significantly increasing overall non-pup trend for 15 years with no significant decline in the population trend in any two adjacent sub-regions (NMFS 2008a). The non-pup count data indicate that currently two sub-regions are in decline, the western and central Aleutians, though only the western Aleutians decline is statistically significant. Without abatement, the decline in the central Aleutians will become statistically significant. The eastern Aleutians and western GOA are both increasing, although not significantly, with the central GOA in a slight decline (- 0.14%). At each end of the geographic distribution of the western DPS, the Russia/Asia sub-region and the eastern GOA sub-region, both are increasing significantly (Table 5.8).

In some areas, management measures to mitigate these fishery impacts (e.g., Atka mackerel fishery exclusion zones around rookeries) may not be equally effective in all areas due to differences in the spatial distribution of essential habitat inside and outside of these zones. Exclusion zones appear to be more effective in isolating fishery effects in regions where zone and habitat boundaries are similar (e.g., Seguam Pass). In these areas, aggregations of fish inside the exclusion zone have low exchange rates with aggregations of fish outside, at least over the course of short-term fisheries (i.e., 1-2 weeks). Exclusion zones are less effective in those areas where zone boundaries cut across habitat where fish would be expected to move freely, thus allowing fisheries outside to negatively influence prey populations thought to be protected inside the zone (e.g., Amchitka). FIT and other studies of trawl exclusion zone effectiveness around Cape Sarichef (Unimak Island) have shown similar results with respect to the Pacific cod trawl fishery on the southeastern Bering Sea shelf (Fritz and Brown 2005, Conners and Munro 2008). Fishery exclusion zones within parts of critical habitat, especially in the Aleutian Islands and also in the central GOA, likely have not taken into account the spatial heterogeneity of important prey habitat. This could result in adverse effects on prey fields within closed areas, thus compromising in part the utility of closure zones to protect important Steller sea lion habitat without having taken into account the prey habitat within those areas. In such areas, larger closure zones may be needed to adequately protect the prey field of Steller sea lions.

7.4.3 Summary of Evidence for Risks to the Western DPS

The Steller sea lion population in the western DPS has declined significantly through the 1980s and 1990s, but since 2000 the decline has ceased and in a majority of sub-regions the wSSL population is increasing. Sub-regions where the decline continues at more than 0.5% are the western and central Aleutian Islands (RCAs 1-5). In the other five sub-regions Steller sea lion abundance is approximately stable to increasing, although the rate of increase is statistically significant in only two of these five sub-regions.

The area inhabited by the western DPS is a fished ecosystem, from which large quantities of certain target species have been harvested since the 1960s, initially by foreign fisheries and since 1989 by entirely domestic fisheries. The count of Steller sea lions in the western DPS in the Kenai to Kiska census area was over 100,000 animals (non-pups) by the end of the 1950s and about 90,000 around the end of the 1970s. Then a marked decline commenced with about 22,000 non-pups counted in this census area by 1990, and by 2000 the number of non-pups was at about 15,000. About 17,000 were counted as of 2008 in the Kenai to Kiska census area, the last survey date for non-pup animals.

Many have speculated on reasons for the decline in the 1980s and 1990s, including a climate regime change in the late 1970s that may have altered habitat conditions and prey abundance and diversity, increased predation, intentional and non-intentional human-caused mortalities, and fishery effects. It is generally agreed that the primary factor or factors responsible for the steep decline in the 1980s will never be identified with any assurance. Likely it is a combination of multiple factors (NRC 2003). In this last decade, the available information on birth and death rates indicates that adult and juvenile survival rates are similar to those pre-decline, but that natality has declined on the order of 30% relative to the pre-decline era. Our understanding about changes in these vital rates is limited because the number of sub-regions properly studied in the western DPS is limited to at most three (e.g.,Marmot Is. in RCA 9).

Pup to non-pup ratios are an indicator of reproductive rates (or natality) in sea lion populations. Chapter 3 describes some of the caveats about the interpretations of ratios of counts of pups to counts of non-pups. Values for the ratios of counts of pups to adult females on rookeries are provided in Table 3.6. Pup/adult female ratios in the western AI sub-region are the lowest of any of the western Steller sea lion DPS sub-regions (i.e., 0.29). All other sub-regions show pup/adult female ratios of 0.37-0.42, which on average are about 28% higher than in the western AI. In the central GOA, a sub-region that has experienced a 36% decrease in natality over the past three decades (Holmes *et al.* 2007), the pup/adult female ratio is 0.42. The pup/adult female ratio for the eastern DPS is in excess of 0.8, which is associated with a robust rate of increase in abundance for over 20 years (i.e., 3% per year).

Other measures of SSL natality have been published including models by Holmes *et al.* (2007) and Maniscalco *et al.* (2010). Brand/resight work, particularly in Russia (Burkanov 2010) show that some SSLs move away from branding sites, possibly confounding use of pup/non-pup ratios as indicators of natality. Results of these various studies are equivocal in ascertaining a natality rate that would result in continued population increase. Given population increases in four of the seven sub-regions in the western DPS from 2000-2008, natality appears sufficient to ensure increases in most of the sub-regions. However, interpretation of a successful natality rate inferred from pup/adult female ratios is difficult because the ratio in the central Aleutian Islands (0.39), where the numbers of non-pups and pups are decreasing, is greater than the pup/adult female ratios for the eastern Aleutian Islands and the western and eastern GOA, where numbers of non-pups and pups are increasing (Table 5.8).

Pup to non-pup ratios based on data collected in 2009 are also consistent with the inferences based on available data on pup/adult females ratios. That is, pup to non-pup ratios indicate that natality rates of the western DPS are lower than those in southeast Alaska (DeMaster 2009). The pup to non-pup ratio from the two largest and oldest rookeries in southeast Alaska (Forrester Complex and Hazy Island), was 0.85. Rookery pup to non-pup ratios varied from 0.44 to 0.63 among sub-regions in the western DPS in 2009 and averaged 0.57. (DeMaster 2009). A reasonable explanation consistent with the pattern of natality in the western DPS relative to the eastern DPS is that the western DPS is nutritionally stressed because other hypotheses related to mechanisms associated with decreased natality (e.g., disease and contaminants) have not been supported with the available data.

This Biological Opinion seeks to address the issue of whether the groundfish fisheries in the GOA and BSAI areas are likely to jeopardize the continued existence of the western DPS of Steller sea lions or adversely modify its critical habitat. Given that in the seven sub-regions identified in the down- and de-listing criteria in the Steller sea lion Recovery Plan (NMFS 2008a), trends in abundance are increasing in four sub-regions, our primary focus of this Biological Opinion has been directed at the two sub-regions where declines in abundance have been recorded since 2000. The continued decline in these sub-regions is a considerable concern to NMFS, especially because there are fisheries in these sub-regions that target species that have been identified as Steller sea lion prey. We recognize existing fishery measures are likely mitigating impacts on Steller sea lions in other sub-regions, allowing recovery to continue, and are not recommended for change; our focus here is on what additional measures, or changes in measures, may be required in the two sub-areas that are not increasing in abundance currently.

It appears from the best scientific and commercial data available that the following factors have acted or continue to act individually or together to cause significant declines or otherwise limit the rate of recovery in one or more of the sub-regions that comprise the distribution of this DPS.

- Change in carrying capacity for SSLs; the North Pacific is likely a different ecosystem today than in the 1950s and 1960s. No regime shift has occurred since the decline began in the late 1970s or early 1980s that might suggest another favorable foraging environment for Steller sea lions is on the horizon. Therefore, the habitat for the western DPS in the North Pacific may be close to its maximum capacity to sustain both SSLs, other large piscivorous marine mammals, ichthyo-piscivorous competitors, and current fishing levels.
- Overlapping diets among SSL competitors, particularly arrowtooth flounder (Section 4.2.4; Boyd 2010), which is likely at the highest ever abundance in the North Pacific, may be a factor in placing competitive pressure on common prey items for Steller sea lions, and although data are unavailable, conceivably this competitive pressure could have a role in depressing the rate of recovery of the western SSL population. This interaction may vary by sub-region.
- Killer whale predation may be locally high and could be suppressing Steller sea lion recovery in some parts of the Aleutian Islands (e.g.,Durban *et al.* [2010] reported very high numbers of killer whales in the central and eastern Aleutians) and are thought by some to be the single greatest source of mortality for juvenile SSLs in the eastern GOA region (Horning and Mellish 2010a).
- The importance of nutritional stress in explaining the dynamics of the western DPS of Steller sea lions has been debated for decades, and the current conclusion is that pollock in adequate availability can sustain adult Steller sea lions; SSLs for optimal foraging require a diversity of prey species and have adapted to seasonal sources of high energy prey and to the mix of prey species present in their foraging areas (Rosen 2009, Trites *et al.* 2007, Womble *et al.* 2009, Winter *et al.* 2009, Sigler *et al.* 2009).
- Body condition of Steller Sea lions in the western DPS is relatively good (i.e., similar to body condition in animals from the eastern DPS, a population that is increasing at a robust rate), particularly for pups, indicating nutrition is sufficient to produce healthy young. Another indicator of Steller sea lion condition, skull size, hasn't changed as it likely would have if nutritional stress were the primary issue over the last 2-3 decades (Trites *et al.* 2008).

- Direct mortality of Steller sea lions through a combination of historic commercial harvests, subsistence harvests, intentional kill, and incidental take in fisheries may explain a portion of the western Steller sea lion population decline that occurred through 1980 (Trites and Larkin 1992, Atkinson *et al.* 2008) but these sources have not likely affected the population in the past decade, as these sources of mortality have been reduced or eliminated.
- Disease and contaminants could be a factor, but data are scarce and the preponderance of evidence does not support either as being significant factors in understanding the current dynamics of the western DPS. Interestingly, PCB levels and mercury in Steller sea lion tissues were reported to be higher in the western portion of the range of the western DPS (L. Rea, presentation to Steller sea lion Mitigation Committee, January 2010); and plasma haptoglobin levels are significantly higher in sea lions in the Aleutians and GOA than in SE Alaska (Zenteno-Savin *et al.* 1997).
- Correlations between recent trends in abundance for the western DPS of Steller sea lions and catches of commercial groundfish fisheries are highly varied, with no clear findings of significant positive relationships (e.g., Hennen 2006).

Past Steller sea lion telemetry data indicate Steller sea lions in certain areas have tended to forage close to land, most within 20 nm. Recent modeling work by Gregr and Trites (2008) confirm the importance of nearshore foraging areas, particularly in summer and particularly to juvenile and adult females. New spatial analyses indicate that SSLs indeed forage close to rookeries and haulouts, particularly in the 0 to 10 nm zone but also in the areas further offshore to 20 nm (Boor 2010, AFSC 2010b). In particular, recent telemetry information indicates that in RCAs 1, 2 and 3 an unusually large proportion of telemetered animals forage outside 20 nm in winter months (AFSC 2010b). Further, available data indicate that this may also be the case for parts of the central GOA sub-region. Therefore, it appears that foraging strategies of Steller sea lions vary by sub-region. While specific mechanisms related to competitive interactions between SSLs and commercial fisheries are difficult to verify, it appears that commercial fisheries, at least in the western and parts of the central Aleutian Islands, may remove fish that are prey for SSLs that forage there, or may draw down biomass levels in the general region, affecting prey availability in nearshore areas where SSLs prey most heavily (Ortiz and Logerwell 2010).

Diet information indicates the dependence of SSLs on certain prey species varies by sub-region. Steller sea lions in the western and central AI region heavily depend on Atka mackerel (96% FO winter, 55 % FO summer), which is the major target species harvested in commercial fisheries. Steller sea lions also utilize Pacific cod in the western and central AI sub-regions (6 % FO summer, 26 % FO winter).

The AI region also may be a more rigorous physical environment, as evidenced by frequent stormy conditions, variable temperatures, and complex frontal features related to sea surface temperatures. While these characteristics may also be associated with areas of relatively high productivity, some believe that the Aleutian Islands region may also be less hospitable for Steller sea lions (Lander *et al.* 2010) than other regions.

7.4.4 Peer and Public Review

This Biological Opinion was released as a draft document on August 2, 2010. NMFS requested that the public provide comments on the analyses contained in this Biological Opinion and on the conclusions reached. The Council and its Scientific and Statistical Committee and its Advisory Panel convened a special meeting in August 2010 to review the draft Biological Opinion. Over 10,000 comments were submitted to NMFS including extensive scientific reviews of the draft and fairly rigorous critiques of the scientific underpinnings of the conclusions reached. NMFS also requested an internal Agency review of the scientific information contained in the Biological Opinion by a NMFS scientist familiar with Steller sea lions, the North Pacific Ocean ecosystem, and the commercial fisheries prosecuted in this region.

NMFS considered the comments and reviews when developing this final Biological Opinion. The comments and reviews pointed to a variety of concerns with the scientific information used to derive the conclusions reached, and NMFS reexamined these issues and made changes in the document to better reflect current scientific consensus. The document was updated, minor errors and omissions were corrected, and the conclusions based on the available data were re-examined. After that re-examination, NMFS continues to conclude that fishery removals of important prey items for Steller sea lions remains a large concern, as described below.

7.4.5. Conclusions

The analysis in the preceding chapters, and summarized above in this chapter, forms the basis for conclusions as to whether the proposed actions satisfy the standards of ESA Section 7(a)(2). To do so, the action agency must ensure that their proposed action is not likely to jeopardize the continued existence of any listed species or destroy or adversely modify their designated critical habitat. Chapter 3 defines the biological requirements of the western DPS. Chapter 4 evaluates the relevance of the Environmental Baseline to the status of the western DPS of Steller sea lion. Chapter 5 details the likely effects of the proposed action on the western DPS. Chapter 6 considers the cumulative effects of relevant non-Federal actions reasonably certain to occur within the action area. In this Chapter, NMFS reviews the response of Steller sea lion to these various effects (Section 7.4.1) and assesses the risk to the species (Section 7.4.2). It is on the basis of this information and analysis that NMFS draws it conclusions about the effects of the action on the survival and recovery of the western DPS of Steller sea lion.

One question that must be addressed in this consultation is whether the action is likely to jeopardize the species' continued existence. As the court stated in *National Wildlife Federation*, even if the baseline itself causes jeopardy to the species, only if the action is likely to cause additional harm can it be found to jeopardize the species' continued existence. This determination requires an evaluation of the action's effects, separate from the conditions that would exist if the action were not carried out. Having made such an analysis, and recognizing the baseline condition for the western DPS is one in which there exists an unacceptably high probability of extinction, we find that the proposed action, which in this case is the current action continued into the future, is likely to present such "additional harm". This harm would be due largely to the indirect effects of the action in the western and central Aleutian Island sub-regions of the western DPS of Steller sea lion, and the direct effects of the action on their habitat through the removal and negative modification of important prey species.

Our assessment found that there is considerable new information since the 2000 FMP Biological Opinion was completed. Although such new information provides insight into the habitat requirements of Steller sea lions, it also creates an extremely large body of sometimes-contradictory evidence to examine. The above Summary of Weight of Evidence, Section 7.4.3, compiles the salient information relevant to the opinion presented in this Biological Opinion. Many study results are equivocal or lack the definition necessary to inform management action; others show association but fail to demonstrate cause. Our approach was to assess relationships between fishery policy and implementation and the likelihood of affecting Steller sea lions or their critical habitat through a weight of evidence approach. It is reasonable to conclude that some of the conservation measures implemented since the 2000 FMP Biological Opinion have been effective in ameliorating stressors that existed prior to implementation, especially in some sub-regions. However, it is NMFS' Biological Opinion that the available scientific information also indicates these efforts, which are included in the proposed action, are not adequate to ensure that the likelihood of jeopardy to the western DPS of Steller sea lion is avoided. NMFS' finds that the current fisheries, as modified by the actions and RPAs contained in past Biological Opinions, continue to impede the survival and recovery of the western DPS of Steller sea lion. Thus, additional measures are necessary to avoid the likelihood of jeopardy.

A significant rationale for this conclusion is based on the continued decline in abundance of Steller sea lions within the western and central Aleutians (RCAs 1, 2, 3 and 4). This decline may be associated with reductions in numbers of individuals due to decreases in the habitat functionality as a consequence of the continued fishery operations within these regions, and is not significantly influenced by emigration to other areas (although some redistribution may be expected). Other factors may also be contributing to the lack of recovery. Unfortunately, the data needed to evaluate the impacts of killer whale predation are not available. The proposed action, which is a continuation of past and current Federal and parallel fisheries, would be expected to continue this trend, with the risk of eventual extirpation of Steller sea lions in the western Aleutians.

These consequences, along with the present status of the western DPS, are expected to increase the risk of extinction by their effect on Steller sea lion numbers. While new information indicates other regions of the western DPS is or may be showing positive growth, NMFS believes that the extirpation of Steller sea lions in the western Aleutians would be significant to the western DPS, and is expected to appreciably reduce the likelihood of both their survival and recovery in the wild.

The sum of information available on the Steller sea lion population trends, predation and competition for prey resources, fishery harvest of prey resources, other factors like contaminants and disease, diet, foraging geography, and ecosystem carrying capacity indicate that Steller sea lions in the western and portions of the central Aleutian Islands subareas (RCAs 1,2, 3, and 4) require the implementation of conservation measures that will promote the recovery of SSLs sufficient to eliminate the existing risk of overall extinction.

Therefore, the Agency concludes in this Biological Opinion that, while fisheries cannot be unequivocally shown to be a causative factor in continued Steller sea lion declines in the western portion of the wDPS in Alaska, analysis of available data indicate that an adverse relationship between Steller sea lions and the commercial fisheries may exist in the western Aleutian Islands sub-region and portions of the central Aleutian Islands sub-region where two specific fisheries, for Atka mackerel and Pacific cod, target important Steller sea lion prey. This competition between Steller sea lions and the commercial fisheries may compromise the availability of food resources of Steller sea lions sufficiently to likely jeopardize their continued existence or to adversely modify their critical habitat. Only a small percentage of Steller sea lions remain in the western and central Aleutian Islands sub-regions relative to the pre-decline population level. Fishery removals of prey in the western and central Aleutian Islands sub-region may be adversely affecting the western DPS of Steller sea lions in these areas sufficient to stress animals through longer and less successful foraging trips and foraging trips that require more repetitive dives to acquire prey. The possibility that this interaction may be one of several primary causes of the observed declines in non-pup counts cannot be eliminated. Lack of site specific data on vital rates and food habits in the region where declines in abundance have been reported make our evaluation of what factors are impeding a robust rate of recovery of the western DPS difficult. Nonetheless, because of the weight of evidence described in this Biological Opinion and the requirements of the ESA, we must act in a precautionary and measured approach by changing fishery harvests in the sub-regions that have demonstrated declines in Steller sea lion numbers sufficient to impede recovery.

After reviewing the current status of the endangered western population of Steller sea lions, the environmental baseline for the action area, the proposed actions, and the cumulative effects, it is NMFS' Biological Opinion that the action, as proposed, is likely to jeopardize the continued existence of the western DPS of Steller sea lion.

7.5 Steller Sea Lion Critical Habitat: Western DPS

7.5.1 Summary of Responses to the Environmental Baseline, Effects of the Action, and Cumulative Effects

The overall effects to critical habitat by the Environmental Baseline, Effects of the Action, and Cumulative Effects are presented earlier in this chapter (see Section 7.4.1 above for an expanded discussion which is incorporated here by reference in this assessment).

The essential features of this habitat most relative to this assessment are the principal species of Steller sea lion prey, which include pollock, Atka mackerel, salmon, Pacific cod, cephalopods, arrowtooth flounder, and herring, among other species. These species may be highly dynamic, and heavily influenced by environmental variability (oceanographic conditions, regime shifts, climate change), ecological change, anthropogenic factors including commercial fisheries, and predation.

A potential response of Steller sea lions to reduced availability of prey in portions of critical habitat left open to fisheries would be nutritional stress (see Section 3.1). Reduced prey availability can lead to physiological responses by Steller sea lions that directly (e.g., reduced natality) or indirectly (e.g., increased mortality from predators due to increased foraging) reduce their population rate of growth. A sustained reduction of prey resources across a broad geographic region, or ecosystem, would thus reduce the carrying capacity of the ecosystem for Steller sea lions (see Section 3.1.14).

State-managed fisheries occur almost entirely within critical habitat. State-managed fisheries include salmon fisheries that occur in nearshore and offshore areas. Herring fisheries are also managed by the State. Both of these target high-energy forage species that are considered important components (at least seasonally) of the diet of Steller sea lions. Additional cumulative effects include State-managed groundfish fisheries in state waters and in the EEZ that are not managed under parallel regulations and are not included in an FMP. This includes lingcod, black rockfish, blue rockfish, State waters sablefish, and the State waters Pacific cod fishery. The State of Alaska employs various management measures that indirectly provide some measure of protection to Steller sea lions, and all waters within 3 nm of shore within Steller sea lion critical habitat are closed to vessel entry, including vessels fishing under ~~the~~ State programs. These State fisheries would present similar issues and concerns as those associated with the proposed actions, albeit at much reduced levels. Generally, these fisheries currently are small compared to the Federal fishing associated with the proposed action. For example, the 2000 State pollock harvest for the Gulf of Alaska was 1.7% of the Federal fishery, although the state cod harvest was 22.5% of the total Federal ABC. The amount to which State-managed fisheries may add to the cumulative effects to the western DPS remains unknown, but could be significant in that they take place within near shore areas and often target highly-concentrated, high value prey species for these sea lions.

Section 7.4.1 describes fisheries impacts to RCAs within critical habitat and the response of Steller sea lions and is incorporated here by reference.

7.5.2 Assess Risk to Critical Habitat

The question in assessing the risk of the action to critical habitat is whether the proposed action will reduce the value of critical habitat for the conservation of the species. Thus, NMFS must determine whether affected designated critical habitat is likely to remain functional (or retain the ability to become functional) to serve the intended conservation role for the species in both the near and long term under the effects of the action, the environmental baseline and any cumulative effects. Much of the discussion in Section 7.4.2 above is incorporated here as well, specifically impacts to critical habitat and Steller sea lion responses to habitat changes. Because the jeopardy analysis is primarily a habitat-based assessment, and the extent of critical habitat for the western DPS is so extensive, the two assessments are very similar. Rather than make the same arguments in both places, the arguments supporting the likelihood of jeopardy are incorporated here.

Prey resources are the most essential feature of marine critical habitat for Steller sea lions. The status of critical habitat is best described as the status and availability of the important prey resources contained within those areas, which include pollock, Atka mackerel, salmon, Pacific cod, arrowtooth flounder, Irish lord, rock sole, snailfish, herring, capelin, sand lance, other forage fish, squid, and octopus. Dominant prey items vary with region and season, but the most significant groundfish prey items for Steller sea lions in the western DPS are Atka mackerel, pollock, Pacific cod, and arrowtooth flounder, each of which have at least a 10% frequency of occurrence in the Steller sea lion diet.

A reduction in prey resources in critical habitat may result in a reduction in population growth rate, which would therefore be a reduction in the conservation value of the critical habitat. Specifically, reduced prey availability can lead to physiological responses by Steller 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 that region for Steller sea lions. These impacts have generally been referred to as nutritional stress (see Section 3.1.14) and would be a reduction in the functionality of the conservation role of designated critical habitat.

Despite the many factors and changes to these prey species over the last several decades, the biomass for Pacific cod and walleye pollock are near levels assessed in the mid- to late-1970s. However, data on how prey availability within critical habitat may have changed over time is often unavailable. Chapter 4 describes the great difficulty in assigning causation to changes in prey biomass or local abundance.

Removals of fish by these fisheries could have exacerbated natural changes in carrying capacity and may have contributed to declines in the western DPS. The implementation of conservation measures, in the early 1990s, the late 1990s, and early 2000s, is 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, and likely never will be to the extent of identifying cause and effect. Thus, it is possible that conservation measures implemented in the 2000s had a positive impact on reducing the impacts of the fishery exploitation strategy on Steller lions, but fell short of ensuring that the current strategy combined with the local implementation of the fishery sufficiently allows for the survival and recovery of the western DPS.

We have discussed how Steller sea lions foraging near rookery sites in critical habitat depend on the availability of food supplies in the vicinity to meet the energetic needs involved in reproduction (adult females and males, and pups). The issue of prey availability and nutritional adequacy is highly complex and very likely specific to the eco-region or habitat site in question. Nonetheless, there exists qualitative information that indicates some commercial fisheries are causing reductions in prey species (at least locally) to the extent that they may reasonably be expected to inhibit the recovery of the western DPS (e.g., Figure 3.10).

Differences in western Steller sea lion population response since 2000 is likely reflective of three main factors, (a) overall ecosystem productivity, (b) fishery intensity within critical habitat on important Steller sea lion prey species, and (c) predation pressure (i.e., western Aleutian sea lions may be more susceptible than sea lions in sub-regions that contain more animals). For example, Steller sea lion populations in RCAs located in the Aleutian Islands experienced higher fishery intensities inside critical habitat (RCAs 1-3). Steller sea lions in these RCAs performed poorer than those located in the EBS or GOA, which experienced lower fishery intensities inside critical habitat. Critical habitat areas in the Aleutian Islands west of 178°W are open to Atka mackerel directed fishing up to 60% of their annual catch, and open to the directed Pacific cod fixed gear fisheries with few restrictions on catch (other than BSAI TAC). Steller sea lion populations in these areas (RCAs 1-3) continue to decline and have shown no recovery since conservation measures were implemented in the early 1990s (e.g., no shooting). Critical habitat areas in the central GOA (RCAs 8-9) continue to provide a high proportion of the catches of pollock and Pacific cod, particularly in winter. Steller sea lion populations in these areas continued to decline through the 1990s and while increasing in abundance since 2000, the rate of increase is not robust. By contrast, Steller sea lion populations in regions with lower proportions of catch in critical habitat have stabilized or increased in the 2000s and may depend on the overall productivity of the region. Those areas in the EBS or western GOA (RCAs 6-7) have performed better than those in the Aleutian Islands (RCAs 1 - 4), where significant fishing occurs in critical habitat.

7.5.3 Conclusions

From these data and observations, NMFS concludes that the relative intensity of groundfish fisheries as currently prosecuted within critical habitat is negatively associated with Steller sea lion population response since 2000 and that these adverse effects on the availability of important Steller sea lion prey within critical habitat are exacerbated in areas of low ecosystem productivity and habitat spatial heterogeneity. Based on this analysis of the effects of the action, and considering the ongoing nature of this action, it is unlikely that designated critical habitat within the western DPS of Steller sea lion will remain functional (or retain the ability to become functional) to serve the intended conservation role for the western DPS in both the near and long-term.

After reviewing the current status of critical habitat that has been designated for the western population of Steller sea lions, the environmental baseline for the action area, the proposed action for Alaska Groundfish in the Bering Sea and Aleutian Islands and Gulf of Alaska, and the cumulative effects, it is NMFS' Biological Opinion that the action, as proposed, is likely to adversely modify the designated critical habitat for the western DPS of Steller sea lion.

7.6 Steller Sea Lion: Eastern DPS

7.6.1 Summary of Responses to the Environmental Baseline, Effects of the Action, and Cumulative Effects

Steller sea lion populations in southeast Alaska increased at 3% per year through 2002 and likely at greater rates than that between 2002 and 2009 (DeMaster 2009). NMFS recognizes that there may be some small number of western DPS animals present in southeast Alaska, but that this constitutes an extremely small number of animals relative to the current population. Thus most of the increase in southeast Alaska is due to population increases in the eastern DPS. There are no fisheries for the principal groundfish prey species within critical habitat in southeast Alaska, and only very small fisheries for those species in the range of the eastern DPS. Given migration of some animals from the western DPS to the eastern DPS and evidence of pupping of those females, it is likely that habitat conditions in the eastern DPS provide for adequate survival and the ability to recover based on long-term demographics. There is little in the way of cumulative effects to the eastern DPS.

7.6.2 Assess Risk to the DPS

The eastern DPS is not in danger of extinction or likely to become endangered.

7.6.3 Conclusions

After reviewing the current status of the threatened eastern DPS of Steller sea lion, the environmental baseline for the action area, the proposed action for Alaska Groundfish in the Bering Sea and Aleutian Islands and Gulf of Alaska, and the cumulative effects, it is NMFS' Biological Opinion that the action, as proposed, is not likely to jeopardize the continued existence of the eastern DPS of Steller sea lion.

7.7 Steller Sea Lion Critical Habitat: Eastern DPS

Critical habitat within the eastern DPS of Steller sea lion consists of a 3,000 ft zone seaward of the three designated rookeries, numerous designated haulouts, and some identified terrestrial areas.

7.7.1 Summary of Responses to the Environmental Baseline, Effects of the Action, and Cumulative Effects

Given that there is very little removal of prey and virtually no removal of groundfish prey species of interest to Steller sea lions in this very nearshore area designated as critical habitat in the eastern DPS it is reasonable to conclude that there is no substantial effect to critical habitat within the eastern DPS of Steller sea lion.

7.7.2 Assess Risk to Critical Habitat

It is likely that designated critical habitat within the eastern DPS is likely to remain functional (or retain the ability to become functional) to serve the intended conservation role for the species (eastern DPS) in both the near and long-term.

7.7.3 Conclusions

NMFS concludes that the proposed action does not destroy or adversely modify critical habitat in the eastern DPS of Steller sea lion.

7.8 Humpback Whales

7.8.1 Summary of Responses to the Environmental Baseline, Effects of the Action, and Cumulative Effects

Humpback whales were listed as endangered under the ESA in 1973 due to commercial exploitation that severely depleted its populations. Prior to 1905, there were an estimated 15,000 humpback whales in the entire North Pacific; by 1966, following commercial harvest, the population was estimated to be between 1,000 and 1,200 animals. Following the IWC 1965 ban on commercial whaling in the Pacific Ocean**,** Soviet whalers continued to harvest humpback whales until 1980. Currently, some illegal whaling continues although actual harvest levels are unknown.

Since listing and the prohibition on commercial harvest, the Central and Western North Pacific humpback whale populations have increased substantially. The current annual abundance estimate for the entire North Pacific population is 18,000-20,000 individuals, a significant increase from a previous estimate of approximately 6,010 animals. The overall increasing trend for these populations is probably at least partially the result of protective legislation enacted in both the United States and Canada during the early 1970s that resulted in reduced mortality at a time when the population was below carrying capacity.

Humpback whales exhibit two types of migrations: within-season movement through a portion of the summer range, and long-distance migrations between feeding and breeding grounds. However, data from the Structure of Populations, Levels of Abundance, and Status of Humpbacks (SPLASH) project conducted between 2004-2006 indicate that population structure and migration patterns are much more complex than previously understood. Additionally, humpback whales exhibit a high degree of site fidelity to specific feeding areas, with little interchange among them.

Humpback whales feed mainly on small schooling fishes, euphausiids, and other large zooplankton. Fish prey species in the North Pacific include Pacific herring, capelin, juvenile walleye pollock, and sand lance. Should the animals not get enough food during the time spent in Alaska, compensation will not occur in other locations or at other times of the year, as humpback whales fast while on their breeding grounds.

Effects of contaminants, toxins, and disease on humpback whale populations are largely unknown. Killer whales prey upon humpback whales, although such attacks are observed relatively infrequently. Younger animals may be more vulnerable to this type of predation during migration when group size is smaller than in summering or wintering areas.

7.8.1.1 Western North Pacific Population

The Aleutian Islands and Bering Sea are important foraging habitats for this population. Until recently, the Western North Pacific humpback whale population was estimated at about 400 animals (low sampling effort, recognized as an underestimate); currently, results from the SPLASH project estimate a population of approximately 6,000-14,000 for the Bering Sea and Aleutians, and 100-700 for Russia.

Data from the SPLASH research effort suggested the likely existence of wintering areas that have not been previously described for humpbacks that forage in the Aleutians and in the Bering Sea. These animals were not well-represented in any of the winter sampling areas, indicating that they must be using a different and unknown winter location. Although a reasonable assumption for this breeding ground would be a region in the eastern central North Pacific, the location is uncertain given the complexities around migratory pathways.

7.8.1.2 Central North Pacific Population

Most humpback whales in the Central North Pacific population spend the winter months in Hawaii where they breed, give birth to and nurse their calves. Some animals, however, remain on the feeding grounds year-round.

Humpback whales in this population typically show fidelity to either the southeast Alaska or the Prince William Sound feeding areas. The current humpback whale abundance estimate in the combined feeding areas of southeast Alaska and Northern British Columbia is approximately 3,000-5,000 animals. The current best adult survival rate in the Central North Pacific population is estimated at 0.963, and the rate of increase for this population is currently acknowledged as 7% per year.

Although measures of abundance continue to indicate an increasing trend, abundance estimates alone cannot be relied upon as accurate measures of population recovery without a long-term understanding of demographic parameters and variability in the population and the effects of natural and anthropogenic stressors on the status of the population. In addition, the species may be vulnerable to catastrophic or random events that could result in significant declines and increase the species' risk of extinction. However, on the basis of total abundance, current distribution, and regulatory measures that are currently in place, it is unlikely that this species is in danger of extinction in the foreseeable future, although some stocks may be at risk. NMFS initiated a status review under the ESA on August, 12 2009 (74 FR 40568) to determine whether humpback whales should be delisted or reclassified from endangered to threatened. NMFS is also considering whether to recognize Distinct Population Segments (DPSs) of humpback whales according to the 1996 DPS policy (61 FR 4722). DPSs of vertebrate species, as well as subspecies of all listed species, may be proposed for separate reclassification or for removal from the list. Critical habitat has not been designated for humpback whales anywhere throughout their range.

7.8.2 Assess Risk to the Species

We have found humpback whale prey species are not targeted or taken in significant amounts by the fishery actions evaluated in this opinion. Gear entanglements are not uncommon for these whales, and are associated with the fisheries under the proposed actions. However, it is unclear to what extent entanglements reported to the stranding network in Alaska involve groundfish fishing gear. Overall, the number of entanglements that ~~that~~ might result from interactions with groundfish fisheries appears to be low in contrast to other gear types. For such events that do occur with individual whales, the extent of entanglement from groundfish fisheries is not expected to have negative consequences for humpback whales in the North Pacific.

The incidence of ship strikes and/or serious injury from vessels involved in the groundfish fisheries are likely to be negligible, and unlikely to have population level consequences for humpback whales in the North Pacific.

Humpback whales may be disturbed by noise from fishing vessel engines. Typical measures of a whale’s reaction to the presence of a vessel have been visible changes in behavior, and whales may leave the action area if sufficiently disturbed. However, the effect of such displacement on individual humpback whales, if it were to occur, would not compromise the recovery or survival of the species.

Cumulative effects on humpback whales include state-managed fisheries, which clearly have a direct impact on humpback whales. Over 100 entanglement incidents have been reported to the NMFS Alaska stranding program over the last 30 years, many involving pot gear and/or gill net gear from fisheries in inside waters in southeast Alaska, and areas around Kodiak, Homer and Seward. For many of these incidents, when disentanglement is not possible or the animal is not re-sighted, the ultimate fate of the animal remains unknown. State-managed fisheries represent an additional source of anthropogenic impact beyond those posed by the proposed action through entanglements to the Central North Pacific and Western North Pacific populations of humpback whales.

7.8.3 Conclusions

Based on the above analysis, we do not expect that any elements of the action would affect population viability for humpback whales. If we do not expect an action to have adverse consequences on the viability of the subpopulations or foraging groups represented by humpback whales in the action area, we would not expect the Central North Pacific population or the Western North Pacific population of humpback whales to experience reductions in reproduction, numbers, or distribution that might appreciably reduce their likelihood of surviving and recovering in the wild.

After reviewing the current status of the Central Pacific population and Western Pacific population of humpback whales, the environmental baseline for the action area, the effects of the Federal and parallel groundfish fisheries off Alaska, and the cumulative effects, it is NMFS’ Biological Opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Central North Pacific population or the Western North Pacific population of humpback whales. No critical habitat has been designated for this species; therefore, none will be affected.

7.9 Sperm Whales

Sperm whales were listed as endangered under the ESA in 1973. Commercial whalers between 1947 and 1987 harvested approximately 258,000 sperm whales in the North Pacific. In particular, the Bering Sea population of sperm whales was depleted. They were protected from commercial harvest by the IWC in 1981, although the harvest of sperm whales continued in the North Pacific until 1988. In the past two decades the government of Japan has authorized the killing of small numbers of sperm whales associated with a scientific research program in the western North Pacific.

7.9.1 Summary of Responses to the Environmental Baseline, Effects of the Action, and Cumulative Effects

Sperm whales are found throughout the North Pacific and throughout most Alaskan waters, usually in waters greater than 300 m in depth. Mature female and immature sperm whales of both sexes are found in temperate and tropical waters; sexually mature males join these groups throughout the winter. During the summer, mature male sperm whales are thought to move north into the Aleutian Islands, GOA, and the Bering Sea.

Reliable estimates for population abundance, status, and trends for the Alaska population of sperm whales are not available. However, the number of sperm whales in the eastern North Pacific has been estimated to be 39,200 animals. Alternatively, sperm whale density extrapolations have resulted in a practical working estimate of sperm whale abundance for the entire North Pacific ranging from 100,000-200,000 animals.

Sperm whales feed primarily on mesopelagic squid, but also consume octopus, other invertebrates, and fish. Fish eaten in the North Pacific included salmon, lantern fishes, lancetfish, Pacific cod, pollock, saffron cod, rockfishes, sablefish, Atka mackerel, sculpins, lumpsuckers, lamprey, skates, and rattails. Daily food consumption rates for sperm whales ranges from 2 - 4% of their total body weight. Male sperm whales are known to take sablefish off longline gear in the GOA. A discussion of this behavior can be found in the Environmental Baseline section for this species (Section 4.9.4).

The estimated annual rate of human-caused mortality and serious injury appears to be minimal for this population. On the basis of total abundance, current distribution, and regulatory measures that are currently in place, this species may not be in danger of extinction in the foreseeable future, however a recent ESA status review concluded that due to the lack of sufficient and reliable information on the severity of multiple potential threats to the recovery of sperm whale populations, as well as population structure, species abundance and population trends, the status of the sperm whale should remain as "endangered" (NMFS 2009c). Critical habitat has not been designated for sperm whales anywhere throughout their range.

7.9.2 Assess Risk to Species

There is potential for competition between sperm whales foraging for prey species and groundfish fisheries in the GOA. While the extent of this impact is currently not well understood, there is no evidence that the groundfish fisheries in Alaska compromise sperm whale diet.

Sperm whales are known to pick longline fishing gear in Alaskan waters, and this depredation has been increasing throughout the last decade. Most recorded episodes of predation on catch have occurred in the GOA rather than in the BSAI region; thus, this is where most exposure is likely to occur resulting in fisheries interactions and entanglements. Entanglements have occurred, although available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions although it is possible that whales may break through or carry off trailing gear and become debilitated, injured, or die as a result, with no observation of the event. The incidence of sperm whale entanglement in Alaska appears to be low, and would not be expected to rise to a level that would have population level consequence for sperm whales.

While possible, the incidence of ship strikes and/or serious injury from ship strikes involved in the groundfish fisheries to sperm whales are likely to be minimal and not expected to result in an adverse population level effect for sperm whales in Alaska.

Any impact to sperm whales due to disturbance by vessels is uncertain. Given that many individual sperm whales are attracted to the sound of groundfish vessel engines and gear hauling catch, it would appear that they often do not interpret such noise as disturbance. Additionally, as depredation behavior in Alaska is only known to involve male sperm whales, it is unlikely vessel disturbance would present a concern for the species.

7.9.3 Conclusions

Based on the foregoing analysis, we do not expect that any elements of the action would affect population viability for sperm whales. If we do not expect an action to have adverse consequences on the viability of the subpopulations or foraging groups represented by sperm whales in the action area, we would not expect the population of sperm whales to experience reductions in reproduction, numbers, or distribution that might appreciably reduce their likelihood of surviving and recovering in the wild.

After reviewing the current status of the North Pacific sperm whale population, the environmental baseline for the action area, the effects of the Federal and parallel groundfish fisheries off Alaska, and the cumulative effects, it is NMFS' Biological Opinion that the action, as proposed, is not likely to jeopardize the continued existence of the North Pacific sperm whale population. No critical habitat has been designated for this species; therefore, none will be affected.

7.10 Fin Whales

Fin whales were listed as endangered under the ESA in 1973. The total North Pacific population of fin whales prior to commercial whaling was estimated to be 42,000 – 45,000 whales (Ohsumi and Wada 1974; Omura and Ohsumi 1964). The current fin whale population west of the Kenai Peninsula is estimated to be 5,700 animals (Angliss and Allen 2009). An annual rate of population increase of 4.8% (95% CI: 4.1-5.4%) was estimated for the period 1987-2003 (Zerbini *et al.* 2006).

7.10.1 Summary of Responses to the Environmental Baseline, Effects of the Action, and Cumulative Effects

Information on abundance of fin whales in Alaskan waters has improved considerably in the past few years. Although the full range of the NEP stock of fin whales in Alaskan waters has not been surveyed, a rough estimate of the size of the population west of the Kenai Peninsula could include the sums of the estimates from Moore *et al.* (2002) and Zerbini *et al.* (2006). Using this approach, the provisional estimate of the fin whale population west of the Kenai Peninsula would be 5,700 (NMFS 2010). The most recent estimate of the annual fin whale population growth rate in Alaska was 4.8% (95% CI: 4.1-5.4%) for the period 1987-2003 (Zerbini *et al.* 2006).

Data for serious injury and mortality incidental to commercial fishing operations includes observer data, subsistence harvest, and observations of stranded or entangled NEP stock of fin whales reported to the NMFS AKR stranding network through various sources. Only two fin whale strandings have been reported - both in 2003, the cause of these dead strandings was not determined. Stranding data are opportunistic data that are reported to NMFS from various sources, including the general public, authorized members of the NMFS' marine mammal stranding networks, commercial fishermen, NMFS Enforcement, the U.S. Coast Guard, and others.

The average minimum estimated total of all known serious injury and mortalities to NEP stock of fin whales incidental to commercial and recreational fisheries (in Alaska) is 0.23 fin whale/yr based on fisheries observer data (Angliss and Allen 2009).The current PBR for this stock is conservatively calculated at 11.4 animals (Allen and Angliss 2009). Therefore, the total annual average incidental take in commercial fisheries (0.23) in the US EEZ for the 2002-2006 (5yr) timeframe is < 10% of the PBR (11.4).

7.10.2 Assess Risk to Species

NMFS evaluated the best available information in assessing the interactions between ESA-listed NEP fin whales and Alaska fisheries (including observer data), other fisheries (using primarily stranding and sightings data), and other sources of human-caused serious injury and mortality, in order to determine whether the incidental mortality and serious injury from all commercial fisheries will have a negligible impact on the stock. Allen and Angliss (2009) reported an annual rate of increase of 4.8 percent and a PBR of 11.4 for this stock. Mortality of one NEP fin whale was reported in the Bering Sea/Aleutian Islands pollock trawl fishery during the 2002-2006 period, and the mean annual mortality and serious injury rate incidental to commercial fisheries is 0.23 whales per year. NMFS stranding data contain no reports of fisheries-related NEP fin whale strandings or entanglements in the EEZ offshore of Alaska.

Based on the one mortality reported and investigated during 2002-2006, the minimum mean annual mortality/serious injury from ship strikes is 0.20 fin whales per year in Alaska. The estimated minimum annual total human-caused mortality and serious injury rate for the NEP stock of fin whales in the US EEZ for 2002-2006 is 0.43 whales per year. Accordingly, total human-caused mortality and serious injury is below 10 percent of PBR (1.14) for this stock. Because all total human-related serious injuries and mortalities are less than 0.1 PBR, NMFS determined that mortality and serious injury incidental to commercial fisheries will have a negligible impact on the NEP fin whale stock (75 FR 68767).

There is potential for competition between fin whales foraging for prey species and groundfish fisheries in the GOA and BSAI. While the extent of this impact is currently not well understood, there is no evidence that the groundfish fisheries in Alaska compromise fin whale diet.

7.10.3 Conclusions

Based on the foregoing analysis, we do not expect that any elements of the action would affect population viability for fin whales. If we do not expect an action to have adverse consequences on the viability of the subpopulations or foraging groups represented by fin whales in the action area, we would not expect the population of fin whales to experience reductions in reproduction, numbers, or distribution that might appreciably reduce their likelihood of surviving and recovering in the wild.

After reviewing the current status of the Northeast Pacific fin whale population, the environmental baseline for the action area, the effects of the Federal and parallel groundfish fisheries off Alaska, and the cumulative effects, it is NMFS' Biological Opinion that the action, as proposed, is not likely to

jeopardize the continued existence of the Northeast Pacific fin whale population. No critical habitat has been designated for this species; therefore, none will be affected.

8 REASONABLE AND PRUDENT ALTERNATIVE

Regulations (50 CFR §402.02) implementing section 7 of the ESA define reasonable and prudent alternatives as alternative actions, identified during formal consultation, that: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) avoid the likelihood of jeopardizing the continued existence of listed species or the likelihood of resulting in the destruction or adverse modification of designated critical habitat.

Based on the synthesis discussion in Chapter 7 and the supporting evidence in other chapters of this Biological Opinion, a reasonable and prudent alternative (RPA) to the manner in which the groundfish fisheries are managed in the BSAI is required to avoid: (1) the likelihood of jeopardy to the western DPS of Steller sea lion, and (2) the likelihood of adverse modification of its designated critical habitat.

8.1 Previous Conclusions and Reasonable and Prudent Alternatives

The analysis in the 2000 FMP Biological Opinion supported a determination that certain groundfish fisheries authorized by the FMPs were likely to jeopardize the continued existence of endangered Steller sea lions and adversely modify their critical habitat. These determinations resulted from available evidence of competitive interactions between the fisheries for pollock, Atka mackerel and Pacific cod and Steller sea lions. This competitive interaction, occurring at the global, regional and local scales, was shown to likely jeopardize the continued existence of Steller sea lions by interfering with their foraging opportunities for three major prey species resulting in reduced reproduction and survival. The reduction in survival and reproduction enhanced the decline in the numbers of Steller sea lions relative to an unfished action area. Scientific evidence suggested that the same competitive interaction was likely to also adversely modified critical habitat designated for Steller sea lions by reducing prey availability at temporal and spatial scales relevant to foraging Steller sea lions. Because the competitive interaction was the basis for both the determinations of jeopardy and adverse modification of critical habitat, the RPA in the FMP Biological Opinion avoided the likelihood of jeopardy and adverse modification by requiring FMP amendments that protected both the population from the adverse competitive effects of the fisheries and protected the availability of an adequate prey field inside critical habitat. The following is a summary of the general principles used in the 2000 Biological Opinion RPA to minimize competition between fisheries and Steller sea lions.

