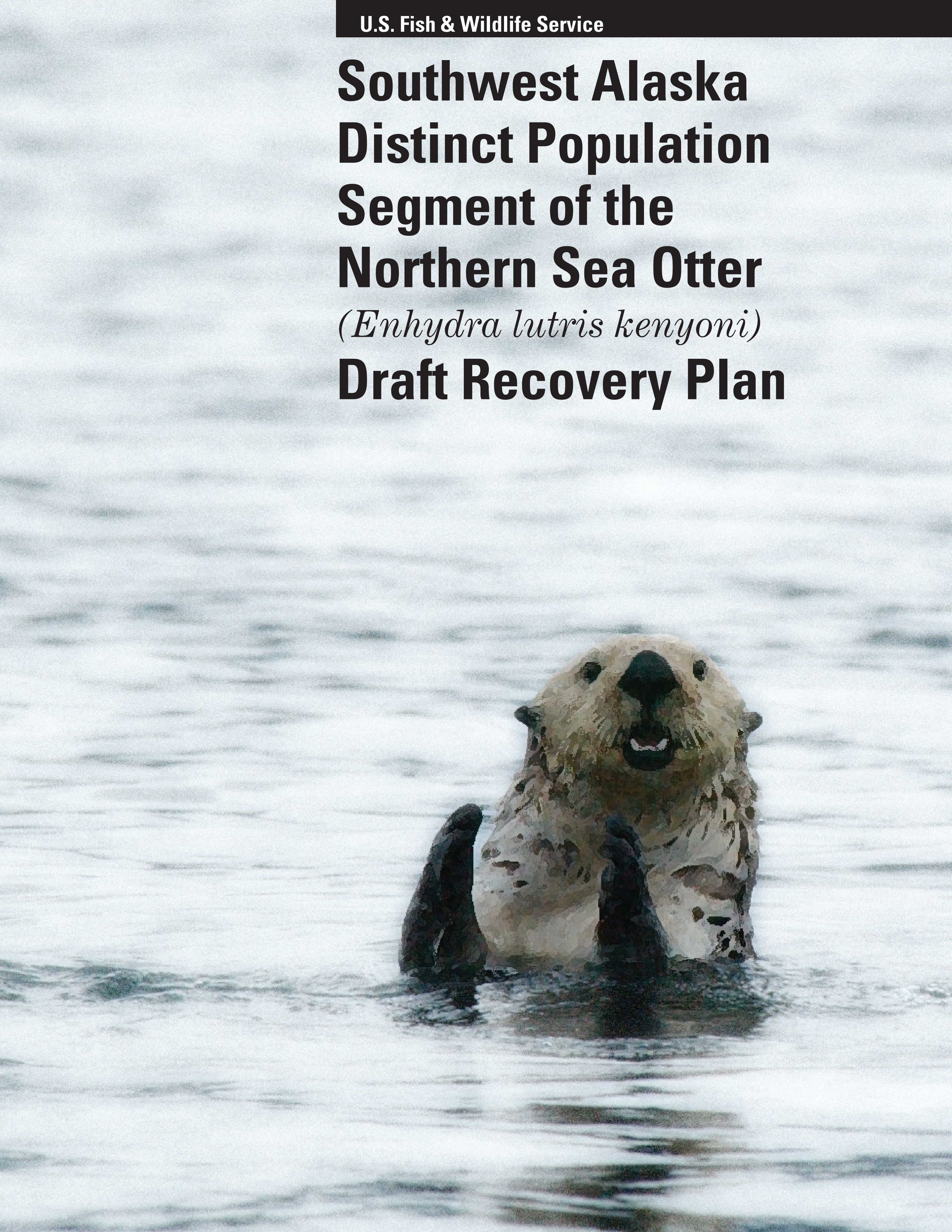
U.S. Fish & Wildlife Service

Southwest Alaska Distinct Population Segment of the Northern Sea Otter

(Enhydra lutris kenyoni)

Draft Recovery Plan



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(Enhyra lutris kenyoni)

Draft Recovery Plan

August 2010

Prepared by

Marine Mammals Management Office
U.S. Fish and Wildlife Service

for

Region 7
U.S. Fish and Wildlife Service
Anchorage, Alaska

Approved: ______________________________
Regional Director, Region 7
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Date: ________________________

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Disclaimer

Recovery plans delineate reasonable actions that are believed to be required to recover and/or protect listed species. We, the U.S. Fish and Wildlife Service (FWS), publish recovery plans, sometimes preparing them with the assistance of recovery teams, contractors, State agencies, and others. Objectives will be attained and any necessary funds made available subject to budgetary and other constraints affecting the parties involved, as well as the need to address other priorities. Recovery plans do not necessarily represent the views, official positions, or approval of any individuals or agencies involved in the plan formulation, other than the FWS. They represent the FWS official position only after they have been signed by the Director or Regional Director as approved. Approved recovery plans are subject to modification as dictated by new findings, changes in species status, and the completion of recovery actions.

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Executive Summary

Based on survey information that indicated that the southwest Alaska population of northern sea otters (*Enhydra lutris kenyoni*) had declined in abundance by more than 50% since the mid-1980s, the U.S. Fish and Wildlife Service (FWS) listed this distinct population segment (DPS) as threatened in August 2005. Section 4(f) of the Endangered Species Act (ESA) directs the Secretary of the Interior to develop and implement plans (generally known as "recovery plans") for the conservation and survival of endangered species and threatened species. In March 2006, the Regional Director for the Alaska Region of the FWS formed a recovery team to serve in an advisory capacity to develop a draft recovery plan for the southwest Alaska DPS of the northern sea otter.

The sea otter is the largest species in the mustelid family, and one of the smallest marine mammals. It possesses a number of unique adaptations allowing it to exist in the nearshore marine environment. As the only marine mammal that lacks a blubber layer, the sea otter relies on a dense coat of fur as insulation from the cold waters where it occurs. To maintain the insulative properties of their fur, sea otters must groom themselves regularly. Their reliance on fur for insulation also makes them highly vulnerable to oil spills. In addition to using fur for insulation, sea otters have a relatively high metabolic rate that helps them maintain their body temperature. This requires them to consume large quantities of prey, as much as 20-33% of their body weight per day. With few exceptions, sea otter prey consists of benthic invertebrates. Sea otter habitat is partially defined by physiological limitations in diving depth, and the animals generally occur in or near shallow waters.

The discovery of large sea otter populations in Alaska by the Russian Bering expedition in 1741 resulted in a commercial fur harvest that lasted 170 years and extirpated sea otters from much of their historic range. When the species was finally given protection under the International Fur Seal Treaty of 1911, the worldwide population may have consisted of fewer than 1,000 individuals in 13 remnant colonies. Throughout much of the 20th century, these remnant colonies grew and expanded their range, eventually recolonizing much of the species' historically occupied habitat. In the late 1960s and early 1970s, the process of recolonization was enhanced by the translocation of otters from areas of high abundance to sites where they had been extirpated by the fur harvest. During the 1990s, sea otter surveys in the Aleutian archipelago indicated that the population trend had shifted from growth and expansion to decline. Additional surveys throughout southwest Alaska helped define the scope and magnitude of the population decline, which led eventually to the listing of this DPS as threatened.

The southwest Alaska DPS ranges from west to east across more than 1,500 miles of shoreline, and the otters occur in a number of distinct habitat types. The magnitude of the population decline has varied over the range. In some areas, numbers have declined by more than an order of magnitude, while in other areas no decline has been detected. To address such differences, this recovery plan identifies five management units (MUs) within the DPS: 1) Western Aleutian Islands; 2) Eastern Aleutian Islands; 3) South Alaska Peninsula; 4) Bristol Bay; and 5) Kodiak, Kamishak, Alaska Peninsula.

The cause of the overall decline is not known with certainty, but the weight of evidence points to increased predation, most likely by the killer whale (*Orcinus orca*), as the most likely cause. Predation is therefore considered a threat to the recovery of this DPS, but other threats, including infectious disease, biotoxins, contaminants, oil spills, food limitation, disturbance, bycatch in fisheries, subsistence harvest, loss of habitat, and illegal take, are also considered in this recovery plan. Threats are summarized in general, and their relative importance is assessed for each of the five MUs. Most threats are assessed to be of low importance to recovery of the DPS; the threats judged to be most important are predation (moderate to high importance) and oil spills (low to moderate importance). Threats from subsistence harvest, illegal take, and infectious disease are assessed to be of moderate importance in the Kodiak, Kamishak, Alaska Peninsula MU, but of low importance elsewhere.

The goal of the recovery program is to control or reduce threats to the southwest Alaska DPS of the northern sea otter to the extent that this DPS no longer requires the protections afforded by the ESA and therefore can be delisted. To achieve this goal, the recovery plan identifies three objectives: 1) achieve and maintain a self-sustaining population of sea otters in each MU; 2) maintain enough sea otters to ensure that they are playing a functional role in their nearshore ecosystem; and 3) mitigate threats sufficiently to ensure persistence of sea otters. Each of these objectives includes explicit criteria to determine if the objective has been met; these are known as "delisting criteria." They stipulate that in order for the DPS to be removed from the Endangered and Threatened Species List, at least three of the five MUs must have met the delisting criteria. The plan also contains criteria to determine if the DPS should be considered for reclassification as endangered; these are known as "uplisting criteria." Delisting should not be considered if any MU meets the criteria specified for uplisting to endangered.

Specific actions to achieve recovery and delisting of the DPS are specified in the recovery action outline and narrative. As demographic characteristics of the population constitute one of the three types of delisting criteria, population monitoring and population modeling are high priorities. Monitoring the status of the kelp forest ecosystem in the Western Aleutian and Eastern Aleutian MUs is also a high priority, as results from such monitoring will be needed to evaluate the ecosystem-based delisting criteria. Other high-priority actions include identifying characteristics of sea otter habitat, and ensuring that adequate oil spill response capability exists in southwest Alaska. As predation is considered to be the most important threat to recovery, additional research on that topic is also a high priority. The recovery implementation schedule provides details regarding the timing, costs, and agencies or entities responsible for implementing each recovery action. The full cost of implementing this recovery plan over the next five years is approximately $15M, of which $2.815M is for Priority 1 actions . Securing adequate funding to implement the plan is therefore also a high priority.

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NOTE: this draft recovery plan is paginated by section, which will allow individual sections of the plan to be updated without requiring the entire document to be reprinted.

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List of Acronyms

CDV	canine distemper virus
DDT	dichlorodiphenyltrichloroethane
DMV	dolphin morbillivirus
DPS	distinct population segment
ESA	Endangered Species Act
FWS	U.S. Fish and Wildlife Service
GPS	global positioning system
HAB	harmful algal bloom
MHC	major histocompatibility complex
MMM	Marine Mammals Management
MMPA	Marine Mammal Protection Act
mtDNA	mitochondrial DNA
MTRP	marine mammal marking, tagging, and reporting program
MU	management unit
NMFS	National Marine Fisheries Service
OC	organochlorine
OLE	Office of Law Enforcement
PCB	polychlorinated biphenyl
PCR	polymerase chain reaction
PDV	phocine distemper virus
PFCs	perfluorinated compounds
PMV	porpoise morbillivirus
POP	persistent organic pollutant
PSP	paralytic shellfish poison
PVA	population viability analysis
PWS	Prince William Sound
TDR	time-depth recorder
UME	unusual mortality event
USGS	U.S. Geological Survey
VE	valvular endocarditis
VHF	very high frequency

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1. Introduction

A. *Brief history of listing*

In April 2000, the U.S. Fish and Wildlife Service (FWS) conducted an aerial survey of sea otters (*Enhydra lutris*) in the Aleutian archipelago. The results of that survey indicated that the population there had declined by an estimated 70% since 1992, which prompted the FWS to designate otters within that portion of their range as a candidate species for listing under the U.S. Endangered Species Act (ESA). Additional aerial surveys revealed that the decline extended beyond the Aleutians, and included much of the southwest Alaska population of northern sea otters (the region extending from the west side of Cook Inlet to Attu Island at the western end of the Aleutians, including Kodiak Island and Bristol Bay).

On 11 February 2004, FWS published a proposed rule (69 FR 6600) to list the northern sea otter in southwest Alaska as a threatened distinct population segment (DPS). The ESA defines "threatened" as "likely to become endangered in the foreseeable future over all or a significant portion of its range." The ESA also defines "endangered" as "likely to become extinct over all or a significant portion of its range." Following a 120-day public comment period, the FWS published a final rule (70 FR 46366) on 9 August 2005, listing the DPS as threatened.

B. *Ecosystem context*

Species often have strong influences on their associated ecosystems. The effects can occur in various ways. In some cases, the effect is self-evident, based on sheer numbers, as for example with a dominant tree species in a forest. Numerical dominants may influence other species through competition for limited resources; by affecting features of the physical environment such as light intensity, temperature, wind, and moisture; through the provision of habitat; or by controlling the flux of energy and matter through the ecosystem.

Comparatively rare species can also play important ecological roles, in some cases having landscape-level effects on the ecosystem that rival or exceed those of numerical dominants. This occurs under two conditions—either when per capita interaction strength with one or more other species is high (Paine 1992, Berlow et al. 1999) or when these direct interactions penetrate the ecosystem's interaction web through indirect effects so as to influence other species and ecosystem processes. Comparatively rare but ecologically important species have been referred to as "keystone species" (Paine 1969, Power et al. 1996). Keystone species often are apex predators and their ecosystem-level effects frequently play out through what have become known as "trophic cascades[1]" (Paine 1980, Carpenter and Kitchell 1993). Both keystone species and trophic cascades are known to occur in many ecosystems (Pace et al. 1999).

Where they inhabit areas with rocky substrates (which includes much of the range of the southwest Alaska DPS), sea otters provide a widely known and well-documented example of a keystone species. The ecosystem-level effects in this case occur through a simple food chain involving sea otters, sea urchins, and kelp. Sea urchins are among the most frequently consumed prey of sea otters, and when sea otters are sufficiently abundant they are capable of limiting sea urchin numbers and biomass. Sea urchins consume kelp and other macroalgae, and when sufficiently abundant are capable of preventing kelp forests from becoming established in extensive areas of shallow rocky reef habitat. These consumer-prey interactions act together to define a trophic cascade, such that sea otters protect kelp forests from destructive overgrazing (Estes and Palmisano 1974, Estes and Duggins 1995). Like forests on land, kelp forests exert important effects on numerous other species and ecosystem-level processes (see Appendix A).

The recent population decline of sea otters in southwest Alaska's Aleutian archipelago has resulted in a wholesale phase shift in the coastal ecosystem from kelp forests to deforested sea urchin barrens (Estes et al. 1998, Estes et al. 2004, Estes et al. in press). In view of the sea otter's keystone role in coastal marine ecosystems, the goal of recovery must be not only to assure the continued survival of sea otters, but also to assure that they are numerous enough to maintain kelp forests through the otter-urchin-kelp trophic cascade. In other words, the objectives of recovery are not only to achieve a demographically viable population, but also to achieve an "ecologically effective" (sensu Soulé et al. 2003, Soulé et al. 2004) sea otter population density.

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[1] Trophic cascades occur when predators in a food web suppress the abundance of their prey, thereby releasing the next lower trophic level from predation (or herbivory if the intermediate trophic level is an herbivore).

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2. Biological Background

For nearly 50 years the United States Department of the Interior (specifically the FWS and U.S. Geological Survey (USGS)) has supported a research program directed at describing and understanding the natural history of sea otters. Research on both wild and captive sea otters has also been supported by the Governments of Russia, Canada, and Japan, the state governments of Alaska, California, and Washington, and numerous academic institutions, aquaria, and non-governmental organizations. The scope of research has been broad and multi-dimensional, but with a focus generally directed toward: 1) understanding basic biology, particularly as it relates to conservation of the species; 2) support of translocations from remnant populations to unoccupied habitats in California, Oregon, Washington, British Columbia, and southeast Alaska; 3) understanding and describing the role of sea otters as keystone predators in structuring nearshore marine communities; 4) describing relations between human uses of nearshore marine resources and sea otters; 5) defining threats to sea otter recovery from human activities, particularly the effects of spilled oil; and 6) understanding the factors responsible for, or contributing to, changes in sea otter populations.

Following the Bering expedition of 1741, sea otters were nearly extirpated for their fur by both Russians and Americans. After the International Fur Seal Treaty of 1911 protected sea otters from further commercial exploitation, sea otter populations generally displayed positive growth rates throughout most of the 20th century (although rates varied among populations), and range expansion along with translocations resulted in recolonization of some previously occupied habitat. The spatial pattern of occupied and unoccupied habitat and the variation in temporal patterns of sea otter recovery enabled comparisons of populations at various stages of recovery, and comparisons of habitats both with and without sea otters. As a result of the long-term dedicated research on this species, and the ability to make "experimental" comparisons owing to the patterns of sea otter presence and absence, a rich, diverse, and extensive body of literature exists on sea otter biology and ecology. Researchers will be able to draw on this foundation of knowledge as they seek to address unanswered and emerging questions in order to further aid sea otter conservation. While this document includes basic background information on the sea otter, it emphasizes those aspects of biology and ecology most likely to be relevant to the conservation of the southwest Alaska population. Other aspects of the species' biology and ecology have been reviewed by Kenyon (1969), VanBlaricom (1988), Riedman and Estes (1990), and Estes and Bodkin (2002), and are described in other references listed in Section 9 of this plan.

A. Species description

The sea otter is a mammal in the order Carnivora. It is the only completely marine species of the aquatic Lutrinae, or otter subfamily of the family Mustelidae (skunks, weasels, minks, badgers, and honey badgers) (Wozencraft 1993). Based on nucleotide sequences of the mitochondrial cytochrome b gene, Koepfli and Wayne (1998) placed Enhydra in one of three reorganized lutrine clades (a group of biological taxa or species that share features inherited from a common ancestor). Two lineages of sea otter are recognized. One led to the extinct *Enhydriodon*; the other to *Enhydritherium* and subsequently to *Enhydra* (Berta and Morgan 1986). Early specimens of *Enhydra*, dating to the early Pleistocene, 1-3 million years ago, have been found along the Pacific Rim and the genus apparently has remained confined to that basin (Riedman and Estes 1990).

Early sea otter taxonomy below the species level was based primarily on comparison of skull morphology between sea otters from Alaska and California. After an exhaustive systematic review and analysis of skull morphology, Wilson et al. (1991) concluded there are three subspecies, *E. lutris lutris* from Asia to the Commander Islands, *E. l. nereis* from California, and *E. l. kenyoni* from Alaska (Figure 1). This taxonomy is largely supported by subsequent molecular genetic data. Analysis of mitochondrial DNA (mtDNA) variation among eight geographically isolated populations identified four major groups (Cronin et al. 1996, Scribner et al. 1997). However, the haplotype frequency (genetic pattern) in the Commander Islands population of *E. l. lutris* is more similar to that observed in the Aleutian-Kodiak grouping, *E. l. kenyoni*, than to the Asian subspecies, *E. l. lutris*, with which it was aligned by skull morphology. Additionally, the Prince William Sound (PWS) population differs from other Alaska populations in haplotype frequency. The distribution of mtDNA haplotypes suggests little or no recent female-mediated gene flow among

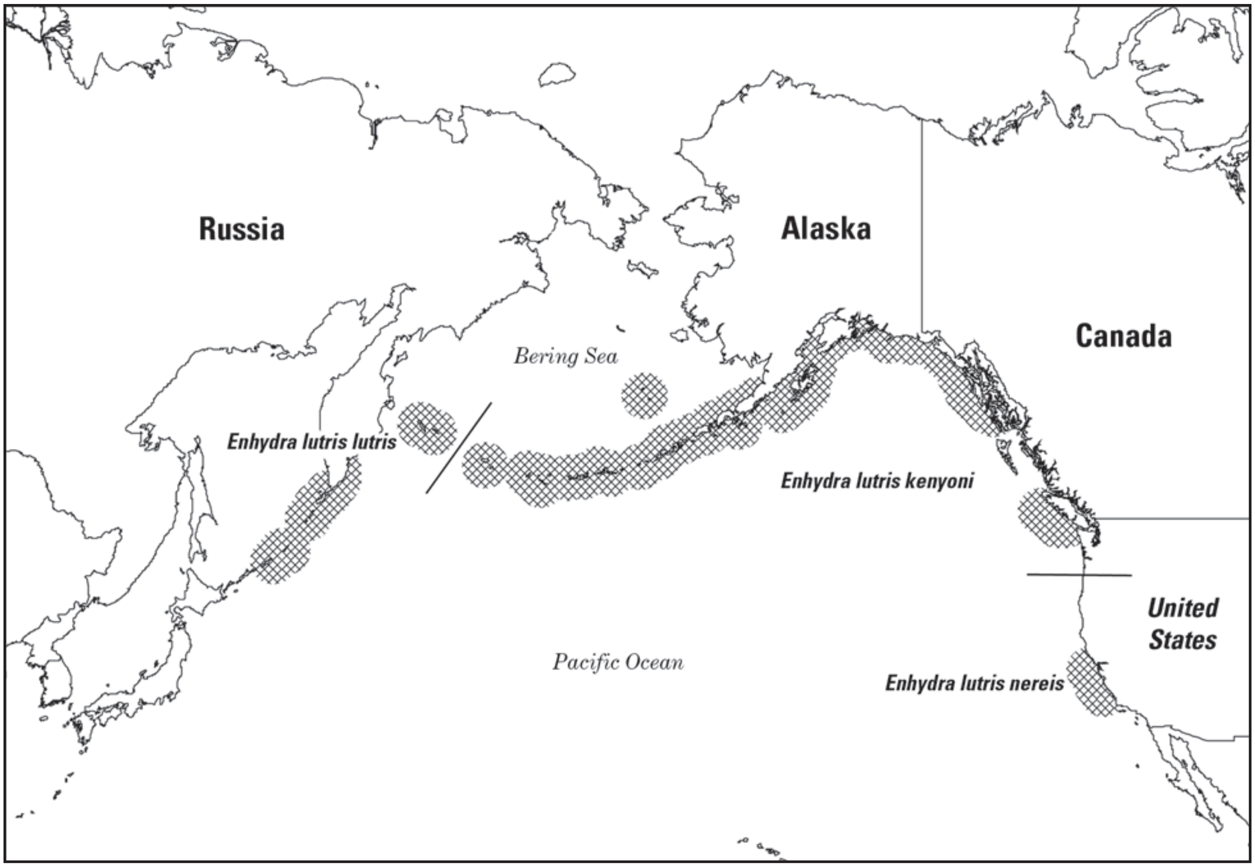

Figure 1. Present distribution of three subspecies of sea otters (hatched areas).

populations sampled. However, populations separated by large geographic distances shared some haplotypes (e.g., the Kuril Islands and Kodiak Island), suggestive of common ancestry and some level of historic gene flow. In a review of sea otter studies using genetic markers, Scribner et al. (1997) concluded that populations are highly differentiated genetically, although limited sequence divergence and lack of phylogeographic concordance suggest an evolutionarily recent common ancestor and some degree of gene flow throughout the range. The differences seen in genetic markers among contemporary sea otter populations likely reflect periods of habitat fragmentation and consolidation during Pleistocene glacial advance and retreat, effects of limited gene flow resulting from natural geographic barriers and limited dispersal capability, and the recent history of harvest-related reductions and subsequent recolonization. The population bottlenecks that resulted from the fur trade harvest may have caused a significant loss of genetic diversity, similar in magnitude to the loss in other species with similar recent histories (Ralls et al. 1983, Larson et al. 2002a, Larson et al. 2002b), and this could reduce long-term population viability. However, Aguilar et al. (2008) concluded that the bottleneck and subsequent loss of genetic diversity in the California sea otter population occurred prior to the onset of the fur trade. Genetic diversity is higher in translocated populations that came from two source populations (e.g., southeast Alaska and British Columbia) than in a population that originated from a single source population (Washington; Larson et al. 2002a). Despite the potential for reduced fitness resulting from population bottlenecks and reduced genetic diversity, rates of increase for translocated populations are significantly higher than those for remnant populations (Bodkin et al. 1999).

Currently, FWS recognizes three stocks of sea otters in Alaska: southeast Alaska, southcentral Alaska, and southwest Alaska (Gorbics and Bodkin 2001; Figure 2). Available data on movements and home ranges of sea otters and findings of divergent population trends at relatively small spatial scales (Bodkin et al. 2002) are suggestive of population structuring at smaller geographic scales than presently recognized.