The RPA from the FMP Biological Opinion (NMFS 2000), focused on 4 key elements:

1. Global Control Rule,
2. Fishery Closures to Eliminate Competition,
3. Spatial Distribution of Fisheries, and
4. Temporal Distribution of Fisheries.

The 2000 Biological Opinion RPA was partially implemented in 2001. Full implementation of the RPA was scheduled for 2002; however, the action was re-considered by the action agency. This reconsideration was based on a request from the Council to immediately convene (January 2001) an RPA committee comprised of members of the fishing community, the conservation community, NMFS, state agencies and the Council's SSC to develop an alternative action to the RPA provided by NMFS in the FMP Biological Opinion that would remove the jeopardy and adverse modification finding.

The Council's RPA Committee developed a revised RPA, and in July of 2001, the action agency (SFD) proposed this alternative RPA to replace the components of the original FMP action that had resulted in the jeopardy and adverse modification finding in the 2000 FMP-level consultation. In 2001, NMFS prepared a project level Biological Opinion (NMFS 2001), which reviewed the revised action and determined that it was not likely to jeopardize or adversely modify critical habitat. State waters, "parallel fisheries," were also included in this Biological Opinion. This RPA was implemented in 2002.

The court reviewed the 2001 Biological Opinion and, although the Court did not find fault with the RPA, the Court found that NMFS was arbitrary and capricious by not fully and adequately explaining the telemetry and zonal approach analyses contained in the Biological Opinion, and remanded the opinion back to NMFS for revision. In response to the court order, NMFS prepared a supplement (NMFS 2003) to the 2001 Biological Opinion, which provided the analyses required in the Court's remand order, and affirmed NMFS' conclusions that the revised FMP actions were not likely to jeopardize ESA-listed species or adversely modify critical habitat.

The 2001 Biological Opinion and its 2003 Supplement provided an extensive analysis of the status of the western DPS, including how the western DPS was expected to respond to the RPA implemented in 2002. That analysis contained expectations for improved growth of the population of Steller sea lions comprising the western DPS based on reduced fishing removals of prey. The 2002 RPA provided an increased availability of the Atka mackerel, Pacific cod, and pollock prey in nearshore foraging areas, in many areas out to 20 nm, and more restricted catch of these three prey species geographically and temporally, all with a "backstop" global control rule. The RPA also included some restrictions on use of trawl gear by requiring reduced levels of trawling within 20 nm of rookeries and important haulouts to slow the pace of prey removals in outer foraging areas. While it appears that the combination of these measures has resulted in an improvement in Steller sea lion growth trends in some sub-regions of the western DPS, other sub-regions are faring poorly. Consensus of scientific research both in the field and in captive-animal settings indicates the importance of accessible prey of sufficient quantities for good Steller sea lion nutrition, growth, physiological functions, and reproduction. No other factor, such as disease, contaminants, predation, or direct mortality, has been identified to cause poor Steller sea lion reproductive performance in these sub-regions. Insufficient prey for nutritional requirements of Steller sea lions is still considered by NMFS to be one of what may include several primary factors responsible for the lack of the expected rate of increase in abundance in the western DPS of Steller sea lion. Since 2000, the overall population has been slowly recovering (i.e., 1.4% per year in US waters), but with an alarming decreasing trend in the western Aleutian Islands and a steadily decreasing trend in the central Aleutian Islands. It is this heterogeneity in population trend and in the underlying vital rates which leads NMFS to conclude that the current fishery, modified by the 2002 protection measures, is likely to jeopardize the continued existence of the western DPS of Steller sea lion and adversely modify its critical habitat. Therefore, an alternative to current management of groundfish fisheries in the action area is required to avoid the likelihood of jeopardy to the western DPS and adverse modification of its designated critical habitat.

8.2 Principles of the Reasonable and Prudent Alternative

This Biological Opinion includes one RPA, which has multiple management measures or elements that are essential to avoid the likelihood of the groundfish fisheries jeopardizing the continued existence of the endangered western DPS of Steller sea lion or adversely modifying its designated critical habitat. Together these measures are designed to ameliorate adverse effects of removing prey biomass in two sub-regions in the range of the western DPS and avoid competition in the short- and long-term.

8.2.1 Objective

The objective of the RPA described below is to conserve the overall forage availability for Steller sea lions and the value of critical habitat by limiting harvest of important prey species in the times and areas where Steller sea lions forage. The measures in this RPA focus on sub-regions where sea lion vital rates and existing fishing practices indicate the likelihood of a compromised prey field under the status quo.

8.2.2 Performance Standards

Changes in fishery management measures resulting from this Biological Opinion should:

- Be commensurate with the rate of Steller sea lion population declines with more stringent measures in those sub-regions, as described in the Recovery Plan (NMFS 2008a), with greater population declines.
- Conserve the value of critical foraging habitat zones and offshore foraging areas in those sub-regions that are used most extensively by foraging Steller sea lions.
- Conserve overall forage biomass for Steller sea lions by limiting fishery removals in areas with low forage biomass availability.
- Disperse fishery removals in time and space to prevent fishery removals from resulting in local depletion of the prey field.
- Consider the distributional effects of time and area closures that are not combined with reductions in total allowable catch such that fishery removals are not concentrated at another time and in another area that may be deleterious to foraging Steller sea lions.
- Conserve prey availability inside trawl exclusion zones in areas where prey (e.g., Atka mackerel) tagging studies indicate high movement of fish from inside to outside of trawl exclusion zones (e.g., Amchitka North in Area 542).
- Consider fishery removals in State waters.
- Maintain 3 nm groundfish fishing closures around currently designated rookeries and establish new 3 nm groundfish fishing closures around emerging rookeries.
- Be implemented in a timely manner given the decline in Steller sea lion abundance in two sub-regions.

8.3 Reasonable and Prudent Alternative

The draft Biological Opinion released for public review and comment in August, 2010 contained an RPA based on the performance standards above. During the comment period, NMFS received many comments on the draft RPA which generally fell into one of three categories: support for the RPA in the draft Biological Opinion; support for the motion adopted by the Council at their August, 2010 special meeting; or suggestions for alternate RPAs to the draft Biological Opinion or Council motion. NMFS reviewed these comments and suggestions, and reanalyzed available data in light of the alternative RPA measures suggested by the public, including the Council’s motion. NMFS analyzed alternate suggestions to the RPA in the August 2, 2010 draft Biological Opinion to determine if the alternate measures conformed to the performance standards and would be likely to remove the likelihood that the fisheries would jeopardize the continued existence of the western DPS of Steller sea lions or adversely modify their designated critical habitat. While the resulting RPA is similar to the RPA in the draft Biological Opinion, several adjustments were made to the RPA in response to public comment where NMFS determined adjustments would not unacceptably compromise the conservation value likely to be achieved by the RPA in the draft Biological Opinion. The resulting RPA is depicted in Figures 8.1 through 8.5. Appendix VII contains the August 2, 2010 draft RPA and the Council’s proposed RPA.

Before prosecuting groundfish fisheries in 2011, NMFS shall amend the regulations implementing the groundfish fisheries in the Bering Sea/Aleutian Islands to include the following RPA. NMFS may implement this amendment by working through the NPFMC, through emergency regulations, or through other action taken by the Secretary of Commerce such that it is in effect January 1, 2011. **Existing fishery management measures as established in previous Biological Opinions must be maintained. The measures required below are in addition to the existing measures contained within regulation (see Table 2.16) as described below.**

Because this Biological Opinion concluded that the groundfish fisheries as authorized are likely to jeopardize the continued existence of the western DPS of Steller sea lions and adversely modify designated critical habitat, the action agency (SFD) is required to notify PRD (NMFS) on the date of the implementation of the RPA contained herein.

8.3.1 Recap of the Indicators and Evidence Underlying RPA Development

In this Biological Opinion, NMFS relied on the following indicators and evidence to develop an RPA that would be sufficient to ensure that the groundfish fisheries are not likely to jeopardize the continued existence of the western DPS of Steller sea lions or adversely modify its designated critical habitat:

Indicators of Concern

- Trends in Steller sea lion populations by Recovery Plan sub-region.
    - Declining population growth rates are due to decreasing survival, decreasing birth rates, or a combination of both.
- Pup/adult female ratios are one indicator of the population birth rate.
    - Reduced birth rates area an indicator of chronic nutritional stress.
    - Chronic nutritional stress results from increased energy expenditures for foraging and insufficient prey availability.

Evidence

- Severe declines in counts of non-pups and pups in the western Aleutian Islands. Recent pup/adult female ratio lowest of all sub-regions.
- Continued declines in abundance of non-pups in the central Aleutian Islands (specifically RCAs 2- 4).
- Continued declines in numbers of pups in parts of the central Aleutian Islands (RCAs 2 and 3).
- No other stressor (or combination of other stressors) identified as primary mechanism for decline of Steller sea lions in the western and central Aleutian Islands; based on dozens of field and captive Steller sea lion studies, prey removals could result in chronic nutritional stress which in turn could be a contributing factor for a lack of a robust rate of population growth in the western DPS.
- Forage biomass of key Steller sea lion prey species have been harvested from the Aleutian Islands historically.
- Harvest rates of key prey species inside Steller sea lion critical habitat have been high in the Aleutian Islands (especially Pacific cod in RCAs 1 and 3).

- Diet information from Steller sea lion scats suggests the importance of Pacific cod, Atka mackerel, and pollock (Table 3.16)
- Foraging distribution as indicated by filtered telemetry data confirm high use of the 0-10 nm zone of important terrestrial sites (rookeries and haulouts). However, Steller sea lion foraging distribution based on updated telemetry information shows movement patterns of tagged juvenile sea lions well outside of 20 nm. RCAs 1-3 have a large proportion of diving locations >4 m depth outside of critical habitat (AFSC 2010b).
- The Boor (2010) analysis of POP dataset shows substantial Steller sea lion foraging offshore in summer, especially south of Attu and Agattu Islands, and an even larger number of encounters offshore in winter throughout the Aleutian Basin. We recognize that this analysis includes sightings data from over the last 40 years. Nonetheless, this analysis suggests the potential importance of areas outside critical habitat for Steller sea lion foraging.
- Fishery measures implemented in 2002 west of 178° W for Atka mackerel in Steller sea lion critical habitat were not as conservative as they were to the east (e.g., a greater percentage of critical habitat was open to groundfish fisheries); Steller sea lion population response is also different east and west of 178° W (RCAs 1-3)(Figures 2.9, 2.10, 2.11 and 3.10).

8.3.2 RPA by Fishery Management Area

In this Biological Opinion, NMFS analyzed Steller sea lion trends in abundance, particularly from 2000-2008, and fishery responses. NMFS accomplished this by looking at trends in abundance by sub-region, and where trends were found to be negative, by then evaluating the action area at a spatial scale equal to the RCAs. This enabled NMFS to identify patterns and signals in responses on a finer scale than Steller sea lion sub-region or NMFS fishery management area. RCA boundaries overlap both Steller sea lion sub-region and fishery management boundaries to a varying degree. In developing the RPA, NMFS relied on the signals by RCA to identify measures needed to conserve Steller sea lions at the DPS level through conservation of Recovery Plan sub-regions, and to understand how fishing patterns might correspond with observed responses in Steller sea lions. Management measures are specified by NMFS fishery management areas to facilitate practical implementation. The relationship between Steller sea lion sub-regions, RCAs, and NMFS fishery management areas is shown in Figure 3.8. A depiction of the RPA is shown in Figure 8.1.

8.3.2.1 Area 543 (RCA 1)

The rationale for mitigation measures in NMFS Fishery Management Area 543 is as follows:

- This is the poorest performing Steller sea lion sub-region; non-pups have declined 45% and pups have declined 43% from 2000-2008 (Table 3.1b and Table 3.4).
- An index of production of animals, i.e., the ratio of pups to adult-females on rookeries, is the lowest of the entire western DPS (Table 3.6).
- The largest fisheries in this area are Atka mackerel and Pacific cod, two species that are important in the Steller sea lion diet. Atka mackerel are most prominent in the diet of Steller sea lions with a frequency of occurrence of 96% and 55%, in summer and winter, respectively. Pacific cod are the second-most prominent with a frequency of occurrence in the diets of Steller sea lions of 6% and 26%, in summer and winter, respectively (Table 3.16).

- Pacific cod is primarily a winter fishery in Area 543 with 66% of the catch harvested in February and March and 25% harvested in October and November (NMFS Catch Accounting Database 2008-2009 average).
- Atka mackerel fisheries are largely fall and winter fisheries in the Aleutian Islands. In Area 543, 51% of the Atka mackerel was harvested in September. The next largest months were October (13%) and February (16%) (NMFS Catch Accounting Database 2008-2009 average).
- The fishery harvested 7% of the available estimated Atka mackerel biomass in 2008; 65% of the harvest occurred in critical habitat. The amount of catch in critical habitat decreased from an average of 87% (1991-1999) to an average of 41% (2000-2008) (Table IV-1).
- The fishery harvested 43% of available estimated biomass of Pacific cod in 2008; 95% of the harvest occurred in critical habitat. The amount of catch in critical habitat decreased slightly from an average of 92% (1991-1999) to an average of 89% (2000-2008) (Table IV-1).
- Sea lions forage both inside and outside of critical habitat in this sub-region (based on recent analyses of telemetry and POP data (Boor 2010); animals appear not to be limited to foraging in nearshore zones (Section 3.1.6).
- There are 10 trend sites and less than 900 animals in the western Aleutian Islands. Down from 1,633 animals surveyed in 2000, this sub-region represents a small remnant sub-population of the western DPS (Table 3.1b).

The severity of the decline in Steller sea lions and the large fishery removals of important Steller sea lion prey across the broad areas where they forage in this sub-region require the following measures to ensure that the groundfish fishery is not likely to jeopardize the survival or recovery of the western DPS of Steller sea lion or is not likely to adversely modify designated critical habitat for the western DPS of Steller sea lion.

RPA for Area 543

Pacific cod –

1. Prohibit retention of Pacific cod by all federally permitted vessels.

Atka mackerel –

1. Prohibit retention of Atka mackerel by all federally permitted vessels.
2. Establish a TAC for Atka mackerel sufficient to support the incidental discarded catch that may occur in other targeted groundfish fisheries (e.g., Pacific ocean perch).

Intended Effect of RPA in Area 543

Complete closures to directed fishing and a prohibition on retention of Pacific cod and Atka mackerel in this sub-region are intended to increase the prey biomass available to Steller sea lions. As reported in Ianelli *et al.* (2010b- see summary table below), the single species models predict a gradual annual increase in Atka mackerel and Pacific cod biomass such that, by the year 2020, this management area will experience a 42% increase in Atka mackerel biomass, a 119% increase in Pacific cod biomass, and a 17% increase in overall groundfish biomass. A 10% increase in biomass is predicted to be achieved after 3 years of the closure for Atka mackerel, after 1 year of the closure for Pacific cod, and after 5 years of the closure for the overall groundfish biomass. In terms of the expected change in the forage ratio, based on the projections in Ianelli *et al.* (2010b), the forage ratio for the combined biomass of pollock, cod, and

Atka mackerel calculated for all of 543 (and not just for critical habitat) relative to the prey biomass requirements of the group of Steller sea lions currently inhabiting this management area would increase from a ratio of 26 to 33. Model results indicate that allowing fishing to occur even at substantially reduced levels would not likely result in a significant increase in biomass of Atka mackerel, which NMFS believes will be required to contribute to both the continued survival and recovery of Steller sea lions in this sub-region (see data tables in Ianelli *et al.* 2010a). It is recognized that the projections reported in Ianelli *et al.* (2010b) are based on single species models, which do not take into account multispecies interactions (e.g., predation).

For a multispecies perspective (as discussed in Section 4.5.4.6), Ayden (2010) reported qualitative inferences made from a combination of whole-Aleutians food web model results and data specific to the western and central Aleutians. Ayden (2010) explored the effect of simultaneously reducing fishing on Pacific cod and Atka mackerel in Area 543 and concluded that it is possible for Atka mackerel mortality to increase relative to a fished scenario due to reduced Pacific cod fishing mortality since Pacific cod prey on Atka mackerel.

The results of the food web model of reduced Pacific cod mortality in the Aleutian Islands from Aydin (2010) are shown in Figure 8.6. In particular, effects on Atka mackerel and Steller sea lions are extremely limited and uncertain, with no clear change in direction relative to the baseline and with the range of responses crossing the baseline condition. Atka mackerel biomass decreased in half and increased in half of the 500 simulations of reduced Pacific cod fishing mortality in the Aleutian Islands (Aydin 2010) with decreases attributed to direct Pacific cod predation on Atka mackerel and increases possibly due to increased cod predation on walleye pollock (a competitor of Atka mackerel). While the food web model used in Aydin (2010) does not adequately simulate all details of spatial interactions in the Aleutian Islands, it can give guidance to area closures in conjunction with area-specific data. Overall, the model predicts that declines in Atka mackerel fishing throughout the Aleutian Islands would lead to increases in prey supply for Steller sea lions and that a simulated closure of Area 543 to Atka mackerel fishing would show similar results. However, the model predictions are mixed on the effects of reducing Pacific cod fishing in Area 543, with limited apparent effects on the total Steller sea lion food supply (Aydin 2010).

Thus, NMFS examined the results of both single-species and multispecies models of reduced Atka mackerel and Pacific cod fishing mortality in the Aleutian Islands overall and in Area 543. As noted in Van Kirk *et al.* (2010), the decision of which approach to use depends on the objectives. Trade-offs must be made between the advantage of greater biological realism via multispecies information and the disadvantage of increased uncertainty due to additional model complexity (Van Kirk *et al.* 2010). Here we note that uncertainties inherent with the assumptions of single-species approaches become magnified in multispecies models. Therefore, NMFS believes that given the information available, it is premature to add more assumptions to the models predicting predator-prey responses and has relied on the results of the single species models to a greater extent than the multispecies models in predicting the effects of the RPA.

Given the severity of the decline in Steller sea lions in this sub-region, NMFS did not identify alternate time or area closures that would be adequate to remove the likelihood of the fisheries causing jeopardy or adverse modification. Prohibiting retention of Pacific cod and Atka mackerel is necessary to eliminate any incentive to catch these species in groundfish fisheries of non-prey species that will continue to occur in this area (e.g., Pacific Ocean perch). The best available science does not allow NMFS to quantify incremental increases in Steller sea lion population responses with the expected increases in prey biomass from foregone fishery harvests. However, given the evidence available for the potential for Atka mackerel and Pacific cod fisheries to compete with Steller sea lions in a manner that limits their reproduction or survival as evidenced in population responses observed to date in this sub-region, NMFS must eliminate this potential competition to meet the ESA's affirmative conservation responsibility for federally authorized actions.

The existing fishery management measures required implementation and maintenance of the HLA platoon management system for Atka mackerel in the Aleutian Islands. Closures to Atka mackerel fishing inside and outside of critical habitat in Area 543 eliminate the need for HLA platoon management of the Atka mackerel fleet.

Summary of results from projection model regarding closure of Area 543 to fishing for Pacific cod and Atka mackerel (from Ianelli *et al.* 2010b).

Groundfish Species	Year 1 (biomass – kt)	Year 11 (biomass – kt)	(% change)
Atka Mackerel	244.1	347.3	+42%
Pacific cod	17.7	38.8	+119%
All Groundfish species	721.7	846.0	+17%

8.3.2.2 Area 542 (RCA s 2&3)

The rationale for mitigation measures in NMFS Fishery Management Area 542 is as follows:

- Continued declines in non-pup and pup abundance in this sub-region from 2000-2008 (i.e., non-pup and pup counts declined by 2% and 4% per year between 2000 and 2008, respectively) (Table 3.9 and Table 5.8).
- The largest fishery in this fishery management area is the Atka mackerel fishery followed by the Pacific cod fishery. In Area 542, 22,328 mt of Atka mackerel and 5,434 mt of Pacific cod were harvested in 2008 (Table 5.2). Both of these species are prominent in the diet of Steller sea lions in the Aleutian Islands. Atka mackerel are particularly prominent with a frequency of occurrence in the diet of Steller sea lions of 96% and 55%, in summer and winter, respectively (Table 3.16).
- The fishery harvested 8% of the estimated Atka mackerel forage biomass in Area 542 in 2008; 58% of the harvest occurred in critical habitat.[28]
- The Atka mackerel harvest in Area 542 was largely harvested in September, October, and February (39%, 19%, and 19%, respectively) (NMFS Catch Accounting Database 2008 and 2009 average).
- The Atka mackerel ABC is specific to fishery management areas in the Aleutian Islands.
- Pacific cod are also prominent in the diet of Steller sea lions with a frequency of occurrence in summer and winter of 6% and 26%, respectively (Table 3.16).
- Pacific cod harvest in Area 542 occurs primarily within Steller sea lion critical habitat (Appendix IV, Table 5.8).
- The Pacific cod harvest was distributed throughout the year in Area 542 with two small peaks in March (23%) and September (16%) (NMFS Catch Accounting Database 2008 and 2009 average).
- The fishery harvested 27% of the estimated forage biomass of Pacific cod in 542 in 2008 (Table 5.3); 91% of the Pacific cod was harvested in critical habitat (Appendix 4, Table 5.8).

---

[28] Calculated by summing values for RCAs 2 and 3 in Tables IV-2 and IV-3.

- Pacific cod TAC is specified for the entire BSAI and not apportioned by fishery management area; thus, there is concern that closing Area 543 to Pacific cod fishing without instituting protection measures in Area 542 would result in a greater proportion of Pacific cod harvested in Area 542 relative to the historic baseline and would exacerbate effects on Steller sea lions.
- Overall harvest of Pacific cod is not particularly of concern in Area 542; rather, the amount harvested in critical habitat, especially within the 3 to 10 nm zone of critical habitat in winter, is of concern.
- Pacific cod is a widely migratory species.
- Sea lions forage both inside and outside of critical habitat in this sub-region (based on recent analyses of telemetry and POP data); animals are not necessarily limited to foraging in nearshore zones (Section 3.1.6).
- FIT studies have shown trawl exclusion zones are not effective in all areas. Around the Amchitka trawl exclusion zone there is high movement of Atka mackerel from inside the exclusion zone to outside. Thus, fishing the outside edge of the protection zone could affect availability of Atka mackerel inside (based on McDermott *et al.* 2010) (Section 4.5.2.5).
- Total production of Atka mackerel may be insufficient to meet foraging needs for Steller sea lions at the Amchitka trawl exclusion zones, especially Amchitka North (Ortiz and Logerwell 2010)(Section 4.5.2.5).
- Given that the decline in Steller sea lions has not been as severe in this sub-region (central Aleutian Islands) relative to the western Aleutian Islands, less severe fishery management measures may be warranted.
- Recent analysis of site counts show that Kanaga Island/Ship Rock (previously identified as a haulout) meets the criteria for a rookery. Currently, groundfish fisheries may be conducted in waters 0-3 nm from this site (Figures 2.9-2.11).

The observed decline in Steller sea lions in this sub-region and the large fishery removals of important Steller sea lion prey across the areas where they forage in this sub-region, require the following measures to ensure that the groundfish fishery does not continue to appreciably reduce the likelihood of Steller sea lion survival or recovery in the wild or adversely modify designated critical habitat:

RPA for Area 542

Groundfish –

1. Close waters from 0-3 nm around Kanaga Island/Ship Rock to directed fishing for groundfish by federally permitted vessels.

Pacific cod –

1. Close 0–6 nm zone of critical habitat to directed fishing for Pacific cod by federally permitted vessels using nontrawl gear year round. For vessels 60 ft or greater, close critical habitat from 6 nm–20 nm to directed fishing for Pacific cod using nontrawl gear by federally permitted vessels from January 1 to March 1, 12:00 noon.
2. Between 177 E to 178 W longitude, close critical habitat from 0-20 nm to directed fishing for Pacific cod by federally permitted vessels using trawl gear year round.
3. Between 178 W to 177 W longitude, close critical habitat from 0-10 nm to directed fishing by federally permitted vessels using trawl gear year round. Between 178 W to177 W longitude, close critical habitat 10-20 nm to directed fishing for Pacific cod using trawl gear by federally permitted vessels from June 10 12:00 noon to November 1, 12:00 noon.

4. Prohibit directed fishing for Pacific cod by all federally permitted vessels from November 1, 12:00 noon to January 1. (This extends the trawl gear restriction to nontrawl gear).
5. Reinitiate ESA consultation if the nontrawl harvest of Pacific cod exceeds 1.5 percent of the BSAI Pacific cod ABC (equivalent to the Area 542 maximum annual nontrawl harvest amount from 2007-2009, source: NMFS Catch Accounting Database). Similarly, reinitiate ESA consultation if the trawl harvest of Pacific cod exceeds 2 percent of the BSAI Pacific cod ABC (equivalent to the Area 542 maximum annual harvest amount from 2007-2009, source: NMFS Catch Accounting Database).

Atka mackerel –

1. Set TAC to no more than 47 percent of the Area 542 ABC.
2. Between 177 E to 179 W longitude and 178 W to 177 W longitude, close critical habitat from 0–20 nm to directed fishing for Atka mackerel by federally permitted vessels year round.
3. Between 179 W to 178 W longitude, close critical habitat from 0-10 nm to directed fishing for Atka mackerel by federally permitted vessels year round. Between 179 W and 178 W longitude, close critical habitat from 10-20 nm to directed fishing for Atka mackerel by federally permitted vessels not participating in a harvest cooperative or fishing a CDQ allocation.
4. Add a 50:50 seasonal apportionment to the CDQ allocation to mirror seasonal apportionments for Atka mackerel harvest cooperatives.
5. Limit the amount of Atka mackerel harvest allowed inside critical habitat to no more than 10 percent of the annual allocation for each harvest cooperative or CDQ group. Evenly divide the annual critical habitat harvest limit between the A and B seasons.
6. Change the Atka mackerel seasons to January 20, 12:00 noon to June 10, 12:00 noon for the A season and June 10, 12:00 noon to November 1, 12:00 noon for the B season.

Intended Effect of RPA in Area 542

Groundfish

Kanaga Island/Ship Rock was previously identified as a haulout and treated as such in the 2003 Steller sea lion protection measures. Recent analysis of site counts shows that this site meets the criteria for a rookery. This RPA is intended to ensure that all rookeries in the Aleutian Islands are provided consistent protection from the effects of groundfish fisheries by prohibiting fishing by federally permitted vessels within 3 nm from Steller sea lion rookeries. The ability to prohibit transit within 0-3 nm from Kanaga by vessels without a Federal permit is outside the scope of the authority of the action agency as it would require a change to federal regulations at 50 CFR 223.202. This RPA would very likely eliminate interaction between Steller sea lions and groundfish fishing vessels, except for vessels transiting the area. Kanaga Island is one of the few locations in Area 542 where Steller sea lion reproduction is occurring, making this site important to the population in Area 542.

Pacific cod

Because the Pacific cod TAC is established for the entire BSAI and because Pacific cod are migratory, there is concern that harvest historically taken in Area 543 may be concentrated within critical habitat in Area 542 if no additional measures were instituted in this area. Non-discretionary measures for Area 542 are designed to conserve the value of critical habitat by reducing the amount of catch historically taken from within the 3-10 nm zone of critical habitat and prevent an intensification of fishing effort in the 10-20 nm zone as a result of displacing harvest from Area 543 and the 3-10 nm zone of Area 542.

Under the status-quo regime, directed fishing for Pacific cod with nontrawl gear in Area 542 is permitted year-round in critical habitat except for the 0-3 nm zone around rookeries; directed fishing for Pacific cod with trawl gear is permitted within 10-20 nm of rookeries part of the year and within 3-20 nm from haulouts part of the year (Table 2.12). Time and area closures for directed fishing for Pacific cod by nontrawl and trawl gear categories instituted with this RPA are intended to conserve the value of critical habitat for Steller sea lions in Area 542.

The RPA in the August 2, 2010 draft Biological Opinion (draft Biological Opinion RPA) would have closed critical habitat within the 0-10 nm zone to directed fishing for Pacific cod for all gear types. The 10-20 nm zone of critical habitat would have been closed to fishing for Pacific cod with trawl gear and nontrawl gear in alternate times of the year. This aspect of the RPA was modified based on public comment received on the draft Biological Opinion. In their August, 2010 motion (Appendix VII) for Area 542, the Council proposed that no additional restrictions beyond the status-quo regime be placed on vessels under 60' in length using nontrawl gear to fish for Pacific cod. Nontrawl vessels over 60' in length would be limited to fishing in the B season (June 10-November 1) and would be prohibited from fishing from 0-4 nm in critical habitat. NMFS reviewed the available information and determined the Council's motion was not likely to provide adequate protection of the prey field inside 10 nm but that some of the exceptions contained in their motion may result in *de minimis* changes relative to the draft Biological Opinion RPA. NMFS' analysis revealed that vessels under 60' in length account for a small proportion of the Pacific cod harvest in Area 542 (~ 2% of historic Pacific cod harvest in Area 542). Moreover, NMFS reviewed ADF&G and NMFS (2001) to understand the importance of the 0-4 nm zone of critical habitat for foraging Steller sea lions. Steller sea lion at-sea locations from satellite-tagged animals summarized by 2 nm bins show high use of adult female Steller sea lions from 0-6 nm, especially in summer, and higher use by juveniles relative to other areas within critical habitat in both summer and winter (ADF&G and NMFS 2001). The data in ADF&G and NMFS (2001) reveal a break in use by adult female Steller sea lions in summer between the 6 and 8 nm bins. In winter, adult females forage to a greater extent in the 8 to 16 nm bins.

Based on review of the information above, NMFS modified the RPA to close the 0-6 nm zone of critical habitat year-round to nontrawl vessels fishing for Pacific cod. For nontrawl vessels over 60' in length fishing for Pacific cod, the 6-20 nm zone of critical habitat will be closed from January 1 through March 1 (12:00 noon). As noted below, directed fishing for Pacific cod will be prohibited from November 1 through December 31 by all gear types. In combination, these measures for nontrawl gear are intended to protect the important forage areas of Steller sea lions. Closures in nearshore areas in winter months are intended to preserve prey during critical times for adult females, which is the focal life-history phase of these conservation measures.

In 2008, over 99% of the nontrawl Pacific cod was harvested in critical habitat in Area 542. Of the 3,739 mt of Pacific cod caught by the nontrawl sector, 52% was caught from 0-6 nm of critical habitat, 39% was caught from 6-10 nm and 9% was taken from 10-20 nm. On average from 2004-2009, 34% (559 mt) of the nontrawl Pacific cod harvest had been taken from 0-6 nm, this harvest would be displaced under this RPA.

A modification was made to the RPA for trawl gear vessels directed fishing for Pacific cod in Area 542 in consideration of comments received on the August 2, 2010 draft. The draft Biological Opinion RPA would have closed 0-20 nm to trawling for Pacific cod in Area 542. For trawl gear in Area 542, the Council's motion would have closed critical habitat west of 178 W longitude to trawling. Under the Council's motion, trawling for Pacific cod would be permitted in critical habitat east of 178 W in the A season (January 20 through June 10) from 3 -20 nm of haulouts and 10-20 nm of rookeries. East of 178 W would be closed to trawling for Pacific cod in critical habitat under the Council's motion. In 2008, nontrawl vessels harvested more Pacific cod in Area 542 than vessels using trawl gear (Table IV-1999-2008 Areas 2 and 3); 36% of the trawl harvest was taken from 3-10 nm and 42% was taken from 10-20 nm of critical habitat, respectively. NMFS sought to identify an area of critical habitat outside of 10 nm for trawl harvest for cod to occur in a manner not likely to appreciably reduce the conservation value of Steller sea lion critical habitat. NMFS modified the RPA in the draft Biological Opinion to close critical habitat from 10 -20 nm for vessels using trawl gear to directed fish for P. cod east of 178 W in Area 542 from January 20 through June 10 (12:00 noon).

Closing the 0-6 nm zone year-round to nontrawl gear, the 0-20 nm zone from November 1 through March 1 for nontrawl vessels over 60', the 0-20 nm zone year-round to trawl gear west of 178 W longitude, and the 10-20 nm zone from June 10 through January 20 from 178 W to 177 W is intended to reduce the amount of catch that has historically been taken within Steller sea lion critical habitat. Seasonal closures will protect the prey field by insuring that fisheries do not expand into seasons they have not typically fished to offset harvest foregone by area closures.

Fishing intensity is not expected to increase to a large extent in the 6-20 nm zone of critical habitat when it is open to nontrawl gear from March 1 through November 1 as gear limitations are expected to be self-limiting in the small area of remaining fishable habitat available to this gear type in Area 542. Moreover, Pacific cod are perhaps less important a prey species for Steller sea lions from June 10- September 30, which should further buffer Steller sea lions from effects of the nontrawl harvest of Pacific cod in 6-20 nm of critical habitat.

The trawl fishery in Area 542 occurs in the A season, which coincides with the time of year in which Steller sea lion energetic needs are high. The 10-20 nm zone of critical habitat would be closed to trawl gear in the B season to prevent the trawl fishery from expanding into a season they have not traditionally fished in Area 542. Therefore, a year-round closure of 0-20 nm to trawl gear in most of Area 542 (177 E to 178 W) is intended to conserve the value of critical habitat and prevent an intensification of harvest, especially in the 10-20 nm zone of critical habitat.

This RPA extends the November 1 – December 31 closure for directed fishing of Pacific cod with trawl gear to nontrawl gear. This measure is intended to protect prey availability in the winter when Steller sea lion energetic needs are high and when Pacific cod comprise a larger proportion of the their diet relative to summer.

Because current assessment and management practices do not apportion the BSAI Pacific cod TAC to fishery management areas, this RPA specifies harvest levels anticipated by NMFS. Should Pacific cod harvest levels in 542 exceed these levels NMFS shall reinitiate ESA section 7 consultation to determine if the groundfish fisheries may be affecting Steller sea lions or their critical habitat in a manner or to an extent not analyzed in this Biological Opinion. For Area 542, NMFS shall reinitiate ESA section 7 consultation if the harvest of Pacific cod by vessels using nontrawl and trawl gear exceeds 1.5% and 2%, respectively, of the BSAI Pacific cod ABC in any given year. These percentages correspond with the maximum harvest amount by the respective sectors from 2007 through 2009 as expressed a percentage of the ABC, as reduced by the State of Alaska GHL harvest (3% of ABC), and rounded to the nearest quarter of a percent.

Based on the projection results reported in Ianelli *et al.* (2010b), fishing at half of the maximum permissible rate for Pacific cod may result in an increase of 59% of biomass between 2009 and 2020 (i.e., change from 40.1 kt to 63.8 kt).

Atka mackerel

Atka mackerel is harvested with trawl gear. In Area 542 under the status-quo regime, Atka mackerel harvest is permitted within the 10-20 nm zone of critical habitat around rookeries and within the 3-10 nm zone around haulouts. In 2008, 58% of the Atka mackerel harvest in Area 542 occurred inside critical habitat, most of it from the 10-20 nm zone (Tables IV- 2 and IV-3). The RPA for Atka mackerel in Area 542 is designed to preserve the prey field inside the 0-20 nm zone of critical habitat where sea lions are most dependent in terms of foraging efficiency and to limit harvest outside of critical habitat to prevent increases in harvest in areas beyond 20 nm for catches that would be displaced from inside critical habitat. Moreover, reductions in harvest outside of critical habitat are warranted to protect availability of Atka mackerel inside of critical habitat.

The limit on harvest instituted by the RPA is based on the average amount of harvest that has occurred outside of critical habitat from 2003-2009 and applied to the current year ABC. The average annual Atka mackerel harvest outside of critical habitat from 2003-2009 was 47% of the total catch in Area 542 (the lowest and the highest years were eliminated in the calculation). Setting TAC at 47% of ABC is intended to preserve historical access to Atka mackerel resources outside of critical habitat while preventing intensification of harvests that would occur if the harvest displaced from the 10-20 nm zone of critical habitat west of 178 W longitude was allowed to be taken in the remaining open area of 542. This limitation on Atka mackerel harvest would be less stringent than Area 543 based on the determination by NMFS that measures should be commensurate with state of Steller sea lions in the particular area.

The RPA in the draft Biological Opinion would have closed 0-20 nm of critical habitat to Atka mackerel trawling in Area 542. The Council's motion for Atka mackerel in Area 542 would have permitted 50% of the TAC to be taken inside critical habitat from 177 E to 178 E longitude and from 180 to 178 W longitude (Appendix VII). NMFS reviewed the available harvest information and modified the RPA to allow trawling for Atka mackerel within the 10-20 nm zone of critical habitat within a one degree longitude zone in the eastern portion of Area 542. Steller sea lion performance is better in RCA 3 (the eastern side of Area 542) than in RCA 2. Therefore, NMFS determined that providing some fishing opportunities within the 10-20 nm zone of critical habitat to buffer against localized depletion of Atka mackerel on Petral Bank (the primary remaining productive Atka mackerel habitat outside of critical habitat). NMFS structured the revised RPA so that the easternmost portion of 10-20 nm of critical habitat in Area 542, not closed to Atka mackerel trawling under status quo (e.g., critical habitat east of 178 W) may provide some opportunity for the Atka mackerel fleet to disperse harvest without exceeding historic harvest levels in the area of critical habitat remaining open to fishing. The revised RPA closes 0-20nm of critical habitat from 179 W to 177 E longitude and maintains the existing 0-20 nm closure from 178 W to 177 W longitude. Under the RPA, only operations participating in a harvest cooperative or fishing CDQ will be permitted to fish within 10-20 nm of critical habitat from 179 W through 178 W and harvest inside critical habitat will be limited to 10% of the operation's annual Atka mackerel allocation. These measures are intended to prevent harvest from exceeding historic harvest levels within this one degree zone of 10-20 nm of critical habitat. Limiting access to 10-20 nm of critical habitat to only operations with an allocation, i.e., operations fishing in harvest cooperatives or operations fishing CDQ, is intended to prevent a race for Atka mackerel in the open area of critical habitat and will guard against exceeding the critical habitat harvest limit. Two additional measures intended to temporally disperse Atka mackerel harvest in Area 542: include adding a 50:50 seasonal apportionment to the CDQ allocation to mirror seasonal apportionments for Atka mackerel harvest cooperatives; and evenly dividing the annual critical habitat harvest limit between the A and B seasons.

Because critical habitat would be closed to open-access directed fishing for Atka mackerel, the HLA platoon management of the Atka mackerel fleet would no longer be needed. The elimination of the platoon system, in conjunction with the Implementation of Amendment 80, provides an additional opportunity to temporally disperse Atka mackerel fishing as NMFS would have the ability to align Atka mackerel seasons with Pacific cod and pollock A and B seasons in Area 542. In effect, the time periods of the seasons would be expanded from January 20 - April 15 and September 1 - November 1 to January 20-June 10 and June 10-November 1. This may further reduce the potential for temporal depletion of prey resources.

Based on the projection results reported in Ianelli *et al.* (2010b), fishing at half of the maximum permissible rate for Atka mackerel (i.e., 50% of FABC) is projected to result in an increase of 7% of biomass between 2009 and 2020 (i.e., change from 290 kt to 311.4 kt). It is recognized that the projections reported in Ianelli *et al.* (2010b) are based on single species models, which do not take into account multispecies interactions (e.g., predation). Given the uncertainties inherent with the assumptions on this single species approach, these become magnified in multispecies models. Therefore the Agency believes that given the information available, it would be premature to add more assumptions predicting predator-prey responses.

A summary of expected biomass changes is presented in the table below. The expected change in forage ratios (using biomass estimates for pollock, cod, and Atka mackerel) in Area 542 would be from a current value of 5 in 2009 to a ratio of 6 in 2020.

Summary of results from projection model regarding closure of Area 542 to fishing for Pacific cod and Atka mackerel (from Ianelli *et al.* 2010b).

Groundfish Species	Year 1 (biomass – kt)	Year 11 (biomass – kt)	(% change)
Atka Mackerel	290	311.4	+7%
Pacific cod	22.4	27.4	+22%
All Groundfish species	702.3	728.7	+4%

8.3.2.3 Area 541 (RCA s 4&5)

The rationale for mitigation measures in NMFS Fishery Management Area 541 is as follows:

- Continued declines in non-pup abundance in the western portion of this Steller sea lion sub-region and fishery management area (RCA 4) from 2000-2008 (Table 3.1b and 3.9).
- Atka mackerel and Pacific cod were harvested in approximately equal amounts in 2008, 18,180 mt and 18,719 mt, respectively, in fishery management Area 541(Table 5.2).
- Pacific cod fisheries are the largest fisheries in the western portion of Area 541 (RCA 4), and Atka mackerel fisheries dominate in the eastern portion (RCA 5) (Tables IV-4 and IV-5).
- Atka mackerel are prominent in the diet of Steller sea lions in summer and winter with a frequency of occurrence of 96% and 55%, respectively (Table 3.16).
- Atka mackerel harvest occurs primarily from January through February and September through October in Area 541 (NMFS Catch Accounting System).
- The fishery harvested 6% of the estimated Atka mackerel forage biomass in Area 541 in 2008; 1% of the harvest occurred in critical habitat.[29]

---

[29] Calculated by summing values for RCAs 4 and 5 in Tables IV-4 and IV-5.

- Pacific cod are also important in the diet of Steller sea lions in summer and winter with a frequency of occurrence of 6% and 26%, respectively (Table 3.16).
- Pacific cod harvest in Area 541 primarily occurs in February and March (NMFS Catch Accounting Database).
- The Pacific cod fishery harvested 46% of the estimated forage biomass of Pacific cod in 541 in 2008; 30% of the Pacific cod was harvested in critical habitat (Table 5.3 and Table IV-4 and Table IV-5). The extent to which the available forage biomass is underestimated due to the biomass survey occurring in the summer when Pacific cod are disaggregated is unknown, although the biomass may be higher when the fishery is active in Area 541 in winter.
- Pacific cod TAC is specified for the entire BSAI and not apportioned by fishery management area; thus, there is concern that closing 543 to Pacific cod fishing and not 541 would result in a greater proportion of Pacific cod being harvested in 541 and would exacerbate effects to Steller sea lions.
- Steller sea lions forage both inside and outside of critical habitat in this sub-region (based on recent analyses of telemetry and POP data); animals are not necessarily limited to foraging in nearshore zones.
- Fishing for Atka mackerel in critical habitat in fishery management Area 541 is prohibited under the status quo.
- With the structure of the fishery under Amendment 80, the Atka mackerel fishery could be further dispersed by extending the duration of the A and B seasons to January 20 to June 10 and June 10 to November 1, respectively.
- Given that the decline in Steller sea lion abundance in Area 541 has not been as severe relative to Areas 543 and 542, less severe fishery management measures in Area 541 may be warranted.

Overall, there has been a decline in the Steller sea lion population in the central Aleutians sub-region and declines in abundance of non-pups continue to occur in the western portion of Area 541. In aggregate, an improvement trends in abundance in the central Aleutian Islands sub-region is needed to meet objectives for population persistence across the range of the western DPS identified by NMFS (2008). The observed decline in Steller sea lions in this sub-region and fishery removals of important Steller sea lion prey in areas where Steller sea lions forage in this sub-region require the following measures to ensure that the groundfish fishery does not appreciably reduce the likelihood of Steller sea lion survival or recovery or adversely modify designated critical habitat:

RPA for Area 541

Pacific cod –

1. Close the 0-10 nm zone of critical habitat to directed fishing for Pacific cod by federally permitted vessels year-round.
2. Close the 10-20 nm zone of critical habitat to directed fishing for Pacific cod using nontrawl gear by federally permitted vessels January 1, noon through noon on March 1, noon.
3. Close the 10-20 nm zone of critical habitat to directed fishing by for Pacific cod using trawl gear by federally permitted vessels from June 10, noon through noon on November 1.
4. Prohibit Pacific cod fishing November 1 through December 31 in Area 541.

5. Reinitiate ESA section 7 consultation if the nontrawl harvest of Pacific cod exceeds 1.5% of the BSAI Pacific cod ABC (equivalent to the Area 541 maximum annual nontrawl harvest amount from 2007-2009). Similarly, reinitiate ESA section 7 consultation if the trawl harvest of Pacific cod exceeds 11.25% of the BSAI Pacific cod ABC (equivalent to the Area 541 maximum annual harvest amount from 2007-2009).

Atka mackerel –

1. The available data do not indicate a need to further modify fishery management measures to conserve Atka mackerel forage availability within this fishery management area. However, the elimination of the platoon management system provides an opportunity to further disperse the Atka mackerel seasons to January 20 through June 10 for the A season and June 10 through November 1 for the B season.
2. Close the EBS subarea to directed fishing for Atka mackerel year-round to provide for status quo opportunities for Atka mackerel harvest inside critical habitat in the EBS subarea (see explanation below).

Intended Effect of RPA in 541

Pacific cod

Under the status-quo regime in Area 541, directed fishing for Pacific cod is closed from 0-10 nm and 0-3 nm around rookeries for trawl gear and nontrawl gear, respectively. The 0-3 nm zone around haulouts is closed to trawl vessels and open to nontrawl gear. In 2008, 84% of the Pacific cod harvested by trawl gear was caught inside of critical habitat (13,768 mt); the majority of the trawl catch in critical habitat was taken within the 10-20 nm zone. Nontrawl gear fisheries harvested 73% of their Pacific cod inside critical habitat (1,506 mt) with approximately equal proportions in the 3-10 and 10-20 nm zones. This RPA would prohibit fishing for Pacific cod with all gear types within the 0-10 nm zone around all rookeries and haulouts to preserve the prey field for Steller sea lions foraging in this area.

Available telemetry data indicate that sea lions utilize the nearshore zone of critical habitat in Area 541 more than sea lions utilize the nearshore zone of critical habitat in Areas 542 and 543. However recent analyses of the POP database show extensive sightings of Steller sea lions outside of 20 nm in the Aleutian Basin in winter (Boor 2010). Therefore, one of the primary objectives of this RPA is to protect prey resources inside the nearshore zone of critical habitat. Gear-specific seasonal closures in critical habitat are designed to prevent fisheries from expanding into seasons they have not fished historically to prevent intensification of fishing effort to harvest displaced TAC from closures in Areas 543 and 542 since the BSAI Pacific cod TAC is not allocated to specific fishery management areas.

All directed fishing for Pacific cod by federally permitted vessels would cease November 1 to preserve Pacific cod prey fields in the November through December 31 time period, when Steller sea lions have higher energetic requirements and consume a greater proportion of Pacific cod relative to summer.

Because current assessment and management practices do not apportion the BSAI Pacific cod TAC to fishery management areas, this RPA specifies harvest levels anticipated by NMFS. Should Pacific cod harvest levels in 541 exceed these levels, NMFS shall reinitiate ESA section 7 consultation to determine if the groundfish fisheries may be affecting Steller sea lions or their critical habitat in a manner or to an extent not analyzed in this Biological Opinion. For Area 541, NMFS shall reinitiate ESA section 7 consultation if the harvest of Pacific cod by vessels using nontrawl and trawl gear exceeds 1.5% and 11.25% of the BSAI Pacific cod ABC in any given year. These percentages correspond with the maximum harvest amount by the respective sectors from 2007 through 2009 as expressed a percentage of the ABC, as reduced by the State of Alaska GHL harvest (3% of ABC), and rounded to the nearest quarter of a percent. This aspect of the RPA has been changed relative to what was proposed in the draft Biological Opinion.

Atka mackerel

Under the status-quo regime Steller sea lion critical habitat is closed to directed fishing for Atka mackerel (0-20 nm around all rookeries and haulouts) in Area 541. Atka mackerel is harvested in the eastern portion of Area 541 which corresponds with RCA 5. Steller sea lions have been increasing in abundance in this RCA. This RPA does not require additional conservation measures for Atka mackerel fisheries in Area 541. However, to be consistent with seasonal management for Area 542 the seasons for Atka mackerel in Area 541 would also be established as January 20-June 10 for the A season and June 10-November 1 for the B season. This aligns the Atka mackerel seasons with those for Pacific cod and pollock and is likely to further disperse Atka mackerel catch temporally.

The closure of the Bering Sea subarea (i.e., east of Area 541) to Atka mackerel directed fishing is necessary to allow for continued harvest of Atka mackerel in a manner similar to historical practices in the Bering Sea. The RPAs are intended to modify fishing practices to protect the prey field for Steller sea lions in areas where Steller sea lions are in decline and not intended to affect historical fishing behavior that has not been identified as a concern in this Biological Opinion. This aspect of the RPA was added subsequent to the draft Biological Opinion upon recognition of the unintended consequences of modifications to the Aleutian Islands Atka mackerel fishery through the RPA. The rationale for closing the Bering Sea subarea to directed fishing for Atka mackerel in order to preserve historic fishing opportunities is as follows:

A portion of the Area 541/Bering Sea apportionment of Atka mackerel is harvested inside critical habitat in the Bering Sea subarea. It is not possible to harvest Atka mackerel outside of critical habitat in the Bering Sea subarea because of where Atka mackerel occurs. Groundfish species likely to be harvested concurrent with Atka mackerel are Arrowtooth flounder, Pacific cod, Greenland turbot, rock sole and yellowfin sole. These groundfish do not occur inside Bering Sea critical habitat in sufficient quantities to support harvest of basis species and of Atka mackerel up to the maximum retainable amount (MRA) for that basis species (NMFS Catch Accounting System, October 21, 2010). This aspect of the RPA would provide for limited fishing for Atka mackerel inside critical habitat using another groundfish species (e.g., yellowfin sole) as the basis species for allowing retention of Atka mackerel up to the MRA specified in Table 11 of 50 CFR part 679. The retention of an incidentally caught species is dependent on the basis species and the directed fishery closures that trigger a “trip” and resetting of the basis species. It is necessary to close the entire Bering Sea subarea to Atka mackerel directed fishing to prevent the triggering of a trip and the resetting of the basis species for purposes of the Atka mackerel MRAs. If the Bering Sea subarea is left open to Atka mackerel directed fishing, a vessel crossing into critical habitat where directed fishing for Atka mackerel is closed would be experiencing a new trip and the resetting of the basis species for determining maximum retainable amounts during that trip. Because groundfish other than Atka mackerel does not occur in large quantities inside critical habitat, any harvest inside critical habitat is likely to be primarily Atka mackerel, which would violate the Atka mackerel directed fishing closure.

8.3.3 How the RPA Avoids Jeopardy and Adverse Modification

The ESA imposes on federal agencies a duty to ensure that their actions are not likely to jeopardize listed species or destroy or adversely modify their designated critical habitat. The following are the definitions of survival and recovery from the ESA Section 7 Handbook:

- Survival is defined as the species' persistence, as a listed or recovery unit, beyond the conditions leading to its endangerment, with sufficient resilience to allow for recovery from endangerment.
- Recovery is the process by which species' ecosystems are restored and/or threats to the species are removed so self-sustaining and self-regulating populations of listed species can be supported as persistent members of native biotic communities.

Recovery is also defined at 50 CFR 402.02 (implementing regulations for the Endangered Species Act):

- Recovery means 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 Endangered Species Act.

NMFS used the requirements of the Endangered Species Act, along with the above definitions and judicial precedents, to guide its development of the RPA.

Overall Intent of RPA

In this Biological Opinion, NMFS has determined through the weight of evidence that competitive interactions between commercial fisheries and the western DPS of Steller sea lions for important prey species could affect survival and natality rates to the point that it prevents the western DPS from achieving survival and recovery goals. The trend in abundance by sub-region is lowest in the western Aleutian Islands sub-region. In this sub-region, the ratio of pups to non-pups is lower than any other sub-region. The number of sea lions in the central Aleutian Islands sub-region, especially the western portion, is also declining. In response to this determination, the information on trends in abundance and pup production in the western Aleutian Islands and central Aleutian Islands sub-regions, and the requirement of the ESA to ensure that its actions are not likely to jeopardize the continued existence of the western DPS of Steller sea lions, NMFS must implement an RPA to the current fishery.

Implementation of this RPA is expected to achieve the following:

- Eliminate local competition between Steller sea lions and the Atka mackerel and Pacific cod fisheries in the western Aleutian Islands sub-region. This is intended to improve foraging success and prey availability for juvenile and adult Steller sea lions, which in turn is expected to lead to higher survival and natality rates.
- Significantly reduce the competitive overlap between Steller sea lions and fisheries for Atka mackerel and Pacific cod in the central Aleutian Islands sub-region. This is intended to improve foraging success and prey availability for Steller sea lions, particularly adult females with dependent young in winter, which in turn is expected to lead to higher survival and natality rates.
- Provide an adaptive management strategy for exploited groundfish forage species that explicitly accounts for the prey requirements of listed species and indicates what actions may be needed to reach the recovery goals for the western DPS in the western and central Aleutian Islands. This is expected to allow NMFS to better evaluate the impact of the conservation measures on the recovery of the western DPS of Steller sea lions in the western and central Aleutian Islands, an area that has seen the least recovery for the western DPS.

This RPA was structured to mitigate effects of the fishery in sub-regions where Steller sea lion abundance continues to decline (western and central Aleutian Islands) and where available information indicates that reproduction may be reduced to a level that cannot support positive population growth (the western Aleutian Islands). The western and central Aleutian Islands were the only two sub-regions where population growth was a significant concern from 2000-2008. Currently, the western DPS of Steller sea lions is growing at a rate of 1.4% per year, although this rate of increase is not statistically significant (P= 0.21). As explained in Chapter 7, the western DPS is not meeting the criteria of a recovering population as recommended in the Revised Recovery Plan (NMFS 2008a).

This RPA was structured to address observed declines in Steller sea lions where the weight of evidence indicates that commercial fisheries for Steller sea lion prey are appreciably reducing the reproduction and thus numbers of Steller sea lions and adversely modifying the value of their critical habitat. This is occurring in the western and central Aleutians by the fisheries removing large quantities of prey species important to Steller sea lions for basic nutrition and reproductive success. It is recognized that competition with fisheries for prey is likely one component of an intricate suite of natural and anthropogenic factors affecting Steller sea lion numbers and reproduction. While natural factors may be contributing, NMFS must ensure that actions authorized by NMFS are not likely to appreciably reduce the likelihood of both survival and recovery of the western DPS of Steller sea lions.

While effects of the RPA on the response of the Steller sea lion population cannot be projected with certainty with the available information, NMFS has determined that conserving important prey species to foraging Steller sea lions in the areas and seasons commensurate with the rate of decline observed in each fishery management area will be adequate to reduce the effects of the fisheries such that they would not be likely to suppress the survival and recovery of the species to an appreciable extent. This determination is supported by two important findings. First, after the implementation of the conservation measures associated with the 2001 Biological Opinion (NMFS 2001), there was a change in the annual population trajectory of the western DPS on the order of 5 to 6 percentage points (i.e., the trend was approximately -4% per year in the 1990s and increased to 1.4% per year between 2000 and 2008). As noted above, this change in the population trajectory was statistically significant, although the experimental design to definitively conclude this change was entirely due to the imposed conservation measures was never implemented. Further, a more fine scale response to conservation measures can be inferred by analyzing change in trends in abundance among the zones identified in NMFS (2001) that were in decline in the 1990s. Seven of the 13 zones identified in NMFS (2001) were declining in abundance prior to the implementation of conservation measures in 2002. The average change in the annual population trajectory for these seven zones between 2000 and 2008 was approximately +6 percentage points. As noted above, no definitive experiment was done to quantify the extent to which the conservation measures implemented were directly responsible for the observed improvement in the population trajectory by zone.

The determination that the RPA is adequate to ensure that the action is not likely to jeopardize the western DPS or adversely modify designated critical habitat is also premised on the expectation that fishery harvests displaced from important times and areas by the RPA are not subsequently concentrated in a manner that would result in effects that are deleterious to the western DPS of Steller sea lions. This is not currently anticipated. However, should modifications to the fisheries result in increases in spatial and temporal concentrations of fishing effort outside the realm of recent fishing patterns (e.g., 2000-2008), that would constitute “new information” requiring NMFS to reinitiate consultation.

Steller sea lion population trends in the central GOA (RCAs 8 and 9) were stable from 2000-2008. NMFS deliberated on whether or not additional fishery conservation measures should be included in the RPA to protect Steller sea lion populations from potential effects of fishing in the central GOA in addition to measures that will be required in the Aleutian Islands. Fisheries for important Steller sea lion prey species occur in RCAs 8 and 9 (Fishery Management Areas 620 and 630). There is evidence that natality has decreased by 36% in the past three decades in at least part of the central GOA, possibly due to nutritional stress from a reduction in quality or quantity of available prey (Holmes *et al.* 2007). Ultimately, NMFS did not require additional non-discretionary fishery management measures for the central GOA for the following reasons:

- Overall population trends for non-pups were stable (i.e., trend in abundance was -0.1%) from 2000-2008 in this sub-region.
- Overall population trends for pups were increasing (i.e., increase in number of pups born between 2001/2002 and 20009 was 6%) in this sub-region.
- Population trends in non-pups in adjacent sub-regions increased by 39% (western GOA) and 58% (eastern GOA) from 2000-2008. The overall number of Steller sea lions in the GOA (i.e., western, central and eastern) increased at a significant rate of 2.8% per year between 2000 and 2008.
- The 2002 fishery management measures implemented in the GOA were more conservative than the 2002 fishery management measures implemented in the Aleutian Islands. The 2002 fishery management measures implemented in the GOA were effective in displacing a substantial amount of Pacific cod and pollock harvest from 0-10 and 10-20 nm zones of critical habitat.
- If measures for the western and central Aleutian Islands implemented in the RPA described above are approximately as effective in reducing fishing effects on Steller sea lion populations in the western and central Aleutian Islands as the measures implemented as a result of the prior two Biological Opinions appear to be at reducing fishing effects on Steller sea lion populations in the eastern Aleutian Islands, and western and eastern GOA, the western DPS would be expected to be increasing at a robust rate (i.e., approaching 3% per year) with no additional conservation measures. Further, all of the demographic criteria for downlisting reported in the Revised Recovery Plan would be met. For example, if the number of sea lions in the western Aleutian Islands sub-region increased by 6.0 percentage points per year by closing the western Aleutian Islands sub-region to commercial fishing and if the number of sea lions in the central Aleutian Islands sub-region increased by 1.5 percentage points per year by adding additional restrictions to commercial groundfish fisheries in the central Aleutian Islands, and if all other trends by sub-regions remained constant through 2018, the resulting rate of increase for the entire western DPS of Steller sea lion would be approximately 2.5% per year, no two juxtaposed areas would be in significant decline, and no single area would have a realized a decline of 50% or more.

It is NMFS' opinion, based on the best scientific and commercial data available, that the RPA described above will ensure that the fisheries described as the proposed action are not to likely jeopardize the continued existence of the western DPS of Steller sea lions or adversely modify their designated critical habitat. However, NMFS should continue to monitor Steller sea lion population trends, especially in the western and central Aleutians Islands and central GOA, to evaluate whether fishery conservation measures continue to be adequate to conserve endangered Steller sea lions and their critical habitat consistent with the requirements of the ESA.

8.3.4 Adaptive Management

NMFS cannot predict with certainty the quantitative impact on the western DPS of Steller sea lions of the planned change in management associated with implementing the RPA in the western and central Aleutian Islands sub-regions. However, the RPA has been designed to be increasingly restrictive going from the eastern edge of the central Aleutian sub-region to the western edge of the western Aleutian sub-region. Therefore, we envision being able to make strong inferences regarding the efficacy of this RPA by testing the hypotheses that trends in both Steller sea lion numbers and prey biomass (i.e., Atka mackerel and cod) in these two sub-regions will diverge over the next 10 years relative to (1) observed trends between 1990 and 2020 and (2) and between the two sub-regions (i.e., the western and central Aleutian Islands).

Changes in numbers of Steller sea lions in the two sub-regions will be tested using approaches similar to those described by Skalski *et al.* (2001), where various “parallelism hypotheses” will be evaluated. This approach allows for differing baseline characteristics between the two sub-regions. As noted in the 2000 Biological Opinion, given the observed level of uncertainty in the biennial surveys for Steller sea lions, we anticipate being able to detect differences in trend lines on the order of 2%-4% per year using appropriate survey data from the 1990s, the five surveys between 2000 and 2010, and five surveys between 2012 and 2020. A formal experimental design will be finalized and reviewed to confirm these expectations by June 2011. The basic premise being tested is that the slope of the non-pup counts over the next 10 years (and perhaps pup counts) will increase faster in Area 543 than in Areas 542 or 541, as well as various tests evaluating changes in rate of change in abundance in the 1990s with observed changes in the 2000s and 2010s.

Changes in biomass of Atka mackerel and Pacific cod in the western and central Aleutian Islands sub-regions (Areas 543, 542 and 541) will be tested in a similar manner to those predicted for Steller sea lion biomass; however, given the uncertainty in the biennial estimates of biomass are considerably greater for Atka mackerel and Pacific cod than they are for Steller sea lions, the statistical power of these analyses may be unsatisfactory. A formal experimental design will be finalized and reviewed by June 2011.

While specific funding commitments cannot be made at this time, it is the intention of NMFS to continue the biennial surveys of abundance for Steller sea lions and biomass for groundfish in the Aleutian Islands through 2020. Further, as funding allows, NMFS will attempt to expand tagging, brand/resight, and foraging studies in the western and central Aleutian Islands through 2020 to better evaluate the efficacy of this RPA. Finally, NMFS will attempt to expand tagging and foraging studies in the central Gulf of Alaska sub-region to better understand factors that may be limiting a more robust rate of recovery of Steller sea lions in this area.

9 INCIDENTAL TAKE STATEMENT

Section 9 of the Act and Federal regulation pursuant to section 4(d) of the Act prohibit the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the Act provided that such taking is in compliance with the reasonable and prudent measures and terms and conditions of this Incidental Take Statement.

The measures described below are non-discretionary, and must be undertaken by NMFS so that they become binding conditions of any grant or permit issued, as appropriate, for the exemption in section 7(o)(2) to apply. NMFS has a continuing duty to regulate the activity covered by this incidental take statement. If NMFS (1) fails to require the applicant to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, and/or (2) fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, NMFS must report the progress of the action and its impacts on the species as specified in the incidental take statement (50 CFR §402.14(i)(3)).

An incidental take statement specifies the impact of any incidental taking of endangered or threatened species. It also provides reasonable and prudent measures that are necessary to minimize impacts and sets forth terms and conditions with which the action agency must comply in order to implement the reasonable and prudent measures.

A marine mammal species or population stock that is listed as threatened or endangered under the ESA is, by definition, also considered depleted under the Marine Mammal Protection Act (MMPA). The ESA allows takings of threatened and endangered marine mammals only if authorized by section 101(a)(5) of the MMPA. Before incidental take of listed marine mammals may be exempt from the taking prohibition of ESA section 9(a), incidental taking must be authorized under section 101(a)(5)(E) of the MMPA, otherwise known as a “negligible impact determination” (NID).

NMFS issued an NID for the Central North Pacific (CNP) stock of humpback whales on May 28, 2010, 75 FR 29984). On October 29, 2010, NMFS issued a draft NID containing a preliminary determination that incidental taking from commerical fishing will have a negligible impact on the endangered Western North Pacific (WNP) stock of endangered humpback whales, North Pacific stock of sperm whales, the Northeast Pacific stock of fin whales, and Western U.S. stock of Steller sea lions; and on the threatened Eastern U.S. stock of Steller sea lions.[30] Subsquent to issuing the draft NID, NMFS published a notice of a proposed permit issuance for a period of three years to authorize the incidental, but not intentional, taking of individuals of the CNP and WNP stocks of humpback whales, sperm whales, the NEP fin whale stock, the western U.S. stock of Steller sea lions, and the eastern U.S. stock of Steller sea lions (75 FR 68767). This proposed authorization was based on determinations that mortality and serious injury of affected stocks of humpback whales, sperm whales, fin whales, and Steller sea lions incidental to the

---

[30] The analysis supporting the NID was based on data from the 2009 Marine Mammal Stock Assessment Reports (Angliss and Allen 2010) whereas, the effects analysis in this Biological Opinion was based on data in the 2008 Marine Mammal Stock Assessment Report (Angliss and Allen 2009). Thus, the estimates of serious injury and mortality vary slightly between the NID and the ITS. However, the differences are *diminimus* and inconsequential to the conclusions of either analysis.

commercial groundfish fisheries in the Bering Sea and Gulf of Alaska will have a negligible impact on the affected, ESA-listed marine mammal stocks; that a recovery plan has been developed, that a monitoring program is established; and that vessels in the fisheries are registered. This ITS will become effective upon completion of a final NID and publication of the permit in the Federal Register to take these threatened and endangered marine mammal species incidental to the commercial Federal and parallel groundfish fisheries in the Bering Sea and the Gulf of Alaska. Therefore, NMFS is not including an incidental take authorization for marine mammals as at this time because the incidental take of marine mammals has not been authorized under section 101(a)(5) of the Marine Mammal Protection Act and/or its 1994 Amendments. Following issuance of such regulations or authorizations, NMFS may amend this Biological Opinion to include an ITS for marine mammals, as appropriate.

9.1 Amount or Extent of Incidental Take (Upon Permit Approval)

Western DPS of Steller Sea Lions

In this Biological Opinion, NMFS determined that direct take of Steller sea lions is reasonably likely to occur in both the Federal and Alaska State managed parallel fisheries for pollock, Pacific cod, and Atka mackerel. Section 4.3.3 explains the extent of take of western DPS Steller sea lions incidental to commercial fisheries in Alaska. The 2008 Marine Mammal Stock Assessment Report (Angliss and Allen 2009) estimates that approximately 26 individuals from the western DPS of Steller sea lions are taken annually incidental to commercial fisheries off Alaska; this estimate includes interaction with the State of Alaska gillnet fishery in Prince William Sound. These estimates are minimum estimates as they are based on observed takes and fisheries are observed at a rate of less than 1.0. The Council, working with industry, has made extensive efforts to reduce the amount of direct take of Steller sea lions to the extent practicable, and therefore, NMFS expects similar direct take levels to continue. The scope of this incidental take statement extends to the parallel fisheries authorized by the State of Alaska in accordance with the requirements contained below.

Eastern DPS of Steller Sea Lions

Section 4.3.3 explains the extent of take of eastern DPS Steller sea lions incidental to commercial fisheries in Alaska. The 2008 Marine Mammal Stock Assessment Report estimates that approximately one individual from the eastern DPS of Steller sea lions is taken annually incidental to commercial fisheries off Alaska (Angliss and Allen 2009). These estimates are minimum estimates as they are based on observed takes and fisheries are observed at a rate of less than 1.0. NMFS expects similar direct take levels to continue.

Central North Pacific Humpback Whales

Direct effects of commercial fisheries on CNP humpback whales are described in section 4.8.2.1. The 2008 Marine Mammal Stock Assessment Report estimates that approximately two individuals from the CNP stock of humpback whales are seriously injured or killed every year incidental to commercial fisheries in the U.S. (Angliss and Allen 2009).These estimates are minimum estimates as they are based on observed takes and fisheries are observed at a rate of less than 1.0.

Western North Pacific Humpback Whales

Direct effects of commercial fisheries on WNP humpback whales are described in section 4.8.2.2. The 2008 Marine Mammal Stock Assessment Report estimates that approximately one individual from the WNP stock of humpback whales is seriously injured or killed every year incidental to commercial fisheries in the U.S. (Angliss and Allen 2009).These estimates are minimum estimates as they are based on observed takes and fisheries are observed at a rate of less than 1.0.

Sperm Whales

Direct effects of commercial fisheries on sperm whales are described in section 4.9.2. Between 2002 and 2006, there were three observed serious injuries of sperm whales in the Gulf of Alaska sablefish longline fishery (Angliss and Allen 2009). The 2008 Marine Mammal Stock Assessment Report estimates that approximately two individual sperm whales are seriously injured or killed every year incidental to commercial fisheries in the U.S. (Angliss and Allen 2009).These estimates are minimum estimates as they are based on observed takes and fisheries are observed at a rate of less than 1.0.

Northeast Pacific Fin Whales

Direct effects of commercial fisheries on fin whales are described in section 4.10.5. Between 2002 and 2006, there was one observed fin whale mortality in the Bering Sea pollock trawl fishery (Angliss and Allen 2009). The 2008 Marine Mammal Stock Assessment Report estimates that approximately 0.23 NEP fin whales are seriously injured or killed every year incidental to commercial fisheries in the U.S. (Angliss and Allen 2009).These estimates are minimum estimates as they are based on observed takes and fisheries are observed at a rate of less than 1.0.

Interannual Variability in Take

NMFS believes that specifying a take limit over a period of three consecutive years is warranted based on demonstrated interannual variability in the rate of interactions between the fisheries and sea lion and whale populations and because the best available empirical data on the extent of annual take have been used in estimating expected levels of incidental take. Over a period of three consecutive years, the likelihood of the fishery exceeding the specified level of take is extremely low. Whereas, if incidental take levels were specified on an annual basis, the likelihood that the level of interactions occurring in the fishery in a given year could exceed the levels specified in the paragraphs above is high. NMFS expects interaction levels to hover around the annual levels listed above and not to exceed the values in the table below over a period corresponding to 3 consecutive fishing years.

NMFS SFD should evaluate take levels following the 2011 fishery based on the sum of the estimated serious injuries and mortalities incidental to the 2009, 2010, and 2011 fisheries. This level should be reevaluated following the 2012 fishery based on the sum of the 2010, 2011, and 2012 direct takes, and so on. If during the course of the fisheries, the level of take specified in the table below is exceeded, SFD must immediately reinitiate formal consultation pursuant to Criterion 2 of the section 7 regulations (50 CFR 402.16 (a)).

Population/Stock	Incidental Take (Serious Injury or Mortality)
Western DPS of Steller sea lions	78
Eastern DPS of Steller sea lions	3
CNP Humpback Whales	6
WNP Humpback Whales	3
Sperm Whales	6
NEP Fin Whales	3

The number of sea lions and whales expected to be captured or killed in the groundfish fisheries off Alaska over a period of three consecutive years. Calculated by rounding-up the estimated annual incidental serious injury or mortality for each stock to the nearest integer and multiplying by three.

9.2 Effect of the Take

In this Biological Opinion, NMFS determined that the proposed action is likely to jeopardize the continued existence of the western DPS of Steller sea lions and result in the destruction or adverse modification of its designated critical habitat. However, NMFS made a draft determination that impacts of direct take of western and eastern DPSs of Steller sea lions, WNP and CNP stocks of humpback whales, sperm whales, and the NEP stock of fin whales anticipated to occur incidental to the Federal and State of Alaska parallel groundfish fisheries off Alaska are likely to be negligible (75 FR 68767). Incidental serious injury and mortality refers only to direct mortality and serious injury, such as from entanglement or hooking by fishing gear, and does not include indirect effects through competition for resources. Therefore, levels of direct take expected to occur incidental to the groundfish fisheries off Alaska are not likely to jeopardize the continued existence of the marine mammal stocks likely to be affected by the fisheries.

9.3 Reasonable and Prudent Measures and Associated Terms and Conditions in Italics

In order to be exempt from the prohibitions of section 9 of the ESA, NMFS must comply with the following terms and conditions, which implement the reasonable and prudent measures. These terms and conditions are non-discretionary.

1. **NMFS will monitor the take of ESA-listed marine mammals in the groundfish fisheries in the Bering Sea and the Gulf of Alaska.**

   NMFS-trained observers on vessels in these fisheries will be deployed under the existing program for observer coverage based on vessel size and sector.

   NMFS will use observer data to make minimum estimates of mean annual mortality for each fishery.

   NMFS will evaluate the observer coverage that results from existing regulatory requirements to determine if changes in coverage are warranted to better assess take of listed marine mammals.

10 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. NMFS has determined that the following conservation recommendations should be implemented by the appropriate entities in order to facilitate the recovery of listed Steller sea lion populations. In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, NMFS requests notification of the implementation of any conservation recommendations.

10.1 Conservation Programs for State managed Fisheries

Available information in this Biological Opinion indicates that Steller sea lions may continue to depend on areas closer to shore for efficient foraging. Analysis of these data has highlighted the concern that fisheries managed by the State of Alaska, within 3 nm of shore, could have adverse effects on sea lions through both indirect and direct mechanisms. The State of Alaska has been very supportive in providing information to NMFS in order to evaluate potential areas of concern (i.e., Kruse *et al.* 2000, Woodby and Hulbert 2006, Soboleff 2005, and Woodby *et al.* 2005). The State of Alaska should further explore these issues and determine whether any conservation measures are necessary in order to avoid adversely affecting the survival and recovery of Steller sea lions. Numerous options are available for further informal and formal consultations between the State of Alaska and NMFS depending upon the appropriate course of action. The goal should be continued cooperation in minimizing adverse impacts to Steller sea lions in order to facilitate their recovery and remove them from listing under the Endangered Species Act.

10.2 Minimizing the Ecosystem Effects of the “Race for Fish”

Overcapitalized fisheries or fisheries that seek fish during a narrow space/time frame because of fish aggregation, product or bycatch considerations have greater potential to produce localized depletion of fish or to interfere with predators that also take advantage of fish that concentrate at certain times. The comprehensive assessment process described in this Biological Opinion provides a means to identify those fisheries and to develop target fishery-specific mitigation measures. However, NMFS, working with the NPFMC, also should promote other means to reduce concentration of fisheries in time and space and overcapitalization of fisheries. Fishery rationalization programs such as the Individual Fishing Quota (IFQ) program, the Community Development Quota (CDQ) program, BSAI groundfish FMP amendments (i.e., 80 and 85) and the American Fisheries Act (AFA) cooperatives have shown success in reducing the “footprint” of fisheries, especially at smaller time/space scales. NMFS recommends consideration of an expansion of these types of approaches to further rationalize the BSAI cod groundfish fishery.

10.3 Recovery Plan and the Identification of Actions Intended to Promote the Recovery of the Western DPS

In 1992, NMFS published the first recovery plan for Steller sea lions. In 2008, NMFS published a revised Steller sea lion recovery plan (NMFS 2008a). Section II of NMFS 2008a summarizes a variety of conservation measures that could potentially influence the recovery of the western DPS of Steller sea lion. The reader is directed to this section of the Recovery Plan for further detail. Section V of the Recovery Plan (pages V-24 to V-59) provides a detailed summary of actions required to promote the recovery of the western DPS. The plan calls for 78 actions in 5 categories:

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

As noted in the Recovery Plan, while actions identified in the Recovery Plan are not mandated under the ESA, NMFS will incorporate as many of these actions into its annual spending plans as possible, and will work closely with Congress to inform the appropriation process as to the funding requirements needed to fully recover this DPS.

In October 2010, the North Pacific Fishery Management Council passed a motion recommending *inter alia* that NMFS include the following research and management actions in its FY11 Annual Operating Plans, continuing into outlying years as appropriate:

1. Focal studies of Steller sea lion foraging behavior, Steller sea lion diet, fish abundance, fish movement, oceanography, ocean productivity and fisheries impacts in contrasting areas of Steller sea lion population trend in the Aleutian Islands and in areas where Steller sea lions forage. These studies would be conducted in summer and winter. Fish abundance estimates would be from trawling, acoustics, tagging, pots and/or camera surveys depending on the species and habitat. AFSC standard trawl surveys are not appropriate for assessing fish biomass distribution at local scales important to Steller sea lions.
2. Foraging ecology studies of Steller sea lions in the western and central Aleutians. Specifically, this research would include at-sea tracking of adult females and juveniles, and collecting Steller sea lion scat and spew. Supplemental research could include stable isotope analyses, fatty acid analysis, contaminant studies, monitoring of condition and health indices, and additional photogrammetric work.
3. Studies to assess vital rates (i.e., reproduction and survival) of Steller sea lions in the western and central Aleutians. Specifically, this would require longitudinal studies (e.g., branding of pups) to determine rates of age- or size-class specific survival, as well as studies to help evaluate the reproductive performance of adult females and natality, including comparative surveys throughout the western Distinct Population Segment.
4. Studies investigating advancements in methods to estimate sea lion abundance, such as the use of unmanned aerial vehicles, that would increase the probability of acquiring abundance estimates in remote areas. [It should be noted that the Council’s Scientific and Statistical Committee specifically recommended that more robust and statistically reliable methods be developed for converting pup and non-pup counts to estimates of total abundance.]
5. Studies to improve understanding of killer whale predation of Steller sea lions, particularly in the western and central Aleutian Islands.
6. Increased frequency of pup and non-pup surveys.
7. There is a need for studies of localized fishery-protected species interactions. Studies of interactions between Steller sea lions and fisheries are needed in the Central GOA, with an emphasis on seasonal prey fields, diet, and movement of sea lions and their prey. These studies should be conducted at appropriate spatial and temporal scales.

8. Foraging ecology studies of Steller sea lions in the Commander Islands. Research techniques would be similar to item #1.
9. Foraging ecology studies of Steller sea lions in the Gulf of Alaska. In addition to at-sea tracking of older animals, outside of the Kodiak area the primary information needed from this sub-region is updated information on diet composition of Steller sea lions throughout the sub-region.
10. Maintain assessment of Steller sea lion vital rates in the Russian Far East and Commander Islands. Research techniques would be similar to item #4 and include expansion to autumn and winter periods.
11. Aerial photogrammetric survey studies of rookeries and haul-outs in Russia. This survey methodology would provide abundance estimates for sea lions in Russia directly comparable to estimates for Alaska.

In addition to this list of research actions, several other proposals were developed by NMFS staff:

1. Studies to assess the nature and extent of Steller sea lion movements between the western Aleutians and the Commander Islands. Use of branded and satellite tagged animals will clarify movement and foraging patterns, assist in determining source and location of mortality, possible emigration, and stock structure.
2. Studies to assess the nature and extent of movements and breeding fidelity between stocks in southeast Alaska and Prince William Sound. These data will help to clarify trends in abundance between the eastern DPS and western DPS.
3. Development and implementation of new and improved procedures for monitoring subsistence harvest levels of western DPS Steller sea lions throughout the U.S. range.
4. Re-initiation of Marine Mammal Observer Program studies in the Gulf of Alaska to assess the significance of mortality incidental to Category II commercial fisheries. Special emphasis should be placed on evaluating mortalities associated with the Prince William Sound salmon drift gillnet fishery.
5. Continue to develop multi-species modeling approaches to better account for ecosystem processes in groundfish stock assessments and harvest recommendations.

Similarly to actions identified in the revised Steller Sea Lion Recovery Plan (NMFS 2008a), NMFS will endeavor to include as many of these actions as possible into its annual spending plans, and will work with Congress to receive appropriations necessary to support the research activities described above.

10.4 Co-management of Steller Sea Lions with Alaska Native Organizations

The Tribal Governments of St. Paul and St. George Islands in the Pribilof Islands and the Aleut Marine Mammal Commission represent Alaska Native co-management partners for the subsistence harvest of Steller sea lions for western Alaska. The Alaska Sea Otter and Steller Sea Lion Commission has worked with some coastal Alaska Native communities to further conservation, local management and local research on Steller sea lions. Historical harvest of sea lions has been documented since 1992 by retrospective surveys conducted by the Alaska Department of Fish and Game Subsistence Division.

11 REINITIATION NOTICE

This concludes formal consultation on the continued authorization of the groundfish fisheries under the Fishery Management Plan for Groundfish of the Bering Sea and Aleutian Islands Management Area; the groundfish fisheries under the Fishery Management Plan for Groundfish of the Gulf of Alaska and the State of Alaska parallel groundfish fisheries. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of the incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action.

12 CITATIONS

Abookire, A., J.T. Duffy-Anderson, and C.M. Jump. 2007. Habitat Associations and Diet of Juvenile Pacific Cod, *Gadus macrocephalus* in Chiniak Bay, Alaska. Mar. Biol. 150(4):713-726.

Adams, C.F., R.J. Foy, D.S. Johnson, and K.O. Coyle. 2008. Seasonal Changes in Pelagic Fish Biomass Around the Chiswell Island Steller Sea Lion Rookery in 2003. Fish Res. 93:179-185.

Adams, T.C. 2000. Foraging Differences and Early Maternal Investment in Adult Female Alaskan Steller Sea Lions (*Eumetopias jubatus*). Unpublished Ph.D. dissertation, Texas A & M University, Galveston.

Agler, B.A., R.L. Schooley, S.E. Frohock, S.K. Katona, and I.E. Seipt. 1993. Reproduction of photographically identified fin whales, *Balaenoptera physalus*, from the Gulf of Maine. J. Mamm. 74:577–587.

Ainley, D.G., R.P. Henderson, H.R. Huber, R.J. Boelkelheide, S.G. Allen, and T.L. McElroy. 1985. Dynamics of White Hake/Pinniped Interactions in the Gulf of the Farallones. Memoirs of the Southern California Academy of Sciences 9 pp. 109-122.

Alaska Fisheries Science Center (AFSC). 2009. Alaska Fisheries Science Center review (April 2009) of Alaska SeaLife Center (ASLC) Technical Report ASLC-0901 “Evidence for high natality rates among Steller sea lions in the Gulf of Alaska” (Maniscalco, Hennen and Parker). AFSC, Seattle, WA. 5p.

Alaska Fisheries Science Center (AFSC). 2010a. Untitled. AFSC comparisons of SSL population changes 1991-2009 relative to the spatial and temporal distribution of SSL prey species, fisheries for these prey species, and various oceanographic measures of the North Pacific. AFSC manuscript. 74 p.

Alaska Fisheries Science Center (AFSC). 2010b. Distribution of At-Sea Juvenile Steller Sea Lions Location: Analysis for BiOp 2009. 26 pp. Unpublished report, available Alaska Fisheries Science Center, NMFS, NOAA, 7600 Sand Point Way, NE, Seattle, Washington 98115.

Alaska Fisheries Science Center (AFSC). 2010c. Regional Trends in Adult and Juvenile Western Steller Sea Lion Counts, 2000-2008. October 2010. 7 pp. Unpublished report, available Alaska Fisheries Science Center, NMFS, NOAA, 7600 Sand Point Way, NE, Seattle, Washington 98115.

Alaska Department of Fish and Game (ADF&G). 1998. The Subsistence Harvest of Harbor Seal and Sea Lion by Alaska Natives in 1994. ADF&G Technical Paper, 246, Alaska Department of Fish and Game: Prepared for NMFS: Juneau.

Alaska Department of Fish and Game (ADF&G) and NMFS. 2001. Satellite Telemetry and Steller Sea Lion Research. Unpublished, 48 pgs. Accessed online September 29, 2010: http://www.fakr.noaa.gov/protectedresources/stellers/biop2002/supportdoc/ADFGandNMFS2001.pdf.

Alaska Fisheries Science Center (AFSC). 2010a. Steller Sea Lion Fishery and Oceanographic Analysis BiOp 2009 (October 22, 2009). 78 pages. Unpublished report, available Alaska Fisheries Science Center, NMFS, NOAA, 7600 Sand Point Way, NE, Seattle, Washington 98115.

Alaska Sea Grant. 1993. Is it food? Alaska Sea Grant Report, 93-1, Alaska Sea Grant Program, 304 Eielson Building, University of Alaska Fairbanks, Fairbanks, Alaska 99775. p. 59.

Alaska-BC Whale Foundation. 1996. http://www.sfu.ca/biolgoy/berg/whale/abcwhales.html

Albers, P.H., and T.R. Loughlin. 2003. Effects of PAHs on Marine Birds, Mammals, and Reptiles. pp 243-261, *In:* P.E.T. Douben (Ed.) PAHs: An Ecotoxicological Perspective. John Wiley and sons, London.

Alderdice, D.F., and A.S. Hourston. 1985. Factors influencing development and survival of Pacific herring (*Clupea harengus pallasi*) eggs and larvae to beginning of exogenous feeding. Can. J. Fish. Aquat. Sci. 42 (Suppl. 1):56-68.

Alexander, A.B. 1898a. Observations During a Cruise on the Seal Schooner, *Louis Olsen,* August 1894. *In:* Seal and Salmon Fisheries and General Resources of Alaska. Vol. IV. U.S. Government Printing Office, Washington, D.C.

Alexander, A.B. 1898b. Observations During a Cruise on the *Dora Siewerd*, August-September 1895. *In:* Seal and Salmon Fisheries and General Resources of Alaska. Vol. IV. U.S. Government Printing Office, Washington, D.C.

Allen, P.C. 2009. Seasonal oscillations in the mass and food intake of Steller sea lions. M.S. Thesis, University of British Columbia, Vancouver, B.C., Canada. Abstract.

Alverson, D.L. 1991. Commercial Fisheries and the Steller Sea Lion (*Eumetopias jubatus*): the Conflict Arena. University of Washington, Fisheries Research Institute FRI-UW-9106. Seattle, Washington.

Alverson, D.L. 1992. A Review of Commercial Fisheries and the Steller Sea Lion (*Eumetopias jubatus*): The Conflict Arena. Rev. Aquat. Sci. 6:203-256.

Anderson, P.J. 2003. Gulf of Alaska small Mesh Trawl Survey Trends. *In:* J.L. Boldt (Ed.) Ecosystem Considerations for 2004. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Anderson, P.J. and J.F. Piatt. 1999. Community Reorganization in The Gulf of Alaska Following Ocean Climate Regime Shift. Marine Ecol. Progr. Series 189:117-123.

Andrews, R.D., D.G. Calkins, R.W. Davis, B.L. Norcross, K. Peijnenberg, and A.W. Trites. 2002. Foraging Behavior and Energetics of Adult Female Steller Sea Lions. *In:* D. DeMaster and S. Atkinson. (Eds.) Steller Sea Lion Decline: Is it food II? Alaska Sea Grant, AK-SG-02-02, Fairbanks. pp. 19-22.

Angliss, R.P., and B.M. Allen. 2009. Alaska Marine Mammal Stock Assessments, 2008. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-193, 258 p.

Angliss, R.P. and R. Outlaw. 2006. Alaska Marine Mammal Stock Assessments, 2005. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-168, 255 p.

Angliss, R.P., A. Lopez, and D.P. DeMaster. 2001. Draft Alaska Marine Mammal Stock Assessments, 2001. National Marine Mammal Laboratory, 7600 Sand Point Way NE, Seattle, Washington 98115. p. 181.

Angliss, R.P., and D.P. DeMaster. 1998. Differentiating Serious and Non-Serious Injury of Marine Mammals Taken Incidental to Commercial Fishing Operations: Report of the serious injury workshop 1-2 April 1997, Silver Spring, Maryland. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-13, 48 p.

Angliss, R.P., and K.L. Lodge. 2004. Alaska Marine Mammal Stock Assessment, 2003. U.S. Dep. Commer. NOAA Tech. Memo. NMFS-AFSC-144, 230 p.

Angliss, R.P., and R. Outlaw. 2005. Alaska Marine Mammal Stock Assessments, 2004. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-161.

Arctic Climate Impact Assessment (ACIA). 2004. Impacts of a Warming Arctic. Cambridge University Press, New York, NY. pp 1046.

Ashwell-Erickson, S., and R. Elsner. 1981. The Energy Cost of Free Existence for Bering Sea Harbor and Spotted Seals. Pages 869-899. *In:* D.W. Hood and J.A. Calder (Eds.) The eastern Bering Sea shelf: Oceanography and Resources. Volume Two. Office of Marine Pollution Assessment, National Oceanic and Atmospheric Administration. University of Washington Press; Seattle, Washington.

Atkinson, S., D. Calkins, V. Burkanov, M. Castellini, D. Hennen, and S. Inglis. 2008. Impact of Changing Diet Regimes on Steller Sea Lion Body Condition. Mar. Mammal Sci. 24(2):276-289.

Atkinson, S., D.P. DeMaster, and D.G. Calkins. 2008. Anthropogenic Causes of the Western Steller Sea Lion *Eumetopias jubatus* Population Decline and Their Threat to Recovery. Mammal Review 38:1-18.

Aydin, K. 2010. Multispecies modeling of Atka mackerel/Pacific cod interactions in the Aleutian Islands. Unpublished manuscript, AFSC Seattle. 11 p.

Aydin, K., S. Gaichas, I. Ortiz, D. Kinzey, and N. Friday. 2007. A Comparison of the Bering Sea, Gulf of Alaska, and Aleutian Islands Large Marine Ecosystems Through Food Web Modeling. NOAA Tech Memo NMFS-AFSC-178, 298p.

Aydin, K.Y., G.A. McFarlane, J.R. King, and B.A. Megrey. 2003. The Bass/Model Report on Trophic Models of the Subarctic Pacific Basin Ecosystems. N. Pac. Mar. Sci. Org. Rep 25. Sydney, California. 97 pp.

Aydin, K.Y., J. Boldt, S. Gaichas, J. Ianelli, J. Jurado-Molina, I. Ortiz, J. Overland and S. Rodinov. 2006. Ecosystem Assessment of the Bering Sea/Aleutian Islands and Gulf of Alaska Management Regions. *In:* Stock Assessment and Fishery Evaluation Report for the Groundfish Resources or the Bering Sea/Aleutian Islands Regions. North. Pac. Fish. Mgmt. Council, Anchorage, Alaska.

Baba, N., H. Nitto, and A. Nitta. 2000. Satellite Tracking of Young Steller Sea Lion Off the Coast of Northern Hokkaido. Fisheries Sci. 66:180-181.

Bailey, K.M. 2000. Shifting Control of Recruitment of Walleye Pollock (*Theragra chalcogramma*) After a Major Climate and Ecosystem Change. Mar. Ecol. Prog. Ser. 198:215-224.

Bailey, K.M., A.B. Hollowed, and W.S. Wooster. 2004. Complexity in Marine Fisheries Dynamics and Climate Interactions in the Northeast Pacific Ocean. Pages 147-152 *In:* N.C. Stenseth, G. Ottersen, J.W. Hurrel, and A. Belgrano (Eds.) Marine Ecosystems and Climate Variation. Oxford Univ. Press.

Bailey, K.M., and D.G. Ainley. 1982. The Dynamics of California Sea Lion Predation On Pacific Hake. Fisheries Research 1:163-176.

Bailey, K.M., and L. Ciannelli. 2007. Ecosystem Structure and Strategies for Survival of Walleye Pollock. Pages 85-92, *In:* R.B. Spies, (Ed.) Long-Term Ecological Change in The Northern Gulf of Alaska. Elsevier, Oxford.

Bailey, K.M., and S.A. Macklin. 1994. Analysis of Patterns in Larval Walleye Pollock (*Theragra chalcogramma*) Survival and Wind Mixing Events in Shelikof Strait, Gulf of Alaska. Mar. Fish. Ecol. Prog. Ser. 113:1-12.

Bailey, K.M., L. Ciannelli, N. Bond, A. Belgrano, and N.C. Stenseth. 2005. Recruitment of Walleye Pollock in a Complex Physical and Biological Ecosystem. Progress in Oceanography 67:24-42.

Bailey, K.M., T.J. Quinn II, P. Bentzen, and W.S. Grant. 1999. Population Structure and Dynamics of Walleye Pollock. Advances in Marine Biology 37:179-255.

Baird, T.A. and B.L. Olla. 1991. Social and Reproductive Behavior of a Captive Group of Walleye Pollock, *Theragra chalcogramma*. Env. Biol. Fish. 30:295-301.

Battaile, B.C. and T.J. Quinn, II. 2006. A DeLury depletion estimator for walleye pollock *(Theragra chalcogramma)* in the eastern Bering Sea. Nat. Res. Modeling 19:655-674.

Baker, C.S., A. Perry, and L.H. Herman. 1987. Reproductive Histories of Female Humpback Whales *Megaptera novaeangliae* in the North Pacific. Marine Ecology Progress Series 41:103-114.

Baker, C.S., and L.M. Herman. 1987. Alternative Population Estimates of Humpback Whales (*Megaptera novaeangliae*) in Hawaiian Waters. Canada Journal Zoology 65:2818-2821.

Baker, C.S., L. Flórez-González, B. Abernethy, H.C. Rosenbaum, R.W. Slade, J. Capella, J.L. Bannister. 1998. Mitochondrial DNA Variation and Maternal Gene Flow Among Humpback Whales of the Southern Hemisphere. Mar Mamm Sci 14:721-737.

Baker, C.S., S.R. Palumbi, R.H. Lambertsen, M.T. Weinrich, J. Calambokidis, and S.J. O'Brien. 1990. Influence of Seasonal Migration on Geographic Distribution of Mitochondrial DNA Haplotypes in Humpback Whales. Nature. 344:238-240.

Baker, C.S., T.R. Loughlin, V. Burkanov, C.W. Matson, R.G. Trujillo, D.G. Calkins, J.K. Wickliffe, J.W. Bickham. 2005. Variation of Mitochondrial Control Region Sequences of Steller Sea Lions: The Three-Stock Hypothesis. Journal of Mammalogy 86(6):1075-1084.

Bakkala, R.G., J. Traynor, K. Teshima, A.M. Shimada, and H. Yamaguchi. 1985. Results of Cooperative U.S.-Japan Groundfish Investigations in the Eastern Bering Sea During June-November 1982. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-F/NWC-87, 448 p.

Ban, S. 2005. Modelling and Characterization of Steller Sea Lion Haulouts and Rookeries Using Oceanographic and Shoreline Type Data. Graduate thesis, University of British Columbia, Vancouver, BC. 103 pp.

Ban, S., and A. Trites. 2007. Quantification of Terrestrial Haul-Out and Rookery Characteristics of Steller Sea Lions. Marine Mammal Science 23(3):496-507.