B. Morphology

The sea otter is the largest mustelid. Adult males attain weights of 45 kg and total lengths of 148 cm and adult females attain weights of 36 kg and total lengths of 140 cm. Size appears to vary among populations and to a large extent may reflect the status of the population relative to available food resources. Weights reported from populations below equilibrium density exceed those from populations at or near equilibrium density by 28% for males and 16% for females (Kenyon 1969). At Bering Island, Russia, mean weights of adult male sea otters declined from 32.1 kg in 1980 to 25.1 in 1990, coinciding with the population

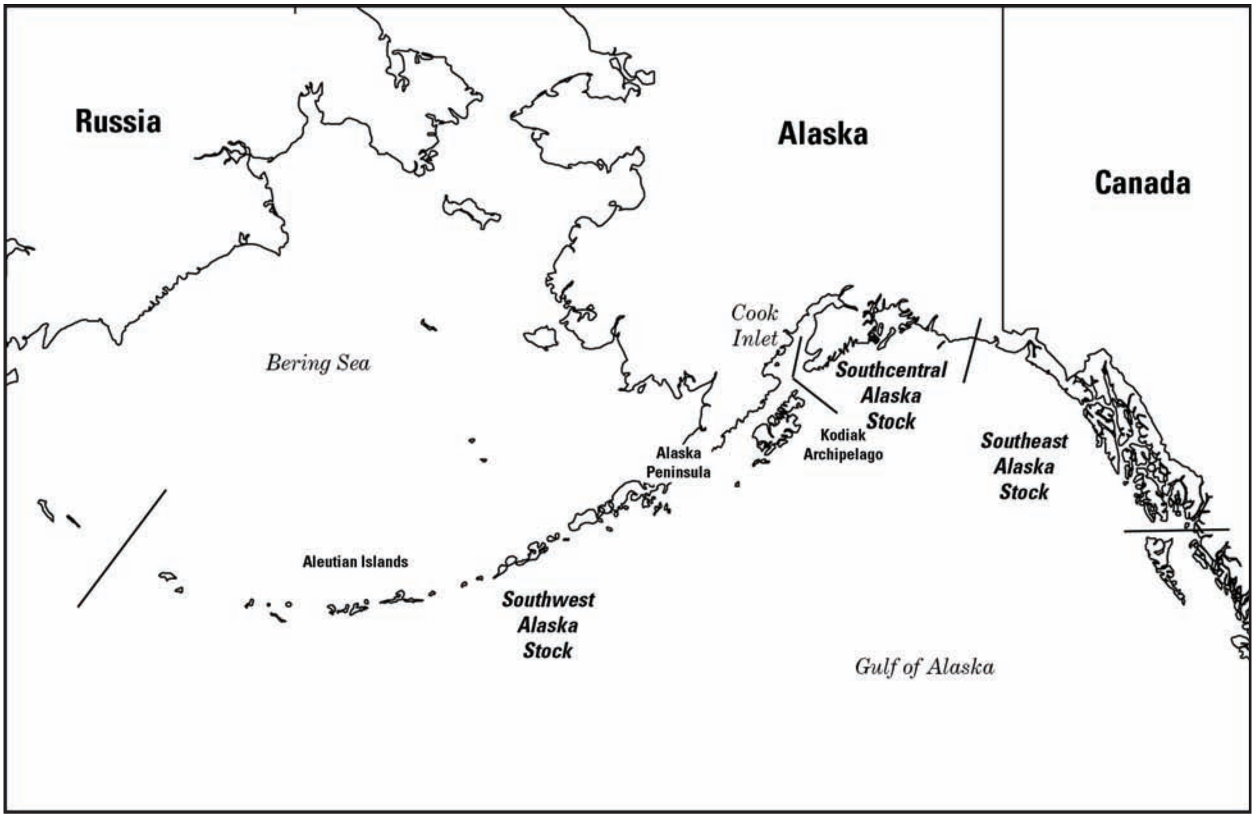

Figure 2. Northern sea otter stock boundaries in Alaska, from Gorbics and Bodkin (2001).

exceeding carrying capacity and a 41% reduction in population size (Bodkin et al. 2000). In California, otters in a small, translocated population at San Nicolas Island (where food is abundant) are significantly larger than otters in the source population (where food is limiting): female mass is 31% greater and male mass is 35% greater (Bentall 2005). At birth, pups weigh 1.7-2.3 kg and are about 60 cm in total length.

The sea otter skull is broad and blunt and with dentition (teeth) that differs from that of most other carnivores in being adapted to crush prey, as opposed to shearing. The canines are long, rounded, and blunt, and used to puncture and pry open prey. The molars are broad and flat, with rounded crowns effective in grinding. The incisors and canines are used to scrape tissues out of shelled prey. A vestigial premolar is present and can be used to estimate age based on annual deposits of cementum (Bodkin et al. 1997). Dental problems associated with tooth wear and breakage leading to systemic infection may be a common contributing cause of mortality, particularly in old individuals (Kenyon 1969).

Fur and the air trapped within it provide the primary sources of insulation and buoyancy for the sea otter. In contrast to most other marine mammals (which rely on blubber for insulation), sea otters have little or no sub-cutaneous fat. The pelage (fur) consists of relatively sparse outer guard hairs and a shorter, very dense underfur at a ratio of about 1:70. Hair densities range from nearly 26,000/cm2 on the hind flipper to 65,000/cm2 on the foreleg (Williams et al. 1988). Sebaceous glands secrete oil that aids in water repulsion. The absence of arrector pili muscles in the epidermis permits the guard hairs to lie nearly parallel to the skin, and this allows the underfur to remain dry even when submerged in water. The ability of the sea otter to thermoregulate is dependent on maintaining the integrity of the pelage, in conjunction with an extremely high metabolic rate. This requires a nearly constant, yet gradual, molt, as well as frequent and vigorous grooming. The color of the pelage ranges from light brown to nearly black. As animals age, they may attain a grizzled appearance, with whitening occurring in the head, neck, and torso regions. Newborn pups have a pale brown, woolly natal pelage until about three months of age.

The forelegs of the sea otter are short and powerful with sensitive paws and extrudable claws used to locate, acquire, and manipulate prey. Forelegs are not used in propulsion. A fold of skin at the axilla (armpit) of each forelimb is used to store and transport prey gathered while foraging. Prey organisms are always consumed at the surface, where they are held and manipulated with the forepaws. The sea otter is one of the few non-human species known to use tools, often using rocks or shells as anvils or hammers to break open hard-shelled prey. The hind limbs are flattened and flipper-like. While swimming, the posterior margins of the hind flippers approximate the lunate (crescent-shaped) pattern and undulating movement of the flukes of cetaceans (whales, dolphins, and porpoises). The tail is long, horizontally flattened, and used in swimming, particularly during slow movements while on the surface. The ears are short and usually held erect while at the surface. While diving, the ears are held downward, presumably to exclude water. Rapid swimming occurs in a face-down position and generally includes diving and may include porpoising for brief periods. Slower swimming generally occurs with the otter on its back with propulsion provided by the hind flippers and/or tail.

C. Physiology

The general mammalian problem of maintaining a constant and elevated body temperature is exacerbated in the sea otter because of its small body size (relative to other marine mammals) and the resulting high surface to volume ratio, as well as the generally cold water temperatures of high-latitude marine environments. The sea otter appears to be one of the few mammals to exist in ambient temperatures outside its thermal-neutral zone (Yeates 2006). In addition to using air in the pelage as an external insulator to reduce heat loss, metabolic heat production in the sea otter is 2.4–3.2 times that predicted in a terrestrial mammal of similar size (Costa and Kooyman 1982). To maintain an average body temperature of about 38° C, a standard metabolic rate of about 0.72 cm3 O2/gm body weight/hour has been measured in the sea otter (Morrison et al. 1974). To maintain the elevated metabolic rate, energy intake must also be elevated, requiring consumption of prey equal to about 20-33% of body weight per day (Kenyon 1969, Costa 1982). Although the air layer in the fur is an efficient insulator, it is also inflexible, requiring a mechanism to dissipate heat during periods of intense exercise. This appears to be accomplished through the broad, highly vascularized, sparsely furred hind flippers.

Some of the physiological adaptations evident in sea otters result from their residing solely in a salt-water environment and foraging under hyperbaric (pressurized) conditions. Sea otters have little access to fresh water and feed primarily on marine invertebrates that are isotonic with seawater and may contain relatively high concentrations of nitrogen, iodine, and other electrolytes. The otters are able to cope with these conditions by consuming sea water (thereby increasing their urinary osmotic space) and producing large volumes of moderately concentrated urine from large, highly efficient kidneys (Costa 1982). The lungs are nearly 2.5 times larger than in other similar-sized mammals, serving to store oxygen needed for diving and buoyancy (Costa and Kooyman 1982). Oxygen-hemoglobin affinity is relatively high, thus increasing blood-oxygen storage capacity. Hemoglobin, red blood cell, and hematocrit values in sea otters are similar to values in pinnipeds (seals, sea lions, and walruses) and cetaceans (Bossart and Dierauf 1990). The trachea is relatively wide compared to other otters, allowing rapid and complete air replacement between dives.

D. Distribution and habitat

The sea otter occurs only in the North Pacific Ocean, and its historical range includes coastal habitats around the Pacific rim between central Baja California and northern Japan. The range currently occupied extends from southern California to northern Japan, with extralimital sightings in central Baja California and near Wrangel Island in the Chukchi Sea. The northward limits in distribution appear related to the southern limits of sea ice, which can preclude access to foraging habitat. Seasonal and inter-annual variation in the southern extent of sea ice results in constriction and expansion of the sea otter's northern range. During periods of advancing winter sea ice along their northern range, sea otters occasionally become trapped and sometimes die (Nikolaev 1965, Schneider and Faro 1975). Sea otters attempting to travel tens of kilometers over the Alaska Peninsula to gain access to the ice-free Pacific were observed in 1971 and 1972 (Schneider and Faro 1975) and again in 1982, 1999, and 2000 (USGS unpublished data). Although some otters may succeed in such efforts, many apparently die from starvation or predation by wolves (*Canis lupus*), red foxes (*Vulpes vulpes*), and wolverines (*Gulo gulo*). Southern range limits are less well understood but appear to coincide with the southern limits of coastal upwelling, associated canopy-forming kelp forests, and the 20-22° C sea surface isotherm (Kenyon 1969).

Sea otters occupy and use all coastal marine habitats within their range, from protected bays and estuaries to exposed outer coasts and offshore islands. Because they need to dive to the sea floor to forage (Bodkin 2001), the seaward limit of their usual distribution is defined by their diving ability and is approximated by the 100 m depth contour. While sea otters can be found at the surface in water deeper than 100 m, either resting or swimming, they are most commonly observed in waters within a few kilometers of shore (Riedman and Estes 1990), and higher densities are frequently associated with shallow water (Laidre et al. 2002). Bodkin and Udevitz (1999) found 80% of the otters in PWS over water depths of < 40 m although the proportion of the seafloor habitat within this bathymetric zone was only about 33%. Sea otters can also be found in high densities where relatively shallow waters or islands occur far offshore (Kenyon 1969). While they periodically haul out on intertidal or supratidal (above the high tide line) shores (particularly during winter months) and generally remain close to the sea/land interface, no aspect of their life history requires leaving the ocean (Kenyon 1969, Riedman and Estes 1990).

Sea otters forage in diverse bottom types, from fine mud and sand to rocky reefs. Recent research employing archival time-depth recorders recovered from sea otters in southeast Alaska showed that 84% of foraging occurred in depths of 2-30 m, and that 16% of all foraging was between 30 and 100 m (Bodkin et al. 2004). Females dove to depths < 20 m on 85% of their foraging dives while males dove to depths > 45 m on 50% of their foraging dives. Recent research from California suggests these patterns are similar among populations (Tinker et al. 2007), but additional research would improve confidence in the generality of these results.

Although it had been speculated that sea otters do not occupy protected inside waters such as occur in southeast Alaska and Puget Sound (Kenyon 1969; Kvitek et al. 1991, Kvitek et al. 1993), recent colonization of such habitat has been observed in PWS and southeast Alaska, including glacial fiords and inside passages such as Icy Straits (Esslinger and Bodkin 2009). Although densities of sea otters are clearly not uniform, within the geographic range currently occupied there is little evidence that particular habitat types are unsuitable. However, there is also clear evidence that some types of habitat are preferred and capable of supporting high densities over extended periods of time (Riedman and Estes 1990). In particular, where canopy-forming kelps occur (including species of *Macrocystis*, *Eularia*, and to a lesser extent *Nereocystis*), they provide preferred resting habitat (Kenyon 1969, Riedman and Estes 1990). However, canopy-forming kelp is not a required habitat element, and high densities of sea otters occur and persist in nearshore habitat where kelp is absent, either seasonally or entirely, as a consequence of unconsolidated substrates. This is particularly evident along coastal mainland Alaska (Kenyon 1969, Riedman and Estes 1990). There also appears to be a positive relationship between shoreline complexity and sea otter density (Kenyon 1969, Riedman and Estes 1990). Again, although not obligatory, headlands, coves, and bays appear to offer preferred resting habitat, particularly to females with pups, presumably because they provide protection from high wind and sea conditions. Another recently recognized habitat attribute is refuge from predators. In a shallow lagoon that limited killer whale access at Adak Island, Estes et al. (1998) found relatively stable sea otter numbers during a period when declines of up to 90% were detected outside the lagoon. A shift in distribution toward very shallow (1-3 m), nearshore, highly protected areas in the Aleutian Islands (USGS unpublished data) that has been detected since the recent decline may represent a response to the risk of predation (see Section 3.A.).

Comparatively few data are available to describe relations between sea otter densities and habitat characteristics, however it is generally recognized that rocky habitats support higher sea otter densities compared to soft sediment habitats (Riedman and Estes 1990, Laidre et al. 2001, Laidre et al. 2002). Estimates of equilibrium densities (those that are approximately stable and consistent over time) across various populations within the sea otters range indicate values that range from about 1.5-17/km^2 and generally consider habitats out to the 40-55 m isobath (depth contour). Burn et al. (2003) and Estes (1977) provide estimates of equilibrium densities for the Aleutian Archipelago and Amchitka Island of 16 and 17/ km2 respectively. Equilibrium densities likely vary among habitats, with reported values specific to rocky habitats of about 1 to 8/km^2 (Laidre et al. 2001, Laidre et al. 2002, Lowry and Bodkin 2005, Gregr et al. 2008). Equilibrium densities from soft-sediment habitats are generally lower at < 1/ km^2. In predominantly soft sediment habitats in PWS sea otter densities vary among areas, averaging about 1.5/km^2 and ranging from fewer than 1 to about 6/km2 (Bodkin and Udevitz 1999, USGS unpublished data), although maximum densities in Orca Inlet, a shallow soft-sediment habitat, are up to 16/km^2. Other estimates of maximum sea otter densities of about 12/km^2 have been reported from the Aleutian and Commander Islands (Kenyon 1969, Bodkin et al. 2000) where habitats are predominantly rocky. In 2003, densities throughout the Aleutian Archipelago are estimated to be approximately 3% of equilibrium density (Estes et al. 2005).

E. Population biology

As in other sexually reproducing species, sea otter populations are ultimately regulated by age- and sex-specific rates of reproduction and survival. The life history patterns of sea otters are more similar to those of the pinnipeds and other marine mammals (with whom they share the ocean as a common environment) than to those of the other lutrines (with whom they share a more recent common ancestor) (Estes and Bodkin 2002). These patterns limit the potential for population increase.

Male sea otters can attain sexual maturity by age 3 but likely do not attain the social maturity required for successful reproduction until >5 years old (Garshelis 1983). Variation in reproductive success among males and the different reproductive strategies they may employ are largely unexplored. Female sea otters attain sexual maturity as early as age 2, and by age 3 most females are sexually mature (Bodkin et al. 1993, Jameson and Johnson 1993). Where food resources limit population growth, sexual maturation may be delayed to 4-5 years of age. Reported reproductive rates of adult females range from 0.80 to 0.98 births/yr (Siniff and Ralls 1991, Bodkin et al. 1993, Jameson and Johnson 1993, Riedman et al. 1994, Monson and DeGange 1995, Monson et al. 2000a, Tinker et al. 2006a). In areas where sea otter reproduction has been studied, reproductive rates appear to be fairly consistent despite differences in resource availability. Gestation, including a period of delayed implantation, requires about six months. Although copulation and pupping can take place at any time of year, there appears to be a positive relation between latitude and reproductive synchrony. In California, pupping is weakly synchronous to nearly uniform across months (Siniff and Ralls 1991, Riedman et al. 1994, Tinker et al. 2006a), whereas in PWS a distinct peak in pupping occurs in late spring. In recently colonized habitats in Alaska where food is plentiful, pupping may be less synchronous than in populations at or near equilibrium density (Bodkin and Monson 2002).

Females give birth to a single pup, although rare instances of twinning have been observed (Jameson and Bodkin 1986). The average duration of birth to weaning is about six months, resulting in a reproductive interval of approximately one year from copulation to weaning. If a female loses her pup prior to weaning, she will soon enter estrus and breed again. Copulation occurs in the water, and a male may remain with an estrous female for several days (Riedman and Estes 1990), although a female may breed with more than one male. Distinctive and sometimes severe wounds can result from the male biting the nose of the female during copulation.

Whereas reproductive output remains relatively constant over a broad range of ecological conditions, pup survival appears to be more strongly influenced by resource availability. At Amchitka Island, where the population was at or near equilibrium density prior to the recent decline, dependent pup survival ranged from 22-40%, compared to nearly 85% at Kodiak Island, where food was not limiting and the population was increasing (Monson et al. 2000a). Female experience apparently is important to the survival of offspring, with primiparous females (new mothers) generally less successful at weaning their pups (Riedman et al. 1994, Monson and DeGange 1995, Tinker et al. 2006a). Body condition (weight/unit length) of the mother at the time of birth also influences pup survival (Bodkin and Monson 2002). Female sea otters must spend large amounts of time grooming and nursing their newborn pups, and keeping them warm and dry on their chest or hauled out on rocks. This necessarily restricts the amount of time they can spend foraging. A female in poor condition will not be able to restrict her feeding time to the extent a female in good condition can, and her pup will be exposed to longer periods in the water, and less grooming and nursing, while she feeds. The result is poorer pup survival during the first few weeks of life, the period during which most pre-weaning pup mortality occurs. This effect may be exaggerated during winter when conditions are particularly harsh. In sea otter populations with limited food resources, pups born in winter are more likely to die soon after birth. This trend, combined with the tendency for females to enter estrus soon after losing a pup, tends to result in greater synchronization of pup production.

Sea otter populations generally consist of more females than males (Kenyon 1969, Bodkin et al. 2000). Age-specific survival of sea otters is generally lower among males (Kenyon 1969, 1982; Siniff and Ralls 1991, Monson and DeGange 1995, Bodkin et al. 2000), although variation in the post-weaning survival of females appears to be the primary mechanism of population regulation around carrying capacity. Populations living with an abundance of food exhibit relatively high survival rates in all age classes compared to food-limited populations, with especially high relative survival in juvenile age-classes. Alternatively, populations at or above equilibrium density, with limited food, show high variation in survival in the weeks following weaning. Post-weaning survival is variable among populations and years, ranging from 18% to 86% (Monson et al. 2000a, Ballachey et al. 2003). In general, once a sea otter survives its first year of life, there appears to be a relatively good probability that it will survive to senescence (old age), where density-dependent mechanisms structure population abundance. Such may not be case where density-independent factors such as predation are important. Survival of sea otters more than 2 years of age is generally high, approaching or exceeding 90%, but gradually declines over time (Bodkin and Jameson 1991, Monson et al. 2000a). In Alaskan and other northern populations, most mortality (other than human-related) occurs during late winter and spring, presumably associated with harsh winter environmental conditions and seasonal declines in prey availability (Kenyon 1969, Bodkin and Jameson 1991, Bodkin et al. 2000, Watt et al. 2000). Maximum ages achieved by sea otters outside captivity are about 22 years for females and 15 years for males.

The fetal sex ratio does not differ from 50:50 (Kenyon 1982, Bodkin et al. 1993), yet sea populations may exhibit unequal sex ratios. Survival of juvenile (post-weaning) males in California exceeded that of juvenile females (Siniff and Ralls 1991) although the opposite was found at Amchitka Island (Kenyon 1969).

Causes of mortality in sea otter populations are difficult to determine. The probability of detecting and assigning cause of death depends on the cause. For example, the carcass of a sea otter that dies of starvation is more likely to be recovered than that of one killed by a predator. Documented sources of mortality include predation, starvation, disease, oil spills, incidental take in fisheries, harvest, and intra-specific aggression. In California, infectious disease was implicated as the cause of death for nearly 40% of 195 carcasses analyzed between 1992 and 1995 (Thomas and Cole 1996) and 63% of 105 carcasses analyzed between 1998 and 2001 (Kreuder et al. 2003). However, these estimates may not accurately reflect the causes of death in the population as recovered carcasses are likely not an unbiased sample (as mentioned above) and the cause is unknown for the majority of deaths: an estimated 40-60% of carcasses are not recovered and the cause of death cannot be determined for about 72% of those that are recovered (Estes et al. 2003, Gerber et al. 2004). Disease factors contributing to mortality included peritonitis caused by acanthocephalan parasites, protozoan encephalitis, coccidioidomycosis, and bacterial infections. As noted previously, sea ice can be a mortality factor as well. Recognized sea otter predators include the white shark (*Carcharodon carcharias*), brown bear (*Ursus arctos*), wolf, red fox, wolverine, killer whale, and bald eagle (*Haliaeetus leucocephalus*) (Kenyon 1969, Ames and Morejohn 1980, Riedman and Estes 1990, Monson and DeGange 1995, Hatfield et al. 1998, Bodkin et al. 2000). Bald eagles prey primarily on young pups. Gelatt (1996) found that mothers with young pups fed less often overall, and when they did feed it was often nocturnally, apparently to avoid exposing their pup to eagle predation. Declining sea otter populations across the Aleutian archipelago during the 1990s have been attributed to increased predation by killer whales (Estes et al. 1998).

Prior to the sea otter decline in the Aleutian Islands late in the 20th century, population densities were generally thought to be limited by prey availability, with mortality being density dependent and increasing during periods of food shortage and severe weather. This pattern of elevated winter mortality, particularly among juveniles and old individuals, was initially observed at Amchitka Island in the 1950s and 1960s (Kenyon 1969). In the Commander Islands, the sea otter population declined by 41% in a single year, following 10 years of increasing density, declining prey populations, and declining weights of adult male otters (Bodkin et al. 2000). It is possible that some of the disease-related mortality in California is ultimately linked to prey availability, and the cessation of population growth in the regions of highest sea otter density in California is associated with declining body condition and increased percentage of time spent foraging, both suggestive of food limitation (Bentall et al. 2005, Tinker et al. 2006a, Tinker et al. 2006b).

Relatively few studies have investigated the relations between the physical and biological attributes that contribute to variation in productivity of nearshore marine invertebrates, such as the clams, mussels, and crabs that sea otters consume, and how that variability in productivity affects variation in annual sea otter survival (Lowry and Bodkin 2005). Given the observed variation in sea otter survival and the recognized role of food in regulating sea otter populations, understanding those relations would facilitate empirical measures of the relative contributions of predation and primary production as controlling factors in structuring nearshore marine communities. Due to the relatively small size of their home ranges, sea otters integrate physical and biological attributes of the ecosystem over small spatial scales. Further, because they occur near shore, sea otters, their prey, and physical and biological ecosystem attributes can be accurately and efficiently monitored, providing a strong foundation for understanding mechanisms and interactions among factors that regulate sea otter populations.

Our understanding of the frequency, magnitude, causes, and consequences of changes in sea otter populations is constrained by the brief temporal perspectives imposed by the short window of human observation and written history. One consequence of this narrow time perspective is that we may view causes and consequences of change as novel, even when they are not. Relatively unexplored evidence from archeological remains suggests that local abundance of sea otters and other nearshore marine species has varied significantly over millennial time scales (Simenstad et al. 1978). An improved understanding of long-term population changes would provide added context for evaluating and responding to contemporary fluctuations in sea otter populations.

F. Behavior

Reproduction

Male sea otters gain access to estrous females by establishing and maintaining territories from which other males are excluded (Kenyon 1969, Garshelis et al. 1984, Jameson 1989). Territories may be located in or adjacent to female resting or feeding areas, or along travel corridors between those areas. Territories are occupied continuously or intermittently over time (Loughlin 1980, Garshelis et al. 1984, Jameson 1989). Male occupancy in a territory may extend over 6 to 9 years (Riedman and Estes 1990). Male territoriality results in partial segregation of the sexes, and males that do not occupy territories tend to reside in dense "bachelor" aggregations (Kenyon 1969, Bodkin et al. 2000). Males that do not defend territories (transients) may gain access to receptive females by traveling through or adjacent to male territories and female areas. Male aggregation areas identified by Kenyon in 1962 at Amchitka Island persisted through at least 1995 (USGS unpublished data). In California, many males that defend territories for part of the year may periodically move to bachelor aggregation areas, apparently to take advantage of seasonally and locally abundant food resources (Jameson 1989, Tinker et al. 2006b, Kage 2004). Female choice in mate selection is facilitated by females traveling among male territories, although males may try to sequester estrous females within their territory. Adult male sea otters in California maintain territories that average about 0.4 km^2 (Jameson 1989). Adult females apparently move freely among these territories, but the territory holder aggressively excludes juvenile males. Although the role of male territoriality in regulating population density is largely unexplored, territories likely serve to increase individual reproductive success.

Adult males frequently harass females with large pups in an apparent effort to force separation, thus inducing the female to enter estrus so that he can inseminate her and she will bear his offspring. Copulation occurs repeatedly during a brief consortship, after which the male and female separate.

The sea otter's polygynous mating system (one male can mate with more than one female) likely evolved in response to the high densities they achieved in prehistoric populations not affected by humans, which promoted male competition for females. Polygynous mating systems are typical of all otariids (eared seals) and some phocids (true seals) in temperate-latitude systems but apparently are rare or absent in other species of otters. As is true for many polygynous species, male sea otters provide no parental care (Estes 1989).

Diving

Diving occurs during grooming, traveling and foraging. Grooming dives usually occur before feeding or resting periods, and are of short duration and shallow depth. Because locomotion is more efficient under water than on the surface (Williams 1989), otters frequently make relatively long (30-60 s), shallow dives while traveling between resting and feeding areas. Foraging dives are predominantly to the sea floor, although canopy foraging in kelp forests for snails and crabs is common in California. Foraging dives are typically longer in duration with greater rates of descent and ascent than dives of other function. Attributes of dives (duration, depth, ratio of bottom time to dive time, and rates of descent and ascent) are significantly different for traveling, grooming, and foraging dives (Bodkin et al. 2004), and can be used to classify dives according to their function.

Three general types of diving data have been obtained through direct visual observation (Estes et al. 1981, Riedman and Estes 1990, Calkins 1978, Garshelis 1983, Doroff and Bodkin 1994, Doroff and DeGange 1994), radio-telemetry with remote information acquisition (Ralls et al. 1996), and most recently archival time-depth recorders (TDRs) (Bodkin et al. 2004, Tinker et al. 2006b). Dive attributes from visual observations include dive duration, surface intervals between dives, and approximate water depths at the estimated dive locations. These data are inherently biased against animals foraging well away from shore (Ralls et al. 1996). Thus, estimates of average and maximum dive duration from animals instrumented with radio-transmitters are substantially longer than those obtained visually (Ralls et al. 1996, Bodkin et al. 2004).