Bannister, J.L. and E. Mitchell. 1980. North Pacific Sperm Whale Stock Identity: Distributional Evidence From Maury and Townsend Charts. Rep. int. Whal. Commn. (Special Issue) 2:219-30.

Baraff, L.S., and T.R. Loughlin. 2000. Trends and Potential Interactions Between Pinnipeds and Fisheries of New England and the U.S. West Coast. Marine Fisheries Review 62(4):1-39.

Barbeaux, S., J.N. Ianelli, E. Brown. 2005. Aleutian Islands Walleye Pollock SAFE. *In:* Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions. North Pac. Fish. Mgmt. Council, Anchorage, Alaska, Section 1A.

Barbeaux, S., J.N. Ianelli, S. Gaichas, and M. Wilkins. 2007. Aleutian Islands Walleye Pollock SAFE. *In:* Stock Assessment and Fishery Evaluation Report for t he Groundfish Resources of the Bering Sea/Aleutian Islands Regions. North Pac. Fish. Mgmt. Council, Anchorage, Alaska, Section 1A.

Barbeaux, S.J., and M. Dorn. 2003. Spatial and Temporal Analysis of Eastern Bering Sea Echo Integration trawl Survey and Catch Data of Walleye Pollock, *Theragra chalcogramma*, for 2001 and 2002. Unpublish Manuscript.

Barlough, J.E., E.S. Berry, E.A. Goodwin, R.F. Brown, R.L. DeLong, and A.W. Smith. 1987. Antibodies to Marine Caliciviruses in the Steller Sea Lion (*Eumetopias jubatus schreber*). J. Wildl. Dis. 23:34-44.

Barlow, J. 2006. Cetacean Abundance in Hawaiian Waters Estimated From a Summer/Fall Survey in 2002. Marine Mammal Science 22(2):446-464.

Barlow, J. and B.L. Taylor. 1998. Preliminary Abundance of Sperm Whales in the Northeastern Temperate Pacific Estimated from a Combined Visual and Acoustic Survey. Unpublished Report submitted to Int. Whal. Comm. (SC/50/CAWS/20). 19 p.

Barlow, J., and B.L. Taylor. 2005. Estimates of Sperm Whale Abundance in the Northeastern Temperate Pacific from a Combined Acoustic and Visual Survey. Marine Mammal Science 21(3):429-445.

Barlow, J., K. Forney, A. Von Saunder, and J. Urban-R. 1997. A Report of Cetacean Acoustic Detection and Dive Interval Studies (CADDIS) Conducted in the Southern Gulf Of California, 1995. NOAA Tech. Memo. NOAA-TM-NMFS-SWFSC-250.

Barrett-Lennard, L.G. 2000. Population Structure and Mating Patterns of Killer Whales (*Orcinus orca*) As Revealed by DNA Analysis. PhD thesis, University of British Columbia, 97 pp.

Barrett-Lennard, L.G., K. Heise, E. Saulitis, G. Ellis, and C. Matkin. 1995. The Impact of Killer Whale Predation on Steller Sea Lion Populations in British Columbia and Alaska. North Pacific Universities Marine Mammal Research Consortium, University of British Columbia, Vancouver, British Columbia. Unpublished report.

Barron, M.G., R. Heintz and M.M. Krahn. 2003. Contaminant Exposure and Effects in Pinnipeds: Implications for Steller Sea Lion Declines in Alaska. Science of the Total Environment 311:111-133.

Bartholomew, G.A. 1970. A Model for the Evolution of Pinniped Polygyny. Evolution 24:546-559.

Bartkiw, S., J. Boldt, P. Livingston, G. Walters, and J. Hoff. 2007. Indicators of Size Diversity in the Eastern Bering Sea. International Council for the Exploration of the Sea (ICES) Annual Science Conference 2007, Helsinki, Finland. ICES CM Documents 2007, Theme Session on Marine Biodiversity: A Fish and Fisheries Perspective (E). ICES CM 2007/E:21. http://www.ices.dk/products/CMdocs/CM-2007/E/E2107.pdf

Baumann, P., S.C. West. 1998. DNA end-joining catalyzed by human cell-free extracts. Proc Natl Acad Sci USA 95:14066-14070.

Bax, N.J. 1991. A Comparison of the Fish Biomass Flow to Fish, Fisheries, and Mammals in Six Marine Ecosystems. ICES Mar. Sci. Symp. 193:217-224.

Beck, C.A., L.D. Rea, S.J. Iverson, J.M. Kennish, K.W. Pitcher, and B.S. Fadely. 2007. Blubber Fatty Acid Profiles Reveal Regional, Seasonal, Sex and Age-Class Differences in the Diet of Young Steller Sea Lions in Alaska. Marine Ecology Progress Series 338:269-280.

Beckmen, K.B., J.E. Blake, G.M. Ylitalo, J.L. Stott, and T.M. O'Hara. 2003. Organochlorine Contaminant Exposure and Associations with Hematological and Humoral Immune Functional Assays with Dam Age as a Factor in Free-Ranging Northern Fur Seal Pups (*Callorhinus ursinus*). Marine Pollution Bulletin 46:594-606.

Beckmen, K.B., L.K. Duffy, X. Zhang, and K.W. Pitcher. 2002. Mercury Concentrations in the Fur of Steller Sea Lions and Northern Fur Seals from Alaska. Marine Pollution Bulletin 44(10):1130-1135.

Benson, A.J., A.W. Trites, 2002. Ecological Effects of Regime Shifts in the Bering Sea and Eastern North Pacific Ocean. Fish and Fisheries 3:95-113.

Berkson, J.M., and D.P. DeMaster. 1985. Use of Pup Counts in Indexing Population Changes in Pinnipeds. Can. J. Fish. Aquat. Sci. 42:873-879.

Berman, M.D. 2008. Endangered species, threatened fisheries: Science to the rescue! Evaluating the Congressionally designated Steller sea lion research program. Marine Policy 32:580-591.

Berzin, A.A. 1971. Kashalot. [The sperm whale.] [In Russian.] Izd. Pishchevaya Promyshlennost, Moscow, 367 pp. (Transl. by Israel Program Sci. Transl., 1972, 394 pp.) Available Natl. Tech. Inf. Serv., Springfield, Virginia as TT71-50152/50160.

Berzin, A.A. and A.A. Rovnin. 1966. Distribution and Migration of Whales in the Northeastern Part of the Pacific Ocean, Bering and Chukchi Seas. Izv. Tikhookean. Nauchno-issled, Inst. Rybn. Khoz. Okeanogr. (TINRO) 58:179-207. [In Russian]. Translated by U.S. Dep. Inter., Bur. Commer. Fish., Seattle, Washington, 1966, pp. 103-106 *In:* Panin, K.I. (Ed.) Soviet Research on Marine Mammals of the Far East.

Berzin, A.A., 1972. The Sperm Whale. V, 394 pp. Jerusalem: Israel Program for Scientific Translations Ltd. E. Hoz and Z. Blake. (Trans.).

Bianchi, G., H. Gislason, K. Graham, L. Hill, X. Jin, K. Koranteng, S. Manickchand-Heileman, I. Paya, K. Sainsbury, F. Sanchez, and K. Zwanenburg, 2000. Impact of Fishing on Size Composition and Diversity of Demersal Fish Communities. ICES J. Mar. Sci., 57:558-571.

Bickham, J.W., J.C. Patton, and T.R. Loughlin. 1996. High Variability for Control-Region Sequences in a Marine Mammal: Implications for Conservation and Biogeography of Steller Sea Lions (*Eumetopias jubatus*). J. Mammal. 77(1):95-108.

Bickham, J.W., T.R. Loughlin, D.G. Calkins, J.K. Wickliffe, and J.C. Patton. 1998b. Genetic Variability and Population Decline in Steller Sea Lions from the Gulf of Alaska. J. Mammal. 79:1390-1395.

Bickham, J.W., T.R. Loughlin, J.K. Wickliffe, and V.N. Burkanov. 1998a. Genetic Variation in the Mitochondrial DNA of Steller Sea Lions: Haplotype Diversity and Endemism in the Kuril Islands. Biosphere Conservation 1:107-117.

Bigg, M.A. 1985. Status of Steller Sea Lion (*Eumetopias jubatus*) And California Sea Lion (*Zalophus californianus*) in British Columbia. Canadian Special Publication of Fisheries and Aquatic Sciences 77:20.

Bigg, M.A. 1988. Status of the Steller sea lion, *Eumetopias jubatus*, in Canada. Canadian Field-Naturalist 102:315-336.

Bigg, M.A., Ellis, G.M., Ford, J.K.B., and Balcomb, K.C. 1987. Killer Whales, a Study of Their Identification, Genealogy and Natural History in British Columbia and Washington State. West Coast Whale Research Foundation and Phantom Press and Publishers Inc., Nanaimo, British Columbia, Canada.

Bishop, D.H., and J.F. Morado. 1995. Results on Blood Cell Morphology and Differential Blood Cell Counts from Seventeen Steller Sea Lion Eumetopias Jubatus Pups. Dis. Aquat. Organisms 23:1-6.

Blood, D.M. 2002. Low-Temperature Incubation of Walleye Pollock (*Theragra chalcogramma*) Eggs from the Southeast Bering Sea Shelf and Shelikof Strait, Gulf Of Alaska. Deep-Sea Research II 49:6095-6108.

Boldt, F.-M. J. Heinze, M. Diez, J. Petersen, M. Börsch. 2004. Real-Time Ph Microscopy Down to the Molecular Level by Combined Scanning Electrochemical Microscopy/Single Molecule Fluorescence Spectroscopy. Anal. Chem. 76:3473-3481 [PubMed].

Boldt, J. (Ed.). 2007. Ecosystem Considerations for 2008. Appendix C. Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Region. North Pacific Fishery Management Council, 605 West 4th Avenue, Suite 306, Anchorage, Alaska.

Boldt, J.L. (Ed.). 2004. Ecosystem Considerations for 2005. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Boldt, J.L. (Ed.). 2005a. Fisheries and the Environment: Ecosystem Indicators for the North Pacific and Their Implications for Stock Assessment. AFSC Processed Rep. 2005-04, 94 p. Alaska Fish. Sci. Cent., Natl. Mar. Fish. Serv., NOAA, 7600 Sand Point Way NE, Seattle Washington 98115.

Boldt, J.L. (Ed.). 2005b. Ecosystem Considerations for 2006. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Boldt, J.L, and E. Conners. 2004. Trends in Groundfish Biomass and Recruits per Spawning Biomass. In J.L. Boldt (Ed.) Ecosystem Considerations for 2005. Appendix C of the BSAI\GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, AK 99501.

Boldt, J.L., C.N. Rooper and M. Auburn-Cooke. 2005. The Diet of Juvenile Pacific Ocean Perch in the Aleutian Islands. Lowell-Wakefield Symposium on Biology, Assessment and Management of North Pacific Rockfishes, Anchorage, Alaska.

Boldt, J.L., S. Bartkiw, P.A. Livingston, J. Hoff, and G. Walters. 2008. Community Size Spectrum of the Bottom Trawl-Caught Fish Community of the Eastern Bering Sea. *In:* J.L. Boldt (Ed.) Ecosystem Considerations for 2009. Appendix C of the BSAI\GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Boldt, J. and S. Zador (Ed). 2009. Ecosystem Considerations for 2010. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Boldt, S., U.H. Weidle, W. Kolch. 2002. The Role of MAPK Pathways in the Action of Chemotherapeutic Drugs. *Carcinogenesis* 23:1831-1838.

Bond, N.A., J.E. Overland, M. Spillane, and P.J. Stabeno. 2003. Recent Shifts In The State Of The North Pacific. Geophys. Res. Lett. 30(23):2183. DOI: 10.1029/2003GL018597.

Boness, D.J. and W.D. Bowen. 1996. The Evolution of Maternal Care in Pinnipeds. Bioscience 46:645-654.

Boness, D.J., O.T. Oftedahl and K.A. Ono. 1991. The Effect Of El Nino On Pup Development In The California Sea Lion (Zalophus Californianus) I. early postnatal growth. Pages 173-179 in Trillmich, F. and K.A. Ono (Eds.) Pinnipeds and El NiZo: Responses to Environmental Stress. Springer-Verlag, Berlin.

Boor, G.K.H. 2010. A Poisson Bayesian analysis of opportunistic Steller sea lion at-sea observations from the platform of opportunity dataset. PhD. thesis, Montana State University. 36 p.

Borobia, M., P.J. Gearing, Y. Simard, J.N. Gearing, and P. Beland. 1995. Blubber fatty acids of finback and humpback whales from the Gulf of St. Lawrence. Mar. Biol. 122:341–353.

Bowen, W.D., J. Harwood, D. Goodman, and G.L. Swartzman. 2001. Review of the November 2000 Biological Opinion and Incidental Take Statement with Respect to the Western Stock of the Steller Sea Lion. North Pacific Fishery Management Council.

Bowles, E. 2009. Determining the relative amounts of prey in Steller sea lion (*Eumetopias jubatus*) diet using real-time PCR. M.S. Thesis, University of British Columbia, Vancouver, BC. 46 p.

Boyd, I.L. 1997. The Behavioural and Physiological Ecology of Diving. Trends in Ecology and Evolution 12:213-217.

Boyd, I.L. 2000. State-Dependent Fertility in Pinnipeds: Contrasting Capital and Income Breeders. Functional Ecology 14:623-630.

Boyd, I.L. and A.W.A. Murray. 2001. Monitoring a Marine Ecosystem Using Responses of Upper Trophic Level Predators. Journal of Animal Ecology 70:747-760.

Boyd, I.L. and T.S. McCann. 1989. Pre-Natal Investment in Reproduction by Female Antarctic Fur Seals. Behav. Ecol. Sociobiol. 24:377-385.

Boyd, I.L., J.P.Y. Arnould, T. Barton, and J.P. Croxall. 1994. Foraging Behaviour of Antarctic Fur Seals During Periods of Contrasting Prey Abundance. J. Anim. Ecol. 63:703-713.

Boyd, I.L. 2010 (in press). Assessing the effectiveness of conservation measures: resolving the “wicked” problem of the Steller sea lion. Biological Conservation. 40 p.

Bradshaw, C.J.A., L.S. Davis, C. Lalas, and R.G. Harcourt. 2000. Geographic and Temporal Variation in the Condition of Pups of The New Zealand Fur Seal (*Arctocephalus forsteri*): Evidence for Density Dependence and Differences in the Marine Environment. J. Zool., Lond. 252:41-51.

Braham, H.W., G.W. Oliver, C. Fowler, K. Frost, F. Fay, C. Cowles, D. Costa, K. Schneider, and D. Calkins. 1982. Marine Mammals. Pages 55-81 *In:* M.J. Hameedi (Ed.) The St. George Basin Environment and Possible Consequences of Planned Offshore Oil and Gas Development. Alaska OCSEAP, Juneau, Alaska.

Braham, H.W., R.D. Everitt, and D.J. Rugh. 1980. Northern Sea Lion Decline in the Eastern Aleutian Islands. J. of Wildl. Management 44:25-33.

Braham, H.W., R.D. Everitt, B.D. Krogman, D.J. Rugh, and D.E. Withrow. 1977. Marine Mammals of the Bering Sea: A Preliminary Report on Distribution and Abundance, 1975-76. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest and Alaska Fisheries Center, Seattle, Washington 98112. 90 p.

Brander, K.M. 2003. What Kinds of Fish Stock Predictions Do We Need and What Kinds of Information Will Help Us to Make Better Predictions? Scientia Marina 67:21-33.

Brandon, A.A. and R.W. Davis.1999. Neonatal Growth and Condition and Female Attendance Patterns. Steller Sea Lion Research Peer Review Physiology Workshop.

Brandon, E.A.A. (2000). Maternal Investment in Steller Sea Lions in Alaska. Ph.D. dissertation, Texas A & M University, Galveston.

Brandon, E.A.A., D.G. Calkins, T.R. Loughlin, and R.W. Davis. 2005. Neonatal Growth of Steller Sea Lion (*Eumetopias jubatus*) Pups in Alaska. Fish Bulletin 103:246-257.

Briggs, H.B., D.G. Calkins, and R.W. Davis. 2005. Habitat Associations and Diving Patterns of Juvenile Steller Sea Lions in The North-Central Gulf of Alaska. Pages 59-67 *In:* T.R. Loughlin, S. Atkinson and D.G. Calkins, (Eds.) Synopsis of research on Steller sea lions: 2001-2005. Alaska SeaLife Center’s Steller Sea Lion Research Program, Seward, Alaska.

Bright, D.B. 1959. The Occurrence and Food of the Sleeper Shark, *Somniosus pacificus*, in a Central Alaska Bay. Copeia. pp. 76-77.

Brock, V.E., and R.N. Riffenburgh. 1960. Fish Schooling: a Possible Factor in Reducing Predation. Extrait du Journal du Conseil, Conseil International pour l'Exploration de la Mer 25:307-317.

Brodeur, R.D., and D.M. Ware. 1992. Interannual and Interdecadal Changes in Zooplankton Biomass in the Subarctic Pacific Ocean. Fish. Oceanogr. 1:32-38.

Brownell, R.L., Jr., A.V. Yablokov and V.A. Zemmsky. 1998. USSR Pelagic Catches of North Pacific Sperm Whales, 1949-1979: Conservation Implications. Paper SC/50/CAWS27 presented to the International Whaling Commission, June 1998 (unpublished).

Brownell, R.L.J., K.M. Robertson, S. Kang, and A.E. Dizon. 2000. Molecular Identification of Cetacean Products from the Markets of Korea, 1997 and 1999. Unpublished report to the Scientific Committee of the International Whaling Commission (SC/52/SD6). Available from the International Whaling Commission, The Red House, 135 Station Road, Impington, Cambridge CB4 9NP, United Kingdom.

Brueggeman, J.J., L.A. Green, R.A. Grotefendt, and D.G. Chapman. 1987. Aerial Surveys of Endangered Cetaceans and Other Marine Mammals in the Northwestern Gulf of Alaska and Southeastern Bering Sea. *In:* NOAA/OCSEAP Environmental Assessment of the Alaskan Continental Shelf. Final Reports of Principal Investigators 61:1-124.

Bryant, B.C. 2009. Adapting to uncertainty: Law, science, and management in the Steller sea lion controversy. Stanford Environmental Law Journal 28:171-211.

Buck, E.H. 1973. Alaska and the Law of the Sea. National Patterns and trends of fishery development in the North Pacific. University of Alaska, Arctic Environmental Information and Data Center, Anchorage, Alaska. 65 p.

Burbacher, T.M., C. Monnett, K.S. Grant, and, N.K. Mottet, 1984. Methylmercury Exposure and Reproductive Dysfunction in the Nonhuman Primate. To.vicol Appl. Pharmacol. 75:18-24.

Burbacher, T.M., M.K. Mohammed, N.K. Mottett. 1988. Methylmercury effects on reproduction and offspring size at birth. Reproductive Toxicology l:267-2 78.

Burdin, A.M., D.R. Hennen, D.G., Calkins, V.V. Burkanov, V.S. Nikulin, and T.Y. Lisitsina. In press. Can we see a cohort effect on survival of Steller sea lions (*Eumetopias jubatus*) at Kozlova Cape rookery (eastern Kamchatka, Russia)? Mar. Mamm. Sci.. 14 p.

Burek, K.A., F.M.D. Gulland, G. Sheffield, D. Calkins, E. Keyes, T.R. Spraker, A.W. Smith, D.E. Skilling, J. Evermann, J.L. Stott and A.W. Trites. 2003. Disease Agents in Steller Sea Lions in Alaska: A Review and Analysis of Serology Data from 1975-2000. Fisheries Centre Reports 11(4):26 pp.

Burek, P., R. Hoehndorf, F. Loebe, J. Visagie, H. Herre, and J. Kelso. 2006. A Top-Level Ontology of Functions and Its Application in the Open Biomedical Ontologies. Bioinformatics 22(14):e66-e73.

Burkanov, V.N., and T.R. Loughlin. 2005. Historical Distribution and Abundance of Steller Sea Lions on the Asian Coast. Marine Fisheries Review. 67(2):1-62.

Burkanov, V.N., V.V. Vertyankin, and E.G. Mamaev. 1997. Migration of Sea Lions on North-West Pacific. Page 15 *In:* Migratory Ecology of Steller Sea Lions in the Far East Waters. Todo Symposium, Hokkaido University. Abstract only.

Burkanov, V. 2010. Russian SSL studies update. Presentation to NPFMC's Steller Sea Lion Mitigation Committee, January 2010, Seattle, WA. Available at http://www.fakr.noaa.gov/npfmc/current_issues/ssl/JanuarySSL_mtg2010/BurkanovRussiastudies.pdf

Byrd, G.V. 1989. Observations of Northern Sea Lions at Ugamak Island, Buldir, and Agattu Islands, Alaska in 1989. Unpublished report, U.S. Fish and Wildlife Service. Alaska Maritime National Wildlife Refuge.

Calambokidis, J., and J. Barlow. 2004. Abundance of Blue and Humpback Whales in the Eastern North Pacific Estimated by Capture-Recapture and Line-Transect Methods. Marine Mammal Science 20(1):63-85.

Calambokidis, J., E.A. Falcone, T.J. Quinn, A.M. Burdin, P.J. Clapham, J.K.B. Ford, C.M. Gabriele, R. LeDuc, D. Mattila, L. Rojas-Bracho, J.M. Straley, B.L. Taylor, J. Urbán R., D. Weller, B.H. Witteveen, M. Yamaguchi, A. Bendlin, D. Camacho, K. Flynn, A. Havron, J. Huggins, and N. Maloney. 2008. SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific. Final Report for Contract AB133F-03-RP-00078 for NOAA Fisheries, Seattle, Washington. 57 pp.

Calambokidis, J., G.H. Steiger, and J.R. Evenson. 1993. Photographic Identification and Abundance Estimates of Humpback and Blue Whales Off California in 1991-92. Final Contract Report 50ABNF100137 to Southwest Fisheries Science Center, P.O. Box 271, La Jolla, California 92038. 67 pp.

Calambokidis, J., G.H. Steiger, J.C. Cubbage, K.C. Balcomb III, and P. Bloedel. 1989. Biology Of Humpback Whales in the Gulf of the Farallones. Report to Gulf of the Farallones National Marine Sanctuary, San Francisco, California by Cascadia Research Collective, 218½ West Fourth Avenue, Olympia, Washington. 93 pp.

Calambokidis, J., G.H. Steiger, J.M. Straley, L.M. Herman, S. Cerchio, D.R. Salden, J. Urbán R., J.K. Jacobsen, O. von Ziegesar, K.C. Balcomb, C.M. Gabriele, M.E. Dahlheim, S. Uchida, G. Ellis, Y. Miyamura, P. Ladrón de Guevara, P., M. Yamaguchi, F. Sato, S.A. Mizroch, L. Schlender, K. Rasmussen, J. Barlow and T.J. Quinn II. 2001. Movements and Population 19 Structure of Humpback Whales in the North Pacific. Marine Mammal Science 17(4):769-794.

Calambokidis, J., G.H. Steiger, J.M. Straley, T. Quinn, L.M. Herman, S. Cerchio, D.R. Salden, M. Yamaguchi, F. Sato, J.R. Urban, J. Jacobson, O. Von Zeigesar, K.C. Balcomb, C.M. Gabriele, M.E. Dahlheim, N. Higashi, S. Uchida, J.K.B. Ford, Y. Miyamura, P. Ladrón de Guevara, S.A. Mizroch, L. Schlender, and K. Rasmussen. 1997. Abundance and Population Structure of Humpback Whales in the North Pacific Basin. Final Contract Report 50ABNF500113 to Southwest Fisheries Science Center, P.O. Box 271, La Jolla, California 92038. 72 pp.

Calkins, D.G. (1998). Prey of Steller Sea Lions in the Bering Sea. Biosphere Conservation, 1(1):33-44.

Calkins, D.G. 1985. Steller Sea Lion Entanglement in Marine Debris. Proceedings of the workshop on the fate and impact of marine debris, 26-29 November 1985, Honolulu, Hawaii. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFC-54.

Calkins, D.G. 1986. Sea Lion Investigations in Southern Alaska. Final report to the National Marine Fisheries Service, Alaska Region, Contract 81-ABC-00280. AK Dep. Fish and Game, Anchorage, AK. 23 pp.

Calkins, D.G. 1988. Marine Mammals. Pages 527-558, *In:* D.W. Hood and S.T. Zimmerman (Eds.) The Gulf of Alaska: Physical Environment and Biological Resources. NOAA Ocean Assessments Div., Anchorage, Alaska.

Calkins, D.G. 1996. Movements and Habitat Use of Female Steller Sea Lions in Southeastern Alaska. Pages 110-134, 166 *In:* Steller Sea Lion Recovery Investigations in Alaska, 1992-1994. Report from ADF&G, Juneau, Alaska to NOAA, Wildlife Technical Bulletin 13, May 1996.

Calkins, D.G. 2006. Fixed gear-marine mammal interactions study. Final Report for NOAA Fisheries, Juneau, AK. Report from Texas A&M University Research Foundation prepared by North Pacific Wildlife Consulting, LLC.

Calkins, D.G., and E. Goodwin. 1988. Investigation of the Declining Sea Lion Population in the Gulf Of Alaska. Alaska Department of Fish and Game, 333 Raspberry Road, Anchorage, Alaska 99518. 76 pp.

Calkins, D.G., and K.W. Pitcher. 1982. Population Assessment, Ecology and Trophic Relationships of Steller Sea Lions in the Gulf of Alaska. U.S. Dep. of Commer., NOAA. OCSEAP Final Report 19 1983, pp. 445-546.

Calkins, D.G., D.C. McAllister, and K.W. Pitcher. 1999. Steller Sea Lion Status and Trend in Southeast Alaska: 1979-1997. Marine Mammal Science 15: 462-477.

Calkins, D.G., E. Becker, T.R. Spraker, and T.R. Loughlin. 1994. Impacts on Steller Sea Lions. Pages 119-139 *In:* T.R. Loughlin (Ed.) Marine Mammals and the Exxon Valdez. Academic Press, N.Y.

Calkins, D.G., E.F. Becker, and K.W. Pitcher. 1998. Reduced Body Size of Female Steller Sea Lions from a Declining Population in the Gulf of Alaska. Mar. Mamm. Sci. 14:232-244.

Calkins, D.G., M. Castellini, V. Burkanov, S. Atkinson, S. Inglis, and D. Hennen. 2005. Impact of Changing Diet Regimes in Steller Sea Lion Body Condition. Pages 6-18 *In:* T.R. Loughlin, D.G. Calkins, and S. Atkinson (Eds.) Synopsis of Research on Steller Sea Lions: 2001-2005. Alaska Sea Life Center's Steller Sea Lion Research Program. Alaska Sea Life Center, Homer, Alaska.

Calkins, D.G., J.A. Mellish, S. Atkinson, and D. Hennen. 2005. Does consuming pollock truly have negative impacts on free-ranging Steller sea lions? In: 16th Biennial Conference on the Biology of Marine Mammals, December 12-16, San Diego, CA.

Calkins, M.J., R.J. Jakel, D.A. Johnson, K. Chan, Y.W. Kan, and J.A. Johnson. 2005. Protection from Mitochondrial Complex II Inhibition in Vitro And in Vivo by Nrf2-Mediated Transcription. Proc Natl Acad Sci U.S.A. 102:244-249.

Call, K.A., and T.R. Loughlin. 2005. An Ecological Classification of Alaskan Steller Sea Lion (*Eumetopias jubatus*) Rookeries: A Tool For Conservation/Management. Fish Oceanogr. 14:212-222 Suppl. 1.

Call, K.A., B.S. Fadely, A. Greig, and M.J. Rehberg. 2007. At-Sea and On-Shore Cycles of Juvenile Steller Sea Lions (Eumetopias Jubatus) Derived from Satellite Dive Recorders: A Comparison Between Declining and Increasing Populations. Deep-Sea Res. II 54:298-310.

Campbell, R.A., B.L. Chilvers, S. Childerhouse, and N.J. Gales. 2006. Conservation Management Issues and Status of the New Zealand (*Phocarctos hookerii*) and Australian (*Neophoca cinerea*) Sea Lion. Pages 455-469 in Trites, A.W., S.K. Atkinson, D.P. DeMaster, L.W. Fritz, T.S. Gelatt, L.D. Rea and K.M. Wynne. Sea lions of the world. Alaska Sea Grant, Fairbanks.

Carretta, J.V. 2002. Preliminary Estimates of Cetacean Mortality in California Gillnet Fisheries for 2001. Paper SC/54/SM12 presented to the International Whaling Commission (Unpublished). 22:57

Carretta, J.V. and S.J. Chivers. 2003. Preliminary Estimates of Marine Mammal Mortality and Biological Sampling if Cetaceans in California Gillnet Fisheries for 2002. Paper SC/55/SM3 presented to the International Whaling Commission (unpublished). 21pp.

Carretta, J.V. and S.J. Chivers. 2004. Preliminary Estimates of Marine Mammal Mortality and Biological Sampling of Cetaceans in California Gillnet Fisheries for 2003. Paper SC/56/SM1 presented to the IWC Scientific Committee, June 2004 (Unpublished). Available from Southwest Fisheries Science Center, National Marine Fisheries Service, 8604 La Jolla Shores Drive, La Jolla, California 92037, USA.

Carretta, J.V., K.A. Forney, M.S. Lowry, J. Barlow, J. Baker, D, Johnston, B. Hanson, M.M. Muto, D. Lynch, and L. Carswell. 2009. U.S. Pacific Marine Mammal Stock Assessments: 2008. U.S. Department of Commerce Technical Memorandum, NOAATM-NMFS-SWFSC-434, 340 p.

Castellini, J.M., K. Beckmen, L. Rea, and T. O'Hara. 2009. Mercury and Selenium Levels in Steller Sea Lion Pups in Alaska. Poster Presentation, Society for Marine Mammalogy 18th Biennial Conference, Quebec City, Canada.

Castellini, M., R.W. Davis, T.R. Loughlin, and T.M. Williams. 1993. Blood Chemistries and Body Condition of Steller Sea Lion Pups at Marmot Island, Alaska. Marine Mammal Science 9:202-208.

Castellini, M.A. 1991. The Biology of Diving Mammals: Behavioral, Physiological, and Biochemical Limits. Advances in Comparative and Environmental Physiology, 8:105-134.

Castellini, M.A. 1999. Assessing Heavy Metals in Populations of Marine Mammals. EPA Symposium on Western Ecological Systems. San Francisco, California. April, 1999.

Castellini, M.A. 2002. Captive Studies With Steller Sea Lions at the Alaska Sealife Center. *In:* Steller Sea Lion Decline: Is It Food II, Vol. AK-SG-02-02. D. DeMaster and S. Atkinson. (Eds.) University of Alaska Sea Grant, Fairbanks, Alaska. pp. 80.

Castellini, M.A., and D.G. Calkins. 1993. Mass Estimates Using Body Morphology in Steller Sea Lions. Marine Mammal Science, 9:48-54.

Causey, D., D.G. Corbett, C. Lefevre, D.L. West, A.B. Savinetsky, N.K. Kiseleva, and B.F. Khassanov. 2005. The Palaeoenvironment of Humans and Marine Birds of the Aleutian Islands: Three Millennia of Change. Fisheries Oceanography 14:s1, 259-276.

Chambellant, M., G. Beauplet, C. Guinet, and J-Y. Georges. 2003. Long-Term Evaluation of Pup Growth and Preweaning Survival Rates in Subantarctic Fur Seals, *Arctocephalus tropicalis*, on Amsterdam Island. Canadian Journal of Zoology 81:1222-1232.

Chiba, I., A. Sakakibara, Y. Goto, T. Isono, Y. Yamamoto, H. Iwata, *et al.* 2001. Negative Correlation Between Plasma Thyroid Hormone Levels and Chlorinated Hydrocarbon Levels Accumulated in Seals from the Coast of Hokkaido, Japan. Environ. Toxicol. Chem. 20:1092-1097.

Chilvers, B.L., I.S. Wilkinson, P.J. Duignan, and N.J. Gemmel. 2005. Summer foraging areas for lactating New Zealand sea lions (*Phocarctos hookeri*). Mar. Ecol. Prog. Ser. 304:235-247.

Christensen, J.R., M. MacDuffee, R.W. Macdonald, M. Whiticar, and P.S. Ross. 2005. Persistent Organic Pollutants in British Columbia Grizzly Bears: Consequence of Divergent Diets. Environ. Sci. Technol. 39:6952-6960.

Chumbley, L., J. Sease, M. Strick, and R. Towell. 1997. Field Studies of Steller Sea Lions (*Eumetopias jubatus*) at Marmot Island, Alaska 1979 through 1994. NOAA Tech. Mem. NMFS-AFSC-77.

Chumbley, K. 2007. Prey biomass abundance, distribution, and availability to the endangered Steller sea lion (*Eumetopias jubatus*) population at Ugamak Island, Alaska, 1995-1999. M.E.S. thesis, The Evergreen State College. 119 p.

Christensen, I., T. Haug, and N. 0ien. 1992. A review of feeding and reproduction in large baleen whales (Mysticeti) and sperm whales *Phvseter macrocephalus* in Norwegian adjacent waters. *Fauna norvegica Series A* 13:39–48.

Ciannelli , L., K.S. Chan, K.M. Bailey, and N.C. Stenseth. 2004. Non-Additive Effects of Environmental Variables on the Survival of a Large Marine Fish Population. Ecology 85:3418-3427.

Ciannelli, L., K.M. Bailey, K.S. Chan, and N.C. Stenseth. 2007. Phenological and Geographical Patterns of Walleye Pollock Spawning in the Western Gulf of Alaska. Can. J. Fish. Aquat. Sci. 64:713-722.

Ciannelli, L., K.M. Bailey, N.C. Stenseth, K.S. Chan, and A. Belgrano. 2005. Climate Change Causing Phase Transition of Walleye Pollock (*Theragra chalcogramma*) Recruitment Dynamics. Proc. Royal Soc. B. 272:1735-1743.

Clark, W.G. 1991. Groundfish Exploitation Rates Based on Life History Parameters. Can. J. Fish. Aquat. Sci. 48:734-750.

Clark, W.G. 1993. The Effect of Recruitment Variability on the Choice of a Target Level of Spawning Stock Biomass Per Recruit. Pages 233-246 *In:* G. Kruse, D.M. Eggers, R.J. Marasco, C. Pautzke, and T.J. Quinn II, (Eds.) Proceedings of the International Symposium on Management Strategies for Exploited Fish Populations. Alaska Sea Grant College Program Report 93-02. University of Alaska, Fairbanks, Alaska, USA.

Clark, W.G., S.R. Hare, A.M. Parma, P.J. Sullican, R.J. Trumble. 1999. Decadal Changes in Growth and Recruitment of Pacific Halibut (*Hippoglossus stenolepis*). Can J Fish Aquat Sci 56:242-252.

Coffing, M., C.L. Scott, and C.J Utermole. 1998. The Subsistence Harvest of Seals and Sea Lions by Alaska Natives in Three Communities of the Yukon-Kuskokwim Delta, Alaska, 1997-98. ADF&G, Juneau, AK, Subsistence Div. Tech. Paper No. 255. 48 pp.

Comparative Physiology B. 176:589-598.

Conners, M.E. and P. Munro. 2008. Effects of Commercial Fishing on Local Abundance of Pacific Cod (*Gadus macrocephalus*) in the Bering Sea. Fishery Bulletin 106:281-292

Conners, M.E., A.B. Hollowed and E. Brown. 2002. Retrospective Analysis of Bering Sea Bottom Trawl Surveys: Regime Shift and Ecosystem Reorganization. Progress in Oceanography 55:209-222.

Conners, M.E., P. Munro, and S. Neidetcher. 2004. Pacific Cod Pot Studies 2002-2003. AFSC Processed Rep. 2004-04, 64 p.+ Appendix. Alaska Fish. Sci. Cent., Natl. Mar. Fish. Serv., NOAA, 7600 Sand Point Way NE, Seattle, Washington 98115.

Cooney, R.T. 1993. A Theoretical Evaluation of the Carrying Capacity of Prince William Sound, Alaska, for Juvenile Pacific Salmon Fish. Res. 18:77-87.

Costa, D.P. 1993. The Relationship Between Reproductive and Foraging Energetics and the Evolution of the Pinnipedia. *In:* I.L. Boyd (Ed.) Recent Advances in Marine Mammal Science. Zoological Society of London, Oxford University Press.

Costa, D.P., G.A. Antonelis, and R.L. DeLong. 1991. Effects of El NiZo on the foraging energetics of the California sea lion. Pages 156-165 *In:* F. Trillmich, and K.A. Ono (Eds.) Pinnipeds and El NiZo: Responses to Environmental Stress. Springer-Verlag, Berlin.

Csepp, D.J., J.J. Vollenweider, and M.F. Sigler. 2010. Seasonal abundance and distribution of potential prey species for marine predators in southeastern Alaska. Manuscript. 44 p.

Cunningham, K.M., M.F. Canino, I.B. Spies, L. Hauser. 2009. Genetic Isolation by Distance and Localized Fjord Population Structure in Pacific Cod (*Gadus Macrocephalus*): Limited Effective Dispersal in the Northeastern Pacific Ocean. Can J Fish Aquat Sci 66:153-166.

Cunningham, P.M. 2007. A Sensitivity Analysis of an Individual-Based Model. M.S. thesis, Humboldt State University, Arcata, California.

Cury, P.M., and V. Christensen. 2005. Quantitative Ecosystem Indicators for Fisheries Management: An Introduction. ICES J. Mar. Sci. 62:307-310.

D'Vincent, C.G., R.M. Nilson, and R.E. Hanna. 1985. Vocalization and Coordinated Feeding Behavior of the Humpback Whale in South-Eastern Alaska. Scientific Reports of the Whales Research Institute, 36, 41-47.

Dabin, W., G. Beauplet, E.A. Crespo, and C. Guinet. 2004. Age Structure, Growth, and Demographic Parameters in Breeding-Age Female Subantarctic Fur Seals, Arctocephalus Tropicalis. Canadian Journal of Zoology 82:1043-1050.

Daniel, D.O., and J.C. Schneeweis. 1992. Steller Sea Lion, Eumetopias Jubatus, Predation on Glaucous-Winged Gulls, *Larus glaucescens*. Can. Field-Natur. 106:268.

Daniel, R.G. 2003. The Timing of Moulting in Wild and Captive Steller Sea Lions (*Eumetopias jubatus*). M.Sc. thesis, University of British Columbia, Vancouver, British Columbia, Canada. 64 pp.

Dans, S.L., E.A. Crespo, S.N. Pedraza, and M.K. Alonso. 2004. Recovery of the South American Sea Lion (*Otaria flavescens*) Population in Northern Patagonia. Canadian Journal of Aquatic Science 61:1681-1690.

Darling, R.W.R. 1991. Estimating Probabilities of Hurricane Wind Speeds Using a Large-Scale Empirical Model. J. Climate, 4:1035-1046.

Darroch, J.N. 1961. The Two-Sample Capture-Recapture Census When Tagging and Sampling are Stratified. Biometrika 48:241-260.

Davis, M.W. 2001. Behavioral Responses of Walleye Pollock, Theragra Chalcogramma, Larvae to Experimental Gradients of Sea Water Flow: Implications for Vertical Distribution. Environ. Biol. Fish. 61, 253-260.

Davis, R.W., A.A. Brandon, D. Calkins, and T.R. Loughlin. 2004. Indices of Reproductive Effort and Nutritional Health in Lactating Steller Sea Lions and Pups in Areas of Declining and Stable Population. 22nd Wakefield Fisheries Symposium, Sea Lions of the World, Anchorage, Alaska. Sept 30-Oct 3, 2004. Alaska Sea Grant College Program, Fairbanks. p. 36.

Davis, R.W., A.A. Brandon, T.C. Adams, T.M. Williams, M.A. Castellini, T.R. Loughlin, and D.G. Calkins. 1996. Indices of Reproductive Effort, Body Condition and Pup Growth for Steller Sea Lions (*Eumetopias jubatus*) in Alaska. Wildlife Technical Bulletin, 13, ADF&G, Division of Wildlife Conservation, Anchorage, Alaska. pp. 53-61.

Davis, R.W., E.A.A. Brandon, D.G. Calkins, and T.R. Loughlin. 2006. Female Attendance and Neonatal Pup Growth in Steller Sea Lions (*Eumetopias jubatus*). pp. 13-29. *In:* A.W. Trites, S.K. Atkinson, D.P. DeMaster, L.W. Fritz, T.S. Gelatt, L.D. Rea, and K.M. Wynne (Eds.) Sea Lions of the World. Alaska Sea Grant Program Report AK-SG-06-01, University of Alaska, Fairbanks.

Davis, R.W., G.S. Fargion, N. May, T.D. Leming, M. Baumgartner, W.E. Evans, L.J. Hansen, K.D. Mullin. 1998. Physical Habitat of Cetaceans Along the Continental Slope in the North-Central and Western Gulf of Mexico. Marine Mammal Science 14, 490-507.

de Swart, R.L., P.S. Ross, J.G. Vos, and A.D.M.E. Osterhaus. 1996. Impaired Immunity in Harbor Seals (*Phoca vitulina*) Exposed to Bioaccumulated Environmental Contaminants: Review of a Long Term Feeding Study. Environmental Health Perspectives, 104, pp. 823-828.

de Swart, R.L., P.S. Ross, L.J. Vedder, H.H. Timmerman, S.H. Heisterkamp, H. van Loveren, J.G. Vos, P.J. H. Reijnders, and A.D.M.E. Osterhaus. 1994. Impairment of Immune Function in Harbour Seals (*Phoca vitulina*) Feeding on Fish from Polluted Waters. Ambio 23:155-159.

de Wit, C.A. 2002. An Overview of Brominated Flame Retardants in the Environment. Chemosphere 46:583-624.

Debier, C., G.M. Ylitalo, M. Weise, F. Gulland, D.P. Costa, B.J. Le Boeuf, T. de Tillesse, and Y. Larondelle. 2005. Pcbs and DDT in the Serum of Juvenile California Sea Lions: Associations with Vitamins A and E and Thyroid Hormones. Environ. Pollut. 134:323-332.

DeLong, R.L., W.G. Gilmartin, and J.G. Simpson. 1973. Premature Births in California Sea Lions: Association with High Organochlorine Pollutant Residue Levels. Science, 81:1168-1169.

DeMaster, D. 2009. Aerial survey of Steller sea lions in Alaska, June-July 2009 and update on the status of the western stock in Alaska. Memorandum to D. Mecum, K. Brix, and L. Rotterman, NMFS AKR PR. December 2, 2009. 2009. 30 p.

DeMaster, D.J., C.W. Fowler, S.L. Perry, and M.E. Richlen. 2001. Predation and Competition: The Impact of Fisheries on Marine Mammal Populations Over the Next One Hundred Years. Journal of Mammalogy 82:641-651.

DeMaster, D.P., A.W. Trites, P. Clapham, S. Mizroch, P. Wade, R.J. Small, and J. Ver Hoef. 2006. The Sequential Megafaunal Collapse Hypothesis: Testing with Existing Data. Prog. Oceanogr. 68:329-342.

Di Natale, A. and G. Notarbartolo di Sciara. 1994. A Review of the Passive Fishing Nets and Trap Fisheries in the Mediterranean Sea and of Cetacean Bycatch. Report International Whaling Commission, Special Issue, 15, 189-202.

Dietz, R., M.P. Heide-Jorgenson, and T. Harkonen. 1989. Mass Death of Harbour Seals Phoca Vitulina in Europe. Ambio, 18:258-64.

Dillingham, P.W., J.R. Skalski, and K.E. Ryding. 2006. Fine-Scale Geographic Interactions Between Steller Sea Lion (*Eumetopias jubatus*) Trends and Local Fisheries. Can. J. Fish. Aquat. Sci. 63:107-119.

Dixon, R.G. 1986. Alaska Division of Geological and Geophysical Surveys, July 1986, public data file 86-79. Alaska Department of Natural Resources, 3601 C Street, PO Box 7028, Anchorage Alaska 99510, Chiniak Bay Archaeological Investigation June 1986, The KOD-350 Site.

Dolphin, W.F. 1987. Ventilation and Dive Patterns of Humpback Whales *Megaptera novaeangliae*, on Their Alaskan Feeding Grounds. Can. J. Zoo. 65:83-90.

Donovan, G.P. 1991. A Review of IWC Stock Boundaries. Report of the International Whaling Commission. Special Issue 13:39-63.

Dorn, M., K. Aydin, S. Barbeaux, M. Guttormsen, B. Megrey, K. Spalinger, and M. Wilkins. 2005. Assessment of Walleye Pollock in the Gulf of Alaska for 2006. North Pacific Fishery Management Council, 605 West 4th Ave., Suite 306, Anchorage, Alaska 99501.

Dorn, M.W. 2002. Advice on West Coast Rockfish Harvest Rates From Bayesian Meta-Analysis of Stock-Recruit Relationships. NAJFM 22:280-300. Martin

Dorn, M.W., K. Aydin, S. Barbeaux, B.M. Guttormsen, B. Megrey, K. Spalinger, and M. Wilkins. 2003. Assessment of the Walleye Pollock Stock in the Gulf of Alaska. *In:* Stock Assessment and Fishery Evaluation Report for Groundfish Resources of the Gulf of Alaska.

Dorn, W., K. Dethloff, A. Rinke. *et al*. 2007. Sensitivities and Uncertainties in a Coupled Regional Atmosphere-Ocean-Ice Model with Respect to the Simulation of Arctic Sea Ice. J Geophys Res; 112: D10118.

Duck, C.D. 1990. Annual Variation in the Timing of Reproduction in Antarctic Fur Seals, Arctocephalus Gazella, at Bird Island, South Georgia. J. Zool., Lond. 222:103-116.

Duffy-Anderson, J.T., K.M. Bailey, and L. Cianelli. 2002. Consequences of a Superabundance of Larval Walleye Pollock, Theragra Chalcogramma, in the Gulf of Alaska in 1981. Mar. Ecol. Prog. Ser. 243:179-190.

Duffy-Anderson, J.T., K.M. Bailey, L. Ciannelli, P. Cury, A. Belgrano, and N.C. Stenseth. 2005. Phase Transitions and Climate-Environmental Variability in Marine Fish Recruitment Processes. Ecol. Complexity. 2(3):143-169.

Duplisea, D.E., S.R. Kerr, and L.M. Dickie. 1997. Demersal Fish Biomass Size Spectra on the Scotian Shelf, Canada; Species replacement at the shelf wide scale. Can. J. Fish. Aquat. Sci. 54:1725-1735.

Durban, J., D. Ellifrit, M. Dahlheim, J. Waite, C. Matkin, L. Barrett-Lennard, G. Ellis, R. Pitman, R. LeDuc, and P. Wade. In Press (2010). Photographic mark-recapture analysis of clustered mammal-eating killer whales around the Aleutian Islands and Gulf of Alaska. Marine Biology. 25 p.

E.E. Holmes, L.W. Fritz, A.E. York, K. Sweeney. 2007. Age-Structured Modeling Reveals Long-Term Declines in the Natality of Western Steller Sea Lions. Ecological Applications 17(8):2214-2232.

Ebbesmeyer, C.C., D.R. Cayan, D.R. McLain, F.H. Nichols, D.H. Person, and K.T. Redmond. 1991. 1976 Step in The Pacific Climate: Forty Environmental Changes Between 1968-1975 and 1977-1984. Proceedings of the Seventh Annual Pacific Climate (PACLIM) Workshop, April, 1990, California Dept. of Water Resources, Interagency Ecological Studies Program Technical Report 26.

Edds, P.L. and J.A.F. Macfarlane. 1987. Occurrence and general behavior of balaenopterid cetaceans summering in the St. Lawrence Estuary, Canada. Can. J. Zool. 65:1363–1376.

Edie, A.G. 1977. Distribution and Movements of Steller Sea Lion Cows (*Eumetopias jubatus*) on a Pupping Colony. Unpublished. M.S. thesis, Univ. British Columbia, Vancouver. 81 pp.

Elliott, H.W. The Seal Islands of Alaska. Kingston, Ontario: The Limestone Press, 1976 [1881].

Estes, J.A., D.P. DeMaster, D.F. Doak, T.M. Williams, and R.L. Brownell, Jr. (Eds.). 2006. Whales, Whaling, and Ocean Ecosystems. University of California Press, Berkeley.

Estes, J.A., M.T. Tinker, T.M. Williams, and D.F. Doak. 1998. Killer Whale Predation on Sea Otters Linking Oceanic and Nearshore Ecosystems. Science 282:473-476.

Estes, J.A., D.F. Doak, A.M. Springer, T.M. Williams, and G.B. Van Vliet. 2009. Trend data *do* support the sequential nature of pinniped and sea otter declines in the North Pacific Ocean, *but* does it really matter? Mar. Mamm. Sci. 25(3):748-754.

Evans, G.T., and J.S. Parslow. 1985. A Model of Annual Plankton Cycles. Biological Oceanography 3(3):327-347.

Fadely, B., B.W. Robson, J.T. Sterling, A. Greig, and K.A. Call. 2005. Immature Steller Sea Lion (*Eumetopias jubatus*) Dive Activity in Relation to Habitat Features of the Eastern and Central Aleutian Islands. Fish. Oceanogr. 14(s1):243-258.

Fadely, B.S., and T.R. Loughlin. 2001. Weak Association Between Steller Sea Lion Pup Condition and Population and Environmental Trends in Western Alaska. Abstract submitted to 14th Biennial Conference on the Biology of Marine Mammals, Vancouver, Canada.

Fadely, B.S., J.A. Zeligs, and D.P. Costa. 1994. Assimilation Efficiencies and Maintenance Requirements of California Sea Lions (*Zalophus californianus*), Walleye Pollock (*Theragra chalcogramma*) and Herring (*Clupea harengus*). Final Report to the National Marine Mammal Laboratory. p. 29.

Fadely, B.S., J.N. Ianelli, A. Grieg, M.W. Dorn, E.A. Loggerwell. 2010. Updated Forage Ratio Estimates for 2000 and 2008. October 14, 2010, 6 pp. Unpublished report, available Alaska Fisheries Science Center, NMFS, NOAA, 7600 Sand Point Way, NE, Seattle, Washington 98115.

Fadely, B.S., T.S. Gelatt, L.D. Rea, J.C. King, and T.R. Loughlin. 2004. Geographic Differences Among Juvenile Steller Sea Lion (*Eumetopias Jubatus*) Growth Rates in Alaska, Pages 567-568 *In:* V.M. Belkovich, I.V.J. Smelova, and A.N. Boltunov (Compilers) Marine Mammals of the Holarctic: collection of scientific reports after the third international conference, Koktebel, Crimea, Ukraine, October 11-17, 2004. Moscow: KMK. 609p.

Fasham, M.J.R. 1995. Variations in the Seasonal Cycle of Biological Production in Subarctic Oceans: A Model Sensitivity Analysis. Deep Sea Research I 42(7) 1111-1149.

Fay, G. 2004. A Bayesian Stochastic Metapopulation Model For Steller Sea Lions In Alaska. M.S. Thesis, University of Washington, Seattle, Washington. 253 pp.

Fay, Gavin and André E. Punt. 2006. Modelling Spatial Dynamics of Steller Sea Lions (/Eumetopias Jubatus/) Using Maximum Likelihood and Bayesian Methods: Evaluation Causes for Population Decline, Pages 405-434. *In:* A.W. Trites, S.K. Atkinson, D.P. DeMaster, L.W. Fritz, T.S. Gelatt, L.D. Rea, and K.M. Wynne, (Eds.) Sea Lions of the World: Proceedings of the Symposium Sea Lions of the World: Conservation and Research in the 21st Century, September 30-October 3, 2004, Anchorage, Alaska, USA. Fairbanks, AK: Alaska Sea Grant College Program, University of Alaska Fairbanks. 653p

Félix, F., B., Haase, D. Chiluiza, P. Amador, and J.W. Davis. 1997. A Note on Recent Strandings and Bycatches of Sperm Whales (*Physeter macrocephalus*) and Humpback Whales (*Megaptera novaeangliae*) In Ecuador. Rep. int. Whal. Commn 47:917-9.

Ferrero, R.C. and L.W. Fritz. 1994. Comparisons of Walleye Pollock, *Theragra chalcogramma*, Harvest to Steller Sea Lion, *Eumetopias jubatus*, Abundance in the Bering Sea and Gulf of Alaska. U.S. Dep. of Commer., NOAA Tech. Memo. NMFS-AFSC.

Ferrero, R.C., D.P. DeMaster, P.S. Hill, M.M. Muto, and A.L. Lopez. 2000. Alaska Marine Mammal Stock Assessments, 2000. NOAA Technical Memorandum, NMFS-AFSC-119, U.S. Department of Commerce, NOAA. p. 191-43.

Ferrero, R.C., J. Hodder, and J. Cesarone. 1994. Recent Strandings of Rough-Toothed Dolphins, *Steno bredanensis*, on the Oregon and Washington Coasts. Mar. Mammal Sci. 10(1):114-6.

Fiscus, C.H. 1961. Growth in the Steller Sea Lion. J. Mamm. 42:195-200.

Fiscus, C.H., and G.A. Baines. 1966. Food and Feeding Behavior of Steller and California Sea Lions. Journal of Mammalogy, 47:195-200.

Ford, J.K.B. and G.M. Ellis. 1999. Transients: Mammal-Hunting Killer Whales of British Columbia, Washington and Southeastern Alaska. University of British Columbia Press, Vancouver.

Ford, J.K.B., G.M. Ellis, and K.C. Balcomb. 2000. Killer Whales. The Natural History and Genealogy of *Orcinus orca* In British Columbia and Washington State. 2nd ed. Vancouver: UBC Press.

Ford, J.K.B., G.M. Ellis, L.G. Barrett-Lennard, A.B. Morton, R.S. Palm, and K.C. Balcomb. 1998. Dietary Specialization in Two Sympatric Populations of Killer Whales (*Orcinus orca*) In Coastal British Columbia And Adjacent Waters. Can. J. Zool. 76:1456-1471.

Forney, K.A., and R.L. Brownell. 1996. Preliminary Report of the 1994 Aleutian Island Marine Mammal Survey. Unpublished document submitted to Int. Whal. Comm. (SC/48/O 11). 15 pp.

Fowler, C.W. 1999. Nature's Monte Carlo Experiments in Sustainability. *In:* Proceedings of the Fifth NMFS Stock Assessment Workshop: Providing Scientific Advice to Implement the Precautionary Approach Under the Magnuson-Stevens Fishery Conservation and Management Act, pp. 25-32. Ed. by V.R. Restrepo. U.S. Department of Commerce, NOAA Technical Memorandum, NMFS-F/SPO-40.

Fowler, S.L., D.P. Costa, J.P.Y. Arnould, N.J. Gales, and C.E. Kuhn. 2006. Ontogeny of Diving Behaviour in the Australian Sea Lion: Trials of Adolescence in a Late Bloomer. J. Anim. Ecol. 75:358-367.

Foy, R.J., and A.J. Paul. 1999. Winter Feeding and Changes in Somatic Energy Content for Age 0 Pacific Herring In Prince William Sound, Alaska. Transactions of the American Fisheries Society. 128:1193-1200.

Foy, R.J., and B.L. Norcross. 1999. Spatial and Temporal Differences in the Diet of Juvenile Pacific Herring (*Clupea pallasi*) In Prince William Sound, Alaska. Can. J. of Zoology. 77:697-706.

Francis, J., D. Boness, and H. Ochoa-Acuna. 1998 A Protracted Foraging and Attendance Cycle in Female Juan Fernandez Fur Seals. Mar. Mamm. Sci. 14:552-574.

Francis, R.C., and S.R. Hare. 1994. Decadal Scale Regime Shifts in the Large Marine Ecosystem of the Northeast Pacific: A Case for Historical Science. Fish. Oceanogr. 3:279-291.

Frankel, A.S. 1994. Acoustic and Visual Tracking Reveals Distribution, Song Variability, and Social Roles of Humpback Whales in Hawaiian Waters. Doctoral Dissertation, University of Hawaii. University Microfilms, Inc.

Frankel, A.S., and C.W. Clark. 1998. Results of Low-Frequency Playback of M-Sequence Noise to Humpback Whales, Megaptera Novaeangliae, In Hawaii. Can. J. Zool. 76:521-535.

Fraser S., Gotceitas V., and Brown J.A. 1996. Interactions Between Age-Classes of Atlantic Cod and Their Distribution Among Bottom Substrates. Can. J. Fish. Aquat. Sci. 53:305-314.

Frid, A., G.G. Baker, and L.M. Dill. 2006. Do Resource Declines Increase Predation Rate on North Pacific Harbor Seals? A Behavior-Based Plausibility Model. Mar. Ecol. Prog. Ser. 312:265-275.

Frid, A., J. Burns, G.G. Baker, and R.E. Thorne. 2008. Predicting Synergistic Effects of Resources and Predators on Foraging Decisions by Juvenile Steller Sea Lions. Oecologia

Fristrup, K.M. and G.R. Harbison. 2002. How Do Sperm Whales Catch Squids? Marine Mammal Science 18(1):42-54.

Fritz, L., M. Lynn, E. Kunisch, and K. Sweeney. 2008. Aerial, Ship, and Land-Based Surveys of Steller Sea Lions (*Eumetopias jubatus*) in Alaska, June and July 2005-2007. U. S. Dep. Commer., NOAA Tech. Memo., NMFS-AFSC-183, 70 p. (.pdf, 4.73 MB). URL: http://www.afsc.noaa.gov/Publications/AFSC-TM/NOAA-TM-AFSC-183.pdf

Fritz, L., T. Gelatt, J. Bengtson, and D. DeMaster. 2009. Survey of Adult and Juvenile Steller Sea Lions, June-July 2008: Response to the Council's 19 December 2008 Letter to Robert D. Mecum, Acting Administrator, NMFS Alaska Region.

Fritz, L.W. (Compiler). 1993. Estimated Catches of Walleye Pollock, Atka Mackerel, and Pacific Cod Within Critical Habitat of the Steller Sea Lion in the Bering Sea, Aleutian Islands, And Gulf Of Alaska From 1977-92. AFSC Processed Rep. 93-13, 66 p. Alaska Fish. Sci. Cent., Natl. Mar. Fish. Serv., NOAA, 7600 Sand Point Way NE, Seattle, WA 98115-0070. (.pdf, 4.46 MB). URL: http://www.afsc.noaa.gov/Publications/ ProcRpt/PR1993-13.pdf

Fritz, L.W. (Compiler). 1993. Observed Catches of Groundfish and Selected Bycatch Species Within Critical Habitat of the Steller Sea Lion in the Bering Sea, Aleutian Islands, and Gulf of Alaska from 1977-92. AFSC Processed Rep. 93-07, 261 p. Alaska Fish. Sci. Cent., Natl. Mar. Fish. Serv., NOAA, 7600 Sand Point Way NE, Seattle, Washington 98115-0070. (.pdf, 11.83 MB).

Fritz, L.W., and C. Stinchcomb. 2005. Aerial, Ship, and Land-Based Surveys of Steller Sea Lions (*Eumetopias jubatus*) in the Western Stock in Alaska, June and July 2003 and 2004. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-153, 56 p.

Fritz, L.W., and E.S. Brown . 2005. Survey-and Fishery-Derived Estimates of Pacific Cod (*Gadus macrocephalus*) Biomass: Implications for Strategies to Reduce Interactions Between Groundfish Fisheries and Steller Sea Lions (*Eumetopias jubatus*). Fish. Bull. 103:501-515.

Fritz, L.W., and R.C. Ferrero. 1998. Options in Steller Sea Lion Recovery and Groundfish Fishery Management. Biosphere Conservation 1:7-20.

Fritz, L.W., and S. Hinckley. 2005. A Critical Review of the Regime Shift - "Junk Food" - Nutritional Stress Hypothesis for the Decline of the Western Stock of Steller Sea Lion. Marine Mammal Science 21(3):476-518.

Fritz, L.W., R.C. Ferrero, and R.J. Berg. 1995. The Threatened Status of Steller Sea Lions, *Eumetopias jubatus*, Under the Endangered Species Act: Effects on Alaska Groundfish Fisheries Management. Mar. Fish. Rev. 57:14-27.

Frost, K.J., R.B. Russell, and L.F. Lowry. 1992. Killer Whales, *Orcinus orca*, In the Southeastern Bering Sea: Recent Sightings and Predation on Other Marine Mammals. Marine Mammal Science 8(2):110-119

Funk, F., and K.A. Rowell. 1995. Population Model Suggests New Threshold for Managing Alaska's Togiak Fishery for Pacific Herring in Bristol Bay. Alaska Fisheries Res. Bull. 2(2):125-136.

Gargett, A.E. 1997. The Optimal Stability 'Window': A Mechanism Underlying Decadal Fluctuations in North Pacific Salmon Stocks? Fish. Oceanogr. 6:109-117.

Gearin, P., S. Jeffries, S. Riemer, L. Lehman, K. Hughes, and L. Cooke. 1999. Prey Of Steller's Sea Lions, Eumetopias Jubatus, in Washington State. *In:* Abstracts of the 13th Biennial Conference on the Biology of Marine Mammals, Wailea, Hawaii November 28 December 3, P. 65. Soc. Marine Mammalogy, Wailea, Hawaii.

Gelatt, T. and A.W. Trites, and K. Hastings, and L. Jemison, and K. Pitcher, and G. O'Corry-Crowe. 2006. Population Trends, Diet, Genetics and Observations of Steller Science Symposium.

Gelatt, T., A.W. Trites, K.K. Hastings, L. Jemison, K.W. Pitcher, and G. O'Corry-Crowe. 2007. Population Trends, Diet, Genetics, and Observations of Steller Sea Lions. I*n:* Glacier Bay National Park. J.F. Piatt and S.M. Gende (Eds.) U.S. Geological Survey Scientific Investigations Report 2007-5047. pp. 145-149.

Gende, S., and M.F. Sigler. 2006. Persistence of Forage Fish 'Hot Spots' and Its Association with Foraging Steller Sea Lions (*Eumetopias jubatus*) in Southeast Alaska. Deep-Sea Research II 53:432-441.

Gentry, R.L. 1970. Social Behavior of the Steller Sea Lion. Unpubl. Ph.D. dissertation, Univ. California, Santa Cruz, 113 pp.

Gentry, R.L., and J.H. Johnson. 1981. Predation by Sea Lions on Northern Fur Seal Neonates. Mammalia 45:423-430.

Gerber J.A., J. Roletto, L.E. Morgan, D.M. Smith, and L.J. Gage. 1993. Findings in Pinnipeds Stranded Along the Central and Northern California Coast, 1984-1990. J. Wild. Dis. 29:423-433.

Gerber, L.R., and G.R. VanBlaricom. 2001. Implications of Three Viability Models for the Conservation Status of the Western Population of Steller Sea Lions (*Eumetopias jubatus*). Biological Conservation 102(2001) 261-269.

Gerrodette, T. and D.P. DeMaster. 1990. Quantitative Determination of Optimum Sustainable Population Level. Mar. Mamm. Sci. 6:1-16.

Gilmartin, W.G., R.L. DeLong, A.W. Smith, J.C. Sweeney, B.W. DeLappe, R.W. Risebrough, L.A. Griner, M.D. Dailey, and D.B. Peakall. 1976. Premature Parturition in the California Sea Lion. J. Wildl. Diseases, 12:104-115.

Gilpin, M.E., and M.E. Soulé. 1986. Minimum Viable Populations: Processes of Species Extinction. Pages 19-34 *In:* M.E. Soulé (Ed.) Conservation biology: the science of scarcity and diversity, Sinauer Associates, Inc., Sunderland, Massachusetts, 584 pp.

Gisiner, R.C. 1985. Male Territorial and Reproductive Behavior in the Steller Sea Lion, (*Eumetopias jubatus*). Ph.D., University of California, Santa Cruz.

Goldstein, T., J.A.K. Mazet, F.M.D. Gulland, T. Rowles, J.T. Harvey, S.G. Allen, D.P. King, B.M. Aldridge, and J.L. Stott. 2004. The Transmission of Phocine Herpesvirus-1 In Rehabilitating and Free-Ranging Pacific Harbor Seals (*Phoca vitulina*) in California. Veterinary Microbiology 103(3-4):131-141.

Goldstein, T., J.A.K. Mazet, V.A. Gill, A.M. Doroff, K.A. Burek, and J.A. Hammond. 2009. Phocine Distemper Virus in Northern Sea Otters in the Pacific Ocean, Alaska, USA. Emerg. Infect. Dis.

Goodman, D., M. Mangel, G. Parkes, T. Quinn, V. Restrepo, T. Smith and K. Stokes. 2002. Scientific Review of the Harvest Strategy Currently Used in the BSAI and GOA Groundfish Fishery Management Plans. North Pacific Fishery Management Council, 605 West 4th Ave., Suite 306, Anchorage, Alaska 99501.

Goto, Y., and Shimazaki, K. 1998. Diet of Steller Sea Lions off the Coast of Rausu, Hokkaido, Japan. *Biosphere conservation*, 1:141-148.

Graham, N.E. 1994. Decadal-Scale Climate Variability in the Tropical and North Pacific During the 1970s and 1980s. Observations and Model Results. Clim. Dyn. 10, 135-162.

Grebmeier, J.M., J.E. Overland, S.E. Moore, E.V. Farley, E.C. Carmack, L.W. Cooper, K.E. Frey, J.H. Helle, F.A. McLaughlin, and S.L. McNutt. 2006. A Major Ecosystem Shift in the Northern Bering Sea. Science 311:1461-1464.

Greenstreet, S.P.R. and S.J. Hall, 1996. Fishing and the Groundfish Assemblage Structure in the Northwestern North Sea: An Analysis of Long-Term and Spatial Trends. J. Anim. Ecol. 65:577-598.

Greenstreet, S.P.R., F.E. Spence, J.A. McMillan, 1999. Fishing Effects in Northeast Atlantic Shelf Seas: Patterns in Fishing Effort, Diversity and Community Structure. V. Changes in Structure of the North Sea Groundfish Species Assemblage Between 1925 and 1996. Fish. Res. 40:153-183.

Guenette, S. and V. Christensen (Eds.) 2005. Food Web Models and Data for Studying Fisheries and Environmental Impacts on Eastern Pacific Ecosystems. Fisheries Centre Research Reports 13(1):237.

Guénette, S., S.J.J. Heymans, V. Christensen, and A.W. Trites. 2006. Ecosystem Models Show Combined Effects of Fishing, Predation, Competition, and Ocean Productivity on Steller Sea Lions (*Eumetopias jubatus*) In Alaska. Canadian Journal of Fisheries and Aquatic Sciences 63:2495-2517.

Guénette, S., S.J.J. Heymans, V. Christensen, and A.W. Trites. 2007. Ecosystem models of the Aleutian Islands and southeast Alaska show that Steller sea lions are impacted by killer whale predation when sea lion numbers are low. Proc. Fourth Glacier Bay Sci. Sympos. pp. 150-154.

Guinet, C., O. Chastel, M. Koudil, J.P. Durbec, and P. Jouventin. 1998. Effects of Warm Sea-Surface Temperature Anomalies on the Blue Petrel at the Kerguelen Islands. Proceedings of the Royal Society London, B265, 1001-1006.

Gulf of Alaska (GOA) Groundfish FMP. Prepared by the Gulf of Alaska Groundfish Plan Team, North Pacific Fishery Management Council, P.O. Box 103136, Anchorage, AK 99510. North Pacific Fisheries Management Council, Anchorage, Alaska.

Haase, B. and F. Felix. 1994. A Note on the Incidental Mortality of Sperm Whales (*Physeter macrocephalus*) In Ecuador. Rep. int. Whal. Commn (Special Issue 15):481-483.

Haddon, M. 2001. Modeling and Quantitative Methods in Fisheries. Chapman and Hall/CRC, Boca Raton.

Haigh, S.P., K.L. Denman, and W.W. Hsieh. 2001. Simulation of the Planktonic Ecosystem Response to Pre-and Post-1976 Forcing in an Isopycnic Model of the North Pacific. Canadian Journal of Fisheries and Aquatic Sciences 58:703-722.

Hain, J.H.W., M.J. Ratnaswamy, R.D. Kenney, and H.E. Winn. 1992. The fin whale, *Balaenoptera physalus*, in waters of the northeastern United States continental shelf. Rep. Int. Whal. Commn. 42:653–669.

Hall, J.D. 1979. A Survey of Cetaceans of Prince William Sound and Adjacent Vicinity-Their Numbers and Seasonal Movements. Pages 63 l-726 *In:* Environmental Assessment of the Alaskan Continental Shelf. Final Reports of the Principal Investigators, Volume 6, Biological Studies. NOAA-OCSEAP, Boulder, Colorado.

Hall, S.J., J.S. Collie, D.E. Duplisea, S. Jennings, M. Bravington, and J. Link. 2006. A Length-Based Multispecies Model for Evaluating Community Responses to Fishing. Can. J. Fish. Aquat. Sci. 63:1344-1359.

Hamai, I., K. Kyushin, and T. Kinoshita. 1971. Effect of Temperature on the Body Form and Mortality in the Development and Early Larval Stages of the Alaska Pollock, *Theragra chalcogramma* (Pallas). Bull. Fac. Fish. Hokkaido Univ. 22:11-29.

Hamilton, P.K., G.S. Stone, and S.M. Martin. 1997. Note on a Deep Humpback Whale (*Megaptera novaeangliae*) Dive Near Bermuda. Bul. Mar. Sci. 61:491-494.

Hansen, J. *et al.*, 2008. Target CO2: Where Should Humanity Aim? http://www.columbia.edu/~jeh1/2008/TargetCO2_20080407.pdf Hansen, J.R. *et al.*, 2006, Global Temperature Change. Proc. Nat. Acad. Sci. 101, 16109-16114.

Hanski, I., and D. Simberloff. 1997. The Metapopulation Approach Its History, Conceptual Domain, and Application to Conservation. Pages 5-26 *In:* I.A. Hanski and M.E. Gilpin (Eds.) Metapopulation Biology. Academic Press, San Diego, CA.

Harding, K.C., T. Härkönen, H. Caswell. 2002. The 2002 European Seal Plague: Epidemiology and Population Consequences. Ecol Lett 5:727-732.

Hare, S.R., and N.J. Mantua. 2000. Empirical Evidence for Northeast Pacific Regime Shifts in 1977 and 1989. Progress in Oceanography, 46 pp. 103-145.

Hare, S.R., N.J. Mantua, and R.C. Francis. 1999. Inverse Production Regimes: Alaskan and West Coast Pacific Salmon. Fisheries 24:6-14.

Harkonen, T., R. Dietz, P. Reijnders, J. Teilmann, K. Harding, A. Hall, *et al.* 2006. The 1988 and 2002 Phocine Distemper Virus Epidemics in European Harbour Seals. Dis Aquat Organ. 68:115-30. PubMed DOI: 10.3354/dao068115.

Harlin-Cognato A., J.W. Bickham, T.R. Loughlin, R.L. Honeycutt. 2006. Glacial Refugia and the Phylogeography of Steller's Sea Lion (*Eumetopias jubatus*) in the North Pacific. J . Evol. Biol. 19:955-969.

Haug, T. 1981. On some reproduction parameters in fin whales *Balaenoptera physalus* (L.) caught off Norway. Rep. Int. Whal Commn. 31:373–378.

Havens, P. 1965. Observations on Sea Lion Harvest, Alaska Peninsula. Unpublished trip report. Available: National Marine Mammal Laboratory, 7600 Sand Point Way, NE, Seattle, Washington 98115. 9 pp.

Haynes, T.L., and C. Mishler. 1991. The Subsistence Harvest and Use of Steller Sea Lions in Alaska. Alaska Department of Fish and Game Technical Paper, 198, Alaska Department of Fish and Game. p. 44.

Heath, C.B., K.A. Ono, D.J. Boness and J.M. Francis. 1991. The Influence of El Nino on Female Attendance Patterns in the California Sea Lion. Pages 138-145 in F. Trillmich, and K.A. Ono (Eds.) Pinnipeds and El NiZo: Responses to Environmental Stress. Springer-Verlag, Berlin.

Heide-Jørgensen, M.P., T. Härkönen, R. Dietz, and P.M. Thompson. 1992. Retrospective of the 1988 European Seal Epizootic. Dis. Aquat. Organisms. 13:37-62.

Heintz, R., M.M. Krahn, G.M. Ylitalo, and F. Morado. 2006. Organochlorines in Walleye Pollock from the Bering Sea and Southeastern Alaska, Pages 561-580. *In:* A.W. Trites, S.K. Atkinson, D.P. DeMaster, L.W. Fritz, T.S. Gelatt, L.D. Rea and K.M. Wynne (Eds.) Sea Lions of the World. Alaska Sea Grant Program Report AK-SG-06-01, University of Alaska, Fairbanks.

Heise, K., L.G. Barrett-Lennard, E. Saulitis, C.G. Matkin, and D. Bain. 2003. Examining The Evidence For Killer Whale Predation on Steller Sea Lions in British Columbia and Alaska. Aquatic Mammals 29:325-334.

Helle, E., Olsson, M., and Jensen, S. 1976. DDT And PCB Levels and Reproduction in Ringed Seal from the Bothnian Bay. Ambio, 5:188-189.

Hennen, D.R. 2006. Associations Between the Alaska Steller Sea Lion Decline and Commercial Fisheries. Ecological Applications. 16(2):704-717.

Herman, D.P., D.G. Burrows, P.R. Wade, J.W. Durban, C.O. Matkin, R.G. Leduc, L.G. Barrett-Lennard, and M.M. Krahn. 2005. Feeding Ecology of Eastern North Pacific Killer Whales from Fatty Acid, Stable Isotope, and Organochlorine Analysis of Blubber Biological Opinionsies. Marine Ecology Progress Series. 302:275-291.

Herman, L.M., P.H. Forestell, R.C. Antinoja. 1980. The 1976/77 Migration of Humpback Whales into Hawaiian Waters: Composite Description. Marine Mammal Commission Report MMC-77/19, Washington, D.C.

Hermann, A.J., D.B. Haidvogel, E.L. Dobbins, and P.J. Stabeno. 2002. Coupling Global and Regional Circulation Models in the Coastal Gulf of Alaska. Progress in Oceanography 53:335-367.

Hester, K.C., E.T. Peltzer, W.J. Kirkwood and P.G. Brewer. 2008. Unanticipated Consequences of Ocean Acidification: A Noisier Ocean. Geophys. Res. Letters 35:L19601, DOI:10.1029/2008GL034913.

Heyning, J.E., and T.D. Lewis. 1990. Entanglements of baleen whales in fishing gear of southern California. Rep. int. Whal. Commn 40:427B431.

Higgins, L.V. and L. Gass. 1993. Birth to Weaning: Parturition, Duration of Lactation, and Attendance Cycles of Australian Sea-Lions (*Neophoca cinerea*). Canadian Journal of Zoology 71:2047-55.

Higgins, L.V., D.P. Costa, A.C. Huntley, and B.J. LeBoeuf. 1988. Behavioural and Physiological Measurements of Maternal Investment in the Steller Sea Lion, *Eumetopias Jubatus*. Marine Mammal Science, 4:44-58.

Hilborn, R., and C.J. Walters, 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty, Chapman and Hall, New York. 570 pp.

Hill, P.S. and D.P. DeMaster. 1999. Alaska Marine Mammal Stock Assessments 1998. National Marine Mammal Laboratory, 7600 Sand Point Way NE, Seattle, Washington 98115-0070.

Hill, P.S., and D.P. DeMaster. 1998. Alaska Marine Mammal Stock Assessments, 1998. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-97, 166 p.

Hill, P.S., J.L. Laake and E.A. Mitchell. 1999. Results of a Pilot Program to Document Interactions Between Sperm Whales and Longline Vessels in Alaska Waters. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-AFSC-108, 42 p.

Hinckley, S. 1987. The Reproductive Biology of Walleye Pollock, *Theragra chalcogramma*, in the Bering Sea, with Reference to Spawning Stock Structure. Fishery Bulletin US, 85:481-498.

Hobbs, L. and C.W. Fowler. 2008. Putting Humans in Ecology: Consistency In Science and Management. Ambio 37:119-124

Hobson, K.A., E.H. Sinclair, A.E. York, J.R. Thomason, and R.L. Merrick. 2004. Retrospective Isotopic Analyses of Steller Sea Lion Tooth Annuli and Sea-Bird Feathers: A Cross-Taxa Approach to Investigating Regime and Dietary Shifts in the Gulf of Alaska. Mar. Mammal Sci. 20(3):621-638.

Hoelzel, A.R., M. Dahlheim, and S.J. Stern. 1998. Low Genetic Variation Among Killer Whales (*Orcinus orca*) in the Eastern North Pacific and Genetic Differentiation Between Foraging Specialists. J. Hered. 89:121-128.

Hoenig, J.M. 1983. Empirical Use of Longevity Data to Estimate Mortality Rates. Fish. Bull. 82:898-903.

Hoff, G.R. 2003. Biodiversity as an Index of Regime Shifts in the Eastern Bering Sea. *In:* J.L. Boldt (Ed.) Ecosystem Considerations for 2004. Appendix C of the BSAI/GOA Stock Assessment and Fishery

Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, AK 99501.

Hoffman, J.I., C.W. Matson, W. Amos, C.D. Phillips, T.S. Gelatt, and W.J. Bickham. 2006. Deep Genetic Subdivision Within a Continuously Distributed and Highly Vagile Marine Mammal, the Steller's Sea Lion (*Eumetopias jubatus*). Molecular Ecology 15:2821-2832.

Hoffman, J.I., K.K. Dasmahapatra, W. Amos, C.D. Phillips, T.S. Gelatt, and W.J. Bickham. 2009. Contrasting Patterns of Genetic Diversity at Three Different Genetic Markers in a Marine Mammal Metapopulation. Molecular Ecology 18(14):2961-2928.

Hollowed, A.B. and B.A. Megrey. 1990. Walleye Pollock. *In:* Stock Assessment and Fishery Evaluation Report for the 1991 Gulf of Alaska Groundfish Fishery. Prepared by the Gulf of Alaska Groundfish Plan Team, North Pacific Fishery Management Council, P.O. Box 103136, Anchorage, Alaska 99510.

Hollowed, A.B., and W.S. Wooster. 1992. Variability of Winter Ocean Conditions and Strong Year Classes of Northeast Pacific Groundfish. ICES Mar. Sci. Symp. 195:433-444.

Hollowed, A.B., and W.S. Wooster. 1995. Decadal-Scale Variations in the Eastern Subarctic Pacific: II. Response of Northeast Pacific Fish Stocks. *In:* Climate Change and Northern Fish Populations. Can. Spec. Pub. of the Fish. Aquat. Sci. 121:373-385.

Hollowed, A.B., J.N. Ianelli, and P.A. Livingston. 2000. Including Predation Mortality in Stock Assessments: A Case Study Involving Gulf of Alaska Walleye Pollock. ICES Journal of Marine Science, 57:279-293.

Hollowed, A.B., S.R. Hare, and W.S. Wooster. 2001. Pacific Basin Climate Variability and Patterns of Northeast Pacific Marine Fish Production. Prog. Oceanogr. 49:257-282

Holmes, A.L., *et al.* (14 co-authors). 2008. Metal tissue levels in Steller sea lion (*Eumetopias jubatus*) pups. Mar. Pollution Bull. 56:1416-1421.

Holmes, E. 2009. Review of "Evidence for high natality rates among Steller sea lions in the Gulf of Alaska". Alaska Fisheries Science Center, Seattle, WA. 4 p.

Holmes, E.E., and A.E. York. 2003. Using Age Structure to Detect Impacts on Threatened Populations: A Case Study With Steller Sea Lions. Conservat. Biol. 17(6):1794-1806.

Holmes, E.E., L.W. Fritz, A.E. York and K. Sweeney. 2007. Age-Structured Modeling Reveals Long-Term Declines in the Natality of Western Steller Sea Lions. Ecological Applications, 17(8) 2007, pp. 2214-2232.

Hong, S.-M., S. Atkinson, K. Hülck, and Q. X. Li. 2005. PCB Concentrations and Profiles in Tissues of Steller Sea Lions from Alaska and the Bering Sea. Chapter 13, pages 110-120, *In:* T.R. Loughlin, S. Atkinson, and D.G. Calkins (Eds.) Synopsis of Research on Steller Sea Lions: 2001-2005. Alaska SeaLife Center's Steller Sea Lion Program. Sea Script Company, Seattle, Washington. 344 p.

Hood, D.W., and J.A. Calder (Eds.). 1981. The Eastern Bering Sea shelf: Oceanography and Resources, Vol. 1 and 2. Office of Marine Pollution Assessment, NOAA and BLM. Univ. of Washington Press, Seattle, Washington. 1,339 pp

Hood, D.W., and S.T. Zimmerman, (Eds.) 1986. The Gulf of Alaska: Physical Environment and Biological Resources. Minerals Management Service, Anchorage, Alaska. 655 pp.

Hood, W.R., and K.A. Ono. (1997). Variation in Maternal Attendance Patterns and Pup Behavior in a Declining Population of Steller Sea Lions (*Eumetopias jubatus*). Canadian Journal of Zoology, 75:1241-1246.

Hoover, A.A. 1988. Steller Sea Lion (*Eumetopias jubatus*). Pages 159-193 *In:* J.W. Lentfer (Ed.). Selected Marine Mammals of Alaska: Species Accounts with Research and Management Recommendations. U.S. Marine Mammal Commission, Washington, D.C. 275 pp.

Horning, M. and J.E. Mellish. 2009. Spatially explicit detection of predation on individual pinnipeds from implanted post-mortem satellite data transmitters. Endang. Species Res. Preprint 2009. 9 p.

Horning, M. and J.E. Mellish. 2010a. Consummate or consumed predators? High post-weaning predation on Steller sea lions (Eumetopias jubatus) may be limiting recovery in Prince William Sound/Kenai Fjords region. Poster paper presented at Alaska Marine Science Symposium, January 18-22, 2010.

Horning, M. and J.E. Mellish. 2010b. Satellite-linked life history transmitters in Steller sea lions: Assessing the effects of health, status, foraging ability, environmental variability and predation on juvenile survival and population trends. January 2010 Annual Project Update. Oregon State University, Newport, OR. 25 p.

Horning, M., and F. Trillmich. 1997. Development of Hemoglobin, Hematocrit, and Erythrocyte Values in Galapagos Fur Seals. Marine Mammal Science, 13:100-113.

Howe, A.L., G. Fidler, A.E. Bingham, and M.J. Mills. 1996. Harvest, Catch, and Participation in Alaska Sport Fisheries During 1995. Alaska Department of Fish and Game, Fishery Data Series No. 96-32, Anchorage.

Hui, T., R. Flinn, E. Gregr, R. Joy, and A. Trites. 2010. Steller sea lions versus fisheries: competition in the high seas? Poster paper presented at Alaska Marine Science Symposium, January 18-22, 1020, Anchorage, Alaska.

Hulbert, L.B., M.F. Sigler, and C.R. Lunsford. 2006. Depth and Movement Behaviour of the Pacific Sleeper Shark in the Northeast Pacific Ocean. Journal of Fish Biology 69:406-425

Hulson, P.F., S.E. Miller, T.J. Quinn, G.D. Marty, S.D. Moffitt and F. Funk. 2008. Data Conflicts in Fishery Models: Incorporating Hydroacoustic Data into the Prince William Sound Pacific Herring Assessment Model. ICES J. Mar. Sci. 65:25-43.

Hunt, G.L., Jr. and P.J. Stabeno. 2005. Oceanography and Ecology of The Aleutian Archipelago: Spatial and Temporal Variation Fisheries Oceanography 14(1):292-306.

Hunt, G.L., Jr., P.J. Stabeno, G. Walters, E. Sinclair, R.D. Brodeur, J.M. Napp, and N.A. Bond. 2002. Climate Change and Control of the Southeastern Bering Sea Pelagic Ecosystem. Deep-Sea Research II 49:5821-5853.

Ianelli, J., J. Gauvin, D. Stram, and P. Stabeno. 2009. Opportunistic Temperature-At-Depth Recorders on Bering Sea Pollock Trawls to Evaluate Linkages Between Location-Specific Temperatures and Pollock, Salmon, and Other Species. Paper presented at the Alaska Marine Science Symposium, Anchorage, Alaska, January 2009.

Ianelli, J.N., Barbeaux, S., Honkalehto, T., Lauth, B., and N. Williamson. 2005. Assessment of Alaska Pollock Stock in the Eastern Bering Sea. November, 2005 Plan Team Draft Assessment. Alaska Fisheries Science Center, 7600 Sand Point Way, N.E., Seattle, Washington 98115.

Ianelli, J.N., Honkalehto, T., and N. Williamson. 2005. An Age-Structured Assessment of Pollock (*Theragra chalcogramma*) from the Bogoslof Island Region. September, 2005 Plan Team Draft. Alaska Fisheries Science Center, 7600 Sand Point Way, N.E., Seattle, Washington 98115.

Ianelli, J.N., L. Fritz, T. Honkalehto, N. Williamson, and G. Walters. 1999. Eastern Bering Sea Walleye Pollock Assessment. *In:* Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Region. Bering Sea/Aleutian Islands Plan Team, pp. 37-150. North Pacific Fishery Management Council, 605 W. 4th Avenue, Suite 306, Anchorage, Alaska 99501.

Ianelli, J.N., S. Barbeaux, T. Honkalehto, B. Lauth, and N. Williamson. 2005. Assessment of Alaska Pollock Stock in the Eastern Bering Sea. Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions For 2006. North Pacific Fishery Management Council, 605 West 4th Ave., Suite 306, Anchorage, Alaska 99501.

Ianelli, J.N., S. Barbeaux, T. Honkalehto, S. Kotwicki, K.Y. Aydin, and N. Williamson. 2007. Eastern Bering Sea Walleye Pollock. *In:* Stock Assessment and fishery Evaluation Report for the Groundfish Resources or the Bering Sea/Aleutian Islands Regions. North. Pac. Fish. Mgmt. Council, Anchorage, Alaska.

Ianelli, J.N., S. Barbeaux, T. Honkalehto, S. Kotwicki, K.Y. Aydin, and N. Williamson. 2008. Eastern Bering Sea Walleye Pollock. Stock Assessment and Fishery Evaluation (SAFE) Report for the Bering Sea and Aleutian Islands Groundfish, North Pacific Fishery Management Council, Anchorage, Alaska.

Ianelli, James N., S. Barbeaux, T. Honkalehto, S. Kotwicki, K. Aydin and N. Williamson. 2006. Assessment of Alaska Pollock Stock in the Eastern Bering Sea. Alaska Fisheries Science Center http://www.afsc.noaa.gov/refm/docs/2006/EBSpollock.pdf.

Ianelli, J., S. Lowe, G. Thompson, and L. Logerwell. 2010a. Projections of Atka mackerel and Pacific cod catch reductions. Unpublished manuscript, AFSC, Seattle. 20 p.

Ianelli, J., S. Lowe, G. Thompson, and L. Logerwell. 2010b. Aleutian islands trawl survey biomass summary. Unpublished manuscript, AFSC, Seattle. 1 p.

IAP (InterAcademy Panel on International Issues). 2009. IAP Statement on Ocean Acidification. June http://www.interacademies.net/Object.File/Master/9/075/Statement_RS1579_IAP_05.09final2.pdf.

Ikonomou, M.G., S. Rayne and R.F. Addison. 2002. Exponential Increases of the Brominated Flame Retardants, Polybrominated Diphenyl Ethers, in the Canadian Arctic from 1981 to 2000. Environmental Science & Technology 36(9):1886-1892.

Imler, R.H. and H.R. Sarber. 1947. Harbor Seals and Sea Lions in Alaska. U.S. Dept of the Interior, Fish and Wildlife Service, Spec. Sci. Rep. 28, 23 pp.

Innes, S., D.M. Lavigne, W.M. Earle, and K.M. Kovacs. 1987. Feeding Rates of Seals and Whales. J. Anim. Ecol. 56:115-130.

Intergovernmental Panel on Climate Change (IPCC) (2007) Summary for policymakers, *In:* Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the

Intergovernmental Panel on Climate Change, edited by S. Solomon *et al.*, pp. 1-18, Cambridge Univ. Press, New York.

IPCC. 2007a. Climate Change 2007: Impacts, Adaptation and Vulnerability. Brussels: Intergovernmental Panel on Climate Change. Accessed 24 March 2008. Available: http://www.ipcc.ch/ipccre-ports/ar4-wg2.htm.

IPCC. 2007b. Climate Change 2007: Synthesis Report; Summary for Policymakers. Accessed 24 March 2008. Available: http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr_spm.pdf.

International Pacific Halibut Commission (IPHC). 2000. Pacific Halibut Stock Assessment and Fishery Evaluation. *In:* Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions. North Pacific Fishery.

International Whaling Commission (IWC). (1980). Chairman's Report of the Thirty-First Annual Meeting. Rep. Int. Whal. Commn 30:25-41.

Jackson, J.B.C., M.X. Kirby, W.H. Berger, K.A. Bjorndal, L.W. Botsford, B.J. Bourque, R.H. Bradbury, R. Cooke, J. Erlandson, J.A. Estes, T.P. Hughes, S. Kidwell, C.B. Lange, H.S. Lenihan, J.M. Pandolfi, C.H. Peterson, R.S. Steneck, M.J. Teneger, and R.R. Warner. 2001. Historical Overfishing and the Recent Collapse of Coastal Ecosystems. Science 293:629-638.

Jahoda, M., C.L. Laforuna, N. Biassoni, C. Almirante, A. Azzellino, A.S. Panigada, M. Znardelli., G.N. Di Siara. 2003. Mediterranean fin whale's (*Balaenoptera physalus*) response to small vessels and biopsy sampling assessed through passive tracking and timing of respiration. Mar. Mamm. Sci. 19(1):96–110.

Jaquet, N. 1996. How Spatial and Temporal Scales Influence Understanding of Sperm Whale Distribution: A Review. Mammal Review 26:51-65.

Janout, M.A., T.J. Weingartner, T.C. Royer, and S.L. Danielson. 2010. On the nature of winter cooling and the recent temperature shift on the northern Gulf of Alaska shelf. J. Geophys. Res. 115, C05023, doi:10.1029/2009JC005774.

Janout, M.A., T.J. Weingartner, S.R. Okkonen, T.E. Whitledge, and D.L. Musgrave. 2009. Some characteristics of Yakutat Eddies propagating along the continental slope of the northern Gulf of Alaska. Deep-Sea Res. II 56(2009):2444-2459.

Jaquet, N., and H. Whitehead. 1996. Scale-Dependent Correlation of Sperm Whale Distribution with Environmental Features and Productivity in the South Pacific. Mar Ecol Prog Ser 135:1-9

Jaquet, N., S. Dawson and L. Slooten. 1998. Seasonal Distribution and Diving Behavior of Male Sperm Whales Off Kaikura: Foraging Implications. Canadian Journal of Zoology 78:407-419.

Jeanniard du Dot, T., D.A.S. Rosen, and A.W. Trites. 2008. Energy reallocation during and after periods of nutritional stress in Steller sea lions: Low-quality diet reduces capacity for physiological adjustments. Physiological and Biochemical Zoology 82(5):516–530.

Jeanniard du Dot, T., D.A.S. Rosen, J.P. Richmond, A.S. Kitaysky, S.A. Zinn, and A.W. Trites. 2009. Changes in glucocorticoids, IGF-I and thyroid hormones as indicators of nutritional stress and subsequent refeeding in Steller sea lions (*Eumetopias jubatus*). Compar. Biochem. Physiol., Part A 152:524-534.

Jensen, A.S., and G.K. Silber. 2003. Large Whale Ship Strike Database. U.S. Department of Commerce. National Oceanic and Atmospheric Administration. Technical Memorandum NMFS-OPR-25, 37 pp.

Jensen, T., M. van de Bildt, H.H. Dietz, T.H. Andersen, A.S. Hammer, T. Kuiken, A. Osterhaus. 2002. Another Phocine Distemper Outbreak in Europe. Science 297:209

Johnson, D.S. 2009. Estimating natality rate of Steller sea lions at Chiswell Island rookery, Alaska. Alaska Fisheries Science Center, Seattle, WA. 7 p.

Johnson, D.S. 2010. An Analysis of Steller Sea Lion Population Trends in the Western and Asian Stocks in the 1990s and 2000s. Alaska Fisheries Science Center manuscript. 7 p.

Johnson, J.H., and A.A. Wolman. 1984. The Humpback Whale, *Megaptera novaeangliae*. Marine Fisheries Review 46(4):30-37.

Johnson, S.R., J.J. Burns, C.I. Malme, and R.A. Davis. 1989. Synthesis of Information on the Effects of Noise and Disturbance on Major Haulout Concentrations of Bering Sea Pinnipeds. Rep. to U.S. Minerals Management Service, Anchorage, Alaska. No. MMS 88-0092.

Jordan, D.S., and Evermann, B.W., American food and game fishes, New York, Doubleday, Page and Co., 1902.

Kajimura, H., and T.R. Loughlin. 1988. Marine Mammals in the Oceanic Food Web of the Eastern Subarctic Pacific. Bull. Ocean Res. Inst. 26:187-223.