Mean swimming speeds during descent and ascent in foraging dives average about 1 m/s. In California, average dive times are longest for juvenile males and shortest for adult females with dependent pups (Ralls et al. 1996). These differences are likely because juvenile males forage in deeper water offshore while females with pups forage in shallower water near shore. Maximum reported dive durations are 246 s in California (Ralls et al. 1996) and 386 s in Alaska (USGS unpublished data). Dive times and surface intervals correlate with water depth, although the deepest dives are not necessarily associated with maximum dive times. Surface intervals are highly correlated with prey size and type, with the longest intervals associated with the largest prey, thus reflecting handling and consumption times (Ralls et al. 1996, Tinker et al. 2007). Sea otters commonly dive to depths exceeding 40 m in the Aleutian Islands and there is one record of a sea otter drowned in a crab pot set in 91 m of water (Newby 1975).

In California, it was recently demonstrated that TDR data alone are sufficient to detect dietary differences between individuals (Tinker et al. 2007), and information recently obtained from southeast Alaska using TDRs confirms the foraging specializations documented in California (Estes et al. 2003a, Tinker et al. 2007). Archival TDRs identified individual- and sex-related differences in mean and maximum foraging dive attributes and depth distributions (Bodkin et al. 2004). A bimodal pattern in forage depth distribution was detected for most of the individuals sampled, with peaks in foraging between 5-15 m and 30-60 m. Generally, adult females dove to shallower depths than adult males, although some females regularly dove to depths exceeding 60 m. Most adult males foraged at depths between 40 and 60 m, although several repeatedly dove to depths exceeding 60 m. Maximum dive depths were 76 m for females and 100 m for males (Bodkin et al. 2004). Similar patterns appear to occur in California (Tinker et al. 2006b, USGS unpublished data), but more research is needed to document the generality of these findings in other populations.

Activity budgets

Time budgets describe the allocation of time to specific categories of behavior, such as resting, grooming, foraging, or social interaction. The fundamental premise is that food resources frequently limit population abundance, and that the proportion of time individuals allocate to foraging reflects food availability (Gelatt et al. 2002). Sea otter activity budgets have been estimated at 7 different locations and during 13 different time periods, including populations that were increasing, stable, and decreasing (Gelatt et al. 2002). Despite this breadth of research, conclusions about the utility of time budgets to assess sea otter population status have been inconsistent (Shimek and Monk 1977, Loughlin 1979, Estes et al. 1982, Estes et al. 1986, Garshelis et al. 1986, Ralls and Siniff 1990, Gelatt et al. 2002). Some of the divergence in results and conclusions stems from differences in accuracy and precision produced by various methods (visual vs. telemetry), sampling designs (day vs. day and night, and sample sizes), uncertainty in population status relative to carrying capacity at the time of study, and the sometimes overwhelming influence of sampling variance (including within-individual variation across time, as well as variation among individuals, sexes, and age cohorts).

In general, adult male sea otters spend less time foraging (37%) than younger males (44%) or females, adult females spend less time foraging (36-47%) than younger females (50%), and females with young pups spend less time feeding (21%) than females with older pups (52%) (Gelatt et al. 2002). Recent calculations of time budgets based on TDR data from southeast Alaska (Bodkin et al. 2007a) and comparisons between telemetry- and TDR-based activity budgets in California (Tinker et al. 2006b, Bentall 2005, USGS unpublished data) support the premise that the amount of time allocated to foraging reflects prey availability, and thus TDR-derived activity budgets may be of use in evaluating population status relative to equilibrium density.

Movements and home ranges

Sea otters exhibit complex movement patterns related to habitat characteristics, social organization, and reproductive biology. It is likely that movements differ depending on whether a population is at or near carrying capacity or has access to unoccupied suitable habitat into which it can expand (Riedman and Estes 1990). Most research on sea otter movements has been conducted where unoccupied habitat is available to dispersing animals. For example, dominant adult males in California generally occupy and defend relatively small territories, which vary seasonally between summer-fall (4.0 km^2; 1.1 km of coastline) and winter-spring (7.8 km^2; 2.2 km of coastline), frequently moving relatively long distances to reach male aggregations (average = 80.1 km; maximum = 418 km) (Jameson 1989, Ralls et al. 1996, Kage 2004, USGS unpublished data). Juvenile females may also occasionally make very long movements (maximum = 235 km; USGS unpublished data). Adult females rarely move farther than 20 km, although they generally occupy home ranges larger than the territories of territorial males (Ralls et al. 1988). Early research in the Aleutian Islands by Lensink (1962) and Kenyon (1969) found that males had larger home ranges than females, and those authors described the female sea otter's home range as including 8-16 km of contiguous coastline. Adult male home ranges in PWS are 4.6-11.0 km^2 and adult female home ranges are 1.0-4.8 km^2 (Garshelis and Garshelis 1984). In PWS, a telemetry study documented movements by adult males of up to 100 km between male aggregation areas and breeding areas (Garshelis and Garshelis 1984a). Recent estimates of annual home range sizes in western PWS were significantly smaller for territorial males[1] (50% kernel home range = 5.4 km^2, and 90% kernel home range = 9.6 km^2) than for adult females (50% kernel home range = 6.8 km^2, and 90% kernel home range = 23.8 km^2) (Ballachey and Bodkin 2006). Additional telemetry studies on juveniles in PWS, and adult males and females along the Alaska Peninsula and in the Kodiak archipelago (USGS unpublished data, Monnet et al. 1988), documented movements typically of 50 km or less. Comparable data are not available from other areas in Alaska.

Although annual and lifetime home ranges suggest limited movements throughout much of their range, sea otters can move much greater distances when translocated. In California, translocated animals returned as much as 318 km from release sites back to capture sites (Ralls et al. 1992). In Alaska, a female treated and then released after the *T/V Exxon Valdez* oil spill traveled 400 km from her release site (Monnet et al. 1990).

While sea otters are somewhat constrained by the 100 m depth profile, they can navigate some distance over deep water. Translocated sea otters have traveled distances of up to 50 km, over water deeper than their maximum foraging depth, from San Nicolas Island to the mainland along southern California (Rathbun et al. 1990). Similar travel, but unrelated to translocation, was documented by Bodkin et al. (2000) from Medny to Bering Island in the Commander Islands.

Conventional very high frequency (VHF) telemetry has most commonly been used to estimate sea otter home ranges and movements. Because relocation probabilities diminish as distance from last location increases, movement estimates from such studies may be negatively biased. Additional research employing technologies that reduce recognized sources of bias (e.g., global positioning system, GPS) should provide better estimates. Also, in the event of localized extinctions within geographically isolated segments of habitat (e.g., island groups in the Aleutian Islands), recolonization may become more problematic as the required dispersal distance increases.

G. *Foraging ecology*

Sea otters are generalist predators, known to consume more than 150 different prey species (Kenyon 1969, Riedman and Estes 1990, Estes and Bodkin 2002). With few exceptions, their prey consists of sessile or slow-moving benthic invertebrates such as mollusks, crustaceans, and echinoderms. Foraging occurs in habitats with rocky and soft-sediment substrates from the high intertidal to depths slightly in excess of 100 m. Preferred foraging habitat is generally in depths of less than 40 m (Riedman and Estes 1990), although studies in southeast Alaska have found that some animals forage mostly at depths of 40-80 m (Bodkin et al. 2004).

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[1] The "% kernel home range" is the area where an animal can be found that % of the time.

The diet of sea otters is usually studied by observing prey items brought to the surface for consumption, and therefore diet composition is usually expressed as a percentage of all identified prey that belong to a particular prey species or type. Although the sea otter is known to prey on a large number of species, only a few tend to predominate in the diet of individual otters in any particular area. Prey type and size depend on location, habitat type, season, and length of occupation. In California, otters foraging over rocky substrates and in kelp forests mainly consume decapod crustaceans, gastropod and bivalve mollusks, echinoderms, and worms (Ebert 1968, Estes et al. 1981, Tinker et al. 2008). In protected bays with soft sediments, otters mainly consume infaunal clams (*Saxidomus nuttallii* and *Tresus nuttallii*) (Kvitek et al. 1988). Along exposed coasts with soft sediments, the Pismo clam (*Tivela stultorum*) is a common prey (Stephenson 1977). Important prey in Washington State include crabs (*Cancer* spp., *Pugettia* spp.), octopus (*Octopus* spp.), intertidal clams (*Protothaca* spp.), sea cucumbers (*Cucumaria miniata*), and red sea urchins (*Strongylocentrotus franciscanus*) (Kvitek et al. 1989). The predominantly soft-sediment habitats of southeast Alaska, PWS, and Kodiak Island support populations of clams that are the primary prey of sea otters. Throughout most of southeast Alaska, burrowing clams (species of *Saxidomus*, *Protothaca*, *Macoma*, and *Mya*) predominate in the sea otter's diet (Kvitek et al. 1993). They account for more than 50% of the identified prey, although urchins (*S. droebachiensis*) and mussels (*Modiolis modiolis*, *Mytilus* spp., and *Musculus* spp.) can also be important. In PWS and Kodiak Island, clams account for 34-100% of sea otter prey (Calkins 1978, Doroff and Bodkin 1994, Doroff and DeGange 1994). Mussels (*Mytilus trossulus*) apparently become more important as the duration of occupation by sea otters increases, ranging from 0% at newly occupied sites at Kodiak to 22% in long-occupied areas (Doroff and DeGange 1994). Crabs (*C. magister*) were once important sea otter prey in eastern PWS, but apparently have been depleted by otter foraging and are no longer eaten in large numbers (Garshelis et al. 1986). Sea urchins are minor components of the sea otter diet in PWS and the Kodiak archipelago. In contrast, the diet in the Aleutian, Commander, and Kuril Islands is dominated by sea urchins and a variety of fin fish (including those in the taxomic families of hexagrammidae, gadidae, cottidae, cyclopteridae, and scorpaenidae; Kenyon 1969, Estes et al.1982). Sea urchins tend to dominate the diet of low-density sea otter populations, whereas more fish are consumed in populations near equilibrium density (Estes et al. 1982). For unknown reasons, fish are rarely consumed by sea otters in regions east of the Aleutian Islands.

Sea otters also exploit episodically abundant prey such as squids (*Loligo* spp.) and pelagic red crabs (*Pleuroncodes planipes*) in California and smooth lumpsuckers (*Aptocyclus ventricosus*) in the Aleutian Islands (Watt et al. 2000). On occasion, sea otters attack and consume sea birds, including at least teals (*Anas crecca*), scoters (*Melanita perspicillata*), loons (*Gavia immer*), gulls (*Larus* spp.), grebes (*Aechmophorus occidentalis*), and cormorants (*Phalacrocorax* spp.; Kenyon 1969, Riedman and Estes 1990).

Diet diversity generally increases through time as otter populations colonize new habitats and grow toward resource limitation (Estes et al. 1981, Estes and Bodkin 2002). This has been chronicled in the Aleutian Islands, PWS, and California. Diversification is probably the consequence of otters reducing the abundance of their preferred or most abundant prey. Studies of marked sea otters in California have shown extreme individual variation in diet and foraging behavior (Estes et al. 2003b, Tinker et al. 2007), although such individual variation apparently does not occur when food is very abundant, such as at San Nicolas Island (Bentall 2005, Tinker et al. 2008). In the center of the range where food is most limiting, adult otters tend to specialize on 1-3 prey types (Estes et al. 2003a, Tinker 2004, Tinker et al. 2007, 2008). This individual variation cannot be attributed to phenotypic or environmental variation, as different individuals of the same sex and age class often consume different prey at the same time and place. Dietary patterns, which appear to be passed from mothers to their offspring, are known to persist for years and may be life-long characteristics of individuals. The causes and consequences of individual foraging patterns remain uncertain.

Sea otters are well known for the effects their foraging has on the structure and function of coastal marine communities. They provide an important example of the "keystone species" concept (Power et al. 1996). In the absence of sea otter foraging during the 20th century, populations of several species of urchins became extremely abundant. Grazing activities of urchins effectively limited kelp populations, resulting in deforested areas known as "urchin barrens" (Lawrence 1975, Estes and Harrold 1988). Because sea urchins are a preferred prey item, as otter populations recovered they dramatically reduced the sizes and densities of urchins, as well as other prey such as mussels. Released from the effects of urchin herbivory, populations of macroalgae responded, resulting in diverse and abundant populations of understory and canopy-forming kelp forests. Although other factors, both abiotic and biotic, can also limit sea urchin populations (Foster and Schiel 1988, Foster 1990), the generality of the sea otter effect in reducing urchins and increasing kelp forests is widely recognized (Estes and Duggins 1995). Further cascading effects of sea otters in coastal rocky subtidal communities may stem from the proliferation of kelp forests. Kelp forests provide food and habitat for other species, including fin fish (Simenstead et al. 1978, Ebeling and Laur 1988), that in turn provide forage for birds and mammals. Furthermore, kelp constitutes a source of detrital organic carbon for the coastal marine community (Duggins et al. 1989). Additional evidence of the strong trophic cascade precipitated by sea otter predation on urchins is provided by the significant declines in kelp abundance in the Aleutian Islands as a result of increased urchin grazing following the sea otter decline in the 1990s (Estes et al. 1998).

Effects of sea otter foraging also are documented in rocky intertidal and soft-sediment marine communities. The size-class distribution of mussels in the rocky intertidal zone was strongly skewed toward animals < 40 mm shell length where otters were present, whereas mussels > 40 mm comprised a large component of the population where sea otters were absent (VanBlaricom 1988). In soft-sediment coastal communities, sea otters forage on epifauna (crustaceans, echinoderms and mollusks) and infauna (primarily clams). They generally select the largest individuals. This causes declines in prey abundance and reductions in size-class distributions, although the deepest-burrowing clams (e.g., *Tresus nuttallii* and *Panopea generosa*) may gain refuge from some sea otter predation (Kvitek and Oliver 1988, Kvitek et al. 1992). Community-level responses to reoccupation of habitat by sea otters are much less well studied in soft-sediment habitats that dominate much of the North Pacific, and our knowledge of sea otter ecology will benefit from additional research in this area.

H. Population history

Little is known about the long-term patterns of sea otter abundance prior to the 18th century, or the factors that contributed to variation in populations over pre-historic time scales. There is evidence that the indigenous people of the North Pacific made use of the sea otter as a resource, and possibly affected abundance. It is apparent that coastal people, particularly in the northern regions of the Pacific, had developed sophisticated and efficient methods for hunting sea otters long before the advent of modern firearms (Bodkin 2000). Further, skeletal remains of sea otters are common in coastal middens, the accumulated refuse at sites of human occupation. In the Aleutian Islands, variation in the composition and abundance of marine species suggests that sea otter populations were depleted or eliminated (at least locally) by human harvests (Simenstad et al. 1978).

The discovery of large numbers of sea otters in Alaska by Vitus Bering in 1741 initiated a commercial sea otter harvest that lasted 170 years. By the beginning of the 20th century, the species was on the brink of extinction, with a pre-harvest population of approximately 300,000 reduced to perhaps a few thousand individuals, distributed in 13 isolated colonies. Following protection in 1911, two of the remnant colonies disappeared, while the others began the process of recovery. Otter density at the Rat Islands in the central Aleutians apparently reached the pre-exploitation level in the mid 1940s (Kenyon 1969) and fluctuated around an equilibrium density until late in the 20th century (Estes 1990). Throughout much of the 20th century, remnant sea otter populations experienced growth, although annual rates of increase varied, averaging 9% and ranging from 6-13% (Bodkin et al. 1999). Range expansion of remnant populations into unoccupied habitat resulted in nearly complete occupation of northern habitats between the Kuril Islands and PWS by late in the 20th century.

Reintroductions were undertaken in the 1960s and 1970s in southeast Alaska, British Columbia, Washington, and Oregon (Jameson et al. 1982), and sea otters were moved to San Nicolas Island (southern California) in 1987 (Rathbun et al. 1990). Most of the translocated populations increased at rates of 17-20% per year, about the theoretical maximum for sea otters (Estes 1990), and growth rates were significantly greater than the rates observed among remnant populations (Bodkin et al. 1999). The reasons for greater growth rates in translocated populations likely include high prey densities resulting from the prolonged absence of sea otters and minimal impacts from humans. Southeast Alaska, British Columbia, and Washington now support viable sea otter populations. Large amounts of habitat remain available for recolonization in the inside waters of southeast Alaska and south to central California, and along the Kamchatka Peninsula in Russia.

While sea otter populations generally increased in number and range during the 20th century, there are exceptions. Most notable is the dramatic decline in the Aleutian archipelago (Estes et al. 1998, Doroff et al. 2003, Burn and Doroff 2005), the impetus for this recovery plan. Several thousand sea otters were killed by the *Exxon Valdez* oil spill in 1989. While the overall impact of this event on the population in that area is unclear, chronic detrimental effects appear to have persisted through the 1990s (Dean et al. 2000, Monson et al. 2000b, Bodkin et al. 2002). The California sea otter population has generally increased slowly throughout most of the 20th century (Estes et al. 2003b). Since the early 1990s it has grown at a slow and intermittent rate, averaging, 2-3% per year overall but with periods of decline in the late 1990s and 2007-2009 (Bentall 2005). Reintroduced sea otter populations in British Columbia and Washington have generally increased in numbers and range (Bodkin et al. 1999), while in southeast Alaska rates of increase appear to have moderated beginning in the 1990s, except in Glacier Bay where the population increased from near zero in 1995 to >2,400 in 2004 (Esslinger and Bodkin 2009, Bodkin et al. 2007b).

I. Population abundance and trends

Sea otter survey methods vary in different locations due to habitat, geography, and logistics.

In some surveys of southwest Alaska, sea otters have been counted from boats or aircraft within a band of water adjacent to the shoreline; in others, transects have been used to sample an area, and the resulting sea otter density estimate has been extrapolated to generate a population estimate for the entire study area. Sea otters spend considerable time foraging and traveling under water, and it is often not possible to count all individuals that are below the surface at the time a survey is conducted. Also, observers may not see every individual present on the surface. Some survey designs include methods that allow for calculation of a correction factor to adjust for the estimated proportion of otters not detected by observers. One way to make this adjustment requires an independent estimate of the actual number of otters present in an area, also known as "ground-truth," combined with the regular survey data in order to calculate a correction factor to adjust for sea otters not detected. Thus, there are four types of survey results: 1) censuses where it is assumed that all otters are counted by observers; 2) censuses where counts are adjusted for otters not detected; 3) estimates extrapolated from sample counts made in a portion of the study area, without adjustment for otters not detected; or 4) estimates extrapolated from sample counts and adjusted for detection. In cases where uncorrected sea otter counts or estimates are compared over time, the assumption is made that the proportions of otters not detected by observers is constant.

The southwest Alaska sea otter DPS inhabits a vast region, with a linear distance along the coast from the eastern edge (Kodiak Island) to the western edge (Attu Island) of approximately 2,500 km. For several reasons, the Recovery Team concluded that it would be impractical to develop a recovery plan for this region as if it were a single homogeneous unit. Across this range there are major differences in physical habitat features, for example: steep rocky islands with deep water very near shore in the Aleutians; extensive shallow waters with predominantly sandy substrates in Bristol Bay; and a mix of features along the southern Alaska Peninsula and in the Kodiak area. While biological communities show similarities across this region, there are differences in species dominance and presumably also in ecosystem processes. Furthermore, there are regional differences in the pattern of decline of sea otters, and perhaps also in the causes of decline and the threats to recovery. Therefore, the Team decided to develop this recovery plan using five management units (MUs), as listed below and shown in Figure 3:

- Western Aleutian Islands MU
- Eastern Aleutian Islands MU
- Bristol Bay MU
- South Alaska Peninsula MU
- Kodiak, Kamishak, Alaska Peninsula MU

In the following discussion of population abundance and trends, results are presented separately for each MU. Unless otherwise specified, the survey results are unadjusted for otters not detected by observers. Within each study area, recent surveys were conducted using methods similar to those used prior to the recent population decline, so that counts or estimates would be as comparable as possible with baseline information for that area. Although there may be slight differences in the time of year that surveys were conducted, such timing differences do not hinder comparisons of survey results because otters have relatively small home ranges and seasonal redistribution over large areas is unlikely.

Western Aleutian Islands MU (Figure 4)

The first systematic, large-scale population surveys of sea otters in the Aleutian Islands were conducted from 1957 to 1965 (Kenyon 1969). At that time, the descendants of two remnant colonies had expanded throughout the Rat, Delarof, and western Andreanof Island groups. The total unadjusted count for this MU during the 1965 survey was 9,657 sea otters. In 1965, sea otters were believed to have reached equilibrium densities throughout roughly one-third of the Aleutian archipelago, ranging from Adak Island in the east to Buldir Island in the west (Estes 1990). Within this MU, the Near Islands had few sea otters in 1965, and additional population growth and range expansion was expected.

In a 1992 aerial survey of the entire Aleutian archipelago, Evans et al. (1997) counted a total of 6,518 otters in the Western Aleutian MU, which was 3,139 (33%) fewer than the total reported for the 1965 survey. Surveys conducted in the 1970s and 1980s, documented range expansion into the Near Islands, in the far western Aleutians, with a population growth rate of nearly 20% per year in that island group (Estes 1990). The results of the 1992 aerial survey provided the first indication of a sea otter population decline in southwest Alaska. Subsequent surveys conducted from skiffs during the mid-1990s indicated that substantial declines had occurred at several islands in the western and central Aleutians (Estes et al. 1998).

In April 2000, the FWS repeated the 1992 aerial survey of the Aleutian archipelago and counted 1,750 sea otters in this MU, indicating a 73% decline from the 1992 count (Doroff et al. 2003) and an 80% decline from 1965. Aerial and skiff survey data both indicate that the decline began in the late 1980s or early 1990s and proceeded at an average rate of -17.5% per year (Doroff et al. 2003). Although otters declined in all island groups within the archipelago, the greatest declines were observed in island groups within the Western Aleutian MU. This result was unexpected, as the remnant colonies at those island groups had been the first to recover from the effects of commercial

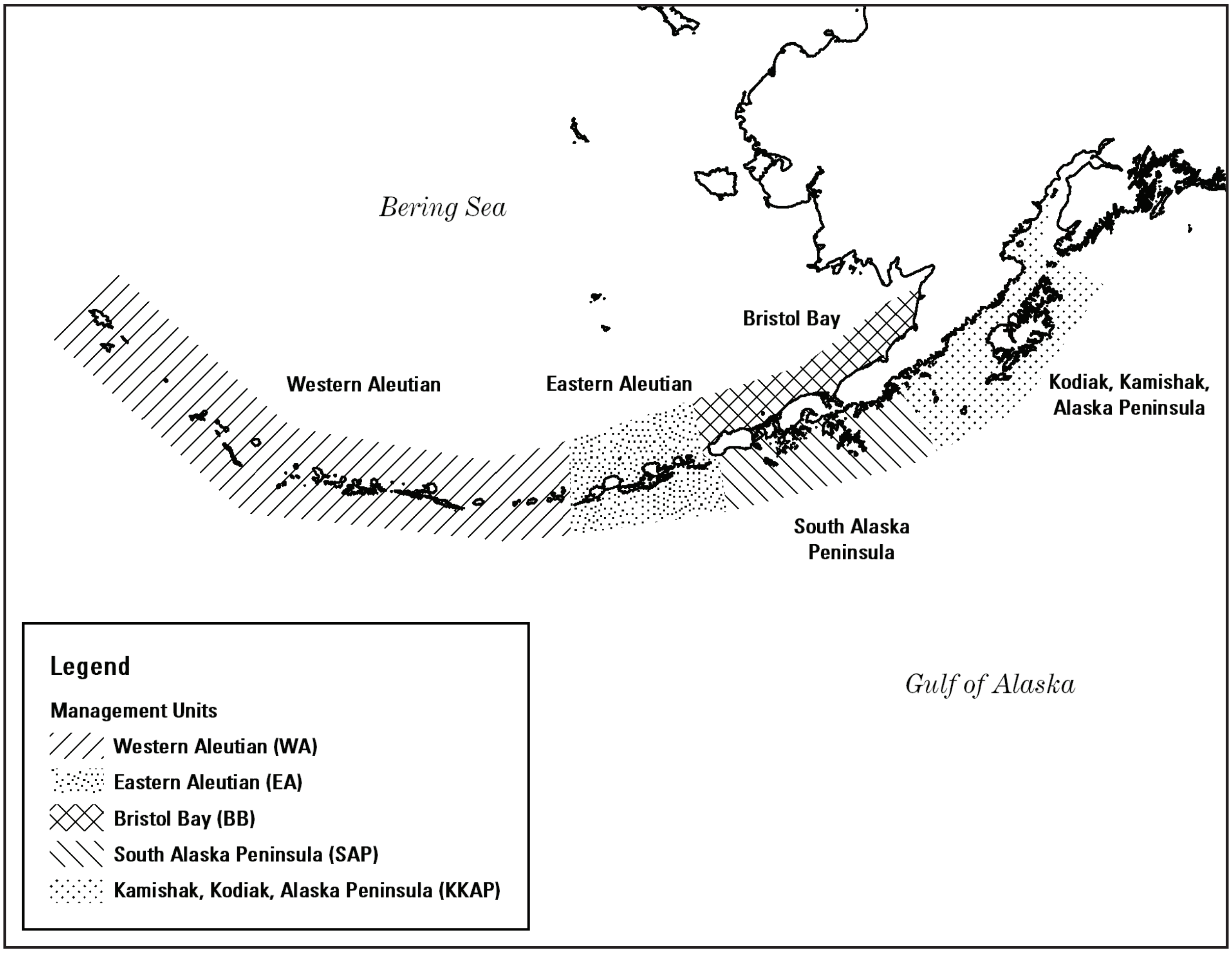

Figure 3. Management units for the southwest Alaska DPS of the northern sea otter.

harvest, and sea otters were believed to have been at equilibrium density at most of these islands in the mid-1960s.

Doroff et al. (2003) used skiff-based counts at six islands in the western and central Aleutians to develop a minimum correction factor, and calculated that aerial observers detected roughly 28% of the sea otters present. Adjusting for otters not detected by observers, the estimated population size of this MU in April 2000 was 6,250. Skiff surveys at these islands conducted in 2003 indicated that the population had declined by a further 63% at an estimated annual rate of -29% per year (Estes et al. 2005). FWS conducted additional skiff surveys of sea otters in the Near Islands and Rat Islands in 2005 and 2007. Counts at most islands appeared to have leveled off or slightly increased as of 2007. Skiff surveys conducted by the USGS at Adak Island indicated that the population size there had stabilized for the past 5 years at around 7-8% of the pre-decline value. Although this MU has not been surveyed comprehensively in recent years, trends from skiff survey data suggest that current sea otter abundance is considerably lower than the 6,250 estimated during the April 2000 aerial survey.

Of the five MUs, the record of survey data for the Western Aleutian MU is the most detailed. In addition to broad-scale aerial surveys conducted in 1965, 1992, and 2000, there is a long history of skiff surveys around several islands throughout the area. The islands that have been surveyed have a wide range in physical size, available habitat, and sea otter populations that have been supported historically. It is therefore difficult to represent sea otter population trends graphically, as changes that have occurred at large islands tend to dwarf those at smaller islands. In order to standardize the changes in population size, survey results can be represented as the proportion of the population that remains, relative to some baseline level. Figure 5 illustrates the population trend in the Near Island group based on time series of skiff survey data collected at Attu and the Semichi Islands, and aerial survey data collected for the entire group in 1992 and 2000. The data indicate that the population in this island group underwent a rapid decline during the mid-1990s, with a current abundance that is approximately 5% of baseline levels. It is not clear, however, if these baseline levels from the early 1990s represent a population that was at equilibrium density.

Population trends for the Rat Island group are shown in Figure 6. Using aerial and skiff survey

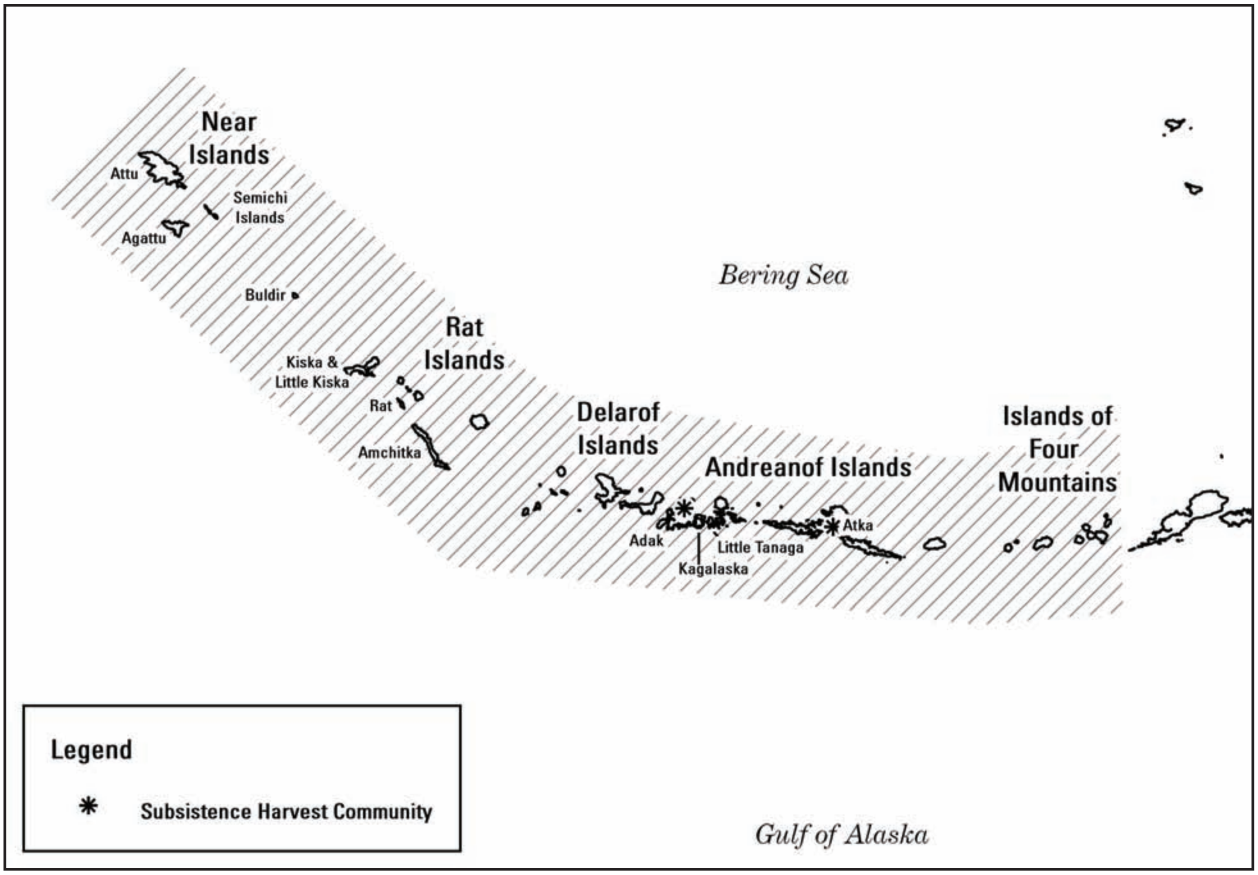

Figure 4. Western Aleutian Management Unit (WA).

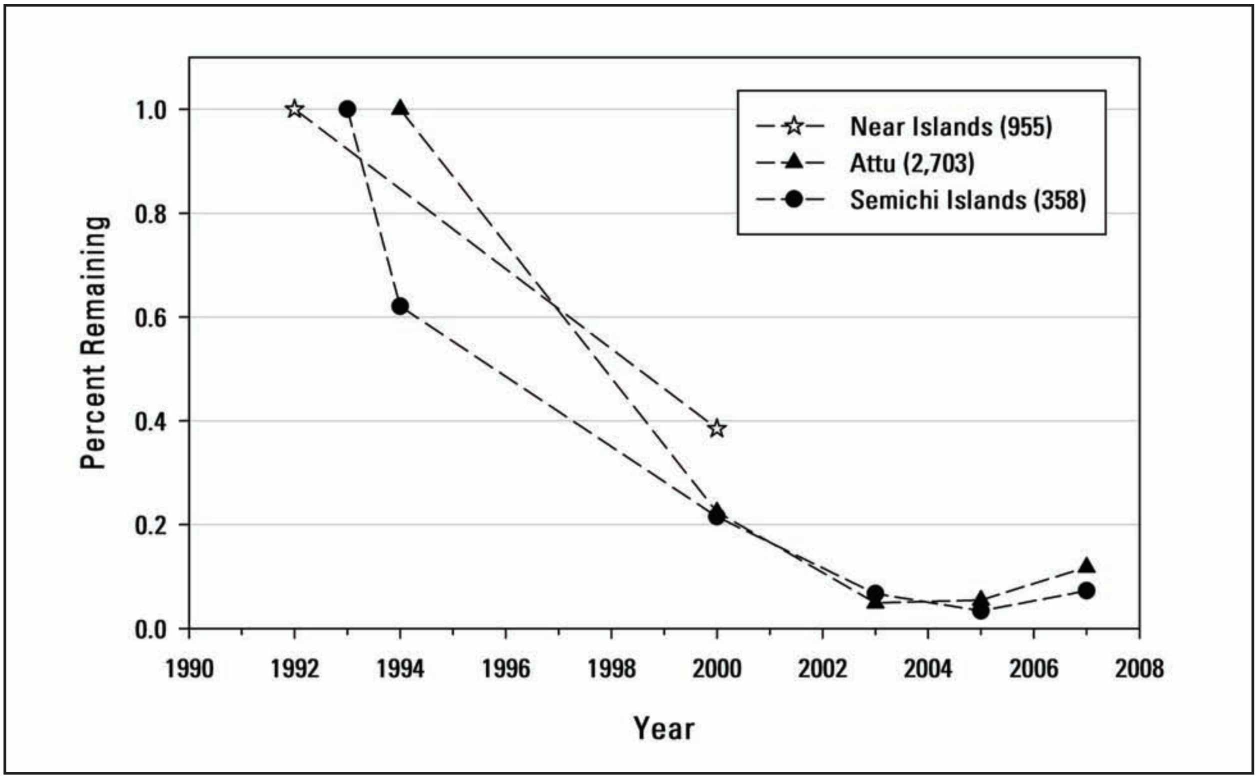

Figure 5. Sea otter survey results for the Near Island group.
Results expressed as the percentage of the population remaining relative to baseline levels (in parentheses). Solid symbols are skiff survey data; hollow symbols are aerial survey data.

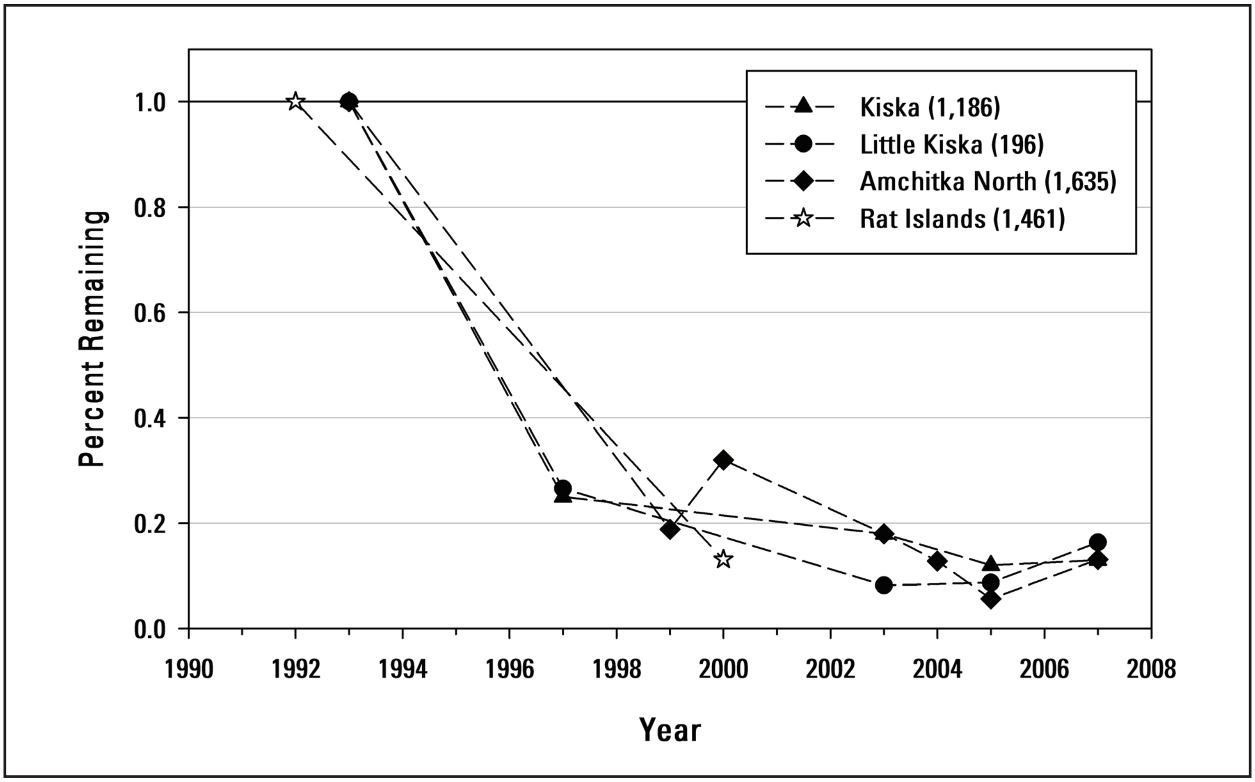

Figure 6. Sea otter survey results for the Rat Island group.
Results expressed as the percentage of the population remaining relative to baseline levels (in parentheses). Solid symbols are skiff survey data; hollow symbols are aerial survey data.

data from the early 1990s as the baseline, it is clear that this group also underwent rapid population declines in the first half of the 1990s. The most recent surveys conducted at Kiska and on the north side of Amchitka in 2007 suggest that sea otter abundance has leveled off, with current abundance at less than 10% of baseline values. As the sea otter population in the Rat Islands was believed to have been at equilibrium density in 1965, and the population had shown a marked decline by the time of the 1992 aerial survey, it is likely that the baseline levels presented in Figure 6 represent a population that was already below equilibrium density.

The most detailed skiff survey data for this MU pertain to Adak Island in the Andreanof Island group, with 14 surveys conducted between 1991 and 2007 (Figure 7). Although the sea otter population in this area appears to have been relatively stable since 2003, the remaining population is less than 10% of the baseline value. Similar to the Rat Islands, much of the Andreanof Island group was believed to have been at equilibrium density in 1965, and it had declined to below that level by the time of the 1992 aerial survey. Therefore, it is likely that these baseline levels from the early 1990s represent a population that was below equilibrium density.

In summary, both skiff and aerial survey data in the Western Aleutian MU indicate rapid population declines during the early to mid-1990s, with current levels approximately an order of magnitude below those at the onset of the declines.

Eastern Aleutian Islands MU (Figure 8)
Similar to the western Aleutians, the island groups in the Eastern Aleutian MU were surveyed from 1957 to 1965 (Kenyon 1969). No remnant colonies of sea otters were present in this area, and recolonization had barely begun by the 1965 survey when only 43 otters were counted (Kenyon 1969). By the time of the next aerial survey in 1992, otters were present throughout the Fox Islands and Krenitzen Islands, where Evans et al. (1997) counted 1,530 individuals. By 2000, however, sea otter abundance had declined from the 1992 level by an estimated 55%, to 692 otters (Doroff et al. 2003). Adjusting this count for detection gives an abundance estimate of 2,492. Additional aerial surveys at several sites in the Krenitzen Islands group in July 2004 indicated that further population declines had occurred. Figure 9 illustrates sea otter population trends in the Eastern Aleutian MU relative to the 1992 aerial survey baseline. Burn et al. (2003) classified the islands in this group as not having reached equilibrium density at the onset of the population decline.

In addition to the aerial survey data record, the Qawalangin Tribe of Unalaska has conducted skiff surveys at Unalaska Island since 2002 as part of a cooperative agreement between FWS and the Alaska Sea Otter and Steller Sea Lion Commission.

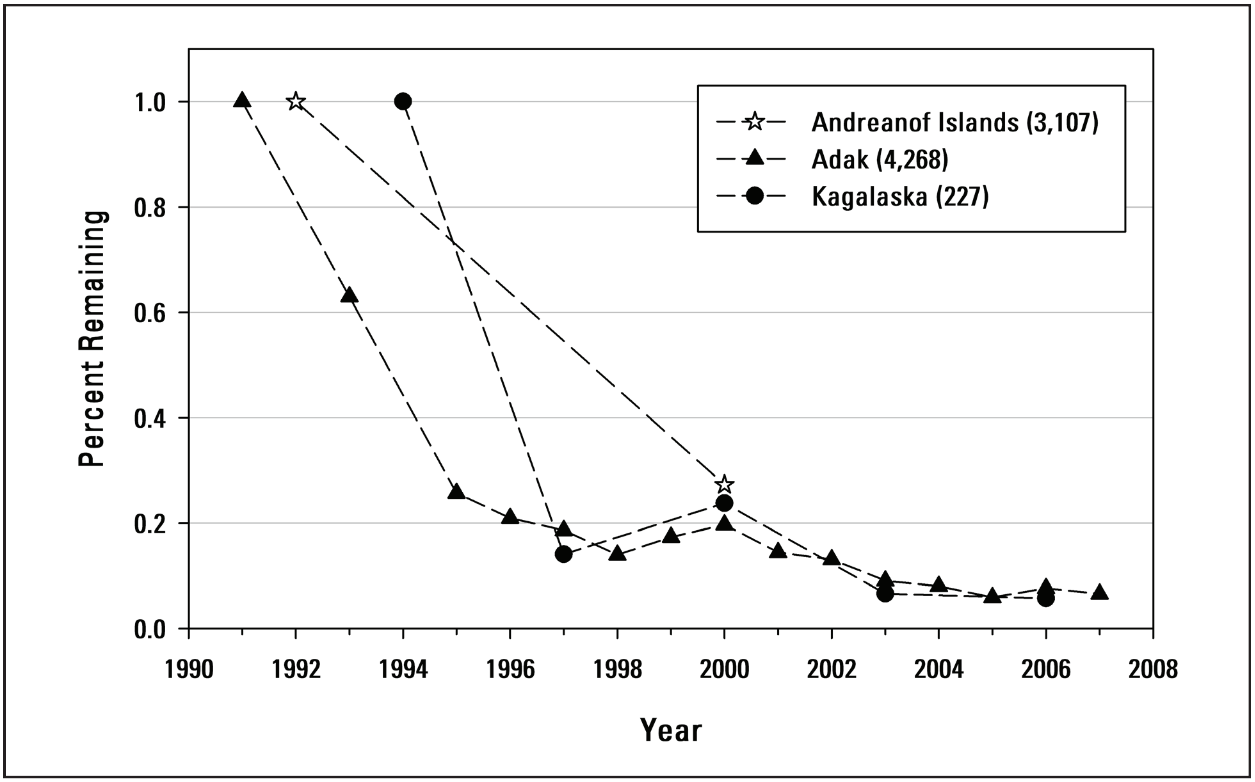

Figure 7. Sea otter survey results for the Andreanof Island group.
Results expressed as the percentage of the population remaining relative to baseline levels (in parentheses). Solid symbols are skiff survey data; hollow symbols are aerial survey data.

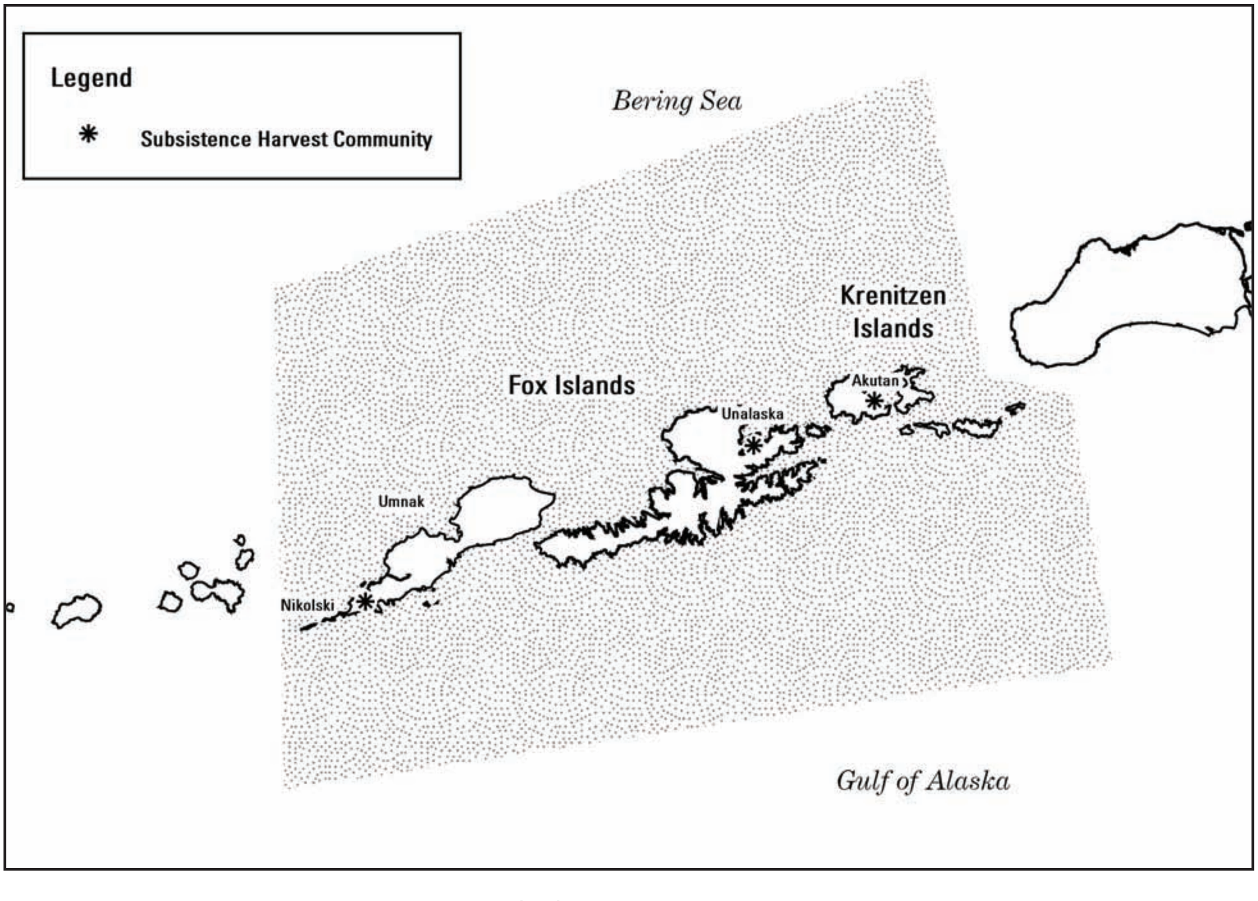

Figure 8. Eastern Aleutian Management Unit (EA).

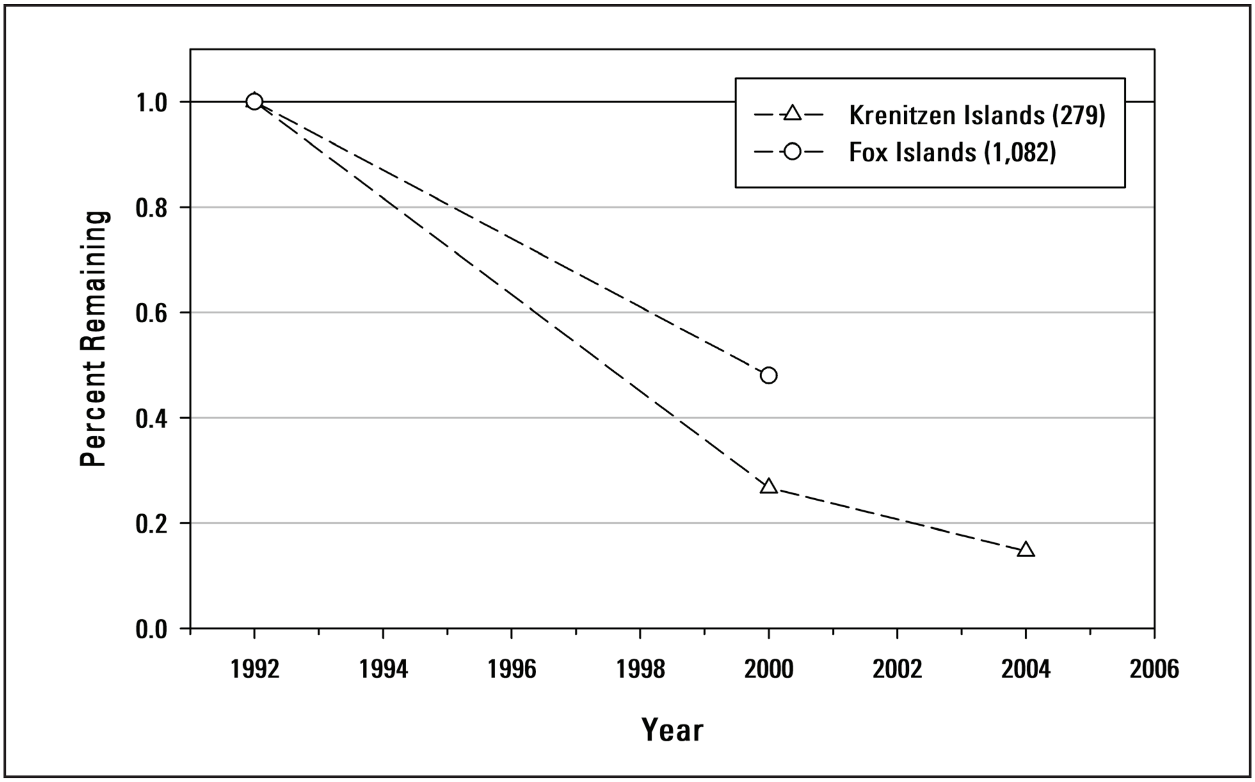

Figure 9. Sea otter survey results for the Eastern Aleutian MU.
Results expressed as the percentage of the population remaining relative to initial levels (in parentheses).