Kannan, K., N. Kajiwara, B.J. Le Boeuf, and S. Tanabe. 2004. Organochlorine Pesticides and Polychlorinated Biphenyls in California Sea Lions. Environ. Pollut. 131:425-434.

Kaplan, C.C., G.C. White, and B.R. Noon. 2008. Neonatal survival of Steller sea lions (*Eumetopias jubatus*). Mar. Mamm. Sci. 24(3): 443–461.

Kastelein, R.A., N. Vaughan, and P.R. Wiepkema. 1990. The Food Consumption of Steller Sea Lions (*Eumetopias jubatus*). Aquat. Mamm. 15:13.

Kasuya T (1999) Examination of the Reliability of Catch Statistics in the Japanese Coastal Sperm Whale Fishery. J Cetacean Res Manag 1:109-122.

Kasuya, T (1991). Density-Dependent Growth in North Pacific Sperm Whales. Mar. Mammal Sci. 7:230-257

Kato, H., and T. Miyashita. 1998. Current Status of North Pacific Sperm Whales and Its Preliminary Abundance Estimates. Unpublished report submitted to Int. Whal. Comm. 6 pp.

Kawakami, T. 1980. A Review of Sperm Whale Food. Scientific Report of the Whales Research Institute Tokyo 32:199-218.

Kawamura, A. 1982. Food habits and prey distributions of three rorqual species in the North Pacific Ocean. Sci. Rep. Whales Res. Inst. 34:59–91.

Kendall, A.W., and S.J. Picquelle. 1990. Egg and Larval Distributions of Walleye Pollock, Theragra Chalcogramma, In Shelikof Strait, Gulf Of Alaska. Fish. Bull. U.S. 88:133-154.

Kendall, A.W., L.S. Incze, P.B. Ortner, S. Cummings, and P.K. Brown. 1994. Vertical Distribution of Eggs and Larvae of Walleye Pollock, *Theragra chalcogramma*, In Shelikof Strait, Gulf Of Alaska. Fishery Bulletin, U.S. 92:540-554

Kenney, R.D. 2007. Right Whales and Climate Change: Facing the Prospect of a Greenhouse Future. Pages 436-459 *In:* S.D. Kraus and R.M. Rolland, (Eds.) The Urban Whale: North Atlantic Right Whales at the Crossroads. Harvard University Press, Cambridge, Massachusetts.

Kenney, R.D., G.P. Scott, T.J. Thompson, and H.E. Winn. 1997. Estimates of Prey Consumption and Trophic Impacts of Cetaceans in the USA Northeast Continental Shelf Ecosystem. J. Northw. Atl. Fish. Sci. Vol. 22:155-171.

Kenyon, K.W. 1962. History of the Steller Sea Lion at the Pribilof Islands, Alaska. J. Mammal. 43:68-75.

Kenyon, K.W., and D.W. Rice. 1961. Abundance and Distribution of the Steller Sea Lion. J. of Mammal. 42:223-234.

Kenyon, K.W., V.B. Scheffer, D.G. Chapman. 1954. A Population Study Of The Alaska Fur Seal Herd. Special Scientific Report of the United States Fish and Wildlife Service 12. Washington, D.C. 77 pp.

Ketchen, K.S. 1968. Trends in marine fisheries along the Pacific coast of North America. Trans. Am. Fish. Soc. 97(1):82-88.

Ketten, D.R. 1998. Marine Mammal Ears: An Anatomical Perspective on Underwater Hearing. *In:* Joint Meeting of the Acoustical Society of America and the International Acoustical Society, Plenary Lecture. Seattle, Washington.

Keyes, M.C. 1968. The Nutrition of Pinnipeds. Pages 359-399 *In:* R.J. Harrison, R.C. Hubbard, R.S. Peterson, C.E. Rice and R.J. Shusterman (Eds.) The Behavior and Physiology of Pinnipeds. Appleton-Century-Crofts, New York, NY.

Kim, G.B., S. Tanabe, R. Tatsukawa, T.R. Loughlin, and K. Shimazaki. 1996. Characteristics of Butyltin Accumulations and its Biomagnification in Steller Sea Lion (*Eumetopias jubatus*) Environmental Toxicology and Chemistry 15(11):2043-2048.

King, J.E. 1954. The Otariid Seals of the Pacific Coast of America. Bull. British Mus. (Nat. Hist.) Zool. 2:311-337.

King, J.R. 2005. Report of the Study Group on Fisheries and Ecosystem Responses to Recent Regime Shifts. PICES Scientific Report No. 28, 162 pp.

Kirsch, P.E., S.J. Iverson, and W.D. Bowen. 2000. Effect of a low-fat diet on body composition and blubber fatty acids of captive juvenile harp seals (*Phoca groenlandica*). Physiological and Biochemical Zoology 73:45-59.

Kitts, D.D., M.D. Huynhl, C. Hu, and A.W. Trites. 2004. Season Variation in Nutrient Composition of Alaskan Walleye Pollock. Canadian Journal of Zoology 82:1408-1415.

Kooyman, G.L. 1985. Physiology Without Restraint in Diving Mammals. Marine Mammal Science, 1, pp. 166-178.

Kotenev, A. and A.I. Glubokov. 2007. Walleye Pollock Theragra Chalcogramma from the Navarin Region and Adjacent Waters of the Bering Sea: Ecology, Biology and Stock Structure. VNIRO Publishing, Moscow, Russia. 180 pp.

Krahn, M.M., D.P. Herman, C.O. Matkin, J.W. Durba, L. Barrett-Lennard, D.G. Burrows, M.E. Dahlheim, N. Black, R.G. LeDuc, and P.R. Wade. 2007. Use of chemical tracers in assessing the diet and foraging regions of eastern North Pacific killer whales. Marine Environmental Research 63 (2007) 91–114.

Krieger, K. and B.L. Wing. 1986. Hydroacoustic Monitoring of Prey to Determine Humpback Whale Movements. NOAA Tech. Memo. NMFS-F/N/NWC-98, 62 pp.

Kruse, G.H. and H.P. Huntington. 2009. Local and traditional knowledge of the nature and extent of interactions between fishermen and Steller sea lions in the Gulf of Alaska and Bering Sea. Final Report, Pollock Conservation Cooperative Research Center, University of Alaska Fairbanks, Juneau, AK. Grant No. G00003412. 30 p.

Kruse, G.H., F.C. Funk, H.J. Geiger, K.R. Mabry, H.M. Savikko, and S.M. Siddeek. 2000. Overview of State-managed Marine Fisheries in the Central and Western Gulf of Alaska, Aleutian Islands, and the Southeastern Bering Sea, with Reference to Steller Sea Lions. ADF&G, Division of Commercial Fisheries, P.O. Box 25526, Juneau, Alaska 99802.

Kruse, G.H., P.A. Livingston, J.E. Overland, G.S. Jamieson, S. McKinnell, and R.I. Perry. (Eds.). 2006. Report of the PICES/NPRB Workshop on Integration of Ecological Indicators of the North Pacific with Emphasis on the Bering Sea. PICES Scientific Report No. 33, 109p.

Kucey, L. 2005. Human Disturbance and the Hauling Out Behaviour of Steller Sea Lions (*Eumetopias jubatus*). MSc thesis, University of British Columbia, Vancouver. 67 pp.

Kucey, L., and A.W. Trites. 2006. A Review of the Potential Effects of Disturbance on Sea Lions: Assessing Response and Recovery. *In:* A.W. Trites, S. Atkinson, D.P. DeMaster, L.W. Fritz, T.S. Gelatt, L.D. Rea, and K. Wynne (Eds.) Sea lions of the World, Alaska Sea Grant.

Kuker, K. and L. Barrett-Lennard. 2010. A re-evaluation of the role of killer whales Orcinus orca in a population decline of sea otters Enhydra lutris in the Aleutian Islands and a review of alternative hypotheses. Mammal Rev. (2010):1-18.

Kumagai, S. 2004. Seasonal Differences in Physiology of Captive Steller Sea Lions (*Eumetopias jubatus*) In Response To Short-Term Low Energy Intake. MSc thesis, University of British Columbia. 112 p.

Kumagai, S., D.A.S. Rosen, and A.W. Trites. In Press. 2006. Body Mass and Composition Responses to Short-Term Low Energy Intake are Seasonally Dependent in Steller Sea Lions (*Eumetopias jubatus*). Journal of

Kurle, C.M., and C.J. Gudmundson. 2007. Regional Differences in Foraging of Young-Of-The Year Steller Sea Lions Eumetopias Jubatus in Alaska: Stable Carbon and Nitrogen Isotope Ratios in Blood. Mar. Ecol. Prog. Ser. 342:303-310.

Kurle, C.M., and E.H. Sinclair. 2003. Estimation of Steller Sea Lion (*Eumetopias jubatus*) Foraging Ecology Using Stable Nitrogen and Carbon Isotope Ratio Analyses of Blood Components. Marine Mammal Biennial Symposium, 2003.

Ladd, C., G.L. Hunt, Jr., C.W. Mordy, S.A. Salo, and P.J. Stabeno. 2005b. Marine Environment of the Eastern and Central Aleutian Islands Fisheries Oceanography Volume 14(1):22-38.

Laevastu, T., and H.A. Larkins. 1981. Marine Fisheries Ecosystem, Its Quantitative Evaluation and Management. Fishing News Books Ltd., Farnham, Surrey, England.

Laist, D. 1997. Impacts of Marine Debris: Entanglement of Marine Life in Marine Debris Including a Comprehensive List of Species with Entanglement and Ingestion Records. Marine Debris, Sources, Impacts and Solutions, J. Coe and D.B. Rogers, (Eds.) Springer-Verlag New York Inc., New York, pp. 99-140.

Laist, D.W., A.R. Knowlton, J.G. Mead, A.S. Collet, and M. Podesta. 2001. Collisions Between Ships And Whales. Mar. Mammal Sci. 17, 35-75.

Lambertsen, R.H., B.A. Kohn, J.P. Sundberg, and C.D. Buergelt. 1987. Genital Papillomatosis in Sperm Whale Bulls. J. Wildlife Diseases. 23(3):361-367.

Lammers, M.O., W.W.L. Au, & D.L. Herzing. 2003. The Broadband Social Acoustic Signaling Behavior of Spinner and Spotted Dolphins. Journal of Acoustical Society of America,114:1629-1639

Lander, M.E., T.R. Loughlin, M.G. Logsdon, G.R. VanBlaricom, B.S. Fadely, and L.W. Fritz. 2009. Regional differences in the spatial and temporal heterogeneity of oceanographic habitat used by Steller sea lions. Ecol. App. 19(6):1645-1659.

Lander, M.E., T.R. Loughlin, M.G. Logsdon, G.R. VanBlaricom, and B.S. Fadely. 2010. Foraging effort of juvenile Steller sea lions *Eumetopias jubatus* with respect to heterogeneity of sea surface temperature. Endang. Species Res. 10:145-158.

Latif, and Barnett. 1996. Decadal Climate Variability Over the North Pacific and North America: Dynamics And Predictability. J. Climate 9:2407-2423.

Laur, D., and L. Haldorson. 1996. Coastal Habitat Studies: The Effect of the Exxon Valdez Oil Spill on Shallow Subtidal Fishes in Prince William Sound. *In:* Rice S.D., Spies R.B., Olfe D.A., and Wright B.A. (Eds.) American Fisheries Society Symposium 18: Proceedings of the Exxon Valdez Oil Spill Symposium. American Fisheries Society, Bethesda Maryland, pp 659-670.

Law, R. 2000. Fishing, Selection, and Phenotypic Evolution. ICES Journal of Marine Science 57:659-668.

Law, R., and D.R. Grey. 1989. Evolution of Yields from Populations with Age-Specific Cropping. Evolutionary Ecology 3:343-359.

Laws, R.M. (1956). Growth and sexual maturity in aquatic mammals. *Nature*, 178, pp. 193-194.

Laws, R.M. 1977. Seals and whales of the Southern Ocean. Philisophical Transactions of the Royal Society of London, B. 279:81-96.

Laws, Richard M. 1985. The ecology of the Southern Ocean. American Scientist 73(1):26-40.

Lea, M.A., C. Guinet, Y. Cherel, G. Duhamel, L. Dubroca, P. Pruvost and M. Hindell. 2006. Impacts Of Climatic Anomalies on Provisioning Strategies of a Southern Ocean Predator. Mar. Ecol. Prog. Ser. 310:77-94.

Leatherwood, S., A.E. Bowles, and R.R. Reeves. 1983. Aerial Surveys of Marine Mammals in the Southeastern Bering Sea. *In:* NOAA/OCSEAP Environmental Assessment of the Alaskan Continental Shelf. Final Reports of Principal Investigators 42:147-473.

Leboeuf, J.N., T. Tajima, and J.M. Dawson. 1982. Dynamic Magnetic X-point, Phys. Fluids, 25:784-798.

Lee, J.S., S. Tanabe, H. Umino, R. Tatsukawa, T.R. Loughlin, and D.G. Calkins. 1996. Persistent Organochlorines in Steller Sea Lion (*Eumetopias Jubatus*) from the Bulk of Alaska and the Bering Sea (1976-1984). Marine Pollution Bulletin, 32:535-544.

Lee, P.C., P. Majluf, and I.J. Gordon. 1991. Growth, Weaning and Maternal Investment from a Comparative Perspective. Journal of Zoology (London) 225:99-114.

Lewis, J.P. 1987. An Evaluation of a Census-Related Disturbance of Steller Sea Lions. M.S. Thesis, University of Alaska, Fairbanks. 93 p.

Lima, M. and E. Paez. 1995. Growth and Reproductive Patterns in the South American Fur Seal. J. Mammal. 76:1249-1255.

Lindsay, R.W., and J. Zhang. 2005. The Thinning of Arctic Sea Ice, 1988-2003: Have We Passed a Tipping Point? J. Climate 18:4879-4894.

Livingston, P.A. 1993. Importance of Predation by Groundfish, Marine Mammals and Birds on Walleye Pollock and Pacific Herring in the Eastern Bering Sea. Marine Ecology Prog. Ser. 102:205-215.

Lockyer, C. 1976. Body Weights of Some Species of Large Whales. J. Conseil International l'Exploration de la Mer 36(3):259-273.

Logerwell, E.A., and L.E. Schaufler. 2005. New Data on Proximate Composition and Energy Density of Steller Sea Lion (*Eumetopias jubatus*) Prey Fills Seasonal and Geographic Gaps in Existing Information. Aquatic Mammals 31(1):62-82.

Logerwell, E.A., K. Aydin, S. Barbeaux, E. Brown, M.E. Conners, S. Lowe, J.W. Orr, I. Ortiz, R. Reuter, and P. Spencer. 2005. Geographic Patterns in the Demersal Ichthyofauna of the Aleutian Islands. Fisheries Oceanography 14(s1):93-112

Logerwell, E.A., S.J. Barbeaux, and L.W. Fritz. 2009. A Cooperative Pollock Acoustic Biomass Survey for Management of Fisheries Interactions with Steller Sea Lions in the Aleutian Islands. NPRB Project 730 Final Report. 86 pp.

Løkkeborg, S. 1989. Longline Bait: Fish Behaviour and the Influence of Attractant Release Rate and Bait Appearance. Dr. Scient. Thesis, Univ. Bergen, Norway.

Lokkeborg, S. 1998. Feeding Behaviour of Cod, Gadus Morhua: Activity Rhythm and Chemically Mediated Food Search Animal Behaviour. Anim. Behav. 56(2):371-378, Aug 1998.

Lokkeborg, S., A. Bjordal, and A. Ferno. 1989. Responses of Cod (*Gadus morhua*) and Haddock (*Melanogrammus aeglefinus*) to Baited Hooks in the Natural Environment. Can. J. Fish. Aquat. Sci. 46:1478-1483.

Lokkeborg, S., and A. Ferno. 1999. Diel Activity Pattern and Food Search Behaviour in Cod, Gadus Morhua. Environmental Biology of Fishes. The Hague Environ. Biol. Fish. 54(3): 345-353.

Long, D.J., K. Hanni, P. Pyle, J. Roletto, R.E. Jones, and R. Bandar. 1996. White Shark Predation on Four Pinniped Species in Central California Waters: Geographic and Temporal Patterns Inferred from Wounded Carcasses. Pages 263-274. *In:* A.P. Klimley and D.G. Ainley (Eds.) Great White Sharks, the Biology of Carcharadon Carcarias. Academic Press, San Diego, California.

Longhurst, A. 1999. Does the Benthic Paradox Tell Us Something About Surplus Production Models? Fisheries Research 867:1-7.

Longhurst, A. 2002. Murphy's Law Revisited: Longevity as a Factor in Recruitment to Fish Populations. Fisheries Research 56:125-131.

Loughlin T.R., N. Williamson, R. Methot, and S. Zimmerman.1992. Marine Mammals-Fisheries Interactions: The Steller Sea Lion Issue. A Management and Science Colloquium, Alaska Region and Fisheries Science Center, June 4, 1992.

Loughlin, T., and R. Nelson, Jr. 1986. Incidental Mortality of Northern Sea Lions in Shelikof Strait, Alaska. Mar. Mamm. Sci. 2:14-33.

Loughlin, T.R. 1993. Status and Pelagic Distribution of Otariid Pinnipeds in the Bering Sea During Winter. OCS study, MMS 93-0026. 58 pp.

Loughlin, T.R. 1997. Using the Phylogeographic Method to Identify Steller Sea Lion Stocks. Molecular Genetics of Marine Mammals (Spec. Pub.) 3:159-171.

Loughlin, T.R. 1998. The Steller Sea Lion: A Declining Species. Biosphere Conservation 1(2):91-98.

Loughlin, T.R. and R.L. Merrick. 1989. Comparison of Commercial Harvest of Walleye Pollock and Northern Sea Lion Abundance in the Bering Sea and Gulf of Alaska. Proceedings of the International Symposium on the Biological Management of Walleye Pollock.

Loughlin, T.R., A.S. Perlov, and V.A. Vladimirov. 1990. Survey of Northern Sea Lions (*Eumetopias jubatus*) in the Gulf of Alaska and Aleutian Islands During June 1989. U.S. Dep. Comm., NOAA Tech. Memo. NMFS F/NWC-176. 26 pp.

Loughlin, T.R., A.S. Perlov, and V.V. Vladimirov. 1992. Range-Wide Survey and Estimation of Total Number of Steller Sea Lions in 1989. Marine Mammal Science, 8:220-239.

Loughlin, T.R., A.S. Perlov, J.D. Baker, S.A Blokhin, and A.G. Makhnyr. 1998. Diving Behavior of Adult Female Steller Sea Lions Iin The Kuril Islands, Russia. Biosphere Conservation 1:21-31.

Loughlin, T.R., and A.E. York. 2000. An Accounting of the Sources of Steller Sea Lion, *Eumetopias jubatus*, Mortality. Marine Fisheries Review 62(4).

Loughlin, T.R., and A.E. York. 2001. Rethinking Alaskan Fishery Impacts on Steller Sea Lion Mortality. Unpublished NMFS white paper.

Loughlin, T.R., and J.V. Tagart. 2006. Compendium of Steller Sea Lion Related Research, 2000-2006. Unpublished report dated May 2006, available form the North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501-2817

Loughlin, T.R., and Ohtani, K.E. Dynamics of the Bering Sea. *AK-SG-99-03*, University of Alaska Sea Grant, Fairbanks. p. 838.

Loughlin, T.R., and R.L. DeLong. 1983. Incidental Catch of Sea Lions During the 1982 and 1983 Walleye Pollock Joint Venture Fishery in Shelikof Strait, Alaska. NWAFC Processed Report. 83-15, 37 pp.

Loughlin, T.R., D.J. Rugh, and C.H. Fiscus. 1984. Northern Sea Lion Distribution. Journal of Wildlife Management 48:729-740.

Loughlin, T.R., Gearin, P.J., DeLong, R.L., and Merrick, R.L. (1986). Assessment of Net Entanglement on Northern Sea Lions in the Aleutian Islands, 25 June-July 1985. NWAFC Processed Report, 86-02, DOC, NOAA, NMFS, Alaska Fisheries Science Center, 7600 Sand Point Way NE, Seattle, Washington 98115. p. 50.

Loughlin, T.R., J.L. Bengtson, and R.L. Merrick. 1987. Characteristics of Feeding Trip S of Female Northern Fur Seals. Can. J. Zool. 65(2):2079-2084.

Loughlin, T.R., J.T. Sterling, R.L. Merrick, J.L. Sease, and A.E. York. 2003. Diving Behavior of Immature Steller Sea Lions (*Eumetopias jubatus*). Fishery Bulletin.

Loughlin, T.R., M.A. Perez, R.L. Merrick. 1987. *Eumetopias jubatus*. Mammalian Species 283:1-7.

Lowe, S.A., and L.W. Fritz. 1997. Atka Mackerel. *In:* Stock Assessment and Fishery Evaluation Report for Groundfish Resources in the Bering Sea/Aleutian Islands Region as Projected for 1998. North Pacific Fishery Management Council. 605 W. 4th Ave., Suite 306, Anchorage, Alaska.

Lowe, S., J. Ianelli, H. Zenger, K. Aydin, and R. Lauth. 2005. Stock Assessment of Aleutian Islands Atka Mackerel for 2006. *In:* Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions. North Pacific Fisheries Management Council, P.O. Box 103136, Anchorage, Alaska.

Lowe, S., J. Ianelli, M. Wilkins, K. Aydin, R. Lauth, and I. Spies. 2009. North Pacific Groundfish Stock Assessment and Fishery Evaluation Reports for 2009. 16. Stock Assessment of Bering Sea/aleutian Islands Atka Mackerel.

Lowe, S.A., J. Ianelli, M. Wilkins, K. Aydin, R. Lauth and I. Spies. 2007. Stock Assessment of Aleutian Islands Atka Mackerel for 2006. North Pacific Fishery Management Council, 605 West 4th Ave., Suite 306, Anchorage, Alaska 99501.

Lowry, L.F., K.J. Frost, D.G. Calkins, G.L. Swartzman, and S. Hills. 1982. Feeding Habits, Food Requirements, and Status of Bering Sea Marine Mammals. Final report to the North Pacific Fishery Management Council, P.O. Box 3136 DT, Anchorage, Alaska 99510. Contract No. 81-4.

Lunn, N.J. and I.L. Boyd. 1993a. Influence of Maternal Characteristics and Environmental Variation on Reproduction in Antarctic Fur Seals. Symposium of the Zoological Society of London 66:115-129.

Lunn, N.J., and I.L. Boyd. 1993b. Effects of Maternal Age and Condition on Parturition and the Perinatal Period of Antarctic Fur Seals. J. Zool. Lond. 229:55-67.

Lunn, N.J., Boyd, I.L. and Croxall, J.P. 1994. Reproductive Performance of Female Antarctic Fur Seals: The Influence of Age, Breeding Experience, Environmental Variation and Individual Quality. J. Anim. Ecol. 63:827-840.

Mackas, D.L., R. Goldblatt, and A.G. Lewis. 1998. Interdecadal Variation in Developmental Timing of Neocalanus Plumchrus Populations at Ocean Station P in the Subarctic North Pacific. Can. J. Fish. Aquat. Sci. 55:1878-1893.

Malavear, M.Y.G. 2002. Modeling the Energetics of Steller Sea Lions (*Eumetopias jubatus*) Along the Oregon Coast. M.S. Thesis, Oregon State University, Corvallis, OR. 114 p.

Malme, C.I., P.I. Miles, C.W. Clark, P. Tyack, and J.E. Bird. 1984. Investigations of the Potential Effects of Underwater Noise from Petroleum Industry Activities on Migrating Gray Whale Behavior - Phase 2: January 1984 Migration. Final Report No. 5586 report prepared by Bolt, Beranek and Newman Inc., Cambridge, MA for the US Minerals Management Service, Anchorage, Alaska. BBN, Inc. NTIS PB-86-218377. 297pp. Available from http://www.mms.gov.

Malme, C.I., P.R. Miles, C.W. Clark, P. Tyack, and J.E. Bird. 1983. Investigations of the Potential Effects of Underwater Noise From Petroleum Industry Activities on Migrating Gray Whale Behavior. Unpublished Report from Bolt Beranek and Newman, Inc., Cambridge, Massachusetts, for U.S. Minerals Management Service, Reston, Virginia. Variously paginated. Available from Minerals Management Service, Alaska Outer Continental Shelf Region, 949 East 36th Avenue, Room 110, Anchorage, Alaska 99508-4302, U.S.A.

Malme, C.I., P.R. Miles, P. Tyack, C.W. Clark, and J.E. Bird. 1985. Investigations of the Potential Effects of Underwater Noise from Petroleum Activities On feeding Humpback Whale Behaviour. Report from Bolt Beranek & Newman Inc., Cambridge, Massachusetts for U.S. Minerals Management Service, Anchorage, Alaska. Var. pag. NTIS PB86-218385, Unpublished.

Maniscalco, J., S. Atkinson, and P. Armato. 2002. Early Maternal Care and Pup Survival in Steller Sea Lions: A Remote Video Monitoring Project in the Northern Gulf of Alaska. Arctic Research of the United States 16:36-41.

Maniscalco, J.M., D.G. Calkins, P. Parker, and S. Atkinson. 2008. Aquatic Mammals 34(3):277-287.

Maniscalco, J.M., C.O. Matkin, D. Maldini, D.G. Calkins, S. Atkinson. 2007. Assessing Killer Whale Predation on Steller Sea Lions From Field Observations in Kenai Fjords, Alaska. Marine Mammal Science, 23(2):306-321, April 2007. C 2007 by the Society for Marine Mammalogy DOI: 10.1111/j.1748-7692.2007.00103.x

Maniscalco, J.M., P. Parker and S. Atkinson. 2006. Interseasonal and Interannual Measures of Maternal Care Among Individual Steller Sea Lions (*Eumetopias jubatus*). Journal of Mammalogy 87:304-311.

Maniscalco, J.M., D. Hennen, and P. Parker. 2009. Evidence for high natality rates among Steller sea lions in the Gulf of Alaska. Alaska SeaLife Center Technical Report ASLC-0901. Seward, AK. 28 p.

Maniscalco, J.M., A.M. Springer, and P. Parker. 2010. High natality rates of endangered Steller sea lions in Kenai Fjords, Alaska and perceptions of population status in the Gulf of Alaska. PLoS ONE 5(4):1-8.

Mann, K.H., and J.R.N. Lazier. 1991. Dynamics of Marine Ecosystems: Biological-Physical Interactions in the Oceans. Blackwell Scientific Publications, Boston. 466 p.

Mansfield, B. and J. Haas. 2006. Scale framing of scientific uncertainty in controversy over the endangered Steller sea lion. Environ. Politics 15(1):78-94.

Mantua, N.J., S.R. Hare, Y. Zhang, J.M. Wallace, and R.C. Francis. 1997. A Pacific Interdecadal Climate Oscillation with Impacts on Salmon Production. Bulletin of the American Meteorological Society 78(6).

Markussen, N.H., M. Ryg, and C. Lydersen. 1992. Food Consumption of the NE Atlantic Minke Whale (*Balaenoptera acutorostrata*) Population Estimated with a Simulation Model. ICES J. Mar. Sci. 49:317-323.

Martin, A.R., S.K. Katona, D.K. Mattila, D. Hembree, and T.D. Waters. 1984. Migration of Humpback Whales Between the Caribbean Aand Iceland. J. Mammal. 65:330-333.

Martineau, D., P. Beland, C. Desjardins, and A. Lagace. 1987. Levels of Organochlorine Chemicals in Tissues of Beluga Whales (*Delphinapterus leucas*) from the St. Lawrence Estuary, Quebec, Canada. Archives of Environmental Contaminants and Toxicology, 16: 137-147.

Martineau, D., S. Lair, S. DeGuise, T.P. Lipscomb and P. Beland. 1999. Cancer in Beluga Whales from the St. Lawrence Estuary, Quebec, Canada: A Potential Biomarker of Environmental Contamination. Journal of Cetacean Research and Management 1:249-265.

Marty, G.D., T.J.I. Quinn, G. Carpenter, T.R. Meyers, and N.H. Willits. 2003. Role of Disease in Abundance of a Pacific Herring *(Clupea pallasi)* Population. Canadian Journal of Fisheries and Aquatic Sciences, 60(10) 1258-1265.

Maschner, H.D.G, K. Reedy-Maschner, A.M. Tews, and M. Livingston. Undated. Unpublished manuscript. Anthropological Investigations on the Decline of the Steller Sea Lion in the Western Gulf of Alaska and Southern Bering Sea. Idaho State Univ., Pocatello. 47 p.

Maschner, H.D.G, M.W. Betts, K.L. Reedy-Maschner, and A.W. Trites. 2008. A 4500-year time series of Pacific cod (Gasus macrocephalus) size and abundance: archaeology, oceanic regime shifts, and sustainable fisheries. Fish. Bull. 106(4):386-394.

Maschner, H.D.G., A.W. Trites, K.L. Reedy-Maschner, M. Betts, A.M. Tews, and M.L. Livingston. 2010. Indigenous, Ethnohistoric and Archaeological Perspectives on the Decline of Steller Sea Lions (Eumetopias jubatus) in Alaska. Manuscript in review: Fish and Fisheries. 70 p.

Matarese, A.C., D.M. Blood, S.J. Picquelle, J.L. Benson. 2003. Atlas of Abundance and Distribution Patterns of Ichthyoplankton from the Northeast Pacific Ocean and Bering Sea Ecosystems Based on Research Conducted by the Alaska Fisheries Science Center (1972-1996). NOAA Professional Paper 1:281 pp.

Mate, B.R. 1973. Population Kinetics and Related Ecology of the Northern Sea Lion, *Eumetopias jubatus*, and the California Sea Lion, Zalophus Californianus, Along the Oregon Coast. Ph.D. dissertation, University of Oregon, 94 pp.

Mathisen, O.A., and R.J. Lopp. 1963. Photographic Census of the Steller Sea Lion Herds in Alaska, 1956-58. U.S. Fish and Wildl. Serv. Spec. Sci. Rep. Fish. No. 424. 20 pp.

Mathisen, O.A., R.T. Baade, and R.J. Lopp. 1962. Breeding Habits, Growth and Stomach Contents of the Steller Sea Lion in Alaska. Journal of Mammalogy, 43:469-477.

Matkin, C., Barrett-Lennard, L., Yurk, H., Ellifrit, D., Trites, A.W. In Press. Ecotypic Variation and Predatory Behavior of Killer Whales (Orcinus orca) in the Eastern Aleutian Islands, Alaska. Fisheries Bulletin.

Matkin, C., L.G. Barrett-Lennard, H.Yurk, D. Ellifrit, and A.W. Trites. 2006. Ecotypic Variation and Predatory Behavior of Killer Whales (*Orcinus orca*) in the Eastern Aleutian Islands, Alaska. Fishery Bulletin, In Press.

Matkin, C.O., E. Saulitis, D. Maldini, J. Maniscalco and L. Mazzuca. 2005. Steller Sea Lion Predation by Killer Whales in Kenai Fjords/Prince William Sound, Alaska. Pages 212-226 *In:* T.R. Loughlin, S. Atkinson and D.G. Calkins, (Eds.) Synopsis of Research on Steller Sea Lions: 2001-2005. Alaska SeaLife Center's Steller Sea Lion Research Program, Seward, Alaska.

Matkin, C.O., G. Ellis, P. Olesiuk, and E. Saulitis. 1999. Association Patterns and Inferred Genealogies of Resident Killer Whales, Orcinus Orca, In Prince William Sound, Alaska. Fish. Bull. vol. 97:900-919.

Matkin, G.O. and F.H. Fay. 1980. Marine Mammal Fishery Interactions on the Copper River and in Prince William Sound, Alaska, 1978. Prep. U.S. Mar. Mamm. Comm., Report No. MMC-78107. 71pp.

Matthiopoulos, J., S. Smout, A.J. Winship, D. Thompson, I.L. Boyd, and J. Harwood. 2008. Getting beneath the surface of marine mammal-fisheries competition. Mammal Rev. 38(2&3):167-188.

McCafferty, D.J., I.L. Boyd, T.R. Walker, and R.I. Taylor. 1998. Foraging responses of Antarctic fur seals to changes in the marine environment. Mar. Ecol. Prog. Ser. 166:285-299.

McDermott, S.F., L.W. Fritz, V. Haist. 2005. Estimating Movement and Abundance of Atka Mackerel (*Pleurogrammus monopterygius*) with Tag–Release–Recapture Data. Fisheries Oceanography 14(s1):113-130.

McDermott, S.F., E. Logerwell, I. Ortiz, and V. Haist. 2009. Fishery Interaction and Availability of Atka Mackerel Prey for Steller Sea Lions: Results from a Local Abundance and Movement Study of Atka Mackerel. ICES CM 2009 J03.

McDonald, M. A., and C. G. Fox. 1999. Passive acoustic methods applied to fin whale population density estimation. J. Acoust. Soc. Am. 105(5):2643-2651.

McKenzie, J., and K.M. Wynne. 2008. Spatial and Temporal Variation in the Diet of Steller Sea Lions in the Kodiak Archipelago, 1999-2005. Marine Ecology Progress Series 360:265-283.

Meekan, M.G. and L. Fortier. 1996. Selection for Fast Growth During the Larval Life of Atlantic Cod (*Gadus morhua*) on the Scotian Shelf. Mar. Ecol. Prog. Ser. 137:25-37.

Megrey, B.A., A.B. Hollowed, S.R. Hare, S.A. Macklin, and P.J. Stabeno. 1996. Contributions of FOCI Research to Forecasts of Year-Class Strength of Walleye Pollock in Shelikof Strait, Alaska. Fish. Oceanogr. 5:189-203.

Megrey, B.A., and V.G. Wespestad. 1990. Alaskan Groundfish Resources: 10 Years of Management Under the Magnuson Fishery Conservation and Management Act. N. Am. J. Fish. Manage. 10(2):125-143.

Mellinger, D.K., K.M. Stafford, and C.G. Fox. 2004. Seasonal Occurrence of Sperm Whale (*Physeter macrocephalus*) Sounds in the Gulf of Alaska, 1999-2001. Mar. Mammal Sci. 21(1):48-62.

Merrick, R.L. 1987. Behavioral and Demographic Characteristics of Northern Sea Lion Rookeries, M.S., Oregon State University, Corvallis, Oregon.

Merrick, R.L. 1995. The Relationship of the Foraging Ecology of Steller Sea Lions (*Eumetopias jubatus*) to Their Population Decline in Alaska. Unpublished Ph.D. dissertation, Univ. of Washington. 171 pp.

Merrick, R.L. 1997. Current and Historical Roles of Apex Predators in the Bering Sea Ecosystem. J. Northw. Atl. Fish. Sci. 22:343-355.

Merrick, R.L., and D.G. Calkins. 1996. Importance of Juvenile Walleye Pollock, *Theragra chalcogramma*, *In:* The Diet of Gulf of Alaska Steller Sea Lions, *Eumetopias jubatus*. Pages 153-166 *In:* U.S. Dep. of Commer., NOAA Tech. Rep. NMFS, 126 p.

Merrick, R.L., and T.R. Loughlin. 1997. Foraging Behavior of Adult Female and Young-Of-The-Year Steller Sea Lions (*Eumetopias jubatus*) *In:* Alaskan Waters. Can. J. of Zool. 75(5):776-786.

Merrick, R.L., M.K. Chumbley, and G.V. Byrd. 1997. Diet Diversity of Steller Sea Lions (*Eumetopias jubatus*) and Their Population Decline in Alaska: A Potential Relationship. Can. J. Fish. Aquat. Sci. 54:1342-1348.

Merrick, R.L., M.K. Maminov, J.D. Baker, and A.D. Makhnyr. 1990. Results of The U.S.-U.S.S.R. Joint Marine Mammal Research Cruise in the Kuril Aand Aleutian Islands 6 June-24 July 1989. U.S. Dept. Commerce NOAA Tech. Memo. NMFS-F/NWC-177, 63 pp.

Merrick, R.L., P.J., Gearin, S. Osmek, and D. Withrow. 1988. Field Studies of Northern Sea Lions at Ugamak Island, Alaska, During 1985 and 1986 Breeding Seasons. NOAA Technical Memorandum, NMFS-F/NWC-143, DOC, NOAA p. 60.

Merrick, R.L., R. Brown, D.G. Calkins, and T.R. Loughlin. 1995. A Comparison of Steller Sea Lion, *Eumetopias jubatus*, Pup Masses Between Rookeries with Increasing and Decreasing Populations. Fish. Bull. 93:753-758.

Merrick, R.L., T.R. Loughlin, and D.G. Calkins. 1987. Decline in Abundance of the Northern Sea Lion, *Eumetopias jubatus*, in 1956-86. Fish. Bull., U.S. 85:351-365.

Merrick, R.L., T.R. Loughlin, G.A. Antonelis, and R. Hill. 1994. Use of Satellite-Linked Telemetry to Study Steller Sea Lion and Northern Fur Seal Foraging. Pol. Res. 13:105-114.

Methot, R.D. 1990. Synthesis Model: An Adaptable Framework for Analysis of Diverse Stock Assessment Data. INPFC Bull. 50:259-277.

Milette, L.L. and A.W. Trites. 2003. Maternal Attendance Patterns of Lactating Steller Sea Lions (*Eumetopias jubatus*) from a Stable and a Declining Population in Alaska. Canadian Journal of Zoology 81:340-348.

Miller, P.J.O., M.P. Johnson, P.L. Tyack. 2004. Sperm Whale Behavior Indicates the Use of Echolocation Click Buzzes Creaks in Prey Capture. Proc Biol Sci. 2004 November 7; 271(1554):2239-2247.

Minerals Management Service (MMS). (2003) Minerals Management Service: Our ocean role. OCS Report 2003-058. Minerals Management Service, Herndon, Virginia.

MMS. 2006. Draft Proposed Program, Outer Continental Shelf Oil and Gas Leasing Program, 2007-2012: MMS document posted at website: http://www.mms.gov/ooc/press/2006/press0208.htm; Copies may be obtained from: Renee Orr, 5-Year Program Manager, Minerals Management Service (MS-4010) Room 3120, 381 Elden Street Herndon, Virginia 20170, February 2006, 25 p.

Mizroch, S.A., and D.W. Rice. 2006. Have North Pacific Killer Whales Switched Prey Species in Response to Depletion of the Great Whale Populations? Marine Ecology Progress Series 310:235-246.

Mizroch, S.A., L.M. Herman, J.M. Straley, D. Glockner-Ferrari, C. Jurasz, J. Darling, S. Cerchio, C. Gabriele, D .Salden, O. von Ziegesar. 2004. Estimating the Adult Survival Rate of Central North Pacific Humpback Whales. J. Mammal. 85(5).

Mizroch, S.A., D.W. Rice, D. Zwiefelhofer, J. Waite, and W.L. Perryman. 2009. Distribution and movements of fin whales in the North Pacific Ocean. Mamm. Rev. 39(3):193-227.

Mobley, Jr., J. R., M. Smultea, T. Norris, and D. Weller. 1996. Fin whale sighting north of Kaua’i, Hawai’i. Pacific Science. 50(2):230-233.

Mobley, J.A., A.S. Bhat, and R.W. Brueggemeier. 1999. Measurement of Oxidative DNA Damage by Catechol Estrogens and Analogues in Vitro. Chem. Res. Toxicol. 12:270-277.

Mobley, J.R., Jr., S.S. Spitz, R. Grotenfendt, P. Forestell, A. Frankel and G. Bauer. 2001. Abundance of Humpback Whales in Hawaiian Waters: Results of 1993-2000 Aerial Surveys. Report to the Hawaiian Islands Humpback Whale National Marine Sanctuary. 16 pp.

Mohamed M., T. Burbacher, N. Mottet. 1987.Effects of methyl mercury on testicular functions in Macaca fascicularis monkeys. Pharmacol. Toxicol. 60(1):29-36.

Mohl, B., M. Wahlberg, and P.T. Madsen. 2003. The Monopulsed Nature of Sperm Whale Clicks. The Journal of the Acoustical Society of America 114(2):1143-1154.

Moles, A. and R.A. Heintz. 2007. Parasites of forage fishes in the vicinity of Steller sea lion (*Eumetopias jubatus*) habitat in Alaska. Journal of Wildlife Diseases, 43(3):366–375.

Monnett, C., and L.M. Rotterman. 1989. Distribution and Abundance of Sea Otters in Southeastern Prince William Sound. U.S. Fish and Wildlife Service Report. 30 pp.

Moore, S. E., K. M. Stafford, M. E. Dahlheim, C. G. Fox, H. W. Braham, J. J. Polovina, and D. E. Bain. 1998. Seasonal variation in reception of fin whale calls at five geographic areas in the North Pacific. Mar. Mamm. Sci. 14(3):617-627.

Moore, S.E., J.M. Waite, L.L. Mazzuca, and R.C. Hobbs. 2000. Provisional Estimates of Mysticete Whale Abundance on the Central Bering Sea Shelf. J. Cetacean Res. Manag. 2(3):227-234.

Moore, S.E., J.M. Waite, N.A. Friday and T. Honkalehto. 2002. Distribution and Comparative Estimates of Cetacean Abundance on the Central and South-Eastern Bering Sea Shelf with Observations on Bathymetric and Prey Associations. Progr. Oceanogr. 55(1-2):249-262.

Moore, S. E., K. M. Stafford, D. K. Mellinger, C. G. Hildebrand. 2006. Listening for large whales in the offshore waters of Alaska. BioScience. 56(1):49-55.

Morris, B.F., M.S. Alton, and H.W. Braham. 1983. Living Marine Resources of The Gulf of Alaska: A Resource Assessment for the Gulf of Alaska/Cook Inlet Proposed Oil and Gas Lease Sale 88. *In:* U.S. Dep. Commer. NOAA, NMFS.

Mueter, F.J. 2007. Total Annual Surplus Production and Overall Exploitation Rate of Groundfish, P. 216-218. *In:* J.L. Boldt (Ed.) Ecosystem Considerations for 2008. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Mueter, F.J., S.D. Batten, P.J. Brickley, A.R. Childers, S. Danielson, B.A. Megrey, P.R. Mundy, T.C. Royer, and T.J. Weingartner. 2004. Gulf of Alaska. P. 153-175 in Marine Ecosystems of the North Pacific. PICES Special Publication Number 1. North Pacific Marine Science Organization, Sidney, B.C., Canada. 280 p.

Mueter, F.J., C.A. Ladd, M.C. Palmer, and B.L. Norcross. 2006. Bottom-Up and Top-Down Controls of Walleye Pollock (*Theragra chalcogramma*) on the Eastern Bering Sea Shelf. Prog. Oceanogr. 68:152-183.

Mueter, F.J., J. Boldt, B.A. Megrey, and R.M. Peterman. 2007. Recruitment and Survival of Northeast Pacific Ocean Fish Stocks: Temporal Trends, Covariation, and Regime Shifts. Can. J. Fish. Aquat. Sci. 64(6):911-927.

Mundy, P.R. (editor). 2005. The Gulf of Alaska: Biology and Oceanography. Alaska Sea Grant Program, Fairbanks, Alaska. 214 p.

Myers, M.J., and S. Atkinson. 2005. Thyroid and Cortisol Hormones and Contaminants in Steller Sea Lions. Chapter 16, Pages 147-158, *In:* T.R. Loughlin, S. Atkinson, and D.G. Calkins (Eds.) Synopsis of Research on Steller Sea Lions: 2001-2005. Alaska SeaLife Center's Steller Sea Lion Program. Sea Script Company, Seattle, Washington. 344 p.

Myers, M.J., G.M. Ylitalo, M.M. Krahn, D. Boyd, D. Calkins, V. Burkanov, and S. Atkinson. 2008. Organochlorine contaminants in endangered Steller sea lion pups (*Eumetopias jubatus*) from western Alaska and the Russian Far East. Science of the Total Environment 396(1):60-69.

Myers, M.J. 2008. Organochlorines in Steller sea lions (*Eumetopias jubatus*). PhD. dissertation, University of Alaska Fairbanks, Fairbanks, Alaska. 126 p.

Myers, R., S.P. Hume, S. Ashworth, A.A. Lammertsma, P.M. Bloomfield, S. Rajeswaran, and T. Jones. 1996. Quantification of Dopamine Receptors and Transporter in Rat Striatum Using a Small Animal PET Scanner. *In:* Quantification of Brain Function Using PET (R. Myers, V.J. Cunningham, D.L. Bailey, and T. Jones, (Eds.) pp. 12-15. Academic Press, San Diego.

Nakatani, T., and T. Maeda. 1984. Thermal Effect on the Development of Walleye Pollock Eggs and their Upward Speed to the Surface. Bull. Japan. Soc. Sci. Fish 50:937-942.

Napp, J., and N. Shiga. 2005. Bering Sea Zooplankton. *In:* J.L. Boldt (Ed.) Ecosystem Considerations for 2006. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Napp, J.M., C.T. Baier, K.O. Coyle, R.D. Brodeur, N. Shiga, K. Mier. 2002. Interannual and Decadal Variability in Zooplankton Communities of the Southeastern Bering Sea. Deep-Sea Research II 49:5991-6008.

Napp, J.M., L.S. Incze, P.B. Ortner, D.L.W. Siefert, and L. Britt. 1996. The Plankton of Shelikof Strait, Alaska: Standing Stock, Production, Mesoscale Variability and Their Relevance to Larval Fish Survival. Fisheries Oceanography 5(s1):19-38.

National Academy of Science (NAS). 1996. The Bering Sea Ecosystem. National Academy Press, Washington, D.C. 307 pp.

National Marine Fisheries Service (NMFS). 1991. Recovery Plan for the Humpback Whale (*Megaptera novaeangliae*). Prepared by the Humpback Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland. 105 pp.

NMFS. 1992a. Recovery Plan for the Steller Sea Lion (*Eumetopias jubatus*). Prepared by the Steller Sea Lion Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland 20910. p. 92.

NMFS. 1992b. Endangered Species Act - Section 7 Biological Assessment - Groundfish Fisheries in the Bering Sea/Aleutian Islands and Gulf of Alaska Regions Managed Under the North Pacific Fishery Management Council's Fishery Management Plans. DOC, NOAA, National Marine Fisheries Service, P.O. Box 21668, Juneau, Alaska 99802-1668. p. 12.

NMFS. 1995. Status Review of the United States Steller Sea Lion, *Eumetopias jubatus*, Population. U.S. Dep. Commer., NOAA, National Marine Mammal Laboratory, AFSC, 7600 Sand Point Way NE, Seattle, Washington 98115. 61 pp.

NMFS. 1998. Section 7 Consultation on the Authorization of the Bering Sea and Aleutian Islands Groundfish Fishery for Walleye Pollock Under the BSAI FMP, on the Authorization of the Bering Sea and Aleutian Islands Atka Mackerel Fishery Under the BSAI FMP, and the Authorization of the Gulf of Alaska Groundfish Fishery for Walleye Pollock Under the GOA FMP, Between 1999 and 2002. Office of Protected Resources, NMFS. Dec. 3, 1998.

NMFS. 1998a. Final Supplemental Environmental Impact Statement: Groundfish Total Allowable Catch Specifications and Prohibited Species Catch Limits Under the Authority of the Fishery Management Plans for the Groundfish Fishery of the Bering Sea and Aleutian Islands Area and Groundfish of the Gulf of Alaska. DOC, NOAA, National Marine Fisheries Service, Alaska Region, P.O. Box 21668, Juneau, Alaska 99802-1668. p. 692 + Appendices and Comments.

NMFS. 1998b. Endangered Species Act Section 7 Consultation - Biological Opinion: Authorization of an Atka Mackerel Fishery Under the BSAI Groundfish FMP Between 1999 and 2002; Authorization of a Walleye Pollock Fishery Under the BSAI Groundfish FMP Between 1999 and 2002; and Authorization af a Walleye Pollock Fishery Under the GOA Groundfish FMP Between 1999 and 2002 (Marine Mammals Including Steller Sea Lion). DOC, NOAA, National Marine Fisheries Service, P.O. Box 21668, Juneau, Alaska 99802-1668.

NMFS. 1999. Section 7 Consultation on the Bering Sea and Aleutian Islands Groundfish Fishery for Walleye Pollock Under the BSAI FMP and the Gulf of Alaska Groundfish Fishery for Walleye Pollock Under the GOA FMP for 2000. Office of Protected Resources, NMFS. December 1999.

NMFS. 2000. Endangered Species Act Section 7 Consultation Biological Opinion and Incidental Take Statement for Continued Authorization of the Bering Sea/Aleutian Islands Groundfish Fisheries Based on the Fishery Management Plan for the Bering Sea/Aleutian Islands Groundfish and the Gulf of Alaska Groundfish Fisheries Based on the Fishery Management Plan for Groundfish of the Gulf of Alaska. November 2000, NMFS Protected Resources Division. Alaska Region, available from http://www.fakr.noaa.gov/protectedresources/stellers/plb/default.htm.

NMFS. 2000b. Environmental Assessment for an Emergency Rule to Implement Steller Sea Lion Protection Measures in the Pollock Fisheries of the Bering Sea and Aleutian Islands Area and the Gulf of Alaska. FONSI Determination January 11, 2001, DOC, NOAA, National Marine Fisheries Service, Alaska Region, P.O. Box 21668, Juneau, Alaska 99802. p. 282.

NMFS. 2001a. Environmental Assessment/Regulatory Impact Review for the Extension and Revision of the Emergency Interim Rule for 2001 Harvest Specifications for the Alaska Groundfish Fisheries and for Steller Sea Lion Protective Fisheries Management Measures. Juneau, Alaska.

NMFS. 2001b. Supplemental Environmental Impact Statement for Steller Sea Lion Protection Measures for the BSAI and GOA Groundfish Fisheries for 2002 and Beyond, Including Amendments 70/70 to the Groundfish Fishery Management Plans. Alaska Regional Office, Juneau, Alaska.

NMFS. 2001c. Notice of Intent to Prepare a Supplemental Environmental Impact Statement for the EFH Components of North Pacific Fishery Management Council Fishery Management Plans. 66 Federal Register 30396 (June 6, 2001). Alaska Regional Office, Juneau, Alaska.

NMFS. 2002. Final Rule Regarding the Taking and Importing of Marine Mammals; Taking Marine Mammals Incidental to Navy Operations of Surveillance Towed Array Sensor System Low Frequency Active Sonar, Publ. 67 FR 46712, July 16, 2002.

NMFS. 2003. Supplement to the 2001 Endangered Species Act, Section 7 Consultation, Biological Opinion and Incidental Take Statement on the Authorization of the Bering Sea/Aleutian Islands and Gulf of Alaska Groundfish Fishery Management Plan Amendments 61 And 70. NMFS Alaska Region, Protected Resources Division, Juneau, Alaska.

NMFS. 2004. Final Programmatic Supplemental Environmental Impact Statement on the Alaska Groundfish Fisheries. U.S. Dep. Commer., NOAA, NMFS, Alaska Region, Juneau, Alaska.

NMFS. 2006. Fisheries Statistics Division, Foreign Trade Information from National Marine Fisheries Service website: www.st.nmfs.noaa.gov/st1/trade/build_a_database/TradeSelectIndex.html

NMFS. 2006a. Biological Assessment of the Alaska Groundfish Fisheries and NMFS Managed Endangered Species Act Listed Marine Mammals and Sea Turtles. April 2006. NMFS Alaska Region, Sustainable Fisheries Division, P.O. Box 21688, Juneau, Alaska 99802.

NMFS. 2006b. Responses to a request for information on 59 topic areas relevant to the consultation based on "Request for assistance on an Endangered Species Act (ESA) Section 7 consultation on the effects of the Groundfish Fishery Management Plans (FMPs) for the Bering Sea and Aleutian Islands Management Area (BSAI) and Gulf of Alaska (GOA) on Steller sea lions and other listed species, Memorandum for Doug DeMaster, Sue Salveson, and Steve Davis from Robert D. Mecum, March 13, 1006."

NMFS. 2006c. Draft Recovery Plan for the Sperm Whale.

NMFS. 2006d. Draft Negligible Impact Determination for the Central North Pacific Stock of Humpback Whales. NMFS Protected Resources Division, Hawaii and Alaska Regions.

NMFS. 2007. Fisheries of the United States 2005. National Oceanic and Atmostpheric Administrations, Silver Spring, Maryland.

NMFS. 2007b. Fisheries Statistics & Economics: Foreign & Domestic Trade Database. National Marine Fisheries Service, NOAA/DOC. Accessed February 2007. Available at: http://www.st.nmfs.gov/st1/trade/ cumulative_data/TradeDataProduct.html

NMFS. 2008a. Recovery Plan for the Steller Sea Lion (*Eumetopias jubatus*). Revision. NMFS, Silver Spring, Maryland, 325 pp.

NMFS. 2008b. Conservation Plan for the Cook Inlet Beluga Whale (*Delphinapterus leucas*). NMFS, Juneau, Alaska; 122 pp.

NMFS. 2009. Our Living Oceans. Draft Report on the Status of U.S. Living Marine Resources, 6th Edition. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-F/SPO-80. 353 p.

NMFS. 2009a. Grouper Fishery Trends in the Gulf of Mexico, 2004-2008. SERO-LAPP-2009-01. 71 pp.

NMFS. 2009b. Summary of Winter and Summer Eastern Gulf of Mexico Aerial Survey Data for Loggerhead Turtle Distribution (Update: Data summary restricted to areas east of -85.50 degrees longitude). NMFS Southeast Fisheries Science Center Contribution, 25 February 2009. 7 p. Atmospheric Administration, ftp://ftp.cmdl.noaa.gov/ccg/co2/trends/co2_mm_gl.txt.

NMFS. 2009c. Sperm Whale *(Physeter macrocephalus)* 5 Year Review: Summary and Evaluation. January 2009. National Marine Fisheries Service, Silver Spring, MD. 44 pp.

NMFS. 2010. Recovery plan for the fin whale (*Balaenoptera physalus*). National Marine Fisheries Service, Silver Spring, MD. 121 pp.

National Research Council (NRC), Committee on the Bering Sea Ecosystem. 1996. The Bering Sea Ecosystem. National Academy Press, Washington, D.C.

NRC. 1998. Nutrient Requirements of Swine. 10th ed. National Academy Press, Washington, D.C.

NRC. 2003. Decline of the Steller Sea Lion in Alaskan Waters: Untangling Food Webs and Fishing Nets. National Academy Press, Washington, D.C. 204 pp.

NRC. 2006. Guidelines for the Humane Transportation of Research Animals. Washington DC: National Academies Press.

Neilson, J.L., C.M. Gabriele. (2008) Results of Humpback Whale Monitoring in Glacier Bay and Adjacent Waters. Annual Progress Report. Glacier Bay National Park and Preserve. Division of Resource Management.

Neilson, R.P., L.F. Pitelka, A. Solomon, R. Nathan, G.F. Midgley, J. Fragoso, H. Lischke, and K. Thompson. 2005. Forecasting Regional to Global Plant Migration in Response to Climate Change: Challenges and Directions. Bioscience, 55:749-759.

Nelson, E.W., (Ed.) 1887. Mammals. *Eumetopias stelleri*. Pages 266-267 *In:* Natural History Collections Made in Alaska Between the Years 1877 and 1881, Report III, H.W. Henshaw, (Ed.) U.S. Government Printing Office, Washington, D.C.

Nemoto, T. 1957. Foods of Baleen Whales in the Northern Pacific. Sci. Rep. Whales Res. Inst. 12:33-89.

Nemoto, T. 1978. Humpback Whales Observed within the Continental Shelf Waters of the Eastern Bering Sea. Scientific Reports of the Whales Research Institute 30:345-247.

Nemoto, T. 1970. Feeding pattern of baleen whales in the oceans, pp. 241–252 in Marine food chains, ed. J.H. Steele. Univ. of California Press, Berkeley.

Nikulin, P.G. 1937. The Sea Lion of the Okhotsk Sea and Its Hunting. Pages 35-48 *In:* On the Hunting and Utilization of Sea Animals and the Sharks in the Far East. Bull. Pac. Sci. Inst. Fish. and Oceanogr. 10. [In Russian with English summary].

Nikulin, P.G. 1946. About the Distribution of Whales in Seas Surrounding the Chukotka Peninsula. Tikhookeanski nauchno-issledov. inst. rybnogo khoz. i okeanogr Izvestiya, 22:255-7. [In Russian].

Nikulin, P.G. 1946. About the Distribution of Whales in Seas Surrounding the Chukotka Peninsula. Tikhookeanski nauchno-issledov. Inst. Rybnogo khoz. I okeanogr. Izvestiya, 22:255-7. [In Russian].

Nishiwaki, M. 1959. Humpback Whales in Ryukyuan Waters. Sci. Rep. Whales Res. Inst. Tokyo 14:49-86.

Nishiwaki, M. 1966. Distribution and Migration of the Larger Cetaceans in the North Pacific as Shown by Japanese Whaling Results. Pages 171-191 *In:* K.S. Norris (Ed.) Whales, Dolphins and Porpoises. University of California Press, Berkeley, California.

Norcross, B.L., E.D. Brown, R.J. Foy, M. Frandsen, S.M. Gay, T.C. Kline, Jr., D.M. Mason, E.V. Patrick, A.J. Paul, and K.D.E. Stokesbury. 2001. A Synthesis of the Life History and Ecology of Juvenile Pacific Herring in Prince William Sound, Alaska. Fish. Oceanogr. 10(s1):42-57.

Noren, D.P., L.D. Rea, and T.R. Loughlin. 2009. A model to predict fasting capacities and utilization of body energy stores in weaned Steller sea lions (*Eumetopias jubatus*) during periods of reduced prey availability. Can. J. Zool. 87:852-864.

North Pacific Fishery Management Council (NPFMC). 2005a. Fishery Management Plan for Groundfish of the Bering Sea and Aleutian Islands Management Area. Available from http://www.fakr.noaa.gov/npfmc/default.htm.

North Pacific Fishery Management Council (NPFMC). 2009a. Fishery Management Plan for Groundfish of the Gulf of Alaska. North Pacific Fishery Management Council, Anchorage, AK, January 2009. 126 pp.

North Pacific Fishery Management Council (NPFMC). 2009b. Fishery Management Plan for Groundfish of the Bering Sea and Aleutian Islands Management Area. NPFMC, Anchorage, Alaska, January 2009. 139 pp.

NPFMC. 2005b. Fishery Management Plan for Groundfish of the Gulf of Alaska. Available from http://www.fakr.noaa.gov/npfmc/default.htm.

NPFMC. 2009. North Pacific Fishery Management Council, Stock Assessment and Fishery Evaluation (SAFE) Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions. http://www.afsc.noaa.gov/refm/stocks/assessments.htm

NPFMC. 2009. North Pacific Fishery Management Council, Stock Assessment and Fishery Evaluation (SAFE) Report for the Groundfish Resources of the Gulf of Alaska. http://www.afsc.noaa.gov/refm/stocks/assessments.htm

North Pacific Marine Science Organization. 2010. PICES North Pacific Ecosystem Status Report. http://www.pices.int.

O'Corry-Crowe, G., W. Lucey, C. Bonin, E. Henniger, and R. Hobbs. 2006. The Ecology, Status and Stock Identity of Beluga Whales, *Delphinapterus leucas*, in Yakutat Bay, Alaska. Report to the U.S. Marine Mammal Commission, February 2006. 22p.

O'Corry-Crowe, G., B.L. Taylor, T. Gelatt, T.R. Loughlin, J. Bickham, M. Basterretche, K.W. Pitcher, and D.P. DeMaster. 2006. Demographic independence along ecosystem boundaries in Steller sea lions revealed by mtDNA analysis: implications for management of an endangered species. Can. J. Zool. 84:1796-1809.

Odum, EP. 1971. Fundamentals of Ecology. Third Edition. Saunders College Publishing; Philadelphia, Pennsylvania.

Ohsumi, S. 1986. Yearly change in age and body length at sexual maturity of a fin whale stock in the eastern North Pacific. Sci. Rep. Whales Res. Inst. 37:1–16.

Olesiuk, PF. 2004. Status of Sea Lions (*Eumetopias jubatus* and *Zalophus californianus*) Wintering Off Southern Vancouver Island. NMMRC Working Paper No. 2004-03 (DRAFT).

Olla, B.L., M.W. Davis, C.H. Ryer, and S.M. Sogard. 1996. Behavioral Determinants of Distribution and Survival in Early Stages of Walleye Pollock, *Theragra chalcogramma*: A Synthesis of Experimental Studies. Fisheries Oceanography. 5(s1):167-178.

Omura, H. and S. Ohsumi. 1964. A review of Japanese whale marking in the North Pacific to the end of 1962, with some information on marking in the Antarctic. Norsk Hvalfangst-Tidende 4:90-112.

Orensanz, J.M., J. Armstrong, D. Armstrong, and R. Hilborn. 1998. Crustacean Resources are Vulnerable to Serial Depletion-The Multifaceted Decline of Crab and Shrimp Fisheries in the Greater Gulf of Alaska. Reviews in Fisheries Science. 8:117-176.

Oritz, I., and E.A. Logerwell. In prep. Evaluating the Efficacy of Trawl Exclusion Zones for Preserving Prey Fields of Steller Sea Lions Foraging on Atka Mackerel. II. Small-Scale Food Web Models to Estimate Sea Lion Consumption of Available Atka Mackerel Production.

Orlov, A.M. 1999. Capture of Especially Large Sleeper Shark [*Somniosus pacificus* (Squalidae)] with Some Notes on its Ecology in Northwestern Pacific. Journal of Ichthyology. 39:548-553.

Orr, R.T., and T.C. Poulter. 1967. Some Observations on Reproduction, Growth, and Social Behavior in the Steller Sea Lion. Proc. California Acad. Sci. 35:193-226.

Ortiz, I. 2007. Ecosystem Dynamics in the Aleutian Islands. Ph.D. Dissertation, University of Washington School of Aquatic and Fishery Sciences. March 2007.

Ortiz, I., and L. Logerwell. 2010. Evaluating the Efficacy of Trawl Exclusion Zones for Preserving Prey Fields of Steller Sea Lions Foraging on Atka Mackerel. II. Site Specific Estimates to Evaluate Availability of Atka Mackerel Production for Sea Lion Consumption. Unpublished report. Available at Alaska Fisheries Science Center, NOAA Fisheries, 7600 Sand Point Way, NE, Seattle WA. 28 pp.

Ohsumi, S. and S. Wada. 1974. Status of whale stocks in the North Pacific, 1972. Rep. Int. Whal. Commn. 24:114–126.

Osterhaus, A., J. Groen, H. Niesters, M. van de Bildt, B. Martina, L. Vedder, J. Vos, H. Van Egmond, B. Abou Sidi, and ME. Barham. 1997. Morbillivirus in Monk Seal Mass Mortality. Nature 388:838-839.

Overland, J.E., J.M. Adams, and N.A. Bond. 1999. Decadal Variability of the Aleutian Low and its Relation to High-Latitude Circulation. J. Climate. 12:1542 1548.

Palmer, MC. 2003. Environmental Controls of Fish Growth in the Southeastern Bering Sea. *In:* Boldt, JL. (Ed.) Ecosystem Considerations for 2004. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, AK 99501.

Panina, G.K. 1966. On the Feeding of the Sea Lion and Seals on the Kuril Islands. Izv. TINRO. 58:235-236. [In Russian.] [Transl. by Bur. Commer. Fish., Off. Foreign Fish., U.S. Dep. Interior, Washington, D.C.].

Parsons, T.R. 1987. Ecological Relations. Ch 18, p 561-570 *In:* Hood, DW., and ST. Zimmerman (Eds.) The Gulf of Alaska, Physical Environment and Biological Resources. U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Pascual, M.A., and M.D. Adkison. 1994. The Decline of the Steller Sea Lion in the Northeast Pacific: Demography, Harvest or Environment. Ecol. Applications 4:393-403.

Paulson, A.C., and R.L. Smith. 1977. Latitudinal Variation of Pacific Herring Fecundity. Trans. Am. Fish. Soc. 106(3):244-247.

Pendleton, G.W., K.W. Pitcher, L.W. Fritz, A.E. York, K.L. Raum-Suryan, T.R. Loughlin, D.G. Calkins, K.K. Hastings, and T.S. Gelatt. 2006. Survival of Steller Sea Lions in Alaska: A Comparison of Increasing and Decreasing Populations. Can. J. Zool. 84:1163-1172.

Perez, M. 2003. Compilation of Marine Mammal Incidental Take Data from the Domestic and Joint Venture Groundfish Fisheries in the U.S. EEZ of the North Pacific, 1989-2001. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-138, 145 p.

Perez, M.A. 1990. Review of Marine Mammal Population and Prey Information for Bering Sea Ecosystem Studies. NOAA Tech. Memo., NMFS-F/NWC-186, U.S. DOC, NOAA. pp.81.

Perez, M.A. 2006. Analysis of Marine Mammal Bycatch Data from the Trawl, Longline, and Pot Groundfish Fisheries of Alaska, 1998-2004, Defined by Geographic Area, Gear Type, and Target Groundfish Catch Species. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-167.

Perez, M.A., and B. McAlister. 1993. Estimates of Food Consumption by Marine Mammals in the Eastern Bering Sea. NOAA Technical Memorandum, NMFS-AFSC-14, U.S. DOC, NOAA. pp.36.

Perez, M.A., and T.R. Loughlin. 1990. Incidental Catch of Marine Mammals by Foreign and Jointventure Trawl Vessels in the U.S. EEZ of the North Pacific, 1973-88. NOAA Tech. Rept. NMFS-104. Seattle, WA. 57 pp.

Perez, M.A., and T.R. Loughlin. 1991. Incidental Catch of Marine Mammals by Foreign and Joint Venture Trawl Vessels in the U.S. EEZ of the North Pacific, 1973-88. NOAA Tech. Rep. NMFS. 104, 57 pp.

Perlov, A.S. 1975. The Food of Sea Lions in the Vicinity of the Kuril Islands. Ekologia. 4:106-108. (translated by S. Pearson, AFSC).

Perlov, A.S. 1996. Harvest of Steller Sea Lion as Major Reason its Decline. Izvestiya TINRO. 121:143-149. [In Russian].

Permyakov, P.A. and V.N. Burkanov. 2009. Interactions between killer whales (Orcinus orca) and steller sea Lions (Eumetopias jubatus) in the vicinity of Brat Chirpoev Island, Kuril Islands. Russian Journal of Marine Biology 35(3):255–258.

Perry, A., C.S. Baker, and L.M. Herman. 1990. Population Characteristics of Individually Identified Humpback Whales in the Central and Eastern North Pacific: A Summary and Critique. Report of the International Whaling Commission. Special Issue 12:308-317.

Perry, S.L., D.P. DeMaster, and G.K. Silber. 1999. The Great Whales: History and Status of Six Species Listed as Endangered Under the U.S. Endangered Species Act of 1973. Mar. Fish. Rev. 61(1).

Peterson, H.A. 2001. Whale behavioral responses and human perceptions: an assessment of humpback whales (*Megaptera novaeangliae*) and vessel activity near Juneau, Alaska. M.E.M. project, Duke University. 73 p.

Phillips, C.D., J.W. Bickham, J.C. Patton, and T.S. Gelatt. 2009. Systematics of Steller Sea Lions (Eumetopias jubatus): Subspecies Recognition Based on Concordance of Genetics and Morphometrics. Occas. Pap. Mus. Texas Tech Univ. 283:1-15.

Piatt, J.F., and P.J. Anderson. 1996. Responses of Common Murres to the Exxon Valdez Oil Spill and Longterm Changes in the Gulf of Alaska Ecosystem. AFS Symposium 18, pp. 720-737.

Pike, G.C., and B.E. Maxwell. 1958. The Abundance and Distribution of the Northern Sea Lion (*Eumetopias jubata* [sic]) on the Coast Of British Columbia. Journal of the Fisheries Research Board of Canada. 15:5-17.

Pitcher, K.W. 1981. Prey of the Steller Sea Lion, *Eumetopias jubatus*, in the Gulf of Alaska. Fish. Bull. 79:467-472.

Pitcher, K.W. 1990. Major Decline in Number of Harbor Seals, *Phoca vitulina richardsi*, on Tugidak Island, Gulf of Alaska. Marine Mammal Science. 6:121-134.

Pitcher, K.W., and D.G. Calkins. 1981. Reproductive Biology of Steller Sea Lions in the Gulf of Alaska. J. Mamm. 62:599-605.

Pitcher, K.W., and F.H. Fay. 1982. Feeding by Steller Sea Lions on Harbor Seals. Murrelet. 63:70-71.

Pitcher, K.W., D.G. Calkins, and G.W. Pendleton. 1998. Reproductive Performances of Female Steller Sea Lions from the Gulf of Alaska: Indications of Nutritional Stress? Can. J. Zool. 76:2075-2083.

Pitcher, K.W., D.G. Calkins, and G.W. Pendleton. 2000. Steller Sea Lion Body Condition Indices. Marine Mammal Science. 16:427-436.

Pitcher, K.W., M.J. Rehberg, G.W. Pendleton, K.L. Raum-Suryan, T.S. Gelatt, U.G. Swain, and M.F. Sigler. 2005. Ontogeny of Dive Performance in Pup and Juvenile Steller Sea Lions in Alaska. Can. J. Zool. 83:1214-123.

Pitcher, K.W., P.F. Olesiuk, R.F. Brown, M.S. Lowry, S.J. Jeffries, J.L. Sease, W.L. Perryman, C.E. Stinchcomb, and L.F. Lowry. 2007. Abundance and Distribution of the Eastern North Pacific Steller Sea Lion (Eumetopias jubatus) Population. Fishery Bulletin. 105(1):102-115.

Pitcher, K.W., V. Burkanov, D.G. Calkins, B.J. LeBoeuf, E.G. Mamaev, R.L. Merrick, and G.W. Pendleton. 2001. Spatial and Temporal Variations in the Timing of Births of Steller Sea Lions. Journal of Mammalogy. 82:1047-1053.

Pitcher, T.J. 1998. Back to the Future: A Novel Methodology and Policy Goal in Fisheries. *In:* Back to the Future: Reconstructing the Strait of Georgia Ecosystem. Pages 4-7. D. Pauly, T.J. Pitcher, and D. Preikshot (Eds.) Fisheries Centre Research Reports 6(5).

Pitman, R.L., L.T. Balance, S.I. Mesnick, and S.J. Chivers. 2001. Killer Whale Predation on Sperm Whales: Observations and Implications. Marine Mammal Science. 17:494-507.

Polovina, J.J. 2005. Climate Variation, Regime Shifts, and Implications for Sustainable Fisheries. Bull. Mar. Sci. 76:233-244.

Polovina, J.J., G.T. Mitchum, and G.T. Evans. 1995. Decadal and Basin-Scale Variation in Mixed Layer Depth and the Impact on Biological Production in the Central and North Pacific, 1960-1988. Deep Sea Research. 42(10):1701-1716.

Porter, B. 1997. Winter Ecology of Steller Sea Lions (*Eumetopias jubatus*) in Alaska. M.S. dissertation, University of British Columbia. 84 pp.

Porter, S.M., L. Ciannelli, N. Hillgruber, K.M. Bailey, K.S. Chan, MF. Canino, and L.J. Haldorson. 2005. Analysis of Factors Influencing Larval Walleye Pollock Theragra Chalcogramma Feeding in Alaskan Waters. Mar. Ecol. Prog. Ser. 302:207-217.

Punt, A.E., J.H.M. David, and R.W. Leslie. 1995. The Effects of Future Consumption by the Cape Fur Seal on Catches and Catch Rates of the Cape Hakes. 2. Feeding and Diet of the Cape Fur Seal Arctocephalus pusillus pusillus. S. Afr. J. Mar. Sci. 16:85-99.

Quinn, T.J. II, and H.J. Niebauer. 1995. Relation of Eastern Bering Sea Walleye Pollock (Theragra chalcogramma) Recruitment to Environmental and Oceanographic Variables. Pages 497-507 *In:* RJ. Beamish (Ed.) Climate Change and Northern Fish Populations. Can. Spec. Publ. Fish. Aquat. Sci. 121.

Quinn, T.J. II, and R.B. Deriso. 1999. Quantitative Fish Dynamics. Oxford University Press, New York, New York, USA.

Quinn, T.J., G.D. Marty, J. Wilcock, and M. Willette. 2001. Disease and Population Assessment of Pacific Herring in Prince William Sound, Alaska. *In:* Herring Expectations for a New Millennium. Pages 363-379. Funk, F., J. Blackburn, D. Hay, AJ. Paul, R. Stephenson, R. Toresen, and D. Witherell (Eds.) Alaska Sea Grant College Program, AK-SG-01-04.

Raum-Suryan, K.L., K.W. Pitcher, D.G. Calkins, J.L. Sease, and T.R. Loughlin. 2002. Dispersal, Rookery Fidelity and Metapopulation Structure of Steller Sea Lions (*Eumetopias jubatus*) in an Increasing and a Decreasing Population in Alaska. Marine Mammal Science. 18:746-764.

Raum-Suryan, K.L., L.A. Jemison, and K.W. Pitcher. 2009. Entanglement of Steller Sea Lions (*Eumetopias jubatus*) in Marine Debris: Identifying Causes and Finding Solutions. Mar. Pollut. Bull. DOI:10.1016/j.marpolbul.2009.06.004

Raum-Suryan, K.L., M.J. Rehberg, G.W. Pendleton, K.W. Pitcher, T.S. Gelatt. 2004. Development of Dispersal, Movement Patterns, and Haul-Out Use by Pup and Juvenile Steller Sea Lions (*Eumetopias jubatus*) In Alaska. Marine Mammal Science 20(4)823-850.

Rea, LD. (1995). Prolonged Fasting in Pinnipeds. Ph.D. dissertation, University of Alaska Fairbanks, Fairbanks, Alaska.

Rea. L.D., M. Berman-Kowalewski, D.A.S. Rosen, and A.W. Trites. 2009. Seasonal differences in biochemical adaptation to fasting in juvenile and subadult Steller sea lions (*Eumetopias jubatus*). Physiol. Biochem. Zool. 82(3):236-247.

Rea. L.D., M.A. Castellini, B.S. Fadely, and T.R. Loughlin. 1998a. Health Status of Young Alaska Steller Sea Lion Pups (*Eumetopias jubatus*) as Indicated by Blood Chemistry and Hematology. Comparative Biochemistry and Physiology Part A. 120:617-623.

Rea, L.D., D.A.S. Rosen, and A.W. Trites. 1998b. Blood Chemistry and Body Mass Changes During Fasting in Juvenile Steller Sea Lions (Eumetopias jubatus). Proceedings of the Proceedings of the Comparative Nutrition Society, Banff, Canada.

Rea, L.D., D.A.S. Rosen, and A.W. Trites. 2000. Metabolic Response to Fasting in 6-week-old Steller Sea Lion Pups (*Eumetopias jubatus*). Canadian Journal of Zoology. 78:890-894.

Rea, L.D., D.A.S. Rosen, and A.W. Trites. 2007. Utilization of stored energy reserves during fasting varies by age and season in Steller sea lions. Can. J. Zool. 85:190-200.

Reed, R.K., and J.D. Schumacher. 1987. Physical Oceanography. Ch 3, Pages 57-75 *In:* DW. Hood, and S.T. Zimmerman (Eds.) The Gulf of Alaska, Physical Environment and Biological Resources. U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Reeves, R.R., and H. Whitehead. 1997. Status of the Sperm Whale, *Physeter macrocephalus*, in Canada. Can. Field Naturalist. 111(2):293-307.

Rehberg, M.J. 2005. Pattern Matters: Changes in the Organization of Swimming and Diving Behavior by Steller Sea Lion Juveniles in Alaska. M.Sc. Thesis, University of Alaska, Anchorage, Alaska.

Rehberg, M.J., and J.M. Burns. 2008. Differences in Diving and Swimming Behavior of Pup and Juvenile Steller Sea Lions (*Eumetopias jubatus*) in Alaska. Can. J. Zool./Rev. Can. Zool. 86 pp.

Rehberg, M.J., R.D. Andrews, U.G. Swain, and D.G. Calkins. 2009. Foraging behavior of adult female Steller sea lions during the breeding season in Southeast Alaska. Mar. Mamm. Sci. 25(3):588–604.

Reijnders, P.J.H. 1986. Reproductive Failure in Common Seals Feeding on Fish from Polluted Coastal Waters. Nature. 324:456-457.

Restrepo, V.R., G.G. Thompson, P.M. Mace, W.L. Gabriel, L.L Low, A.D. MacCall, R.D. Methot. J.E. Powers, B.L. Taylor, P.R. Wade, and J.F. Witzig. 1998. Technical Guidance on the Use of Precautionary Approaches to Implementing National Standard 1 of the Magnuson-Stevens Fishery Conservation and Management Act. NOAA Technical Memorandum NMFS-F/SPO-31.

Rice, D. W. 1974. Whales and whale research in the eastern North Pacific. Pp. 170-195 *In* W. E. Schevill (ed.), The whale problem: A status report. Harvard Press, Cambridge, MA.

Rice, D.W. 1986b. Sperm Whales. Pages 94-101 *In:* D. Haley (Ed.) Marine Mammals of the Eastern North Pacific and Arctic Waters, 2nd ed. Pacific Search Press, Seattle, Washington.

Rice, D.W. 1989 Sperm WhaleöPhyseter macrocephalus Linnaeus, 1758. *In:* Handbook of Marine Mammals. 4. River Dolphins and the Larger Toothed Whales. S.H. Ridgway, and R. Harrison (Eds.) pp. 177-233. London: Academic Press.

Richardson, K., W. Steffen, H.J. Schellnhuber, J. Alcamo, T. Barker, D. Kammen, R. Leemans, D. Liverman, M. Munasinghe, B. Osman-Elasha, N. Stern, O. Waever. 2009. Climate Change. Global Risks, Challenges & Decisions. University of Copenhagen, Copenhagen.

Richardson, W.J., and C.R. Greene, Jr. 1993. Variability in Behavioral Reaction Thresholds of Bowhead Whales to Man-Made Underwater Sounds. J. Acoust. Soc. Am. 94 (3 [Part 2]):1848. [Abstr.]

Richardson, W.J., C.R. Greene, Jr., C.I. Malme, and D.H. Thomson. 1995. Marine Mammals and Noise. Academic Press, San Diego. 576 pp.

Richmond, J.P., J.M. Burns, and L.D. Rea. 2006. Ontogeny of Total Body Oxygen Stores and Aerobic Dive Potential in Steller Sea Lions (*Eumetopias jubatus*). Journal of Comparative Physiology B: DOI 10.1007/s00360-006-0076-9.

Richmond, J.P., J.M. Burns, L.D. Rea, and K.L. Mashburn. 2005. Postnatal Ontogeny of Erythropoietin and Hematology in Free-Ranging Steller Sea Lions (*Eumetopias jubatus*). General and Comparative Endocrinology. 141:240-247.

Ricker, W.E. 1954. Stock and Recruitment. J. Fish. Res. Bd. Can. 2:559-623.

Riemer, S.D., and R.F. Brown. 1997. Prey of Pinnipeds at Selected Sites in Oregon Identified by Scat (Fecal) Analysis, 1983-1996. Oregon Department of Fish and Wildlife, Technical Report No.97-6-02.

Robbins, J., and D.K. Mattila. 2004. Estimating Humpback Whale (*Megaptera novaeangliae*) Entanglement Rates on the Basis of Scar Evidence. Final report to the US National Marine Fisheries Service Unpublished. Available from the Center for Coastal Studies, Box 1036, Provincetown, Massachusetts 02657. 22 pp.

Robson, B.W., M.E. Goebel, J.D. Baker, R.R. Ream, T.R. Loughlin, R.C. Francis, G.A. Antonelis, and D.P. Costa. 2004. Separation of foraging habitat among breeding sites of a colonial marine predator, the northern fur seal (*Callorhinus ursinus*). Can J. Zool. 82:20-29.

Rodionov, S., J. Overland, and N. Bond. 2005. Pacific Climate Overview. 2005. *In:* JL. Boldt (Ed.) Ecosystem Considerations for 2006. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Ronholt, L.L. 1989. Atka Mackerel. *In:* Condition of Groundfish Resources of the Gulf of Alaska in 1988. T.K. Wilderbuer (Ed.). NOAA Tech. Memo. NMFS-F/NWC-165.

Rosen, D.A.S. 2009. Steller Sea Lions *Eumetopias Jubatus* and Nutritional Stress: Evidence from Captive Studies. Mammal Review, October 2009. 39(4):284-306(23).

Rosen, D.A.S., A.J. Winship, and L.A. Hoopes. 2006. Thermal and Digestive Constraints to Foraging Behavior in Marine Mammals. Fisheries Centre Working Paper #2006-17, University of British Columbia, Vancouver, Canada.

Rosen, D.A.S., A.W. Trites. 2003. No Evidence for Bioenergetic Interaction Between Digestion and Thermoregulation in Steller Sea Lions, *Eumetopias jubatus*. Physiol. Biochem. Zool. 76:899-906.

Rosen, D.A.S., and A.W. Trites. 1997. Heat Increment of Feeding in Steller Sea Lions, Eumetopias jubatus. Comparative Biochemistry and Physiology. 118A:877-881.

Rosen, D.A.S., and A.W. Trites. 2000a. Digestive Efficiency and Dry-matter Digestibility of Steller Sea Lions Red Herring, Pollock, Squid and Salmon. Can. J. Zoology. 78:234-239.

Rosen, D.A.S., and A.W. Trites. 2000b. Pollock and the decline of Steller Sea Lions: Testing the Junk-food Hypothesis. Can. J. Zool. 78:1243-1250.

Rosen, D.A.S., and A.W. Trites. 2002. What is it About Food? Examining Possible Mechanisms with Captive Steller Sea Lions. Pages 45-48. *In:* D. DeMaster, and S. Atkinson (Eds.). Steller Sea Lion Decline: Is It Food II, University of Alaska Sea Grant, AK-SG-02-02, Fairbanks.

Rosen, D.A.S., and A.W. Trites. 2004. Satiation and Compensation for Short-term Changes in Food Quality and Availability in Young Steller Sea Lions (*Eumetopias jubatus*). Canadian Journal of Zoology. 82:1061-1069.

Rosen, D.A.S., and A.W. Trites. 2005. Examining the Potential for Nutritional Stress in Steller Sea Lions: Physiological Effects of Prey Composition. Journal of Comparative Physiology B. 175:265-273.

Rosen, D.A.S., and S. Kumagai. 2008. Hormone Changes Indicate that Winter is a Critical Period for Food Shortages in Steller Sea Lions. J. Comp. Physiol. B. 178(5):573-583.

Rosen, D.A.S., L. Williams, and A.W. Trites. 2000. Effect of Ration Size and Meal Frequency on Assimilation and Digestive Efficiency in Yearling Steller Sea Lions, *Eumetopias jubatus*. Aquatic Mammals 26(1):76-82.

Ross, P.S., R.L. DeSwart, H.H. Timmerman, P.J.H. Reijnders, J.G. Voss, H. van Loveren, and A.D.M.E. Osterhaus. 1996. Suppression of Natural Killer Cell Activity in Harbour Seals (*Phoca vitulina*) and Baltic Sea Herring. Aquatic Toxicology 34:71-84.

Rowley, J. 1929. Life History of the Sea-lions on the California Coast. Journal of Mammalogy 10:1-39.

Rugen W.C., and A.C. Matarese. 1988. Spatial and Temporal Distribution and Relative Abundance of Pacific Cod (Gadus macrocephalus) Larvae in the Western Gulf of Alaska. Northwest and Alaska Fisheries Center Processed Report 88-18, National Marine Fisheries Service, Seattle, Washington.

Rutenberg, E.P. 1962. Survey of the Fishes of Family Hexagrammidae. [In Russian.] [Trans. By Isr. Program Sci. Trans., 1970] Pages 1-103. *In:* TS. Rass (Ed.) Greenlings: Taxonomy, Biology, Interoceanic Transplantation. Available from U.S. Dep. Commer., Natl. Tech. Inf. Serv., Springfield, VA, as TT 69-55097.

S.K. Gaichas. Development and Application of Ecosystem Models to Support Fishery Sustainability: A Case Study for the Gulf of Alaska. PhD thesis, School of Aquatic and Fishery Sciences, University of Washington, 2006.

Sambrotto, R.N., and C.J. Lorenzen. 1987. Phytoplankton and Primary Production. Ch 9, Pages 249-282 *In:* Hood, DW., and ST. Zimmerman (Eds.) The Gulf of Alaska, Physical Environment and Biological Resources. U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Sandegren, F.E.. 1970. Breeding and Maternal Behavior of the Steller Sea Lion (*Eumetopias jubata*) in Alaska. M.S. Thesis, Univ. Alaska, Fairbanks. 138 pp.

Saulitis, E., C. Matkin, L. Barrett-Lennard, K. Heise, and G. Ellis. 2000. Foraging Strategies of Sympatric Killer Whale (*Orcinus orca*) Populations in PrinceWilliam Sound, Alaska. Marine Mammal Science. 16:94-109.

Scarff, J.E. 1986. Historic and Present Distribution of the Right Whale (*Eubalaena glacialis*) in the Eastern North Pacific South Of 50 N and East of 180 W. Report of the International Whaling Commission. Special Issue 10:43-63.

Schaufler, L., E. Logerwell, and J. Vollenwieder. 2006. Geographical Variation in Steller Sea Lion Prey Quality in Alaska. Sea Lions of the World, Alaska Sea Grant College Program. AK-SG-06-01.

Scheffer, V.B. 1946. Growth and Behavior of Young Sea Lions. J. of Mammal. 26:390-392.

Schulz, T.M., and W.D. Bowen. 2004. Pinniped Lactation Strategies: Evaluation of Data on Maternal and Offspring Lifehistory Traits. Marine Mammal Science. 20:86-114.

Schumacher, J.D., and V. Alexander. 1999. Variability and Role of the Physical Environment in the Bering Sea Ecosystem. *In:* Dynamics of The Bering Sea. T.R. Loughlin, and K. Ohtani (Eds.). Alaska Sea Grant Pub. AK-56-99-03, Fairbanks, AK, pp. 147±160.

Sergeant, D.E. 1977. Stocks of fin whales *Balaenoptera physalus* L. in the North Atlantic Ocean. Rep. Int. Whal. Commn. 27:460–473.

Sease, J.L., and A.E. York. 2003. Seasonal Distribution of Steller's Sea Lions at Rookeries and Haul-Out Sites in Alaska. Marine Mammal Science 19(4):745-763.

Sease, J.L., W.P. Taylor, T.R. Loughlin, and K.W. Pitcher. 2001. Aerial and Land-based Surveys of Steller Sea Lions (*Eumetopias jubatus*) in Alaska, June and July 1999 and 2000. NOAA Tech. Memo. NMFS-AFSC-122, 52 p.

Shallenberger, E. W. (1981) The status of Hawaiian cetaceans. Final report for MMC contract MM7AC028. Available from: National Technical Information Service, Springfield, Virginia. PB82-109398.

Sharpe, F.A. 2002. Social Foraging of the Southeast Alaskan Humpback Whale, *Megaptera novaengliae*. Dissertation Abstracts International Part B: Science and Engineering. 62:3872.

Sharpe, F.A., and L.M. Dill. 1997. The Behavior of Pacific Herring Schools in Response to Artificial Humpback Whale Bubbles. Can. J. Zool. 75:725-730.

She, J.W., M. Petreas, J. Winkler, P. Visita, M. McKinney, and D. Kopec. 2002. PBDEs in The San Francisco Bay Area: Measurements in Harbor Seal Blubber and Human Adipose Tissues. Chemosphere. 46(5):697-707.

Sheffield, G., and R. Zarnke. 1997. Summaries of Serologic Data Collected from Steller Sea Lions in the Bering Sea and Gulf of Alaska, 1978-1996. Steller Sea Lion Recovery Investigations in Alaska, 1995-1996, Alaska Department of Fish and Game, Division of Wildlife Conservation. 74-84.

Shima, M., A.B. Hollowed, and G.R. VanBlaricom. 2000. Response of Pinniped Populations to Directed Harvest, Climate Variability, and Commercial Fishery Activity: A Comparative Analysis. Rev. Fish. Sci. 8(2):89-124.

Shima, M., A.B. Hollowed, and G.R. VanBlaricom. 2002. Changes Over Time in the Spatial Distribution of Walleye Pollock (*Theragra chalcogramma*) in the Gulf of Alaska, 1984-1996. Fish. Bull. 100:3-7-323.

Shimada, A.M., and D.K. Kimura. 1994. Seasonal Movements of Pacific Cod, *Gadus macrocephalus*, in the Eastern Bering Sea and Adjacent Waters Based on Tag-recapture Data. Fish. Bull. U.S. 92:800-816.

Sigler, M.F., C.R. Lunsford, J.T. Fujioka, and S.A. Lowe. 2003. Alaska Sablefish Assessment for 2004. *In:* Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Gulf of Alaska as Projected for 2004. Available from North Pacific Fishery Management Council, 605 West 4th Avenue, Suite 306, Anchorage, Alaska 99501-2252. Http://www.afsc.noaa.gov/refm/stocks/assessments.htm.

Sigler, M.F., D.J. Tollit, J.J. Vollenweider, J.F. Thedinga, D.J. Csepp, J.N. Womble, M.A. Wong, M.J. Rehberg, and A.W. Trites. 2009. Steller Sea Lion Foraging Response to Seasonal Changes in Prey Availability. Marine Ecology Progress Series 388:243-261.

Sigler, M.F., J.N. Womble, and J.J. Vollenweider. 2004. Availability to Steller Sea Lions (*Eumetopias jubatus*) of a Seasonal Prey Resource: A Prespawning Aggregation of Eulachon (*Thaleichthys pacificus*). Can. J. Fish. Aquat. Sci. 61:1475-1484.

Sigler, MF., LB. Hulbert, CR. Lunsford, N. Thompson, K. Burek, G. O'Corry-Crowe, and AC. Hirons. 2006. Diet of Pacific Sleeper Shark, a Potential Steller Sea Lion Predator, in the Northeast Pacific Ocean. J. Fish. Biol. 69:392-405.

Sigurjønsson, J. 1995. On the life history and autecology of North Atlantic rorquals. Pp. 425–441 in A.S. Blix, L. Walloe, and 0. Ulltang (eds), Whales, seals, fish and man. Elsevier Science.

Sinclair, A.R.E., and C.J. Krebs. 2002. Complex Numerical Responses to Top-Down and Bottom-Up Processes in Vertebrate Populations. Phil. Trans. R. Soc. Lond. B 357:1221-1231.

Sinclair, E.H., and T.K. Zeppelin. 2002. Seasonal and Spatial Differences in Diet n the Western Stock of Steller Sea Lions (*Eumetopias jubatus*). Journal of Mammalogy. 83(4):973-990.

Sinclair, E.H., S.E. Moore, N.A. Friday, T.K. Zeppelin, and J.M. Waite. 2005. Do Patterns of Steller Sea Lion (*Eumetopias jubatus*) Diet, Population Trend and Cetacean Occurrence Reflect Oceanographic Domains from the Alaska Peninsula to the Central Aleutian Islands? Fisheries Oceanography. 14:223-242.

Sinclair, E.H., T. Loughlin, and W. Pearcy. 1994. Prey Selection by Northern Fur Seals (*Callorhinus ursinus*) in the Eastern Bering Sea. Fish. Bull. 92:144-156.

Skalski, J.R., D.A. Coats, and A.K. Fukuyama. 2001. Criteria for oil spill recovery: a case study of the intertidal community of Prince William Sound, Alaska, following the *Exxon Valdez* oil spill. Environ. Management 28:9-18.

Smith S.C., and H. Whitehead. 1993. Variations in the Feeding Success and Behaviour of Galápagos Sperm Whales (*Physeter macrocephalus*) as They Relate to Oceanographic Conditions. Can. J. Zool. 71:1991-1996.

Snyder, G.M., K.W. Pitcher, W.L. Perryman, and M.S. Lynn. 2001. Counting Steller Sea Lion Pups in Alaska: an Evaluation of Medium-Format, Color, Aerial Photography. Marine Mammal Science. 17:136-146.

Soboleff, N.J. 2005. Potential interactions between state-managed fisheries and Steller sea lions (*Eumetopias jubatus*). M.S. Thesis, University of Alaska Fairbanks, Fairbanks, AK. 124 p.

Sobolevsky, Y.I. 1996. Species Composition and Distribution of Squids in the Western Bering Sea. *In:* OA. Mathisen, and KO. Coyle. (Eds.) Ecology of the Bering Sea: A Review of Russian Literature. Alaska Sea Grant Report No. 96-01, Fairbanks, Alaska. pp. 135-141.

Sogard, S.M., and B.L. Olla. 1996. Food Deprivation Affect Vertical Distribution and Activity of a Marine Fish in a Thermal Gradient: Potential Energy-Conserving Mechanisms. Mar. Ecol. Prog. Ser. 133:43-55.

Solomon, S., G-K. Plattner, R. Knutti, and P. Friedlingstein. 2009. Irreversible Climate Change Due to Carbon Dioxide Emissions. Proc. National Acad. Sciences. 106:1704-1709. Available At: http://www.pnas.org/content/early/2009/01/28/0812721106.full.

Soto, K.H., A.W. Trites, and M. Arias-Schreiber. 2004. The Effects of Prey Availability on Pup Mortality and the Timing of Birth of South American Sea Lions (*Otaria flavescens*) In Peru. J. Zool. 264(4):419-428.