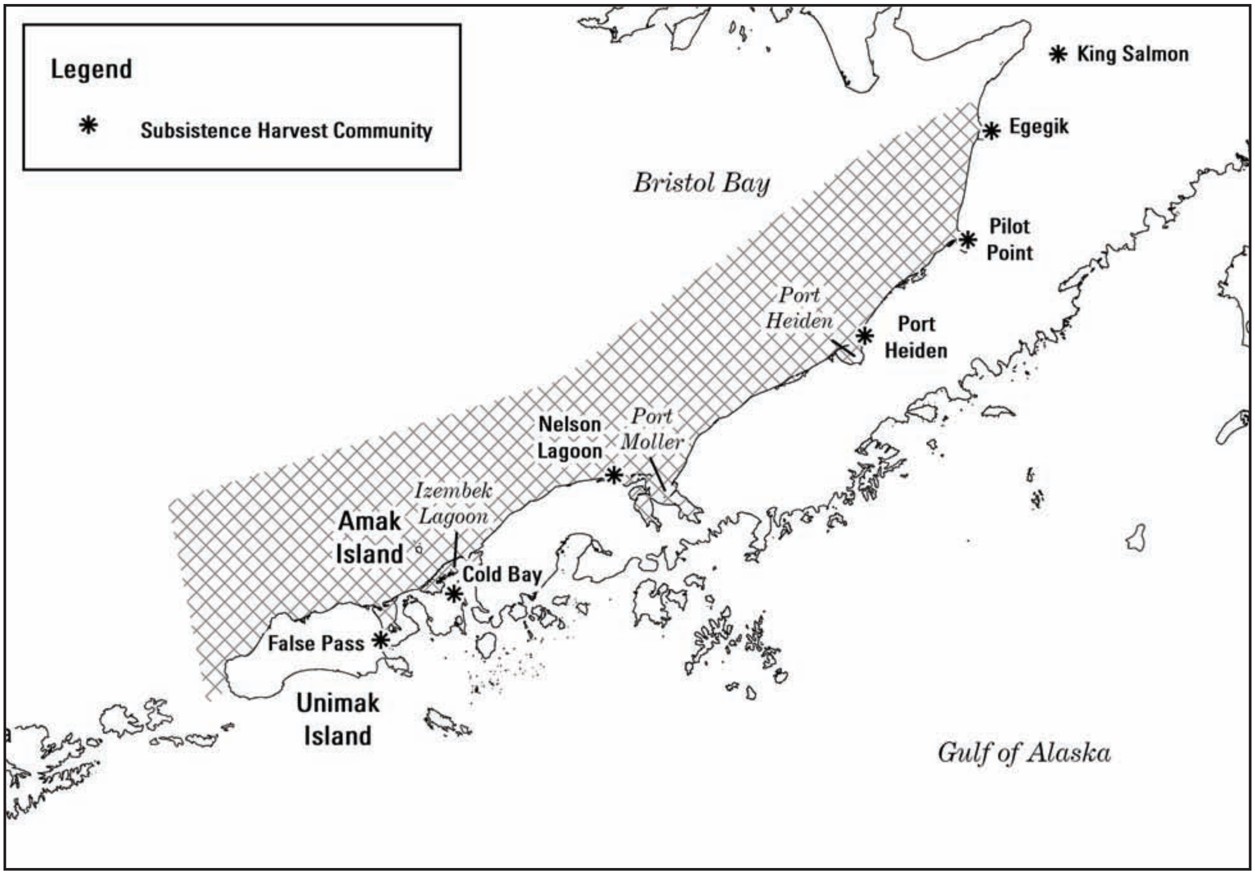

Figure 10. Bristol Bay Management Unit (BB).

In contrast to the Krenitzen Islands, data from these surveys do not indicate a pattern of continued population declines over the past several years (Jack et al. 2007).

Bristol Bay MU (Figure 10)

Within the Bristol Bay MU, one remnant colony in the area of False Pass was believed to have existed after commercial fur harvests ended in 1911 (Kenyon 1969). During surveys in the late 1950s and early 1960s, substantial numbers of sea otters were observed between Unimak Island and Amak Island, with a maximum of 2,892 counted in 1965 (Kenyon 1969).

Sea otter habitat within this MU is somewhat unusual due to the presence of a broad, shallow shelf that extends to the north from the Alaska Peninsula. In the 1970s and 1980s, it was not uncommon to observe large rafts of sea otters more than 25 km from shore (Schneider 1976, Brueggeman et al. 1988). For this reason, aerial or skiff surveys along the shoreline are not an appropriate survey method for this area. Consequently, the northern Alaska Peninsula offshore area has been surveyed from aircraft using parallel transects extending from the shoreline seaward over the shelf. Using this method, Schneider (1976) calculated an unadjusted estimate of 11,681 sea otters on the north side of the Alaska Peninsula in 1976.

Brueggeman et al. (1988) conducted surveys of the same area during three time periods in 1986. Burn and Doroff (2005) re-analyzed the original 1986 survey data to address computational errors in the original survey report; the re-calculated estimates range from 6,474 to 9,215 sea otters (uncorrected) for the three surveys (Burn and Doroff 2005). In May 2000, FWS replicated the survey design of Brueggeman et al. (1988) using identical survey methods. The uncorrected estimate of 4,728 from this survey indicated that abundance on the north side of the Alaska Peninsula had declined by 39% since 1986, and by 60% since 1976 (Burn and Doroff 2005). Data from the Schneider (1976) survey are not available in GIS format, so additional analyses of population declines were made by comparison of survey data from 1986 and 2000.

The population change observed between 1986 and 2000 was not uniform over the entire MU. Brueggeman et al. (1988) had divided their study area into two subareas, with “North Alaska Peninsula A” to the west of 162° W and “North Alaska Peninsula B” to the east. The population in North Alaska Peninsula A declined from an average estimated abundance of 6,303 in 1986 to 374 in 2000 (Burn and Doroff 2005). In contrast, the estimated sea otter abundance in North Alaska Peninsula B increased from an average of 1,703 in 1986 to 4,354 in 2000, which suggests that the population in the Bristol Bay MU underwent a change in distribution as well as abundance during the period between surveys. The largest aggregations of sea otters in May 2000 were observed in Port Moller and Herendeen Bay. This concentration had been described as a seasonal phenomenon, as surveys conducted later in the summer had not recorded similarly large numbers of sea otters (B. Murphy, personal communication). To test this assumption, FWS conducted sea otter surveys in the Port Moller, Herendeen Bay, and Nelson Lagoon areas in May and July 2004. A total of 1,472 otters were counted on May 5, and 936 on July 13 (FWS unpublished data). The surveys were conducted at different tidal states; much of the head of Port Moller was exposed mud flats during the July survey. Although it is not clear to what degree use of these areas changes seasonally, the Port Moller and Herendeen Bay areas continue to support the highest observed sea otter densities within the Bristol Bay MU. Although the decline in this MU is not as severe as those in the Western and Eastern Aleutian MUs, the most recent survey data suggest that current abundance is around 40% of the 1976 value (Figure 11). Schneider (1976) believed that his unadjusted population estimate for 1976 was within the carrying capacity of this area, which would suggest that the 1986 estimate represents a population below equilibrium density.

South Alaska Peninsula MU (Figure 12)

Two remnant colonies – one at Sandman Reefs and the other at the Shumagin Islands – were believed to have existed near the western end of the Alaska Peninsula after commercial fur harvests ended in 1911 (Kenyon 1969). Several island groups within this MU, including the Pavlof and Shumagin Islands, as well as Sanak, Caton, and Deer Islands, have been surveyed several times since the early 1960s. In 1962, Kenyon (1969) counted 1,900 otters along these islands. In 1986, Brueggeman et al. (1988) counted 2,122 in the same area. In 1989, DeGange et al. (1995) counted 1,589 along the shores of the islands that had been surveyed in 1962 and 1986, which was 16–28% fewer sea otters than reported in the earlier counts. This decrease was the first indication of a sea otter population decline in the area of the Alaska Peninsula. The 1989 surveys were conducted using rotary-wing aircraft, as opposed to fixed-wing for all other surveys in this area; it is unclear to what degree the difference in survey platform may have affected survey results. In 2001, only 405 otters were counted from the air in this area of the Alaska Peninsula (Burn and Doroff 2005), indicating an 81% decline from 1986. Additional aerial surveys at 13 of these islands were flown in 2004 and 2007 using methods similar to those used in 2001, and the counts indicated continued declines (FWS unpublished data).

In addition to sea otter counts at islands, similar data are available for the shoreline of the Alaska Peninsula within this MU. In 1989, DeGange et al. (1995) counted 866 sea otters along the southern shoreline of the Alaska Peninsula from False Pass to Castle Cape. In 2001, Burn and Doroff (2005) counted 536 in this same area. The width of the

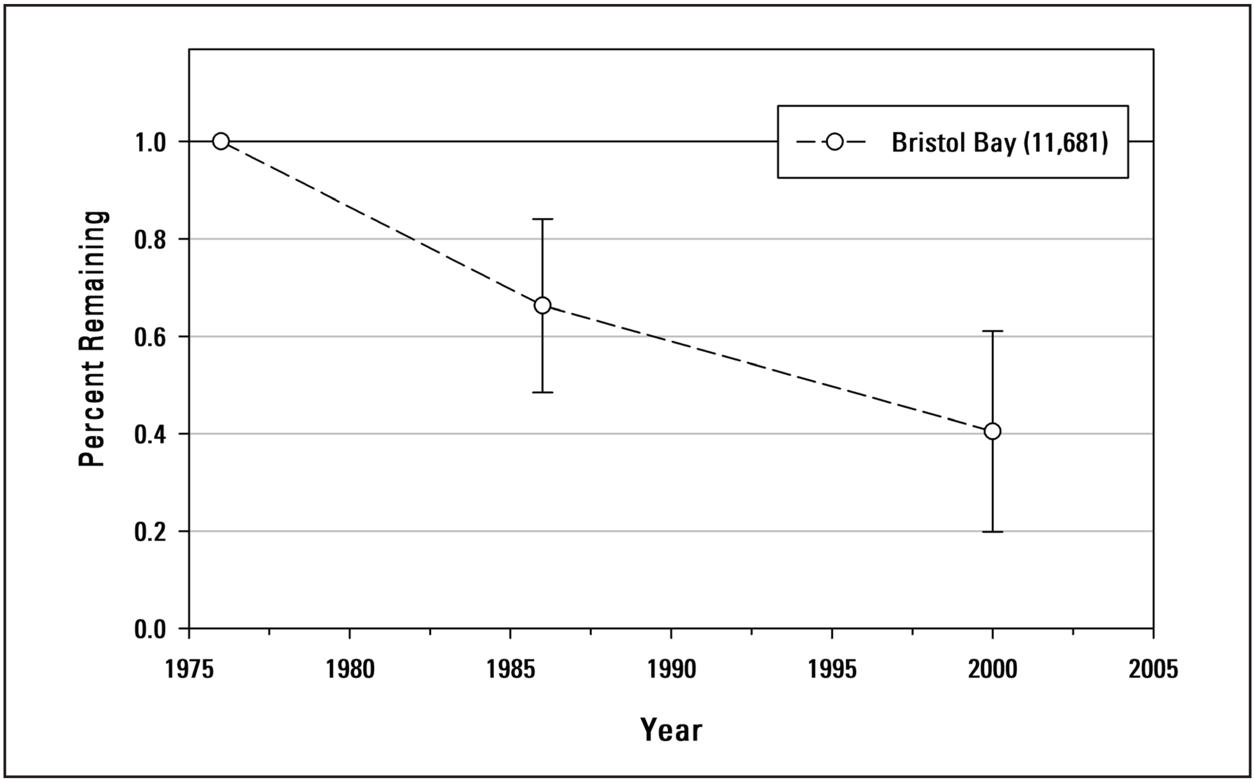

Figure 11. Sea otter survey results for the Bristol Bay MU.
Results expressed as the percentage of the population remaining relative to initial level (in parentheses). Error bars represent coefficients of variation.

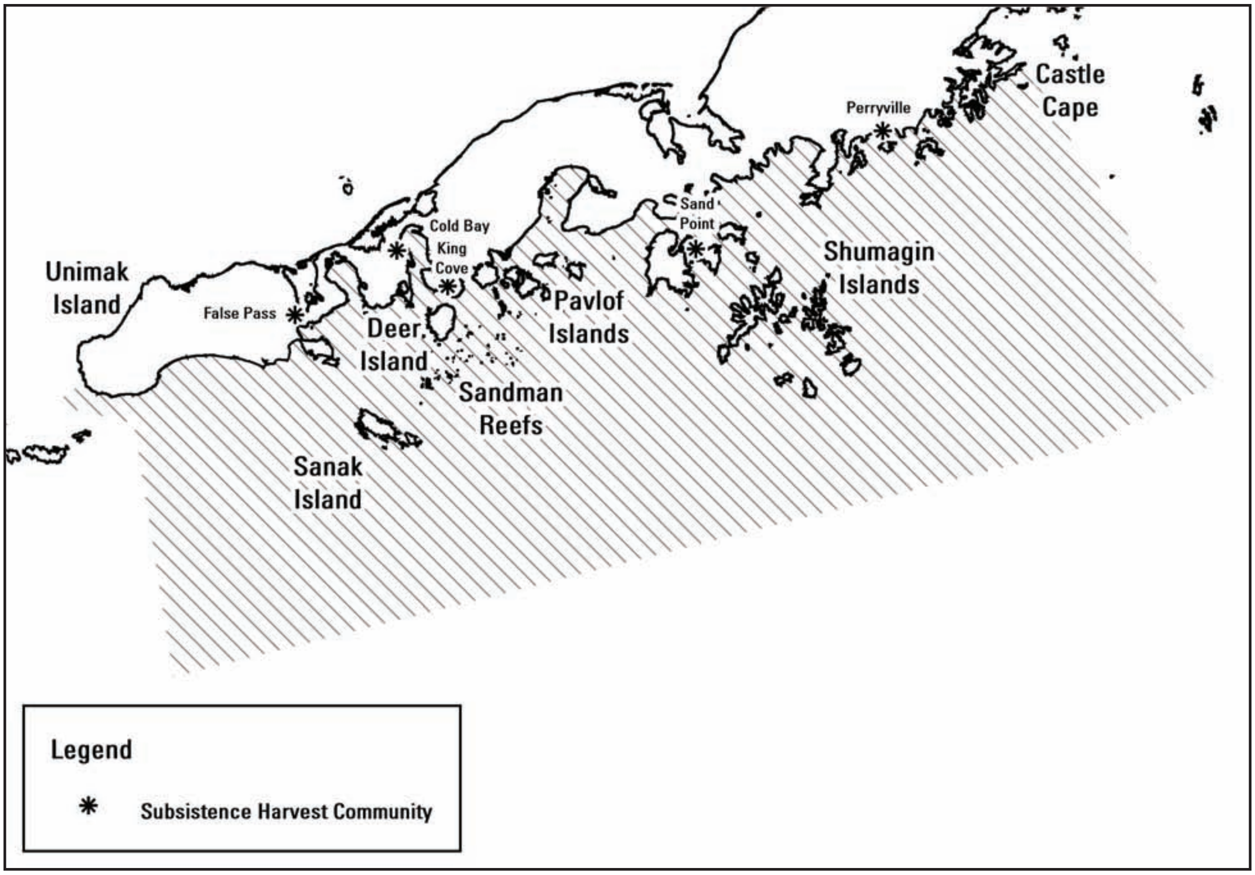

Figure 12. South Alaska Peninsula Management Unit (SAP).

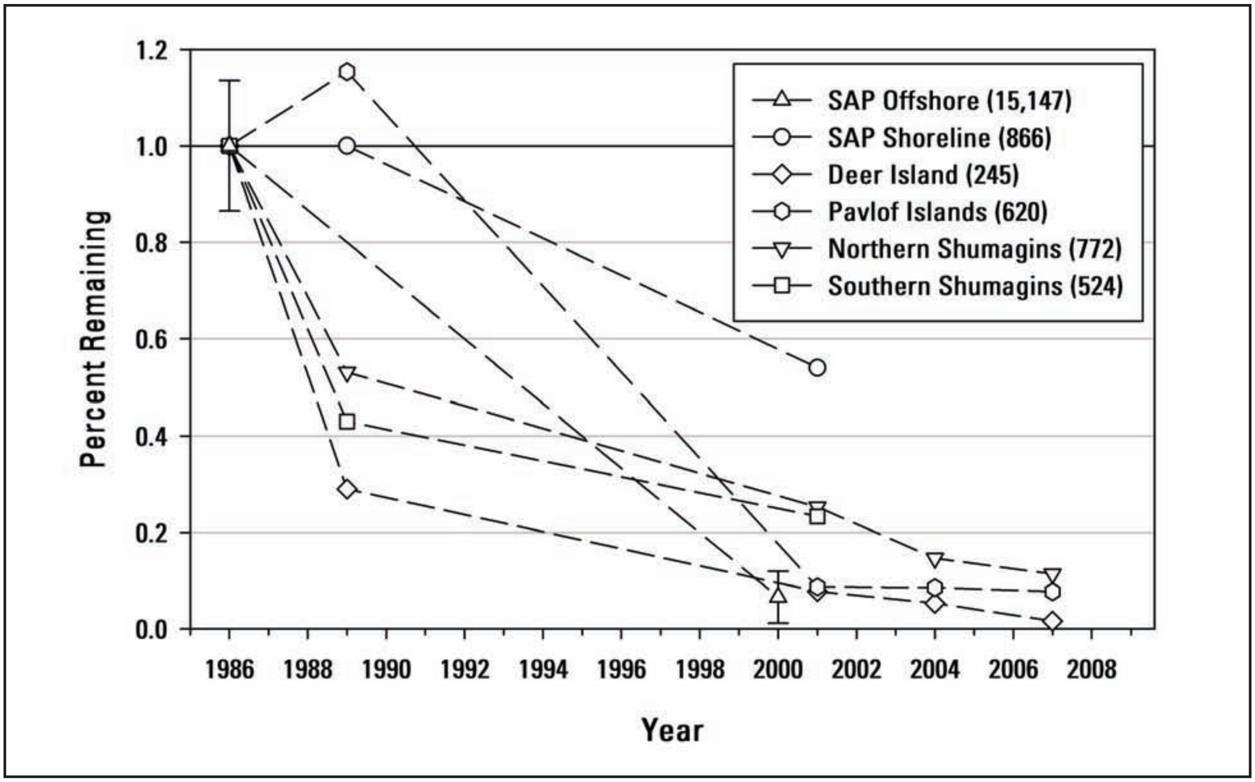

Figure 13. Sea otter survey results for the South Alaska Peninsula MU.
Results expressed as the percentage of the population remaining relative to baseline levels (in parentheses). Error bars represent coefficients of variation for southern Alaska Peninsula offshore estimates. All data collected by aerial survey.

survey zones differed, but a comparison based on sea otter density (otters/km^2) indicates a decline of 46% had occurred.

Similar to the Bristol Bay MU, offshore areas on the south side of the Alaska Peninsula were surveyed by air during three different time periods in 1986 (Brueggeman et al. 1988). Re-calculated abundance estimates for the three surveys were 13,900–17,500 (Burn and Doroff 2005). FWS replicated the surveys in April 2001 and estimated 1,005 otters for the southern Alaska Peninsula offshore area, indicating a decline of at least 93% when compared with the 1986 surveys. Specific areas of high sea otter concentrations in 1986, such as Sandman Reefs, were almost devoid of sea otters when surveyed in 2001 (Burn and Doroff 2005).

Sea otter population trends for the South Alaska Peninsula MU are presented in Figure 13. The data suggest that the population underwent a rapid decline in the late 1980s and early 1990s, and that the decline continued at several island groups at least through 2004. Current abundance is likely less than 20% of the baseline values observed during the latter half of the 1980s. It is unclear if these baseline values represent a population at equilibrium density.

Kodiak, Kamishak, Alaska Peninsula MU (Figure 14)

One of the remnant sea otter colonies in southwest Alaska occurred at the northern end of the Kodiak archipelago at Latax Rocks. In 1959, Kenyon (1969) counted 395 sea otters in the Shuyak Island area. Over the next 30 years, the sea otter population in the Kodiak archipelago grew at about 9% per year (Bodkin et al. 1999), and its range expanded southward around Afognak and Kodiak Islands (Schneider 1976, Simon-Jackson et al. 1986). DeGange et al. (1995) surveyed the Kodiak archipelago in 1989 and calculated a detection-adjusted population estimate of 13,526 sea otters. However, this estimate is now believed to be unreliable[2] (A. DeGange, personal communication) and has been excluded from analysis of population abundance and trends. In July and August 1994, FWS conducted an aerial survey of the Kodiak archipelago using the methods of Bodkin and Udevitz (1999) and calculated an adjusted population estimate of 9,817. In June 2001, FWS surveyed the Kodiak archipelago using the same observer, pilot, and methods as in 1994. That survey resulted in an adjusted population estimate of 5,893 (FWS unpublished data), which represents a 40% decline from the 1994 estimate. In summer 2004, FWS surveyed the Kodiak archipelago

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[2] The specific concerns regarding the 1989 survey include the stratification of shoreline and offshore zones, as well as the methods used to correct for otters not detected by observers.

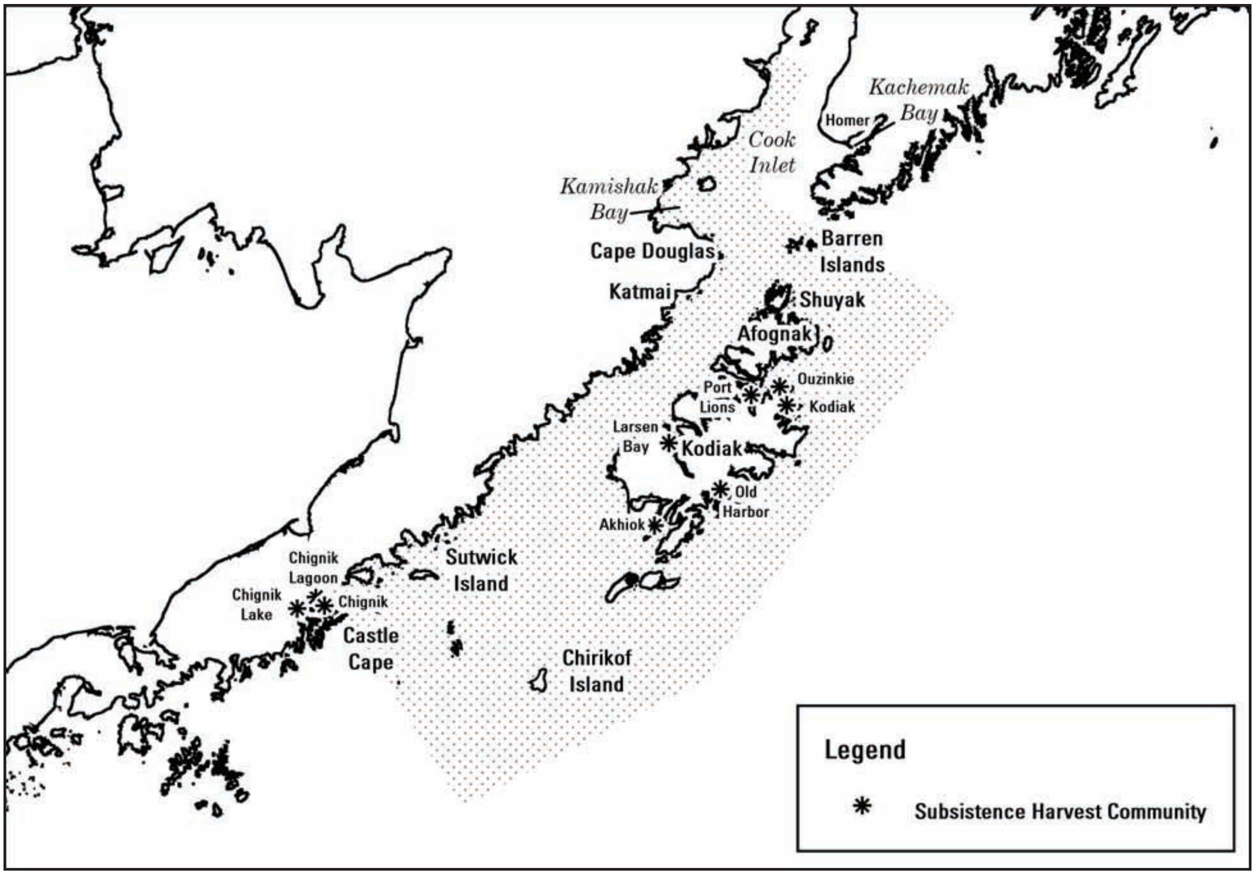

Figure 14. Kodiak, Kamishak, Alaska Peninsula Management Unit (KKAP).

using the same methods as in 1994 and 2001 and estimated the population size at 11,005 sea otters (FWS unpublished data). Although these results do not suggest that the sea otter population in this area has experienced population declines similar to other MUs within the DPS, it remains unclear why the population has not continued to expand its range to occupy more of the available habitat at the southern end of Kodiak Island.

The shoreline of the Alaska Peninsula within this MU (including Sutwick Island) was surveyed in 1989 by DeGange et al. (1995), who counted 1,766 otters between Castle Cape and Cape Douglas. This area was surveyed again in 2001, when Burn and Doroff (2005) counted 2,115. The width of the survey zones differed, but a comparison based on sea otter density (otters/km^2) indicates that the population in this area increased slightly (5%) over this time period. In June 2008, the USGS Biological Resources Division conducted an aerial survey of all presumed sea otter habitat southwest from Cape Douglas to the western extent of Katmai National Park (Katmai Bay). The detection-corrected population estimate was 7,095 (se = 922) with an estimated density of 4.9/ km^2. The area surveyed in 2008 was a small proportion of the area previously surveyed by DeGange et al. (1995) and Burn and Doroff (2005), and the results suggest rapid growth in sea otter abundance at least along the mainland in the eastern part of this MU.

In the summer of 2002, the USGS conducted an aerial survey of lower Cook Inlet and the Kenai Fiords area (Bodkin et al. 2003). The survey was designed, in part, to estimate sea otter abundance in Kamishak Bay, which is located on the west side of lower Cook Inlet. The methods used were identical to the aerial surveys of the Kodiak archipelago described above. Sea otters were abundant in Kamishak Bay during the 2002 survey, with an adjusted estimate of 6,918 (Bodkin et al. 2003). As no previous estimates for Kamishak Bay exist, the population trend for this area is unknown.

Although the survey data for this MU are somewhat fragmentary, there appears to be little evidence that it has experienced population declines similar to the other units (Figure 15).

Overview of population abundance and trends

Due to differences in the years of the various surveys for different areas, it is difficult to combine this information to estimate the overall size of the southwest Alaska sea otter population at the onset of the decline. Calkins and Schneider (1985) provided estimates from survey data collected as of 1976, adjusted for animals not detected by observers, for the Aleutian Islands (55,100–73,700), northern Alaska Peninsula (11,700–17,200), southern Alaska Peninsula (22,000–30,000), and Kodiak archipelago (4,000–6,000). They did not report a specific estimate for the Kamishak Bay area, which presumably was included within their

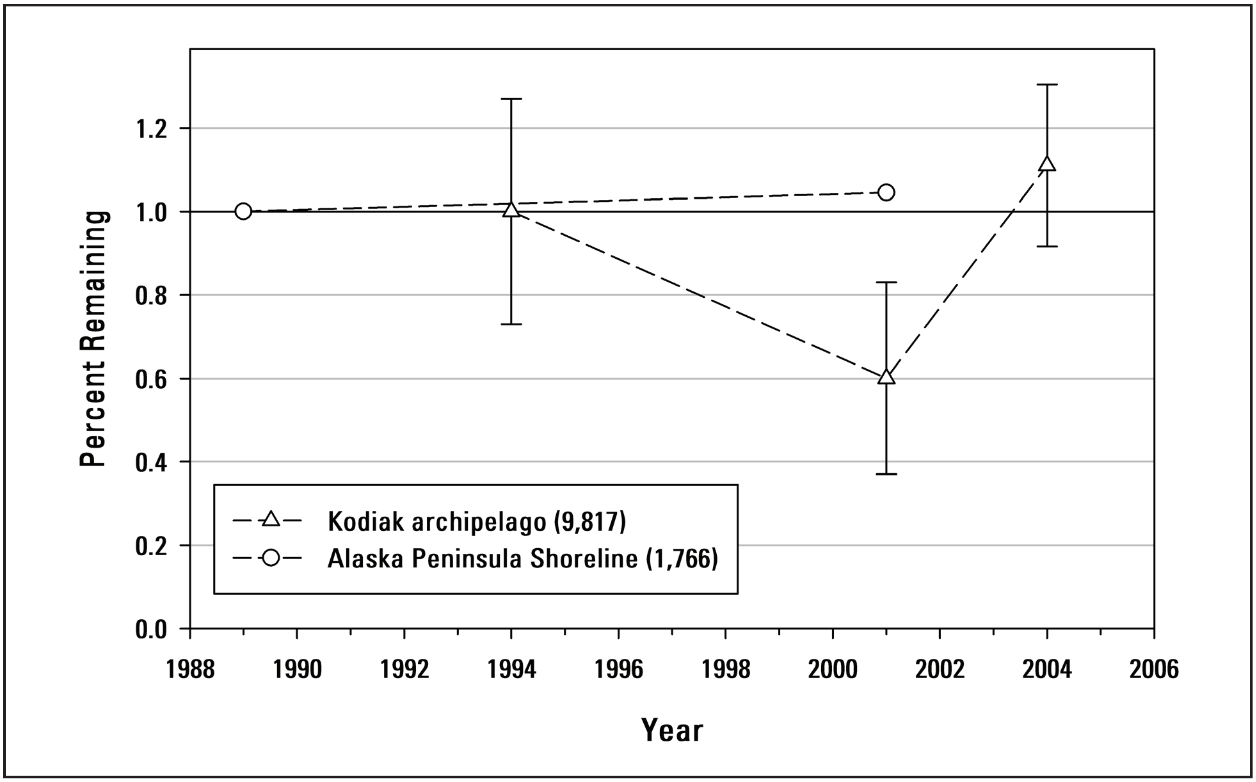

Figure 15. Sea otter survey results for the Kodiak, Kamishak, Alaska Peninsula MU. Results expressed as the percentage of the population remaining relative to baseline levels (in parentheses). Error bars represent coefficients of variation for Kodiak archipelago estimates. All data collected by aerial survey.

estimate for the Kenai Peninsula and Cook Inlet area (2,500–3,500 otters). Assuming that half of the sea otters estimated for Kenai Peninsula and Cook Inlet occurred in Kamishak Bay and combining these estimates, the sea otter population in the area encompassing the range of the southwest Alaska population would have numbered between 94,050 and 128,650 animals as of 1976. As sea otters had not yet fully recolonized southwest Alaska or reached equilibrium density in all areas in 1976, additional population growth was expected. Therefore, the overall population prior to the onset of the decline in the 1980s may have been higher than the population estimate for 1976.

The most recent estimate of the size of the southwest Alaska sea otter population, based on surveys in 2000-2008 and adjusted for animals not detected, is 53,674 (Table 1). As more recent site-specific surveys indicate the decline may have continued in the Western Aleutian, Eastern Aleutian, and South Alaska Peninsula MUs, it is possible that the population size at the time this plan was developed (2008-2010) was actually lower.

The 1976 population estimate, based on the work of Calkins and Schneider (1985), may not be directly comparable to more recent estimates because of different survey approaches and estimation techniques. Nevertheless, the results provide the only basis for comparison of the overall extent of the decline of sea otters in southwest Alaska. Comparing the 1976 estimate to the estimate in Table 1 indicates an overall population decline of 43 to 58%. Using the most comparable survey information as a baseline for each MU, the most severe declines appear to have occurred in the Western Aleutian and South Alaska Peninsula MUs. As the population decline appears to have been underway at the time of the 1992 aerial survey of the Aleutian archipelago, declines in the Western and Eastern Aleutian MUs are likely to have been even greater than estimated here.

Table 1. Recent sea otter population estimates for MUs within the southwest Alaska DPS of the northern sea otter.

Results from the Bristol Bay, South Alaska Peninsula, and portions of the Kodiak, Kamishak, Alaska Peninsula MUs are adjusted using a correction factor of 2.38 following Evans et al. (1997). Counts from the Western Aleutian, Eastern Aleutian, and portions of the Kodiak, Kamishak, Alaska Peninsula MUs have been adjusted using survey-specific correction factors.

Management Unit	Time Period	Most Recent Adjusted Count or Estimate	Population Change	References
Western Aleutian	1992 - 2000	6,451	-73%	Doroff et al. (2003) Estes et al. (2005)
Eastern Aleutian	1992 - 2000	2,291	-56%	Doroff et al. (2003)
Bristol Bay	1986 - 2000	11,253	-39%	Burn and Doroff (2005)
South Alaska Peninsula	1986-2001	4,724	-74%	Burn and Doroff (2005)
Kodiak, Kamishak, Alaska Peninsula	1994 - 2008	28,955	Stable or growing	Burn and Doroff (2005) Bodkin et al. (2003) USGS (unpublished data)
Total		53,674	-43 to 58%	

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3. Threats and Impediments to Recovery

A. Predation

Sea otters are typically viewed as apex predators in near-shore marine communities, yet there have been many reported instances in which sea otters have been killed and/or consumed by other top predators. The predators involved include bald eagles, white sharks , terrestrial carnivores such as brown bears, coyotes (*Canis latrans*) and arctic foxes (*Alopex lagopus*), and killer whales (*Orcinus orca*). Documented instances of predator-caused mortality are summarized below.

Bald eagles are fairly frequent predators of sea otter pups in the Aleutian Islands, with otter pup remains accounting for up to 20% of the prey items found in some eagle nests (Sherrod et al. 1975). However, because this predation is limited to very young pups (< 5 kg), a stage with naturally high mortality, it has generally been considered that eagle predation has a small demographic impact, at least at high sea otter densities (Riedman and Estes 1990, Sherrod et al. 1975). It is possible that eagle predation would pose a more significant threat at very low population densities, such as those found throughout much of the area affected by the decline. However, recent searches of bald eagle nests indicate that eagle diets in the Aleutian Islands have shifted to focus on offshore fish and seabirds, possibly as a result of decreases in nearshore fish populations associated with the loss of kelp (Reisewitz et al. 2006), and sea otters now represent a very small proportion of eagle diets (Anthony et al. 2008). Thus, eagle predation seems unlikely to pose a serious threat to sea otter recovery in southwest Alaska, although there is a potential for impacts at very low population sizes.

White shark attacks represent a significant source of mortality for sea otters in California, particularly at the northern end of that subspecies' range (Ames and Morejohn 1980, Estes et al. 2003b, Kreuder et al. 2003). Based on analysis of the beachcast (deposited on shore) carcasses having evidence of shark bites, it appears that these carcasses do not represent true predation - that is, sharks are not actually consuming the sea otters but rather they are killed incidentally during shark predation on pinnipeds (Ames and Morejohn 1980). This interpretation is consistent with the lack of sea otter remains in white shark stomachs (Riedman and Estes 1990). Regardless of whether or not fatal shark bites represent true predation or just cases of "mistaken identity," sharks have a relatively high potential for limiting population growth in parts of the California sea otter's range (Gerber et al. 2004).

Information on shark predation on sea otters in Alaska is limited to one unconfirmed incident involving a salmon shark (*Lamna ditropis*) that reportedly attacked and consumed a sea otter in Prince William Sound (D. Burn, personal communication). In spite of their apparent rarity, shark attacks have been considered as a potential source of sea otter mortality in this plan.

Martin (2004) found only two occurrences of white sharks in Alaskan waters west of PWS in 40 years of records (1961-2001). Recent satellite telemetry studies showed that none of the 68 white sharks tagged in the coastal concentration area off central California ranged northward far enough to overlap with the southwest Alaska sea otter DPS (Jorgensen et al. 2009). White shark predation does not appear to be a concern for this DPS.

Pacific sleeper sharks (*Somniosus pacificus*), occur in Alaskan waters, and in the 1990s their abundance increased substantially in the Gulf of Alaska and Bering Sea (Gaichas 2002). Sleeper sharks are known to feed on marine mammals other than sea otters (Hulbert et al. 2006, Klimley 1985). Formerly thought to be scavengers, recent evidence shows that they also prey on salmon and other fast-moving species and sometimes make diurnal vertical migrations to waters of 100 m or less in depth (Hulbert et al. 2006), bringing them into the maximum forage depth range of sea otters. Near Unimak Island, sleeper sharks routinely scavenge the carcasses of gray whales killed by killer whales in depths of 50 m or less (L. Barrett-Lennard, personal comunication). During a recent survey, the stomachs of 25 out of 198 sleeper sharks caught in the northern Gulf of Alaska contained marine mammal remains, none of which could be positively identified as from sea otters or determined to have been acquired by predation (Sigler et al. 2006). However, several observations indicate that sleeper sharks do attack live marine mammals. Specifically, Crovetto et al. (1992) provided evidence that a Pacific sleeper shark consumed a live northern right whale dolphin (*Lissodelphis peronii*), van den Hoff and Morrice (2008) reported a southern elephant seal (*Mirounga leonina*) with a bite wound from a closely related sleeper shark species (*Somniosus antarcticus*), and Lucas and Stobo (2000) documented substantial shark predation on harbor seals, some of which was likely by Greenland sharks (*Somniosus microcephalus*; see Taggart et al. 2005).

Abundance of salmon sharks also has increased in Alaska since the 1990s, with schools of hundreds being reported in nearshore areas (Gaichas 2002, Okey et al. 2007). In 2003, fishermen reported increasing densities of sleeper and/or salmon sharks at five locations in the Aleutian Islands (Okey et al. 2007). Salmon sharks are known to feed on salmon and cephalopods (Hulbert et al. 2005), and the only indication that they may prey on sea otters is the unconfirmed report described above. It is also worth noting that in PWS where there have been dramatic increases in salmon shark abundance, sea otter numbers have increased or remained stable (Bodkin et al. 2002).

Terrestrial predators can prey on sea otters in areas where they haul out, although reports of such predation are remarkably few. In the Commander Islands, arctic foxes frequently scavenge sea otter carcasses and also occasionally kill live otters, although such predation is largely limited to young or already moribund (in a dying state) animals (Zagrebel'nyi 2000). In PWS, newly weaned sea otters are occasionally killed and eaten by coyotes (Siniff and Ralls 1988), and brown bears have been reported to be a significant source of predation on sea otters along the Kamchatka Peninsula in the late winter and early spring (Riedman and Estes 1990). There is no potential for predation by terrestrial predators in the Aleutian archipelago because there are no terrestrial carnivores in this region. However, along the Alaska Peninsula and in the Kodiak archipelago, predation by bears, coyotes, and foxes is possible, although there are no published reports of such predation.

Killer whales are recognized as predators of sea otters, yet prior to 1990 there were only a handful of documented interactions between the two species. One aggressive interaction was reported in the Kuril Islands (Nikolaev 1965), and there were a few inconclusive reports of presumed attacks in the Commander Islands (summarized in Hatfield et al. 1998). However, most killer whale-sea otter interactions in Alaska and elsewhere were assumed to be non-aggressive (Kenyon 1969). In the early 1990s, this pattern appeared to change, based on nine documented observations of killer whales apparently attacking sea otters: three of these incidents occurred in PWS and six in the Aleutian Islands (Hatfield et al. 1998). It was speculated that these interactions reflected an increase in killer whale predation on sea otters and that this increase was the result of some "transient" (mammal-eating) killer whales adjusting their diet to include sea otters, presumably as a response to declines in other marine mammal prey, including harbor seals and Steller sea lions (Estes et al. 1998, Hatfield et al. 1998). Since 1995 there have also been multiple anecdotal reports from USGS field staff (M. Tinker, personal communication) and FWS field personnel (J. Williams, personal communication) of apparent killer whale attacks on groups of sea otters; however, because these are unpublished reports, in most cases the details are difficult to confirm. Kuker and Barrett-Lennard (2010) noted that of the six interactions in the Aleutians reported by Hatfield et al. (1998) one was positively identified as a predatory attack that resulted in a confirmed kill of an otter, while the other five could be described as probable kills or harassment. In two of these cases the otters were seen after the incident and no signs of injury were reported, in one case the otter escaped but was apparently injured, and in two cases the otters were not seen after the attack and their fate is uncertain, although the observer believed that they were likely killed and/or consumed. In the most conservative sense, observations of killer whale attacks are classified as predation only when strong evidence of actual consumption is found, such as blood, blubber, or other tissues from the prey floating on the water surface after the attack (Ford et al. 1998). Although strong evidence for actual consumption was found in only one out of the nine observed killer whale attacks on sea otters (Hatfield et al. 1998), the fact that the skeletal remains of at least five sea otters were recovered from the stomach of a male killer whale carcass in PWS confirms that otters are at least sometimes eaten by killer whales (Vos et al. 2006).

Evidence for impacts of killer whale predation on sea otters in southwest Alaska

The cumulative support for predation as a primary cause of the decline actually consists of a number of independent pieces of information, representing both direct and indirect (inferential) evidence, and can be summarized as follows:

1. There was an increase in the rate of observed killer whale attacks around the time the decline began (Estes et al. 1998, Hatfield et al. 1998). For the area between Kiska and Seguam Islands in the Western Aleutian MU, Estes et al. (1998) estimated a loss of 6,788 otters per year between 1991 and 1997. Based on the level of observer effort over that time period, and assuming that any attack within 500 m of an observer during daylight hours would be observed, the expected number of observed attacks was 5.05, consistent with the small number of attacks reported by Hatfield et al. (1998). Thus, it is plausible that the level of killer whale predation needed to produce the observed sea otter decline could have occurred with relatively few of the attacks being observed.

2. Energetic and demographic feasibility analyses indicate that predation by a small number of killer whales (specifically, a pod of five individuals) would have been capable of causing the sea otter population decline in southwest Alaska (Williams et al. 2004).

Similar analyses show the energetic and demographic feasibility that killer whale predation has been a key factor in other marine mammal declines, and it has been hypothesized that a sequential cascade of that was ultimately caused by the reduction of "great whale" populations in the North Pacific by industrial whaling (Springer et al. 2003, Williams et al. 2004). This hypothesis has been the subject of considerable subsequent debate (DeMaster et al. 2006, Wade et al. 2007, Trites et al. 2007, Springer et al. 2008, Wade et al. 2009, Estes et al. 2009).

3. Few beachcast sea otter carcasses were observed during the decline, which is inconsistent with what would be expected if animals were dying from disease, starvation, or other typical causes of sea otter mortality (aside from predation). Systematic searches of beaches at Adak and Amchitka Islands throughout the decline resulted in very few recovered carcasses or skeletal remains. At Amchitka alone, more than 4,000 animals disappeared between 1994 and 1997 (Doroff et al. 2003), yet fewer than 20 skeletal remains (and no fresh carcasses) were found during systematic searches of all beaches on the southern half of the island in the winter of 1993/94 and again in the early spring of 1997 (USGS unpublished data). In contrast, searches of these same beaches in the 1950s during a much smaller mortality event attributed to nutritional stress and disease (Rausch 1953) resulted in the retrieval of hundreds of carcasses and moribund otters (Kenyon 1969). The strength of this line of evidence is reduced, at least somewhat, by the fact that no carcasses or injured otters bearing wounds of either killer whales or sharks were found. In southern Alaska and British Columbia, the inverted skins of seal pups are sometimes cast on beaches or found at low tide when transient killer whales have passed through an area (Barrett-Lennard unpublished data), and (as discussed above), otter carcasses bearing signs of attacks by white sharks are found commonly in California.

4. The rate of premature disappearance of animals that were radio-tagged as part of telemetry-based studies at Amchitka Island (1992-94) and Adak Island (1995-96; Estes et al. 1998, Monson and Degange 1995, Tinker and Estes 1996) greatly exceeded that recorded during radio-telemetry studies in California that used identical techniques and instrumentation (e.g., Siniff and Ralls 1991). Premature disappearance is defined as the unexplained loss of study animals with functioning VHF radio tags, occurring within the period of expected battery life (typically two years), and without confirmation of mortality, carcass recovery, or later resighting of flipper-tagged animals with non-working radios. In contrast to the above observation, the premature disappearance rate of radio-tagged animals in Clam Lagoon at Adak Island (an enclosed embayment with limited ocean access that may serve as a refuge from predators) was significantly lower (Estes et al. 1998) and consistent with the disappearance rate seen in previous studies (e.g., Siniff and Ralls 1991).

5. Between 1995 and 1999, the population inside Clam Lagoon declined only slightly, while numbers dropped by approximately 80% outside the lagoon in Kuluk Bay (Estes et al. 1998). It should be noted that pre-decline sea otter densities (i.e., in the early 1990s) were much higher inside Clam Lagoon, so this discrepancy in rate of decline cannot be explained as the result of a random movement process or of density-dependent immigration into Clam Lagoon.

6. The distribution pattern of sea otters throughout the region of southwest Alaska affected by the decline has shifted from that prior to the decline such that most otters are now closer to shore and tend to be concentrated around protected embayments or areas with extensive shallow reefs (USGS and FWS unpublished data). This likely affords some measure of protection from killer whales.

7. There are unpublished reports in recent years (i.e., post-decline) of behavioral responses of sea otters to disturbance that are both atypical and suggestive of avoidance of marine predators. These include animals feeding and resting in very shallow (<2 m) water, and hauling out on land in response to both shore-based and water-based disturbances (USGS unpublished data).

It should be noted that points 4-7 above do not represent direct evidence of predation, but are indirect evidence insofar as they imply a causal factor consistent with a rapid disappearance of animals of all age classes over a large region, producing almost no beachcast carcasses, and affecting all areas except those with limited accessibility to large aquatic predators.

The available evidence that is inconsistent with predation as a possible cause of the decline is considerably more limited. The lack of published records of sea otter consumption by killer whales in southwest Alaska (i.e., as measured by analyses of stomach contents or blubber fatty acid composition) does not constitute evidence against the hypothesis given the paucity of information about killer whale diets throughout most of this region and the low likelihood that an otter-eating killer whale would have been sampled. One observation that is puzzling in the context of the predation hypothesis is the almost complete lack of carcasses. On one hand, it might be expected that a killer whale would consume most or all of a sea otter and thus leave little in the way of skeletal remains or a floating carcass. If that were the case, predation by killer whales would be consistent with the lack of beachcast carcasses. On the other hand, it may be reasonable to expect that, at times, a sea otter would be only partially eaten by a killer whale, or it would escape an attack with a serious bite and then later die on the beach. If that were the case, we would expect to find at least a few partial carcasses, similar to the pattern with white sharks in California. The lack of even one recovered carcass with evidence of killer whale bites might thus be considered as evidence against the predation hypothesis.

Whether or not the recent sea otter decline was caused by killer whale predation, mortality from predation clearly represents a potentially limiting factor to population recovery. Should killer whale predation on sea otters be expected to continue? On purely theoretical grounds, differing predictions can be made about future impacts depending on the nature of killer whale foraging dynamics. If killer whale predation on sea otters follows a type-II functional response (i.e., predator search rate remains constant irrespective of prey density), then the per-capita mortality rate for sea otters will increase as their density decreases, and the impacts of killer whale predation can be expected to be even more significant at low population densities. Alternatively, if killer whale predation on sea otters follows a type-III functional response (i.e., killer whales are likely to switch to alternative prey at low otter densities), then the per-capita mortality rate for sea otters should decrease at low densities, and we may thus expect a numerical "floor" below which killer whale predation becomes unimportant. These two scenarios represent highly simplified behavioral models, and in reality we should expect a more complex functional response that reflects spatial variation in killer whale habitat use and sea otter abundance, changes in the relative abundance of alternative killer whale prey, and and other factors affecting killer whale foraging decisions.

At the present time, data are insufficient to determine whether the per-capita rate of mortality from killer whale attacks has increased or decreased as a function of decreasing sea otter abundance. The lack of recent data reflects both a decrease in sea otter field research activity since the mid 1990s and a statistical artifact, i.e., even if the per-capita attack rate remained unchanged, the total number of attacks that could be observed would have deceased ten-fold due to decreasing otter abundance. Likewise, there are insufficient data on both killer whale population structure and foraging behavior in southwest Alaska to make informed predictions about functional or numerical responses. As more information becomes available, it will be possible to formulate and test specific hypotheses. For example, if only a small number of transient killer whales have recently added sea otters to their diets, it may be hypothesized that sea otters will be dropped as a diet item if a) more preferred prey species (e.g., Steller sea lions, harbor seals, dolphins, or large whales) increase in abundance, or b) the behavioral specialization for hunting sea otters is lost due to deaths of those few individuals currently using sea otters as prey. However, before any realistic predictions can be made about the likely future impacts of killer whale predation on sea otter recovery, there is a need for both theoretical and empirical work to better understand predator-prey relations between killer whales and sea otters in southwest Alaska. Predictive models should incorporate all the complexities mentioned above and account for the changing environmental conditions and food-web dynamics in the wider North Pacific ecosystem (e.g., Chavez et al. 2003, National Academy of Sciences 2003, Springer et al. 2003).

Conclusions

Predation has not been widely recognized as a limiting factor for sea otter populations in the past, but there are a few specific instances, such as white sharks in central California or killer whales in the Aleutian Islands, in which predation is now suspected to be a significant source of mortality with important population-level effects. In southwest Alaska, a number of predators could impede the recovery of depleted sea otter populations in certain areas, including bald eagles, sharks, terrestrial carnivores, and killer whales. Predation by killer whales has been hypothesized to be the primary driver of the recent decline, and it may pose a serious threat to timely recovery of sea otters. A recent review paper (Kuker and Barrett-Lennard 2010) postulates inconsistencies in the strength of evidence supporting the killer whale predation hypothesis, and emphasizes that there remain uncertainties about the cause of the sea otter decline in southwest Alaska. Such arguments should be evaluated carefully and critically, and it should be noted that all the alternative hypotheses suggested by Kuker and Barrett-Lennard (2010) are addressed in considerable detail in the following sections of this plan.

There are many unanswered questions that must be addressed before reliable predictions can be made about future threats from predation. In particular, more information is needed to answer the following questions:

1. To what extent and how would it be possible to directly test the killer whale predation hypothesis? Is it feasible to deploy "mortality tags" on sea otters that could be used to measure rates and causes of mortality remotely? Could analysis of fatty acid or stable isotope signatures of killer whales be used to measure the intensity of predation on sea otters, or is this method too likely to

result in false negatives (i.e., if only a few killer whales are actually consuming sea otters, they might not be sampled)?

2. What is an acceptable "weight of evidence" in support of the predation hypothesis, given that many of its specific predictions (and of other hypothesized causes of decline) are not amenable to direct testing, given the remoteness and enormous geographic range of the affected populations?

3. What was the specific sequence of events that caused some killer whales to begin to prey on sea otters?

4. How many killer whales are actually feeding on sea otters?

5. Can a functional response be predicted given what is known about killer whale foraging behavior and energetic requirements? If not, what empirical and theoretical work would be needed to do so?

6. How can shark predation (e.g. sleeper or salmon sharks) be evaluated as a factor in the sea otter decline?

In the preceding review of the available evidence for and against predation as an important driver of sea otter population changes in southwest Alaska, and when evaluating the implications of increased predation (both past and future), it will be important to consider all information in the context of food-web interactions. Any attempt to interpret or predict predation impacts on sea otters without considering the broader food web is almost certainly bound to fail. On the other hand, by widening our perspective to encompass the southwest Alaska marine ecosystem (including both natural and anthropogenic perturbations), we can better judge the relative likelihood of a predator-caused decline. If recent population trends have been driven, in whole or in part, by an increase in predation on sea otters, then the best hope for understanding the ultimate cause of these changes, and for predicting how predation is likely to affect sea otter recovery, is to consider the structure and dynamics of the trophic interaction web of which sea otters are a part.

B. Infectious diseases

Disease is a perturbation from the "normal" condition in an animal and can be due to a variety of factors including pathogens, nutritional imbalances or deficiencies, toxins, degenerative changes, and neoplasia (tumors). Infectious diseases occur when a pathogen invades the host and alters normal cell or tissue function. However, host-pathogen-environment interactions that either increase the host's susceptibility, the pathogen's virulence, or its transmission rate may result in a change in the incidence of infectious disease. Epidemiological theory predicts that infectious diseases can have major population-level effects under specific circumstances as follows: 1) when pre-epidemic population sizes are small; 2) if a pathogen is able to use either an abiotic or biotic reservoir; or 3) when a pathogen is evolutionarily novel to a susceptible host (Smith et al. 2006). In many ways, disease agents can be thought of as similar to predators—they are part of the natural system but can cause negative population effects when the natural balance has been disturbed.