Soto, K.H., A.W. Trites, and M. Arias-Schreiber. 2006. Changes in Diet and Maternal Attendance of South American Sea Lions Indicate Changes in the Marine Environment and Prey Abundance. Mar. Ecol. Prog. Ser. 312:277-290.

Spalding, D.J. 1964. Comparative Feeding Habits of the Fur Seal, Sea Lion and Harbour Seal on the British Columbia Coast. Bull. Fish. Res. Board Canada. 146:1-52.

Springer, A.M. 1998. Is it All Climate Change? Why Marine Bird and Mammal Populations Fluctuate in the North Pacific. Pages 109-119 *In:* G. Holloway, P. Muller, and D. Henderson (Eds.) Biotic Impacts of Extratropical Climate Variability in the Pacific. Proceedings 'Aha Huliko'a Hawaiian Winter Workshop. University of Hawaii, Honolulu, Hawaii.

Springer, A.M., G.B. Van Vliet, J.F. Piatt, and E.M. Danner. 2006. Whales and Whaling in the North Pacific: Oceanographic Insights and Ecosystem Impacts. *In:* J.A. Estes, R.L. Brownell, D.P. DeMaster, D.P. Doak, and T.M. Williams (Eds.) Whales, Whaling, and Ocean Ecosystems. University of California Press, Berkeley, California.

Springer, A.M., J.A. Estes, G.B. van Vliet, T.M. Williams, D.F. Doak, E.M. Danner, K.A. Forney, and B. Pfister. 2003. Sequential Megafaunal Collapse in the North Pacific Ocean: An Ongoing Legacy of Industrial Whaling? Proc. National Academy of Sciences. 100:12223-12228.

Springer, A.M., J.A. Estes, G.B. van Vliet, T.M. Williams, D.F. Doak, E.M. Danner, and B. Pfister. 2008. Mammal-eating killer whales, industrial whaling, and the sequential megafaunal collapse in the North Pacific Ocean: A reply to critics of Springer *et al.* Mar. Mamm. Sci. 24(2):414-442.

Stabeno, P.J., and J.E. Overland. 2007. New Climate States During the Last Decade in the North Pacific. *In:* The Changing North Pacific: Previous Patterns, Future Projections and Ecosystem Impacts. p. 199. North Pacific Marine Science Organization (PICES) Annual Meeting, Victoria, B.C. (Canada), 26 Oct-5 Nov 2007. PICES, P.O. Box 6000 Sidney B.C. V8L 4B2 Canada.

Stabeno, P.J., D.G. Kachel, N.B. Kachel, and M.E. Sullivan. 2005. Observations from Moorings in the Aleutian Passes: Temperature, Salinity and Transport. Fish. Oceanogr. 14(s1):39-54.

Stabeno, P.J., N.A. Bond, A.J. Hermann, N.B. Kachel, C.W. Mordy, and J.E. Overland. 2004. Meteorology and Oceanography of the Northern Gulf of Alaska. Continental Shelf Research. 24:859-897.

Stabeno, P.J., N.A. Bond, N.B. Kachel, S.A Salo, and J.D. Schumacher. 2001. On the Temporal Variability of the Physical Environment Over the South-eastern Bering Sea. Fish. Oceanogr. 10:81-98.

Stafford, K. M., D. K. Mellinger, S. E. Moore S.E., Fox C.G. 2007. Seasonal variability and detection range modeling of baleen whale calls in the Gulf of Alaska, 1999–2002. J. Acoust. Soc. Amer. 122(6): 3378-3390.

Stanberry, K. 2003. The Effect of Changes in Dietary Fat Level on Body Composition, Blood Metabolites and Hormones, Rate of Passage, and Nutrient Assimilation Efficiency in Harbor Seals. M.Sc. thesis, University of Hawaii, Honolulu.

Stapleton, H.M., N.G. Dodder, J.R. Kucklick, C.M. Reddy, M.M. Schantz, P.R. Becker, F. Gulland, B.J. Porter, and S.A. Wise. 2006. Determination of HBCD, PBDEs and MeO-BDEs in California Sea Lions (*Zalophus californianus*) Stranded Between 1993 and 2003. Marine Pollution Bulletin. 52:522-531.

Steiger, G.H., J. Calambokidis, J. Straley, L. Herman, S. Cerchio, D.R. Salden, J. Urban-R, J.F. Jacobsen, O. von Ziegesar, K.C. Balcomb, C.M. Gabriele, M.E. Dahlheim, S. Uchida, J.K.B. Ford, P.L. de Guevara-P, M. Yamaguchi, J. Barlow. 2008. Geographic Variation in Killer Whale Attacks on Humpback Whales in the North Pacific: Implications for Predation Pressure. Endang. Species. Res. 4:247-256.

Steiger, G.H., J. Calambokidis, R. Sears, K.C. Bacomb, and J.C. Cubbage. 1991. Movement of Humpback Whales Between California and Costa Rica. Marine Mammal Science. 7:306-310.

Stephens, D.W., and J.R. Krebs. 1986. Foraging Theory. Princeton University Press, New Jersey. 247 pp.

Stewart, B.S., S.A. Karl, P.K. Yochem, S. Leatherwood, and J.L. Laake. 1987. Aerial Surveys for Cetaceans in the Former Akutan, Alaska, Whaling Grounds. Arctic. 40(1):33-42.

Straley J, T. O’Connell, G. Beam, S. Mesnick, A. Allen, and E. Mitchell. 2002. Sperm Whale Depredation in the Demersal Longline Fishery for Sablefish in the Gulf of Alaska: Research Needs and Approaches to Mitigation. *In:* M. Donoghue, R.R. Reeves, and G. Stone (Eds.) Report Of The Workshop On Interactions Between Cetaceans and Longline Fisheries, November 2002. Apia, Samoa, pp 7-8.

Straley, J., T. O'Connell, S. Mesnick, L. Behnken, and J. Liddle. 2005. Sperm Whale and Longline Fisheries Interactions in the Gulf of Alaska. North Pacific Research Board R0309 Final Report, 15 p.

Stone, G.S., S.K. Katona, A. Mainwaring, J.M. Allen, and H.D. Corbett. 1992. Respiration and surfacing rates of fin whales (*Balaenoptera physalus*) observed from a lighthouse tower. Rep. Int. Whal. Commn. 42:739–745.

Swain, U.G. 1996. Foraging Behaviour of Female Steller Sea Lions in Southeast Alaska and the Eastern Gulf of Alaska. Pages 135-166. *In:* Steller Sea Lion Recovery Investigations in Alaska, 1992-1994. Rep from AK. Dep. Fish and Game, Juneau, Alaska to NOAA, Wildlife Technical Bulletin. May 13, 1996.

Szteren, D., D. Aurioles, and L.H. Gerber. 2006. Population Status and Trends of the California Sea Lion (*Zalophus californianus californianus*) in the Gulf of California, Mexico. Pages 369-384. *In:* A.W. Trites, S.K. Atkinson, D.P. DeMaster, L.W. Fritz, T.S. Gelatt, L.D. Rea, and K.M. Wynne. (Eds.) Sea Lions of the World. Alaska Sea Grant, Fairbanks.

Takahashi, N., and K. Wada. 1998. The Effect of Hunting in Hokkaido on Population Dynamics of Steller Sea Lions in the Kuril Islands: A Demographic Modeling Analysis. Biosphere Conservation. 1:49-62.

Takatsu, T., T. Nakatani, T. Miyamoto, K. Kooka, and T. Takahashi. 2002. Spatial Distribution and Feeding Habits of Pacific Cod (Gadus macrocephalus) Larvae in Mutsu Bay, Japan. Fisheries Oceanography. 11(2):90-101.

Tamura, T., and S. Ohsumi. 2000. Regional Assessments of Prey Consumption by Marine Cetaceans in the World. Paper SC/52/E6 presented to the IWC Scientific Committee, June 2000 Unpublished. 42 p.

Tanasichuk, R.W., and D.M. Ware. 1987. Influence of Interannual Varations in Winter Sea Temperature on Fecundity and Egg Size in Pacific Herring (*Clupea harengus pallasii*). Can. J. Fish. Aquat. Sci. 44:1485-1495.

Tarasevich, M.N. 1968. Pishchevye Svyazi Kasholotov v Severnoi Chasti Tikhogo Okeana. [Food Connections of the Sperm Whales of the Northern Pacific.] [In Russian.] Zool. Zhur. 47:595-601. (Transl. by K. Coyle, Univ. Alaska, Fairbanks, 1982, 11 pp.)

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow, and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance of marine mammals. Mar. Mamm. Sci. 23(1): 157–175.

The Royal Society. 2005. Ocean Acidification Due to Increasing Atmospheric Carbon Dioxide. Document 12/05 Royal Society: London.

Theilacker, G.H., K.M. Bailey, MF. Canino, and SM. Porter. 1996. Variations in Larval Walleye Pollock Feeding and Condition: A Synthesis. Fish. Oceanogr. 5:112-123.

Thomas, G., and R. Thorne. 2001. Night-time Predation by Steller Sea Lions. Nature. 411(28):1013.

Thompson, G., J. Ianelli, M. Dorn, D. Nichol, S. Gaichas, and K. Aydin. 2007. Assessment of the Pacific Cod Stock in the Eastern Bering Sea and Aleutian Islands Area. *In:* Plan Team for the Groundfish Fisheries of the Bering Sea and Aleutian Islands (compiler) Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions, p. 209-327. North Pacific Fishery Management Council, 605 West 4th Avenue, Suite 306, Anchorage, AK 99501.

Thompson, G., M. Dorn, and D. Nichol. 2006. Assessment of the Pacific Cod Stock in the Eastern Bering Sea and Aleutian Islands Area. *In:* Plan Team for the Groundfish Fisheries of the Gulf of Alaska (compiler) Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Gulf of Alaska, p. 147-220. North Pacific Fishery Management Council, 605 West 4th Avenue, Suite 306, Anchorage, AK 99501.

Thompson, G.G. 1993. A Proposal for a Threshold Stock Size and Fishing Mortality Rate. *In:* SJ. Smith, JJ. Hunt, and D. Rivard (Eds.) Risk Evaluation and Biological Reference Points for Fisheries Management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Thompson, G.G. 1994. A General Diffusion Model of Stock. Recruitment Systems With Stochastic Mortality. ICES C.M. 1994/ T:18.

Thompson, G.G., and M.W. Dorn. 2005. Assessment of the Pacific Cod Stock in the Eastern Bering Sea and Aleutian Islands Area. *In:* Plan Team for Groundfish Fisheries of the Bering Sea/Aleutian Islands (compiler) Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions. North Pacific Fishery Management Council, 605 W. 4th Avenue Suite 306, Anchorage, Alaska 99501.

Thompson, G.G., and R.D. Methot. 1993. Pacific Cod. *In:* Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Region as Projected for 1994. Bering Sea/Aleutian Islands Plan Team (Eds.) North Pacific Fishery Management Council, 605 W. 4th Avenue, Suite, 306, Anchorage, Alaska 99501.

Thompson, M.H., C.P. Dumont, and C.F. Gaymer. 2008. ISO 14001: Towards International Quality Environmental Management Standards for Marine Protected Areas. Ocean and Coastal Management. 51:727-739.

Thompson, P.O., W.C. Cummings, and S.J. Ha. 1986. Sounds, Source Levels, and Associated Behavior of Humpback Whales, Southeast Alaska. J. Acoust. Soc. Amer. 80:735-740.

Thompson, R.H.S. (1947). Protection of Enzymes by Anticholine-Oxidases Against Inactivation by Nitrogen Mustard. J. Physiol. 105:370.

Thorsteinson, F.V., and C.J. Lensink. 1962. Biological Observations of Steller Sea Lions Taken During an Experimental Harvest. J. Wildl. Management. 26:353-359.

Tojo, S., M. Yoshizawa, T. Motobayashi, and K. Watanabe. 2007. Effects of Loosing Aigamo Ducks on the Growth of Rice Plants, Weeds, and the Number of Arthropods in Paddy Fields. Weed Biol. Manag 7:38-43.

Tollit, D.J., S.G. Heaslip, and A.W. Trites. 2004b. Sizes of Walleye Pollock (*Theragra chalcogramma*) Consumed by the Eastern Stock of Steller Sea Lions (*Eumetopias jubatus*) in Southeast Alaska from 1994 to 1999. Fish. Bull. 102(3):522-532.

Tollit, D.J., A D. Schulze, A.W. Trites, P.F. Olesiuk, S.J. Crockford, T.S. Gelatt, R.R. Ream, and K.M. Miller. 2009. Development and application of DNA techniques for validating and improving pinniped diet estimates. Ecol. App. 19(4):889-905.

Tomlin, A.G. 1967. Mammals of the USSR and Adjacent Countries. Cetacea. Israel Program Sci. Transl. 9(124):717. Natl. Tech. Inf. Serv. TT 65-50086. Springfield, VA [Translation of Russian text published in 1957].

Tønnessen, J.N. and A.O. Johnsen. 1982. The History of Modern Whaling. Univ. of California Press, Berkeley. 798 pp.

Trenberth, K.E., and J.W. Hurrell. 1994. Decadal Atmosphere-ocean Variations in the Pacific. Clim. Dyn. 9:303-319.

Trillmich, F. 1986. Maternal Investment and Sex-allocation in the Galapagos Fur Seal, *Arctocephalus galapagoensis*. Behav. Ecol. Sociobiol. 19:157-164.

Trillmich, F. and D. Limberger. 1985. Drastic effects of El Nino on Galapagos pinnipeds. Oecologia 67:19-22.

Trillmich, F., and K.A. Ono. 1991. Pinnipeds and El Niño: Responses to Environmental Stress. Springer-Verlag, Berlin. 293 pp.

Trippel, E.A., O.S. Kjesbu, and P. Solemdal. 1997. Effects of Adult Age and Size Structure on Reproductive Output in Marine Fishes. *In:* Early Life History and Recruitment in Fish Populations, pp. 31-63. RC. Chambers, and EA. Trippel. (Eds.) Chapman and Hall, New York.

Trites, A.W. 2003. Food Webs in the Ocean: Who Eats Whom, and How Much? Pages 125-143 *In:* M. Sinclair, and G. Valdimarsson (Eds.) Responsible Fisheries in the Marine Ecosystem. FAO, Rome and CABI Publishing, Wallingford.

Trites, A.W., A.J. Miller, H.D.G. Maschner, M.A. Alexander, S.J. Bograd, A. Capotondi, K.O. Doyle, E. Di Lorenzo, E.J. Gregr, C.E. Grosch, B.P. Finney, L. Fritz, G.L. Hunt, J. Jahncke, N.B. Kachel, H-J. Kim, C. Ladd, N.J. Mantua, C. Marzban, W. Maslowski, R. Mendelssohn, D.J. Neilson, J.E. Overland, S.R. Okkonen, K.L. Reedy-Maschner, T.C. Royer, F.B. Schwing, J.X.L. Wang, and A.J. Winship. 2005. Bottom-Up Forcing and the Decline of Steller Sea Lions in Alaska: Assessing the Ocean Climate Hypothesis. Fish. Oceanogr. In Press.

Trites, A.W., A.J. Miller, H.D.G. Maschner, M.A. Alexander, S.J. Bograd, J.A. Calder, A. Capotondi, K.O. Coyle, E.D. Lorenzo, B.P. Finney, E.J. Gregr, C.E. Grosch, S.R. Hare, G.L. Hunt, J. Jahncke, N.B. Kachel, H.J. Kim, C. Ladd, N.J. Mantua, C. Marzban, W. Maslowski, R. Mendelssohn, D.J. Neilson, S.R. Okkonen, J.E. Overland, K.L. Reedy-Maschner, T.C. Royer, F.B. Schwing, J.X.L. Wang, and A.J. Winship. 2006a. Bottom-up Forcing and the Decline of Steller Sea Lions (*Eumetopias jubatas*) in Alaska: Assessing the Ocean Climate Hypothesis. Fisheries Oceanography 16(1):46-67.

Trites, A.W., and B.T. Porter. 2002. Attendance Patterns of Steller Sea Lions (Eumetopias jubatus) and Their Young During Winter. Journal of Zoology, London. 256:547-556.

Trites, A.W., and C.P. Donnelly. 2003. The Decline of Steller Sea Lions Eumetopias jubatus in Alaska: A Review of the Nutritional Stress Hypothesis. Mammal Rev. 33(1):3-28.

Trites, A.W., and D.G. Calkins. 2008. Diets of Mature Male and Female Steller Sea Lions (*Eumetopias jubatus*) Differ and Cannot Be Used as Proxies for Each Other. Aquatic Mammals. 34(1):25-34.

Trites, A.W., and P.A. Larkin. 1992. The Status of Steller Sea Lion Populations and the Development of Fisheries in the Gulf of Alaska and Aleutian Islands. A report of the Pacific States Marine Fisheries Commission pursuant to National Oceanic and Atmospheric Administration Award No. NA17FD0177. Fisheries Centre, University of British Columbia, Vancouver, British Columbia, Canada V6T 1Z4.

Trites, A.W., and P.A. Larkin. 1996. Changes in the Abundance of Steller Sea Lions (*Eumetopias jubatus*) in Alaska from 1956 to 1992: How Many Were There? Aquat. Mamm. 22:153-166.

Trites, A.W., and R.A.H. Jonker. 2000. Morphometric Measurements and Body Condition of Healthy and Starving Steller Sea Lion Pups, (*Eumetopias jubatus*). Aquatic Mammals. 26:151-157.

Trites, A.W., B.P. Porter, V.B. Deecke, A.P. Coombs, M.L. Marcotte, and D.A.S. Rosen. 2006 b. Insights into the Timing of Weaning and the Attendance Patterns of Lactating Steller Sea Lions (*Eumetopias jubatus*) in Alaska During Winter, Spring, and Summer. Aquatic Mammals. 32:(1):85-97.

Trites, A.W., D.G Calkins, and A.J. Winship. 2007. Diets of Steller Sea Lions (*Eumetopias jubatus*) in Southeast Alaska, 1993-1999. Fishery Bulletin. 105:234-248.

Trites, A.W., D.G. Calkins, and A.J. Winship. 2007 (=2006d). Diets of Steller Sea Lions (*Eumetopias jubatus*) in Southeast Alaska from 1993 to 1999. Fishery Bulletin. 105(2):234-248.

Trites, A.W., P.A. Livingston, M.C. Vasconcellos, S. Mackinson, A.M. Springer, and D. Pauly. 1999. Ecosystem Change and the Decline of Marine Mammals in the Eastern Bering Sea: Testing the Ecosystem Shift and Commercial Whaling Hypotheses. Fisheries Centre Research Reports 1999, Vol. 7, University of British Columbia. p. 100.

Trites, A.W., T. Isono, and S. Brunner. 2008. Changes in Steller Sea Lion Skull Sizes: Testing the Nutritional Stress Hypothesis. Presentation abstract from the Alaska Marine Science Symposium.

Trites, A.W., V. Christensen, and D. Pauly. 1997. Competition Between Fisheries and Marine Mammals for Prey and Primary Production in the Pacific Ocean. J. Northw. Atl. Fish. Sci. 22:173-187.

Trites, A.W., V. Christensen, and D. Pauly. 2006. Effects of Fisheries on Ecosystems: Just Another Top Predator? Pages 11-27 *In:* I.L. Boyd, S. Wanless, and C.J. Camphuysen (Eds.) Top Predators in Marine Ecosystems: Their Role in Monitoring and Management. Cambridge: Cambridge University Press. 378p. Available online at: http://www.marinemammal.org/pdfs/Trites_C_P_2006.pdf

Trites, A.W., V. Christensen, and D. Pauly. 2006e. Effects of Fisheries on Ecosystems: Just Another Top Predator? Pages 11-27. *In:* Top Predators in Marine Ecosystems: Their Role in Monitoring and Management. I.L. Boyd, K. Camphuysen, and C.J. Wanless (Eds.) Cambridge University Press, Cambridge: 378p.

Trites, A.W., V.B. Deecke, E.J. Gregr, J.K.B. Ford, and P.F. Olesiuk. 2007. (=2006c). Killer Whales, Whaling and Sequential Megafaunal Collapse in the North Pacific: A Comparative Analysis of the Dynamics of Marine Mammals in Alaska and British Columbia Following Commercial Whaling. Marine Mammal Science. 23(4):751-765.

Trites, A.W., R. Flinn, R. Joy, and B. Battaile. 2010. Was the decline of Steller sea lions in the Aleutian Islands from 2000 to 2009 related to the Atka mackerel fishery? Fisheries Centre Working Paper 2010-10. University of British Columbia, Vancouver, B.C., Canada. 29 p.

Trujillo, R.G., T.R. Loughlin, N.J. Gemmell, J.C. Patton, and J.W. Bickham. 2004. Variation in the Mictosatellites and MtDNA Across the Range of the Steller Sea Lion, Eumetopias jubatus. Journal of Mammalogy. 85:338-346.

Turner, L.M. 1886. Contributions to the Natural History of Alaska. Arctic Series in Connection with the Signal Service, U.S. Army, No. 2. Washington, D.C. Part 5, p. 115-196.

Turnock, B.J. and T.K. Wilderbuer. 2009. Arrowtooth flounder. 2009 Groundfish Stock Assessment and Fishery Evaluation Report for the Gulf of Alaska, North Pacific Fishery Management Council, Anchorage AK. (www.fakr.noaa.gov/npfmc/safes/).

U.S. Global Change Research Program (USGCRP). 2009. Global Climate Change Impacts in the United States. USGCRP. T.R. Karl, J.M. Melillo, and T.C. Peterson (Eds.) Cambridge University Press, New York, NY.

U.S./Russia Mammal Working Group. 2009. Protocol of the 20th Meeting of Project 02.05-61, Marine Mammals, under Area V of the U.S./Russia Agreement on Cooperation in the Field of Environmental Protection (24-27 March 2009, Seattle, WA).

Valiela, I. 1995. Marine Ecological Processes, 2nd Ed. Springer, New York. 696 pp.

Vanderlaan. A. S.M. and C.T. Taggart. 2007. Vessel Collisions with Whales: The Probability of Lethal Injury Based on Vessel Speed. Mar. Mamm. Sci. 23:144–156.

Van Kirk, K.F., T.J. Quinn II, and J.S. Collie. 2010. A multispecies age-structured assessment model for the Gulf of Alaska. Can. J. Fish. Aquat. Sci. 67:1135-1148.

Van Waerebeek, K. and R. Leaper. 2008. Second report of the IWC Vessel Strike Data Standardisation Working Group. Report to the International Whaling Commission's Scientific Committee at the IWC's 60th Annual Meeting, Santiago, Chile, June 2008.

Varanasi, U., J.E. Stein, W.L. Reichert, K.L. Tilbury, M.M. Krahn, and S. Chan. 1992. Chlorinated and Aromatic Hydrocarbons in Bottom Sediments, Fish and Marine Mammals in US Coastal Waters: Laboratory and Field Studies of Metabolism and Accumulation. C.H. Walker and D.R. Livingston, (Eds.) Persistent Pollutants in Marine Ecosystems, Pergamon Press, Oxford, UK, pp. 83-115.

von Ziegesar, O. 1984. A Survey of the Humpback Whales in Southeastern Prince William Sound, Alaska: 1980, 1981, and 1983. Report to the Sate of Alaska, Alaska Council on Science and Technology. 34 pp.

von Ziegesar, O. 1992. A Catalogue of Prince William Sound Humpback Whales Identified by Fluke Photographs Between the Years 1977 and 1991. North Gulf Oceanic Society, P.O. Box 15244, Homer, AK. 29 pp.

von Ziegesar, O., and C.O. Matkin. 1986. Humpback Whales in Prince William Sound in 1985: A Contract Report. Contract No. 41 USC 252, NMFS, National Marine Mammal Laboratory, Seattle.

von Ziegesar, O., B. Goodwin, and R. Devito. 2004. A Catalog of Humpback Whales in Prince William Sound Alaska, 1977-2001. Eye of the Whale Research, Fritz Creek, Alaska.

Vos, D.J., L.T. Quakenbush, and B.A. Mahoney. 2006. Documentation of Sea Otters and Birds as Prey for Killer Whales. Marine Mammal Science. 22(1):201-205.

Wade, P.R., and T. Gerrodette. 1993. Estimates of Cetacean Abundance And Distribution in the Eastern Tropical Pacific. Report of the International Whaling Commission. 43:477-493.

Wade, P.R., J.M. Ver Hoef, and D.P. DeMaster. 2009. Mammal-eating Killer Whales and Their Prey—Trend Data for Pinnipeds and Sea Otters in the North Pacific Ocean Do Not Support the Sequential Megafaunal Collapse Hypothesis. Marine Mammal Science. 25:737-747.

Wade, P.R., V.N. Burkanov, M.E. Dahlheim, N.A. Friday, L.W. Fritz, T.R. Loughlin, S.A. Mizroch, M.M. Muto, D.W. Rice, L.G. Barrett-Lennard, N.A. Black, A.M. Burdin, J. Calambokidis, S. Cerchio, J.K.B. Ford, J.K. Jacobsen, C.O. Matkin, D.R. Matkin, A.V. Mehta, R.J. Small, J.M. Straley, S.M. McCluskey, G.R. VanBlaricom, and P.J. Clapham. 2007. Killer Whales and Marine Mammal Trends in the North Pacific – A Re-examination of Evidence for Sequential Megafauna Collapse and the Prey-switching Hypothesis. Marine Mammal Science 23(4):766-802.

Wade, P.R. and J. Durban. 2010. Abundance of mammal-eating (“transient” type) killer whales by defined Steller sea lion areas. Unpublished manuscript, Alaska Fisheries Science Center, Seattle, WA and Southwest Fisheries Science Center, LaJolla, CA. 6 p.

Waite, J.M., M.E. Dahlheim, R.C. Hobbs, S.A. Mizroch, O. von Ziegesar-Markin, J.M. Straley, L.M. Herman, and J. Jacobsen. 1998. Evidence of a Feeding Aggregation of Humpback Whales (*Megaptera novaeangliae*) Around Kodiak Island, Alaska. Marine Mammal Science. 15:210-220.

Waite, J.M., M.E. Dahlheim, R.C. Hobbs, S.A. Mizroch, O. von Ziegesar-Matkin, J.M. Straley, L.M. Herman, and J. Jacobsen. 1999. Evidence of a Feeding Aggregation of Humpback Whales (*Megaptera novaeangliae*) Around Kodiak Island, Alaska. Mar. Mammal Sci. 15:210-220.

Waite, J.M., V.N. Burkanov, and T.R. Loughlin. 2005. Steller sea lion diet in the Russian far east. In: Synopsis of research on Steller sea lions: 2001-2005. Alaska SeaLife Center, Seward, compiled by T.R. Loughlin, D.G. Calkins, and S. Atkinson. p. 19-26.

Waite, J.N., and V.N. Burkanov. 2006. Steller Sea Lion Feeding Habits in the Russian Far East, 2000-2003. Pages 223-234. *In:* A.W. Trites, S.K. Atkinson, D.P. DeMaster, L.W. Fritz, T.S. Gelatt, L.D. Rea, and K.M. Wynne. (Eds.) Sea Lions of the World. Alaska Sea Grant College Program Report AK-SG-06-01, University of Alaska, Fairbanks.

Walker, P.L., D.J. Kennett, T.L. Jones, R. DeLong. 1999. Archaeological Investigations at the Point Bennett Pinniped Rookery on San Miguel Island. *In:* D.R. Brown, K.C. Mitchell, H.W. Chaney (Eds.) Proceedings of the Wfth California Islands Symposium. U.S. Department of the Interior Minerals Management Service, PaciWc OCS Region. pp 628-632.

Walsh W.J., B.N. Tissot, L. Hallacher. 2005. A Report on the Findings and Recommendations of Effectiveness of the West Hawaii Regional Fishery Management Area. Hawaii State Department of Land and Natural Resources, Honolulu. http://hawaii.gov/dlnr/dar/pubs/ar_hrs188F5.pdf

Walters, C., Christensen, V. and Pauly, D. 1997. Structuring Dynamic Models of Exploited Ecosystems from Trophic Mass-balance Assessments. Reviews in Fish Biology and Fisheries. 7:139-172.

Walters, C.J., and S.J.D. Martell. 2004. Fisheries Ecology and Management. Princeton University Press, Princeton.

Walters, C.J., V. Christensen, S.J. Martell, and J.F. Kitchell. 2005. Possible Ecosystem Impacts of Applying MSY Policies from Single-species Assessment. ICES Journal of Marine Science. 62:448-568.

Wang, M., C. Ladd, J. Overland, P. Stabeno, N. Bond, and S. Salo. 2007. Eastern Bering Sea Climate – FOCI, Pages 103-113. *In:* J.L. Boldt (Ed.) Ecosystem Considerations for 2008. Appendix C of the BSAI/GOA Stock Assessment and Fishery Evaluation Reports. North Pacific Fishery Management Council, 605 W. 4th Ave., Suite 306, Anchorage, Alaska 99501.

Waring, G.T., C.P. Farifield, C.M. Ruhsam and M. Sano. 1993. Sperm Whales Associated with Gulf Stream Features Off the North-eastern USA Shelf. Fisheries and Oceanography. 2:101-105.

Waring, G.T., D.L. Palka, K.D. Mullin, J.H.W. Hain, L.J. Hansen, and K.D. Bisack. 1997. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments, 1996. NOAA Tech. Memo. NMFS-NE-114: 250pp.

Watkins, W.A. 1981. Activities and underwater sounds of fin whales. Sci. Rep. Whales Res. Inst 33:83–117.

Watkins, W.A., K.E. Moore, J. Sigurjønsson, D. Wartzokj and G. Notarbartolo di Sciara. 1984. Fin whale (*Balaenoptera physalus*) tracked by radio in the Irminger Sea. *Rit Fiskideildar* 8(1):1–14.

Watkins, W. A., M. A. Daher, G. M. Reppucci, J. E. George, D. L. Martin, N. A. DiMarzio, and D. P. Gannon. 2000. Seasonality and distribution of whale calls in the North Pacific. Oceanography. 13(1):62-67.

Weingartner, T. 2005. Physical and geological oceanography: coastal boundaries and coastal and ocean circulation. P. 35-48 in The Gulf of Alaska, Biology and Oceanography, P.R. Mundy, ed. Alaska Sea Grant Program, Fairbanks, Alaska. 214 p.

Weingartner, T.J., K. Coyle, B. Finney, R. Hopcroft, T. Whitledge, R. Brodeur, M. Dagg, E. Farley, D. Haidvogel,

L. Haldorson, A. Hermann, S. Hinckley, J. Napp, P. Stabeno, T. Kline, C. Lee, E. Lessard, T. Royer, and S. Strom. 2002. The Northeast Pacific GLOBEC Program: Coastal Gulf of Alaska. Oceanography 15(2):48-63.

Weingartner, T.J., S.L. Danielson, and T.C. Royer. 2005. Freshwater variability and predictability int he Alaska Coastal Current. Deep-Sea Res. II 52 (2005):169-191.

Weingartner, T. L. Eisner, G.L. Eckert, and S. Danielson. 2009. Southeast Alaska: Oceanographic habitats and linkages. J. Biogeogr. 36:387-400.

Wespestad, V. 1991. Pacific Herring Population Dynamics and Early Life History and Recruitment Variation Relative to Eastern Bering Sea Oceanographic Factors. Ph.D. Dissertation, University of Washington, Seattle, WA.

Wespestad, V.G., and J.M. Terry. 1984. Biological and Economic Yields for Bering Sea Walleye Pollock Under Different Fishing Regimes. N. Amer. J. Fish. Manage. 4:204-215.

Wespestad, V.G., L.W. Fritz, W.J. Ingraham, and B.A. Megrey. 1997. On Relationships Between Cannibalism, Climate Variability, Physical Transport and Recruitment Success of Bering Sea Walleye Pollock, *Theragra chalcogramma*. ICES International Symposium, Recruitment Dynamics of Exploited Marine Populations: Physical-biological Interactions. Baltimore, Maryland.

Whitehead H. and T. Arnbom. (1987) Social Organization of Sperm Whales off the Galápagos Islands, February-April 1985. Can. J. Zool. 65:913–919.

Whitehead, H. 2002 Estimates of the Current Global Population Size and Historical Trajectory for Sperm Whales. Mar. Ecol. Prog. Ser. 242:295-304.

Whitehead, H., L. Weilgart, and S. Waters. 1989. Seasonality of Sperm Whales off the Galipagos Islands, Ecuador. Int. Whaling Comm. Rep. Comm. 39:207 -210.

Wilderbuer, T.K., A.B. Hollowed, W.J. Ingraham, Jr., P.D. Spencer, M.E. Conners, N.A. Bond, and G.E. Walters. 2002. Flatfish Recruitment Response to Decadal Climatic Variability and Ocean Conditions in the Eastern Bering Sea. Progress in Oceanography. 55(1-2):235-247.

Wilderbuer, T.K., and T. Sample. 2000. Arrowtooth Flounder. 2000 Groundfish Stock Assessments and Fishery Evaluation Reports for the Gulf of Alaska, North Pacific Fishery Management Council, Anchorage AK. (www.fakr.noaa.gov/npfmc/safes/).

Wilderbuer, T.K., D.G.Nichol, and K. Aydin. 2009. Arrowtooth flounder. 2009 Groundfish Stock Assessment and Fishery Evaluation Report for the Bering Sea/Aleutian Islands, North Pacific Fishery Management Council, Anchorage AK. (www.fakr.noaa.gov/npfmc/safes/).

Willette, M., M. Sturdevant, and S. Jewett. 1997. Prey Resource Partitioning Among Several Species of Forage Fishes in Prince William Sound, Alaska. Pages 11-29. *In:* Forage Fishes in Marine Ecosystems. Proceedings of the International Symposium on the Role of Forage Fishes in Marine Ecosystems. Alaska Sea Grant College Program Report No. 97-01. University of Alaska Fairbanks. 1997.

Williams, E.H. and T.J. Quinn, II. 1998. Age-structured Analysis of Pacific Herring from Norton Sound, Alaska. Alaska Fishery Research Bulletin. 4(2):87-109.

Williams, E.H., and T.J. Quinn, II. 2000a. Pacific Herring, Clupea pallasi, rRecruitment in the Bering Sea and North-east Pacific Ocean, I: Relationships Among Different Populations. Fisheries Oceanography. 9(4):285-299.

Williams, E.H., and T.J. Quinn, II. 2000b. Pacific Herring, Clupea pallasi, Recruitment in the Bering Sea and North-east Pacific Ocean, II: Relationships to Environmental Variables and Implications for Forecasting. Fisheries Oceanography. 9(4):300-315.

Williams, T.M. 2006. Physiological and Ecological Consequences of Extreme Body Size in Whales. Pages 191-201 *In:* JA. Estes, DP. DeMaster, DF. Doak, TM.Williams and RL. Brownell, Jr., (Eds.) Whales, Whaling and Ocean Ecosystems. University of California Press, Berkeley, CA.

Williams, T.W., J.A. Estes, D.F. Doak, and A.M. Springer. 2004. Killer Appetites: Assessing the Role of Predators in Ecological Communities. Ecology. 85(12):3373-3384.

Wilson, C.D., A.B. Hollowed, M. Shima, P. Walline, and S. Stienessen. 2003. Interactions Between Commercial Fishing and Walleye Pollock. Alaska Fishery Research Bulletin. 10(1):61-77.

Wilson, M.T., C.M. Jump, and J.T. Duffy-Anderson. 2006. Comparative Analysis of the Feeding Ecology of Energy-rich and Energy-poor Forage Fishes: Capelin (*Mallotus villosus*) Versus Walleye Pollock (*Theragra chalcogramma*). Mar. Ecol. Prog. Ser. 317:345-358.

Wing, B.L. and K. Krieger. 1983. Humpback Whale Prey Studies in Southeastern Alaska, Summer 1982. Report to the Northwest and Alaska Fisheries Center Auke Bay Laboratory, National Marine Fisheries Service, NOAA, P.O. Box 155. Auke Bay, Alaska 99821. 60 pp.

Winn, H.E, P.J. Perkins, and L. Winn. 1970. Sounds and Behavior of the Northern Bottlenosed Whale. Pages 53-59. *In:* Proceedings of the 7th Annual Conference on the Biology, Sonar and Diving of Mammals. Stanford Research Institute; Menlo Park, California.

Winship A.J. 2000. Growth and Bioenergetic Models for Steller Sea Lions *(Eumetopias jubatus)* in Alaska. MSc thesis, University of British Columbia, Vancouver.

Winship, A.J, A.W. Trites, and D.A.S. Rosen. 2002. A Bioenergetic Model for Estimating the Food Requirements of Steller Sea Lions (*Eumetopias jubatus*) in Alaska, USA. Marine Ecology Progress Series. 229:291-312.

Winship, A.J, and A.W. Trites. 2003. Prey Consumption of Steller Sea Lions (*Eumetopias jubatus*) Off Alaska: How Much Prey Do They Require? Fishery Bulletin. 101:147-167.

Winship, A.J., A.W. Trites, and D.G. Calkins. 2001. Growth in Body Size of the Steller Sea Lion (*Eumetopias jubatus*). Journal of Mammalogy 82:500-519.

Winship, A.J., and A.W. Trites. 2006. Risk of Extirpation of Steller Sea Lions in the Gulf of Alaska and Aeutian Islands: A Population Viability Analysis Based on Alternative Hypotheses for Why Sea Lions Declined in Western Alaska. Marine Mammal Science 22:124-155.

Winter, A., R.J. Foy, and K. Wynne. 2009. Seasonal differences in drey availability around a Steller sea lion haulout and rookery in the Gulf of Alaska. Aquatic Mammals 35(2):145-162.

Withrow, D.E. 1982. Using Aerial Surveys, Ground Truth Methodology, and Haul Out Behavior to Census Steller Sea Lions, *Eumetopias jubatus*. M.S. Thesis, Univ. Washington, Seattle. 102 pp.

Witteveen, B., R. Foy, and K. Wynne. 2006. The Effect of Predation (Current and Historical) by Humpback Whales on Fish Abundance Near Kodiak Island, Alaska. Fish Bull. 104:10-20.

Witteveen, B.H. 2003. Abundance and Feeding Ecology of Humpback Whales (*Megaptera novaeangliae*) in Kodiak, Alaska. Masters thesis, University of Alaska. 109.

Witteveen, B.H., G.A.J. Worthy, and J.D. Roth. 2009b. Tracing Migratory Movements of Breeding North Pacific Humpback Whales Using Stable Isotope Analysis. Marine Ecology Progress Series 393:173-183.

Witteveen, B.H., G.A.J. Worthy, K.M. Wynne, and J.D. Roth. 2009a. Population Structure of North Pacific Humpback Whales on Their Feeding Grounds Revealed by Stable Carbon and Nitrogen Isotope Ratios. Marine Ecology Progress Series 379:299-310.

Witteveen, B.H., J.M. Straley, O. Ziegesar, D. Steel, and C.S. Baker. 2004. Abundance and MtDNA Differentiation of Humpback Whales (*Megaptera novaeangliae*) in the Shumagin Islands, Alaska. Can. J. Zool. 82:1352-1359.

Wolfe, R.J., and L.B. Hutchinson-Scarbrough. 1999. The Subsistence Harvest of Harbor Seal and Sea Lion by Alaska Natives in 1998. Alaska Dep. of Fish and Game, Juneau, Alaska, Subsistence Div. Tech. Paper No. 250. 72 pp.

Wolfe, R.J., J.A. Fall, and M. Riedel. 2008. The Subsistence Harvest of Harbor Seals and Sea Lions by Alaska Natives in 2006. Alaska Department of Fish and Game, Juneau, Alaska, Subsistence Div. Tech. Paper No. 339.

Wolfe, R.J., J.A. Fall, and M. Riedel. 2009. The Subsistence Harvest of Harbor Seals and Sea Lions by Alaska Natives in 2007. Alaska Department of Fish and Game, Juneau, AK, Subsistence Div. Tech. Paper No. 345.

Wolfe, R.J., J.A. Fall, and R.T. Stanek. 2005. The Subsistence Harvest of Harbor Seals and Sea Lions by Alaska Natives in 2004. Alaska Department of Fish and Game, Juneau, AK, Subsistence Div. Tech. Paper No. 303.

Womble, J.N., and M.F. Sigler. 2006. Seasonal Availability of Abundant, Energy-rich Prey Influences the Abundance and Diet of a Marine Predator, the Steller Sea Lion, *Eumetopias jubatus*. Mar. Ecol. Prog. Ser. 325:281-293. (.pdf, 323 KB). URL: http://www.int-res.com/articles/meps2006/325/m325p281.pdf

Womble, J.N., and M.F. Sigler. 2006. Temporal Variation in Steller Sea Lion Diet at a Seasonal Haul-out in Southeast Alaska. Pages 141-154. *In:* A.W. Trites, S.K. Atkinson, D.P. DeMaster, L.W. Fritz, T.S. Gelatt, L.D. Rea, and K.M. Wynne. (Eds.) Sea Lions of the World. Alaska Sea Grant Program Report AK-SG-06-01, University of Alaska, Fairbanks.

Womble, J.N., M.F. Sigler, and M.F. Willson. 2009. Linking Seasonal Distribution Patterns with Prey Availability in a Central-Place Forager, the Steller Sea Lion. J. Biogeogr. 36:1-11.

Woodby, D., and L. Hulbert. 2006. Updated Overview of State-Managed Fisheries in the Central and Western Gulf of Alaska, Aleutian Islands, and Southeastern Bering Sea with Reference to Steller Sea Lion. ADF&G; Division of Sport Fish and Commercial Fisheries. Special Publication No. 06-23.

Woodby, D., D.Carlile, S. Siddeek, F. Funk, J.H. Clark, and L. Hulbert. 2005. Commercial Fisheries of Alaska. Department of Fish and Game Special Publication No. 05-09, Anchorage, Alaska, IV + 66 p.

Wynne, K. 1990. Marine Mammal Interactions with the Salmon Drift Gillnet Fishery on the Copper River Delta, Alaska, 1988 and 1989. Sea Grant Tech. Rep. No. 90-05. Univ. Alaska, Fairbanks.

Wynne, K.M., D. Hicks, and N. Munro. 1992. Marine Mammal Observer Program for the Salmon Driftnet Fishery of Prince William Sound Alaska. Ann. Rep. Contract 50ABNF000036 to NMFS Alaska Region, Juneau, Alaska. 53 pp.

Yablokov, A.V. 1994. Validity of Whaling Data. Nature, Lond. 367, 108.

Yang, M.-S. 2003. Food Habits of Important Groundfishes in the Aleutian Islands in 1994 and 1997. NOAA/AFSC Processed Report 2003-07. US Department of Commerce. Seattle, Washington.

Yang, M.-S., and B.N. Page. 1998. Diet of Pacific Sleeper Shark, Somniosus pacificus, in the Gulf of Alaska. Fish. Bull. 97:406-409.

Ylitalo, G.M., J.E. Stein, T.E. Hom, L.J. Johnson, K.L. Tilbury, A.J. Hall, T. Rowles, D. Greig, L.J. Lowenstine, and F. Gulland. 2005. The Role of Organochlorines in Cancer-associated Mortality in California Sea Lions (*Zalophus californianus*). Marine Pollution Bulletin. 50, 30-39.

York R., E.A. Rosa, T. Dietz. 2008. The Ecological Footprint Intensity Of National Economies. J. Ind. Ecol. 8:139-154.

York, A.E. 1994. The Population Dynamics of Northern Sea Lions 1975-85. Mar. Mamm. Sci. 10:38-51.

York, A.E., R.L. Merrick, and T.R. Loughlin. 1996. An Analysis of the Steller Sea Lion Metapopulation in Alaska. Pages 259-292 *In:* D.R. McCullough (Ed.) Metapopulations and Wildlife Conservation, Island Press, Washington, D.C.

Yoseda, K., D.S. Danjelssen, and E. Moksness. 1993. Recent Progress in Research on Mass Seed Production of the Pacific Cod in Japan. IMR, Bergen, Norway.

Zador, S. and K. Aydin. 2009. Fine-scale analysis of arrowtooth flounder catch rates in the eastern Bering Sea reveals spatial trends in abundance and diet. Poster paper presented at Alaska Marine Science Symposium, Anchorage, AK.

Zarnke, R.L., T.C. Harder, H.W. Vos, J.M. Ver Hoef, and ADME. Osterhaus. 1997. Serologic Survey for Phocid Herpesvirus-1 and -2 in Marine Mammals from Alaska and Russia. J. Wildl. Dis. 33:459-465.

Zavadil, P.A., A.D. Lestenkof, D. Jones, P.G. Tetof, and M.T. Williams. 2005. The Subsistence Harvest of Steller Sea Lions on St. Paul Island in 2004. Unpublished report. Available from Aleut Community of St. Paul Island.

Zenteno-Savin, T., M.A. Castellini, L.D. Rea, and B.S. Fadely. 1997. Plasma Haptoglobin Levels in Threatened Alaska Pinniped Populations. J. Wildl. Dis. 33:64-71.

Zeppelin, T.K., D.J. Tollit, K.A. Call, T.J. Orchard, and C.J. Gudmundson. 2004. Sizes of Walleye Pollock (*Theragra chalcogramma*) and Atka Mackerel (*Pleurogrammus monpterygi*us) Consumed by the Western Stock of Steller Sea Lions (*Eumetopias jubatus*) in Alaska from 1998 to 2000. Fish. Bull., U.S. 102:509-521. (.pdf, 373kb) URL: http://fishbull.noaa.gov/1023/zeppelin.pdf

Zerbini, A.N., J.M. Waite, J.L. Laake, and P.R. Wade. 2007. Abundance, Trends and Distribution of Baleen Whales Off Western Alaska and the Central Aleutian Islands. Deep-Sea Research I. 53:1772-1790.

Zerbini, A.N., P. Wade, J. Waite, J. Durban, R. LeDuc, and M. Dahlheim. 2006. Estimating Abundance of Killer Whales (*Orcinus orca*) in the Nearshore Waters of the Gulf of Alaska and the Aleutian Islands Using Line Transect Sampling. Marine Biology. 150(1):1033-1045.

Zerbini, A. N., J. M. Waite, J. L. Laake and P. R. Wade. 2006. Abundance, trends and distribution of baleen whales off western Alaska and the central Aleutian Islands. Deep Sea Res., Part I:1772-1790.

Zhang, Y., J.M. Wallace, and D.S. Battisti. 1997. ENSO-Like Interdecadal Variability: 1900-93. Journal of Climate. 10:1004-1020.

Zwanenburg, K.C.T. 2000. The Effects of Fishing on Demersal Fish Communities of the Scotian Shelf. ICES J. Mar. Sci. 57:503-509.