The importance of disease at the population level can be difficult to ascertain (Gulland and Hall 2006); documented instances of infectious disease being a major factor in species endangerment or extinction are rare but do exist (Smith et al. 2006). A few examples include: local population extinction of African wild dogs (*Lycaon pictus*) due to rabies from domestic dogs; mortality in several African antelope species due to rinderpest from domestic cattle; mortality of African lions (*Panthera leo*) due to canine distemper virus (CDV) most likely acquired from domestic dogs; and epidemics of canine distemper in Baikal seals (*Phoca sibrica*) and Caspian seals (*Pusa caspica*) transmitted to seals from domestic dogs (Gog et al. 2002, Kennedy et al. 2000, Pollack 2001). Theoretical and historical evidence suggest that infectious disease can temporarily or permanently drive populations to low densities, thus predisposing them to extinction by other forces (DeCastro and Bolker 2005, Gerber et al. 2005). Diseases can also impact populations by effects more subtle than death, such as slower growth, lower fecundity, altered behavior, and lower social status (Lafferty et al. 2005).

The main sources of information on disease in wild sea otters are examinations of carcasses (necropsies) of animals found dead on shore, in fishing gear, or floating (all referred to below as "stranded"), and sampling of living animals in free-ranging populations. There are two main concerns with regard to data from stranded carcasses: 1) the extent that stranded carcasses are representative of overall deaths in the population; and 2) when disease is found, deciding whether it was the primary cause of mortality or the resultant effect of some other stressor. Many samples from live animals are tested using serological screening, in which the presence of antibodies indicates exposure to a disease agent but not necessarily current infection. It is important when evaluating disease impacts to combine data from dead stranded animals with data from live-caught animals and information on population trends.

Infectious disease in sea otters range-wide (Table 2)

<u>California:</u> Causes of mortality have been more thoroughly studied in sea otters in California than in other regions due to extensive efforts to recover carcasses by the public, stranding networks along a heavily populated coastline, and the presence of facilities that specialize in marine mammal stranding response. Some

major causes of death in southern sea otters include protozoal meningoencephalitis (due to *Toxoplasma gondii* and *Sarcocystis neurona*), myocarditis/cardiomyopathy syndrome, and peritonitis due to *Profilicollis* spp. acanthocephalan infestations (Kreuder et al. 2003). Scattered cases of *Coccidioides immitis* have also been reported (Ames 1983). Diseases caused by parasites, bacteria, or fungi, and diseases without a specified cause, were the primary cause of death in 64% of otters examined, and parasitic diseases alone caused death in 38% of otters examined (Kreuder et al. 2003). These causes were observed predominantly in juvenile and prime-aged adult otters (Thomas and Cole 1996, Estes et al. 2003b), which could have negative implications for the overall health and recovery of this population (Kreuder et al. 2003, Gerber et al. 2004).

As mentioned above, of particular concern are infections with pathogens for which sea otters are not typical hosts and that have reservoirs in other species or in the environment because these are the types of situations where population-level effects could occur. Toxoplasmosis is presumed to be an example and it is a relatively common infection in California otters. Seroprevalence analysis in one study showed that 52% of 305 freshly dead sea otters and 38% of 257 live sea otters sampled from 1998 to 2004 were infected with *T. gondii* (Conrad et al. 2005). These are similar prevalences to what is observed in domestic animals and humans, but most of those seropositive cases do not develop disease. To date, felids (both wild and domestic) are the only known definitive hosts for *T. gondii*, with sea otters apparently being highly susceptible to systemic infection as an intermediate host. Terrestrial runoff from urban and agricultural areas into streams and rivers is thought to transport the parasite to coastal marine waters (Miller et al. 2002, Conrad et al. 2005) where it is suspected that invertebrates that are eaten by otters accumulate the parasite (Miller et al. 2002, Arkush et al. 2003). The marine ecology of *T. gondii* is poorly understood, however, as the parasite is also found in pelagic cetaceans and pinnipeds that live far from coastal development (Dubey et al. 2003). *Sarcocystis neurona*, another protozoal parasite with a terrestrial definitive host (opossums), is also causing morbidity and death in southern sea otters (Kreuder et al. 2003). Peritonitis caused by *Profilicollis* spp. has increased between the 1970s and 1990s in California (Thomas and Cole 1996), likely due to changes in sea otter diet or to differences in the intensity of infection in intermediate hosts that increase the rate of transmission (Mayer et al. 2003). The significance of disease at the population level is still not entirely understood since high rates of disease-caused mortality have been noted in California otters for several decades, including during periods of population increase (Estes et al. 2003b). The prevalence and variety of diseases are of concern, however, and it has been speculated that decreased immune function may be a factor. Reduced immune competence could result from environmental toxins, genetic factors, food limitation leading to inadequate diets, new host-pathogen interactions, or habitat degradation leading to nutritional stress (Thomas and Cole 1996, Reeves 2002, Oftedal et al. 2007).

<u>Washington:</u> Sea otters were translocated from Alaska to Washington in the 1960s and the population there is growing and expanding its range (Laidre et al. 2009). Seventy carcasses were collected and necropsied in Washington during 2000 and 2002–04, and 21 were suitable for a full necropsy. Acanthocephalan peritonitis, protozoal encephalitis, and leptospirosis infections were detected (Lance et al. 2004). Of 32 sea otters live-captured in 2000 and 2001, 81% tested positive for exposure to morbillivirus, and there was a fatal case of canine distemper virus in 2004 (Lance et al. 2004). Herpes-like lesions have been seen but are not thought to be clinically significant. Considering that the Washington sea otter population is growing, these findings most likely represent baseline levels of disease that are being balanced by positive population growth factors.

<u>British Columbia:</u> Sea otters were translocated to British Columbia from Alaska in the 1960s, and the population has grown and expanded its range (Nichol et. al 2005). Health surveys are currently being carried out, including serology panels, oropharyngeal swabs for fungal and routine bacterial culture, and polymerase chain reaction (PCR). No fungi or significant bacterial pathogens have been recovered and PCR for herpesvirus has been uniformly negative. In serologic surveys of 42 animals, 8 tested positive for CDV, 2 positive for Toxoplasma, and 6 positive for Brucella sp., and all tested negative for Neospora, Sarcocystis, calicivirus, influenza, and West Nile virus (S. Raverty, personal communication). CDV has been detected serologically in river otters living in the British Columbia marine environment (Mos et al. 2003), but there have been no detected cases of disease or mortality in sea otters due to morbilliviruses. Small numbers of animals had fibrosing cardiomyopathy (S. Raverty, personal communication). As was the case for Washington, considering that the population is growing, these findings most likely represent baseline levels of disease that are being balanced by positive population growth factors.

<u>Russia:</u> Few comprehensive health surveys have been conducted on sea otters in Russia. Birkun and Krivokhizhyn (1991) included starvation and exhaustion (perhaps related to winter storms) as a major cause of death. They frequently found nematode infestations associated with ulcerative gastritis. Other findings included adrenal hemorrhage, "acute total necrotic gastro-enterocolitis, chronic suppurative pancreatitis,

Table 2. Occurrence of major known diseases and disease agents in sea otters by geographic region.

Disease	Russia	Southwest Alaska	Southcentral Alaska	Southeast Alaska	British Coloumbia	Washington & Oregon	California	References
BACTERIAL:								
Vegetative valvular endocarditis	not reported	moderate out of few cases	high prevalence	moderate out of few total cases	negative	negative	rare	K. Burek, S. Raverty, M. Miller (personal communication)
Vibrio sp. associated enteritis/sepsis	not reported	not reported	sporadic	not examined	not reported	not reported	sporadic	M. Miller, K. Burek (personal communication)
Leptospirosis	low prevalence by serology	negative	negative for lesions; 1 high titer	negative	not reported	6 suspect cases in 2002	negative	Lance et al. (2004), Goldstein et al. (in press)
Brucella sp. serology	moderate to high rate of positive serology; negative at Bering Island	not tested	not tested	not tested	negative		moderate with sporadic lesions	Ustinova et al. (2004), Golstein et al. (in press),M. Miller (personal communication)

Table 2. Continued.

Disease	Russia	Southwest Alaska	Southcentral Alaska	Southeast Alaska	British Coloumbia	Washington & Oregon	California	References
PARASITIC:								
Protozoal meningoencephalitis (*T. gondii* /*S. neurona*)	Serology positive for *T. gondii* at Bering Island	negative	rare	negative (few examined)	*T. gondii*, 1 case, genotyped to Type X, 2 animals seropositive	sporadic in WA	high prevalence	Lance et al. (2004), S. Raverty (personal communication), Ustinova et al. (2004)
Acanthocephalan perforations / peritonitis (*Profilicolis* sp.)	Negative	negative	negative	negative	sporadic cases of acanthocephalans, but subacute fibrinous peritonitis common	reported	prevalence 13%, n = 21	Thomas and Cole (1996) , Mayer et al. (2003), Lance et al. (2004)
Nematode-associated peritonitis/ gastric perforation (*Pseudoterranova decipiens*)	sporadic outbreaks (parasites and perforations reported)	sporadic outbreaks	sporadic outbreaks	not reported	sporadic	not reported	negative	Rausch (1953), Margolis et al. (1997), Ballachey et al. (2002), Tuomi and Burek (1999), Burkin (1993)
Cestode impactions (*Diplogonoporus tetrapterus*)	present	not reported	moderate	not reported	negative	not reported	rare	Tuomi (2001), Goldstein et al. (in prep.)

Table 2. Continued.

Disease	Russia	Southwest Alaska	Southcentral Alaska	Southeast Alaska	British Coloumbia	Washington & Oregon	California	References
FUNGAL:								
Coccidioidomycosis	not reported	negative	negative	negative	negative	not reported	Sporadic; esp. in southern half of range	Cornell et al. (1979), Ames et al. (1983)
Histoplasmosis	not examined	rare (1 case)	negative	negative	not reported	not reported	negative	Burek et al. (in prep.)
VIRAL:								
Morbillivirus serology	15% (n=20) positive on Bering Island in 1992; negative in 2004-2005	before 2004 negative; after 2004 high prevalence, low titer	not reported	not reported	19% (n=42) positive, with no evidence of clinical disease	81% positive in 2000/01	high prevalence, low titer	Birkun and Krivokhizhyn (1991), Goldstein et al. (in prep.), S. Raverty (personal communication), Lance et al. (2004), M. Miller (personal communication), Ustinova et al. (2004)
Morbillivirus PCR	not tested	positive nasal swabs from live-captures	positive on a few dead stranded	not tested	dead animals all negative	not tested	Results pending	Goldstein et al. (2009)

Table 2. Continued.

Disease	Russia	Southwest Alaska	Southcentral Alaska	Southeast Alaska	British Coloumbia	Washington & Oregon	California	References
Herpesvirus lesions	low prevalence by serology (phocine herpes-1); suspicious gross lesions	not reported	normally sporadic; high prevalence under stress (*Exxon Valdez* oil spill in PWS)	not reported	negative	sporadic	sporadic and cyclical	Harris et al. (1990), Lipscomb (1998), S. Raverty (personal communication), Lance et al. (2004), Burek and Miller (personal communication), Goldstein et al. (in prep.)
Calicivirus serology	low prevalence by serology	not reported	not reported	not reported	negative	negative	?	Goldstein et al. (in prep.)
UNKNOWN CAUSE:								
Chronic myocarditis/ cardiomyopathy	not reported	negative	rare (2 cases)	negative	rare, 2 cases of fibrosing cardio-myopathy	not reported	high prevalence	Burek (personal communication), Kreuder et al. (2005), S. Raverty (personal communication)
Neoplasia	not reported	not reported	sporadic	not reported	not reported	not reported	rare	Reimer and Lipscomb (1998), Burek (in press)

hepatitis and cholangitis, interstitial and necrotic hepatitis, bullous peritonitis (mesenteritis) and polyserositis, acute and chronic lymphadenitis, catarrhal and productive bronchitis, and ulcerative dermatosis," the causes of which were undetermined. Causes of death listed in a 1935 journal from the Commander Islands include old age, unknown diseases, storms, pathologic deliveries, rock slides, and injuries by killer whales. One animal examined for parasites had a light load of nematodes, all identified as *Porrocaecum decipiens* (renamed now as *Pseudoterranova decipiens*), and cestodes (Barabash-Nikiforov 1947).

On Bering Island from 2004 to 2006, health and disease surveys were conducted jointly by the Alaska SeaLife Center, FWS, and USGS in conjunction with the Marine Mammal Protection Division, Sevvostrybvod, "Komandorsky" Reserve and the Kamchatka Institute of Ecology and Nature Protection (Goldstein et al., in prep). General health was assessed in 89 otters by physical examination and measurement of hematology and serum chemistry parameters. Overall, values did not differ from published ranges for sea otters (Bossart et al. 1990, Hanni et al. 2003) except for a few parameters.

Infectious diseases in Russian sea otters have been studied through serological surveys and examination of fecal parasites. Serologic surveys for common marine and terrestrial pathogens found exposure to *Toxoplasma gondii*, *Sarcocystis neurona*, morbillivirus (CDV, phocine distemper virus (PDV), porpoise morbillivirus (PMV), dolphin morbillivirus (DMV)), phocine herpesvirus-1 and *Leptospira* spp., *Brucella* spp. and San Miguel sea lion virus (calicivirus serotype 1). Serum antibodies to CDV were found in two otters harvested in the Kuril Islands in 1990 (Birkun and Krivokhizhyn, 1991). Otters examined in 2004 and 2005 showed a low prevalence of exposure to *T. gondii*, phocine herpesvirus-1 and *Leptospira interrogans* serovar hardjo, a medium prevalence of Brucella, and negative results for morbilliviruses (Goldstein in review).

Alaska (see Tables 2 and 3): An elevated level of mortality of sea otters was seen at Amchitka Island in February and March of 1951 (Rausch 1953). Many of the animals were old, in poor body condition, and heavily parasitized. Two of the parasites were thought to be clinically significant, including an intestinal trematode associated with hemorrhagic enteritis and a gastric nematode, *Pseudoterranova decipiens* causing gastrointestinal perforations. These parasitic infections apparently resulted from sea otters foraging heavily on fish. Poor body condition due to nutritional stress was thought to have contributed to the high death rate. During 1995-96, dead otters recovered in Orca Inlet, PWS, had died from parasitic infections (consistent with *P. decipiens*) and fishbone impactions associated with consumption of waste products from fish processing (Ballachey et al. 2002). Heating and grinding of the fish waste and disposal farther offshore has eliminated these causes of death. Similar cases occurred in Resurrection Bay when fish waste was being dumped near the harbor, but cases were no longer seen once the fish waste was transported to deeper water (Tuomi and Burek 1999).

Sea otter carcasses resulting from the *Exxon Valdez* oil spill were examined as part of the damage assessment process. The pathologic features of oil spill exposure were described along with lesions associated with stress, including oral ulcers from herpes infection, hemorrhagic enteritis, bullous emphysema, and liver pathology (Lipscomb et al. 1993). Two oil-spill animals that were kept in captivity were necropsied 3-10 years after the spill, and in one there was evidence of long-term liver disease. Rare cases of valvular endocarditis (VE) were described in oil-spill animals, but an etiologic agent was not identified.

Large tumors have been found in sea otters killed by subsistence hunters in the southeast and southcentral Alaska stocks but the incidence appears to be low (Burek et al., in press).

A serologic survey of sea otters from the western Aleutian Islands and southeast Alaska in 1997 (n=72) was negative for exposure to *T. gondii*, CDV, calicivirus, *Leptospira interrogans*, and *Coccidioides immitis*. The study showed some positive titers for *Brucella* spp. (5/65) (Hanni et al. 2003). Another serologic study of 65 Alaskan otters was negative for *T. gondii* (Fayer et al. 2004).

Since 2003, FWS and the USGS, in collaboration with the Alaska SeaLife Center, the California Department of Fish and Game, the University of California at Davis, and Alaska Veterinary Pathology Services, have made a concerted effort to obtain sea otter carcasses through a stranding network and have conducted full diagnostic necropsies on fresh carcasses. Most of the examined carcasses have come from Kachemak Bay, which is within the southcentral stock area. Several carcasses have also been obtained from the southwest stock, including at Kodiak Island and on the Alaska Peninsula. Preliminary results reveal a very different pattern of mortality than has been observed in California (Burek et al. 2004). There has been no evidence of *S. neurona*, three cases of *T. gondii* encephalitis, no intestinal perforations and peritonitis due to *Profilicollis* spp. acanthocephalans, and only one case of chronic myocarditis/cardiomyopathy syndrome. In Alaskan otters examined to date, the most commonly diagnosed cause of death has been VE (Burek et al. 2005a). From 2002-06, VE was the primary cause of death for 26 of 64 fresh, non-frozen beachcast cases, a prevalence of 41%. Of the cases that were cultured for bacteria, 80% were culture positive (in the heart valve and/or blood) for *Streptococcus bovis* complex (Jang et al. 2005). A few cases of VE have been culture negative, a few were mixed cultures including *S. bovis*, *Aeromonas* spp., or other species of Streptococci. The nomenclature of the streptococcal organisms is quite complex and changes very frequently, so "*S. bovis* complex" actually represents a relatively large and diverse group of organisms. Most of the isolates identified further have been *Streptococcus infantarius* spp. *coli*. The same organism has also been found to be the cause of septicemia and meningoencephalitis. This combination of VE, septicemia, and meningoencephalitis due to *S. bovis* complex is now being referred to as "Strep bovis syndrome."

The epidemiology of this disease is not well understood. VE is normally a sporadic disease in human and animal populations. In most cases, a "primary" source of the bacteria for the valvular lesion is found, and is usually due to normal body flora or localized extra-cardiac infection, which results in repeated bacteremia such as with chronic oral infections or other chronic disease. Some possible routes of infection are fecal-oral, direct contact (with broken or unbroken skin), or aerosol. Generally, the sea otters that have died of Strep bovis syndrome were of prime age, male, and without other major primary sources of infection. Classically, with chronic VE, there is often a pre-existing valvular defect or other area of damage to the endothelium that the bacteria colonize, but it seems unlikely that such a large percentage of animals would have pre-existing valvular lesions. Some organisms (i.e., *Bartonella* spp. in dogs, and *Erysipelothrix rhusiopathiae* in pigs) do act as primary pathogens on heart valves. A possible involvement with *Bartonella* spp. is being investigated in sea otters. It is also possible that *S. bovis* complex has a strong predilection to colonize and damage valves/endothelium on its own and is a primary pathogen. Alternatively, there could be some other predisposing condition that damages the valve or endothelium and creates an environment for the *S. bovis* complex to colonize. Cases of VE have been documented from Umnak Island to PWS, with most cases currently originating in Kachemak Bay (the region where the stranding network is strongest).

Historically VE has been reported in sea otters but only sporadically. Out of 282 sea otter carcasses necropsied after the 1989 *Exxon Valdez* oil spill, only one was reported with VE, and it is not known which organism was involved in that case (Lipscomb et al. 1996). An additional case of VE was noted in an animal from PWS that died in captivity in 1990 (Joseph et al. 1990). Approximately 100 sea otter carcasses were examined to various degrees in the decade preceding 2002, and VE was detected in four, all from PWS and Kachemak Bay (FWS unpublished data). In summer 1987 there was a mortality event in the Kodiak archipelago that involved approximately 55 animals. Four animals suitable for necropsy were sampled from this event and one had VE (DeGange and Vacca 1989). The mortality event was described as most likely caused by paralytic shellfish poisoning (PSP), because of an algal bloom and saxitoxin levels detected in other species at the time.

In August 2006, FWS submitted an application to the Marine Mammal Working Group to consider if these circumstances constituted an Unusual Mortality Event (UME) as defined under Section 404 of the Marine Mammal Protection Act (MMPA). The Working Group made a formal recommendation to declare a sea otter UME in Alaska in September 2006, and an investigative team was formed. Population surveys indicated that sea otter numbers were increasing in Kachemak Bay and the adjacent Kenai Peninsula during the UME. This raises the possibility that the Strep bovis syndrome represents a density-dependent disease and not a decline-causing process. As a result of these findings the UME has been closed. Nevertheless, because the disease has been detected in some southwest Alaska DPS animals in Kodiak and the Alaska Peninsula, it is still of concern. Given the information from the Kenai Peninsula, it is unlikely that Strep bovis syndrome was a factor in the decline of the southwest stock; however, it could be a factor limiting population recovery. Studies are ongoing on the epidemiology and pathology of *S. bovis*, Bartonella, and morbilliviruses. Important questions that still need to be addressed include:

- Is *S. bovis* complex a primary pathogen or secondary to another pathogen or toxin?
- What is the source of the *S. bovis* complex or is it a normal part of the flora?
- What is the route of transmission?
- Is it a threat to recovery of the southwest Alaska DPS?
- What is the extent of the disease syndrome and is the incidence increasing or decreasing?

Other causes of mortality documented in recent necropsies of Alaskan otters include boat strike, possible interspecific aggression (killer whale or Steller sea lion), intraspecific aggression, neoplasia, gastrointestinal impactions, emaciation, myocarditis/cardiomyopathy, Vibrio parahemolyticus septicemia/bacteremia and enteritis, and disseminated histoplasmosis (K. Burek personal communication).

The role of morbilliviruses as a cause of morbidity and mortality in sea otters is currently poorly understood. Morbilliviruses in other species of marine mammals have resulted in large and rapid population declines, including PDV epizootics in harbor seals in the North Sea in 1998 and 2002 (Kennedy 1998, Harknonen et al. 2006), CDV

Table 3. Information on disease agents investigated in sea otters in the Gulf of Alaska and the Aleutian Islands.

Agent/syndrome	Data source	Years	Location	Prevalence	Comments	Reference
Valvular endocarditis (general syndrome)	necropsy	1987	Kodiak	1/4	no cultures	DeGange and Vacca (1989), Lipscomb et al. (1996),
		1989	PWS	1/282	no cultures	Joseph et al. (1990), Burek et al. (personal
		1990	PWS	1	no cultures	communication), Jang et al. (2005), Burek et al. (in prep.),
	necropsy of fresh carcasses, culture	2002-06	primarily Kachemak Bay; few from AK Peninsula & Kodiak	26/63[a]	many organisms cultured; 80% S. bovis complex	Gill et al. (in prep.),
Toxoplasma gondii	necropsy and specimens from fresh carcasses, serology and culture	2002-06	primarily Kachemak Bay; few from AK Peninsula & Kodiak	2/ 64[a]	cases positive by serology and culture	Gill et al. (in prep.)
T. gondii	Serology	2004	not indicated	0/65		Fayer et al. (2004),
		2004, 2005	Kodiak, S. Alaska Peninsula, eastern Aleutian Islands	2/74		Goldstein et al. (in prep.)
Sarcocystis neurona	necropsy and specimens from fresh carcasses, serology and culture	2002-06	primarily Kachemak Bay; few from AK Peninsula & Kodiak	0/63[a]		Gill et al. (in prep.)
	serology	2004, 2005	Kodiak, S. Alaska Peninsula, eastern Aleutians	2/74		Goldstein et al. (in prep.)

Table 3. Continued.

Agent/syndrome	Data source	Years	Location	Prevalence	Comments	Reference
Profilicolis sp.	necropsy of fresh carcasses	2002-06		0/63*	associated with gastric and intestinal perforation and peritonitis	
Pseudoterranova decipiens	necropsy	1951	Amchitka Island	? / up to 31	associated with gastric and intestinal perforation and peritonitis in all cases except the 1989 survey	Rausch (1953) Margolis et al. (1997) Ballachey et al. (2002) Tuomi and Burek (1999) Gill et al. (in prep.)
		1989	PWS	1/66 in PWS		
		1995-96	Orca Inlet PWS	5/9		
		1999?	Resurrection Bay	1 case		
		2002-06	primarily Kachemak Bay; few from AK Peninsula & Kodiak	0/63[a]		
Diplogonoporus tetrapterus	necropsy	1989	PWS	8/66	impactions associated with heavy infestations in some cases	Margolis et al. (1997) Gill et al. (in prep.)
	necropsy of fresh carcasses	2002-06	primarily Kachemak Bay; few from AK Peninsula & Kodiak	0/63[a]		
Coccidioides immites	Serology	1997	Aleutians and southeast AK	0/72		Hanni et al. (2003)
herpesvirus	necropsy	1989 1990 1998	PWS	sporadic and episodic	oral ulcers and characteristic lesions, no virus identified	Harris et al. (1990) Reimer and Lipscomb (1998) Goldstein et al. (in prep.)
	serology (phocine herpesvirus-1)	2004, 2005	Kodiak, S. Alaska Peninsula, eastern Aleutians	11/72		

Table 3. Continued.

Agent/syndrome	Data source	Years	Location	Prevalence	Comments	Reference
CDV, calicivirus, *Leptospira interrogans*, *Brucella* sp., *T. gondii*, *Coccidiodes immites*	serology	1997	western Aleutians (61) and SE AK (11)	0/72		Hanni et al. (2003)
		2003-04	Kodiak / AK Peninsula	high prevalence of PDV-like virus	low titers	Goldstein et al. (2009)
		2004, 2005	Kodiak, S. Alaska Peninsula, eastern Aleutians	calici 0/65 *Brucella* 1/72 *Lepto* 1/72		

[a] necropsy data are still preliminary and results approximate due to some pending analyses and the need to re-check data

epizootics in Caspian seals (Kennedy et al. 2000, Pollack 2001, Kuiken et al. 2006), and PMV in dolphins along the U.S. Atlantic coast (Schulman et al. 1997). Populations rebound once enough animals become resistant to disease due to survival and protective immune response. Sea otters that were live-captured and sampled in the Commander Islands, Aleutian Islands, southern Alaska Peninsula, and Kodiak archipelago in 2004 and 2005 were tested by serum neutralization for exposure to CDV, PDV, PMV, and DMV. Preliminary results suggest exposure to a PDV-like morbillivirus in the eastern Aleutians/southern Alaska Peninsula and the Kodiak archipelago, but animals from the Commander Islands and the central and western Aleutians tested negative (Goldstein, in prep.). Archived serum from animals sampled at a variety of sites in the 1980s and 1990s were consistently negative for morbillivirus, and all animals sampled in 1997 from the Aleutians were negative for CDV (Hanni et al. 2003); PDV serologic assays were not run on those animals. Taken collectively, this evidence supports the hypothesis of exposure of sea otters to a novel virus in the eastern Aleutians/Alaska Peninsula/Kodiak area between 1997 and 2004, but not in the western Aleutians where population declines were highest. These data may indicate exposure to a well-characterized morbillivirus or it may represent cross-reactivity to a new pathogen (A. Doroff, V. Gill, and T. Goldstein, personal communications). PCR sequences identical to PDV from the 2002 European harbor seal epizootic have been detected and characterized from live-captured animals from Kodiak and some dead-stranded carcasses from Kachemak Bay (Goldstein et al. 2009). Research is continuing to characterize the virus by culture and PCR, and to determine whether it is causing significant morbidity and mortality in sea otters or other species.

The duration of the decline of southwest Alaska sea otters without signs of a rebound, and its broad geographic distribution, are not typical of a problem caused purely by disease. Additionally, the relative lack of carcasses recovered despite intensive field studies at two locations in the Aleutians during the decline is inconsistent with disease-related mortality observed elsewhere. Although current data suggest that disease has not been the primary factor in this decline, there has been little work on direct causes of mortality in this DPS and the potential for disease to impede recovery cannot be ruled out. Continued studies of diseases and causes of mortality are therefore warranted. Kuker and Barrett-Lennard (2010) provide additional discussion of the possibility of disease as a factor in the sea otter decline.

Immunocompetence

Some researchers believe that the high rate of infectious disease observed in sea otter populations may be due partly to immune deficiency resulting from poor nutrition, anthropogenic toxicants, immunotoxic viruses (such as morbilliviruses, retroviruses, parvoviruses, etc.), or lack of genetic diversity. Defining immune competence and immune deficiency is often problematic, and determining underlying causes of immune deficiency is even more difficult. Some techniques for characterizing immune function parameters in sea otters were reported by Schwartz et al. (2005), but that work mostly characterized the "normal" values and did little to determine whether there actually was immune competence or deficiency.

With all sea otter populations having experienced a population bottleneck in recent history, one possibility for the high rate of infectious disease is a loss of genetic diversity. The concept that loss of genetic diversity may lead to decreased disease resistance is controversial because of the difficulty of performing controlled experiments in the populations at risk. However, several experimental models support this concept (Spielman et al. 2004). Studies of genetic variation within some microsatellite and mitochondrial markers indicate that historical sea otter populations (prior to 1742) had significantly more variation than extant populations (Scribner et al. 1997, Larson et al. 2002a, b). An important question is whether the loss of genetic variation varies geographically in degree and quality, and whether that variation correlates with population-level immune function and/or disease susceptibility. A preliminary examination of major histocompatibility complex (MHC) II genes (which indicate immune system vigor) in sea otters from California, Washington, and Alaska indicated that distinct clades are demarcated by geographic region, but the significance of this for immune vigor or resistance to infectious disease has not been determined (Bowen et al. 2006). Sea otters in California have very low levels of variation at the MHC genes compared to other mammals, where these loci are typically highly variable. One of the MHC genes (DRA) was monomorphic in California sea otters and two of them (DQB and DRB) had only two alleles (Aguilar et al. 2008).

Discussion and Recommendations

Potential causes of the sea otter decline in the Aleutian Islands have been explored by reviewing available data on sea otter reproduction, survival, distribution, habitat, and environmental contaminants. Estes et al. (1998) concluded that the decline was most likely the result of increased adult mortality due to predation by killer whales. However, novel or episodic disease agents have not yet been thoroughly investigated as contributing factors. Furthermore, the number of sea otters remaining in some areas is small and continuing to decline. These remaining small populations are vulnerable to the effects of disease, oil spills, or other catastrophic events.

To determine whether the decline is spreading, surveys are required to estimate population sizes and trends, particularly for populations located at the geographic "margins" of the decline.

Beachcast carcasses throughout the range should be collected and subjected to thorough and consistent necropsies, using protocols established by FWS and collaborators. The possibility that morbillivirus was introduced into a naive population should be confirmed or disproved, and an understanding of the epidemiology of *S. bovis* syndrome should be developed. Investigations of causes of mortality should be part of a comprehensive effort to improve understanding of the factors driving population dynamics, and this, in turn, should provide informed guidance for actions to aid the recovery of southwest Alaska sea otters. Key geographic areas for expanded research include accessible locations within the range of the southwest Alaska DPS (Adak, Kodiak, and Kamishak Bay), along with regions located along the margins of the decline (Commander Islands, Kachemak Bay, and PWS). Comparisons of causes of mortality between the southwest Alaska DPS and populations in Alaska and California would be beneficial.

C. Biotoxins

The potential role of PSP toxins, domoic acid, ciguatoxins, brevetoxin, and other biotoxins in the sea otter decline is poorly understood. These compounds are produced by dinoflagellates and diatoms, which under certain environmental conditions experience increased numbers (blooms). Marine mammal mortality events associated with these “harmful algal blooms” (HABs) have been reported (Bossart et al. 1998, Hernandez et al. 1998, Scholin et al. 2000, O’Hara and O’Shea 2001). There is evidence that the frequency and distribution of toxigenic phytoplankton blooms are increasing (Smayda 1990, Hallegraeff 1993, Van Dolah 2000, Mos 2001). Whether marine mammal mortality events are increasing due to HABs is unclear.

Domoic acid, a biotoxin produced by the alga Pseudonitzschia australis, can accumulate in filter-feeding shellfish and be passed through the food chain, thereby affecting not only animals that prey on invertebrates, but fish-eating species as well. Domoic acid poisoning was first identified in human cases at Prince Edward Island (Canada) where over 100 people became ill and 4 died after eating contaminated mussels (Teitelbaum et al. 1990). First detected on the west coast of North America in 1991, domoic acid has caused several large die-offs of sea birds (Work et al. 1993) and repeated mortality events in California sea lions (*Zalophus californianus*; Scholin et al. 2000). Despite these repeated mass mortality events, the California sea lion population is increasing and therefore domoic acid poisoning is not limiting population growth at this time.

Domoic acid has caused stranding and mortality in California sea otters (Kreuder et al. 2003), but levels of exposure and effects of domoic acid in Alaskan sea otters are unknown (Table 4). There has been minimal monitoring in Alaska, with the exception of one study in which low levels of domoic acid were detected in razor clams in Kachemak Bay (Horner et al. 1997). Current studies of fresh dead otters have detected very low levels of domoic acid in urine, primarily from otters in Kachemak Bay (V. Gill, personal communication). Lesions typical of domoic acid intoxication in other animals have not been detected in Alaskan sea otters. Myocarditis and dilated cardiomyopathy are major causes of death in California sea otters, and exposure to domoic acid and *S. neurona* appears to be a major risk factor for development of this disease syndrome (Kreuder et al. 2005). Cases of chronic myocarditis similar to that described in California are very rare in sea otters in Alaska (K. Burek personal communication).

The compounds responsible for PSP, produced by certain dinoflagellates, can accumulate to toxic levels in filter-feeding bivalves. Butter clams (*Saxidomus giganteus*), which accumulate saxitoxin, are an important component of the sea otter’s diet in Alaska. Although PSP monitoring is performed in some commercial shellfish operations, the presence of these substances has not been monitored consistently throughout Alaska. A mortality event in sea otters at Kodiak Island in 1987 was attributed to PSP exposure, based primarily on the identification of an unknown toxin in two sea otters, a few cases in people at the same time, and the presence of PSP toxins in blue mussels (*Mytilus trossulus*; DeGange and Vacca 1989). However, subsequent studies showed that sea otters are able to avoid consumption of PSP-contaminated prey (Kvitek et al. 1991). In summer 2008, a major saxitoxin mortality event occurred in the St. Lawrence River estuary in Quebec, Canada, with deaths recorded of cetaceans, pinnipeds, fish, and birds (Measures 2007). The intensity of the bloom was thought to have been related to unusually warm ocean temperatures and large amounts of freshwater run-off due to unusual levels of precipitation (L. Measures, personal communication). Since PSP is known to be present in the habitat of the southwest Alaska sea otter DPS and in major food sources of sea otters, studies should be conducted to determine whether PSP is a primary or contributory cause of death. Initial testing of stranded otters for domoic acid and PSP began in 2009.

D. Contaminants

Persistent organic pollutants

Persistent organic pollutants (POPs) may affect sea otters and their habitat. Potential sources of such contaminants include both local point-sources at specific locations in Alaska and more diffuse sources that may affect an area via long-range transport. One study found high levels of polychlorinated biphenyls (PCBs) and dichlorodiphenyltrichloroethane (DDT) in sea otter tissues collected near military installations in the Aleutian Islands (Estes et al. 1997), but

Table 4. Information on biotoxins found in sea otters in the Gulf of Alaska and the Aleutian Islands.

Agent	Data source	Years	Location	Prevalence	Comments	Reference
PSP (suspected)	mortality event	1987	Kodiak	55 otters involved	low levels of PSP in mussels	DeGange and Vacca (1989), K. Burek (personal communication)
	mortality event involving otters (12) and humpback whales (up to 8)	2008	Kodiak / AK Peninsula	preliminary results; 1 otter confirmed positive		
Domoic acid	analysis of urine during disease, monitoring, necropsy	2001	Kachemak Bay	1/30 (low level of toxin)	scattered very low levels of domoic acid found in urine	Gill et al. (in prep.)

the elevated levels were likely an artifact of the samples having come from emaciated, beachcast carcasses (J. Estes, personal communication). Generally, reported levels of total PCBs have been highest in sea otters from the Aleutian Islands followed by California; they were negligible in otters from southeast Alaska (Table 5). High levels of DDT have been recorded in California but DDT levels have been negligible in Alaska (Bacon 1994, Jarman et al. 1996, Giger and Trust 1997, Estes et al. 1997, Bacon et al. 1999).

A study of kidney and liver samples obtained from subsistence-collected animals throughout Alaska from 1993-99 indicated that organochlorine pesticides were either absent, or present only at low concentrations, in most otters. However, this was a biased sample in that hunters are known to target the healthier and "fatter" animals. A few individuals had elevated concentrations of PCB aroclors, indicating that PCB contamination may be occurring in local areas (Comerci et al. 2002). More recent data on blue mussels collected from the Aleutian Islands and southeast Alaska indicated low concentrations of PCBs at most sample locations, with "hot spots" of high PCB concentrations evident at Adak, Dutch Harbor, and Amchitka. With the exception of these "hot spots," PCB levels in mussel samples from southwest Alaska were lower than those from southeast Alaska (Reese 1998).

It is not known whether organochlorine (OC) exposure is having an impact on sea otters at the population level. One documented effect of OCs is immunosuppression (O'Hara and O'Shea 2001), but the effects of different levels and mixtures of OCs in sea otters are unknown. In vitro studies with other species suggest that effects of OCs are species-specific (Mori et al. 2006). Ross (2002) suggested that prior POP exposure can exacerbate morbillivirus-related outbreaks in marine mammals. Immunosuppression resulting from contaminant exposure is considered a potential contributor to the incidence of disease documented in southern sea otters, but this remains unproven (Thomas and Cole 1996, Reeves 2002). The hypothesis that POP contaminants were a factor in the European harbor seal PDV epidemics has not been supported by studies where the samples were properly stratified by age and sex (Rijks et al. 2008).

OCs are also known to cause reproductive effects in a variety of species (Aulerich et al. 1973, Platonow and Karstad 1973). Recent analyses of blood contaminant levels in sea otters indicate much higher overall levels in California than in the Aleutian Islands, yet age-specific reproductive rates in these and other populations are similar (Jessup et al. in review). Such results suggest that OC contamination is not of particular concern for sea otters in the western Alaska DPS, and decreased reproduction (from OC toxicity or other unrelated causes) does not seem to be a factor in the current decline.

Perfluorinated Compounds

Elevated levels of perfluorinated contaminants (PFCs) in livers of sea otters in California were correlated with deaths due to infectious disease (Kannan et al. 2006). Recent studies indicate that levels of PFCs are similar in Alaskan otters to those described in California otters. Over time, perfluorooctanesulfonate levels are decreasing in response to their removal from markets, while the contribution of perfluorononanoate is increasing. No link has been found between PFC levels and causes of death, and further work is needed (Hart et al. 2009).

Heavy Metals

Higher concentrations of cadmium and selenium were present in older sea otters than in juvenile otters sampled in the Aleutian Islands during the 1990s (Giger and Trust 1997). Comerci et al. (2002) analyzed livers and kidneys collected from sea otters from southeast, southcentral, and southwest Alaska for a suite of trace elements and metals. Metals and trace element concentrations were highest in otters from southcentral Alaska, followed by southwest, and then southeast. This is very similar to the pattern seen in Steller sea lions (K. Beckmen, personal communication). Kannan et al. (2006) compared trace element concentrations among diseased, emaciated, and non-diseased sea otters in California. Hepatic concentrations of copper and cadmium were higher in otters in that study than have been reported for any other marine mammal species. Manganese, cobalt, zinc, and cadmium were elevated in the diseased and emaciated otters relative to non-diseased otters. Considering the pattern of low tissue accumulation of metals in Alaskan otters and the lack of industrial or urban development in these areas, heavy metals are unlikely to be a causal factor in the decline of the southwest Alaska DPS.

E. Oil spills and oiling

Sea otters are particularly vulnerable to contamination by oil (Cohen and Aylesworth 1990). Oil contamination can have both immediate and long-term effects on sea otters and on population recovery (Peterson et al. 2003). Five characteristics of sea otter biology help explain their extreme vulnerability to oil contamination:

Sea otters depend on their fur and the air trapped within it for thermal insulation. Oil destroys the water-repellent nature of the fur and it eliminates the air layer, thereby reducing the insulative value by 70% (Williams et al. 1988). The direct result is acute hypothermia.

Once the fur is fouled, sea otters ingest oil as they groom themselves. Ingested oil damages internal organs, resulting in acute and chronic effects on animal health and survival. Based on a mink

Table 5. Information on contaminants found in sea otters in the Gulf of Alaska and the Aleutian Islands.

Agent	Data source	Years	Location	Sample size	Comments	Reference
Polychlorinated Biphenyls Organochlorines Heavy Metals	kidney and liver from subsistence animals	1993-99	throughout Alaska and some of Russia	68 (plus data from other studies)	generally low background levels of all	Comerci et al. (2002)
Polychlorinated Biphenyls Organochlorines	tissues	1991-92	California southeast Alaska and Aleutian Islands	23 otters	high levels in Aleutian Islands, low in southeast Alaska	Estes et al. (1997)
		1993-94	Aleutian Islands	25 eagle eggs		
Polychlorinated Biphenyls Organochlorines	mussels	2005?	Aleutian Islands	39 sites	hot spots in the Aleutian Islands, overall background level low	Reese, Estes, and Tinker (in prep.)
			Southeast Alaska	5 sites		
				30-50 mussels/ sample		
Anthropogenic radionuclides	skulls	1950s and 1990s	Amchitka	18	Pb 210 levels were higher in 1950s than 1990s	Baskaran et al. (2003)
Perfluorinated compounds	livers	1992-2007	Prince William Sound	36	similar to California	Hart et al. (2009)
			Resurrection Bay	7		
			Kachemak Bay	34		

model, oral exposure to low doses of oil can lead to changes in hematology, immune function, and reproductive success (Mazet et al. 2001, Schwartz et al. 2004).

Benthic invertebrates accumulate and store toxic hydrocarbons. Sea otters therefore ingest hydrocarbons when they feed on these organisms during and after an oil spill.

Sea otters are nearshore animals that exhibit strong site fidelity, often remaining in or returning to oiled areas after release. In addition, they often rest in kelp beds, which collect and retain spilled oil.

Sea otters are often found in single-sex aggregations, which can include hundreds of individuals. Thus, large numbers of sea otters (representing a substantial portion of the reproductive potential of a population) can become fouled by oil simultaneously.

The sea otter population in PWS illustrates both short-term and long-term effects of oil contamination. The oil tanker *Exxon Valdez* ran aground in PWS in March 1989, spilling 42 million liters of crude oil. Nearly 1,000 sea otter carcasses were recovered within six months (Ballachey et al. 1994), with the total mortality estimated to be several thousand animals (Tuomi 2001). Lesions described in sea otters exposed to Prudhoe Bay crude oil included gastric erosion and hemorrhage, centrilobular hepatic necrosis, periportal and diffuse hepatic lipidosis, and renal tubular lipidosis (Lipscomb et al. 1993). Several parameters suggest long-term consequences of residual oil. Sea otters in parts of PWS that were most heavily oiled have significantly higher liver levels of cytochrome P4501A, a biomarker for hydrocarbons, than otters from less heavily oiled areas (Ballachey et al. 2002). Differences in blood parameters suggest liver damage in oiled versus unoiled areas (Ballachey et al. 2003). Population growth remains significantly lower in heavily oiled areas, suggesting that population recovery may be constrained by the residual effects of oil, even when the food supply is adequate (Bodkin et al. 2002).

Some safeguards have been established since the *Exxon Valdez* oil spill to minimize the likelihood of another spill of catastrophic proportions in PWS. However, tankers, other vessels, and fuel barges are potential sources of oil and fuel spills that could have a catastrophic effect on sea otters in the southwest Alaska DPS. Since 1990 in Alaska, more than 4,000 spills of oil and chemicals into water have been reported to the U.S. Coast Guard National Response Center. Of these, nearly 1,100 occurred within the range of the southwest Alaska DPS (Table 6). Reported spill quantities have ranged from a few gallons to thousands of gallons, including various refined products (primarily diesel fuel, gasoline, and lubricating oils; National Research Council 2008). It is important to examine patterns of ship traffic within the range of the southwest Alaska DPS, as well as existing regulations, to determine if appropriate controls are in place. A well-designed contingency plan, including procurement of the necessary supplies, protocols, and equipment, should be developed to respond to oil spills as rapidly as possible.

Currently, there is no oil or gas production within the range of the southwest Alaska DPS, but outer continental shelf oil and gas lease sales are planned for lower Cook Inlet and Bristol Bay. Based on a review of the draft Environmental Impact Statement for these sales, it is the opinion of FWS that "the potential impacts of this development on the southwest Alaska DPS will be negligible as sea otters occur primarily in the near shore zone and the lease sale area is at least three miles off shore. Therefore, sea otters do not significantly overlap with the lease sale area." However, within the Bristol Bay MU, many sea otters are found long distances from shore and they could occur within the lease sale area. Also, as demonstrated during the *Exxon Valdez* oil spill, spilled oil can affect wildlife at distances of up to several hundred miles from the initial release site (Loughlin 1994). Studies to predict oil spill trajectories in this area would be beneficial.

Large-volume oil spills, once they have occurred, are nearly impossible to contain or manage with current technology. When large numbers of otters become contaminated with spilled oil, it is not possible to capture and treat the great majority of the animals. Those that become extensively contaminated or ingest large quantities of oil are difficult, if not impossible, to rehabilitate with currently available technology (Estes, 1991). Even with those constraints, however, it should be possible to protect small areas that provide important habitat in specific parts of the southwest Alaska DPS from becoming oiled.

F. Food limitation

As with many apex predators, the abundance or density of sea otter populations may be ultimately limited by the abundance of prey resources (Kenyon 1969, Garshelis et al. 1986, Estes 1990, Estes et al. 1996). It is generally believed that for sea otters, population carrying capacity ("K", the average population density at which total births and total deaths are equal and thus net population growth is zero) is determined by increased mortality due to energetic/nutritional stress, emaciation, or other factors such as disease that may be triggered by energetic/nutritional stress (Estes et al. 1996). While food abundance is certainly not the only factor that can limit sea otter population growth, there are examples from across the sea otter's range that support the general scenario of resource limitation at high population density. These include Bering Island in the early 1990s (Bodkin et al. 2000), Amchitka Island during

Table 6. Summary of reported non-crude oil spills in southwest Alaska by MU, July 1, 1995 – June 30, 2005.

		Quantity Spilled (gallons)							
Management Unit	Source	0-10	10-25	25-100	100-500	500-1,000	1,000-5,000	>5,000	Total
Western Aleutian	Vessel	98	16	22	8	1	1	3	149
	Cannery	16	3	8	1	2	0	1	31
	Harbor/Port	5	0	2	0	0	0	0	7
	Non-crude Terminal	3	1	3	2	0	0	0	9
	Subtotal	122	20	35	11	3	1	4	196
Eastern Aleutian	Vessel	74	11	14	5	1	0	2	107
	Cannery	10	1	1	0	0	0	0	12
	Harbor/Port	4	1	0	0	0	0	0	5
	Non-crude Terminal	0	0	1	1	0	1	0	3
	Subtotal	88	13	16	6	1	1	2	127
Bristol Bay	Vessel	6	1	0	0	0	0	0	7
	Cannery	1	0	0	1	0	0	0	2
	Harbor/Port	0	0	0	0	0	0	0	0
	Non-crude Terminal	0	0	0	1	0	0	0	1
	Subtotal	7	1	0	2	0	0	0	10

		Quantity Spilled (gallons)							
Management Unit	Source	0-10	10-25	25-100	100-500	500-1,000	1,000-5,000	>5,000	Total
South Alaska Peninsula	Vessel	20	2	7	3	0	1	0	33
	Cannery	3	3	1	1	0	0	0	8
	Harbor/Port	1	0	0	0	0	0	0	1
	Non-crude Terminal	0	0	0	0	1	0	0	1
	Subtotal	24	5	8	4	1	1	0	43
Kodiak, Kamishak, Alaska Peninsula	Vessel	100	7	10	0	2	0	1	130
	Cannery	0	1	5	2	1	0	0	9
	Harbor/Port	2	2	0	0	0	0	0	4
	Non-crude Terminal	0	0	1	0	0	0	0	1
	Subtotal	102	10	16	12	3	0	1	144

the 1960s (Kenyon 1969) and early 1970s (Estes 1977), areas of PWS prior to the 1989 *Exxon Valdez* oil spill (Garshelis et al. 1986), and, recently, the central portion of the southern sea otter's range in California (Tinker et al. 2006a, b).

As sea otter populations grow from low to high densities, they can have a strong impact on certain prey species, including sea urchins and other herbivorous invertebrates in rocky habitats (Duggins 1980, Estes and Duggins 1995), and filter-feeding bivalve mollusks in soft-sediment habitats (Kvitek et al. 1989). This tendency to limit abundant prey populations can lead to strong and often complex community-level effects (Estes and Palmisano 1974; Estes et al. 1978, 2004). For example, through a trophic cascade initiated by sea otter predation on urchins, there can be a significant increase in the standing algal biomass and the overall productivity of kelp forests (Duggins 1980), leading to an increase in the abundance of invertebrates and vertebrates that rely on kelp for food or habitat, thereby increasing the suite of potential prey species available to sea otters and other predators. This complex feedback loop can thus elevate the potential sea otter population density (Estes 1990). However, despite adaptive behavioral responses to increasing density, such as increased diet breadth and percent time foraging (Ostfeld 1982, Garshelis et al. 1986, Watt et al. 2000, Gelatt et al. 2002), at high population densities individual sea otters can reach a point at which they are unable to meet their energetic requirements, leading to an increase in starvation-induced mortality. In some instances, mortality associated with food limitation can occur suddenly. Instead of a slow, incremental increase in food-related mortality, there will be a sudden die-off of animals, often at the end of a harsh winter or as a result of some triggering event that pushes many already-weakened animals past the point of energetic balance. Such sudden mortality events associated with nutritional limitation have been reported at Amchitka Island in the 1950s (Kenyon 1969) and Bering Island in 1980 (Bodkin et al. 2000), and they resulted in large numbers of beach-cast sea otter carcasses. In both of these cases, the mortality events appeared to essentially mark the point at which the populations reached carrying capacity, insofar as sea otter numbers subsequently stabilized, The current sea otter population decline in southwest Alaska was first documented in the central and western Aleutian Islands (Estes et al. 1998, Doroff et al. 2003), a region in which sea otters had been present at high density for many years and where populations were thought to be at or near carrying capacity (Kenyon 1969, Estes 1990). Given the rapid onset of this decline, and the fact that there have been instances in the past of sudden die-offs associated with food limitation in sea otter populations, food limitation must be considered a possible factor in the current population decline.

Methods for assessing food limitation

Direct measures of food abundance: Evaluating whether food limitation has contributed to the decline of sea otters in southwest Alaska would seem to be a simple matter of measuring the abundance of sea otter prey. Such a task is complicated however, because: 1) assessments of prey abundance require good information on which prey species are consumed by sea otters, yet diet often varies between locations and habitats; 2) any assessment of prey abundance must be made relative to a sea otter's foraging ability - that is, sampling by researchers must reflect the depths and spatial scales over which sea otters typically forage; and 3) many important prey species are cryptic and difficult to count. Because of these and other difficulties, most studies rely heavily on indirect indices of prey abundance. Nonetheless, in those cases where there are consistent and broad-based sampling regimes for key prey species that are maintained over long periods of time, the resulting time series can provide insight into changes in the abundance of food resources relative to sea otter densities.

Two examples serve to illustrate this approach. In the western Aleutian Islands during the 1980s, Estes and colleagues documented rapid declines in the density and size structure of green sea urchins after sea otters re-colonized Attu Island (Estes 1990, Estes and Duggins 1995). As the otter population approached carrying capacity at one location, Massacre Bay, the size-frequency distribution and density of sea urchins at long-term sampling sites began to approach that of Amchitka Island, where sea otters had been present at high densities for many years. In this case, the combination of direct estimates of urchin size and abundance in conjunction with the time series of sea otter population counts provided a clear picture of resource limitation in sea otters. A second example of the effective use of direct estimates of prey abundance comes from Glacier Bay. After their translocation to southeast Alaska in the 1960s (Jameson et al. 1982), sea otter numbers increased rapidly as the animals re-colonized unoccupied habitat, and otters began to move into Glacier Bay in about 1995. Bodkin et al. (2007) sampled the relative density of infaunal bivalves and other soft-sediment prey species, both before and during the re-colonization event. Clear decreases in bivalve size and abundance have already been seen, and it is anticipated that this time series will eventually provide baseline data on the relative density of prey that exists when this population reaches carrying capacity.

The above examples demonstrate that it is possible to estimate food abundance directly and relate those estimates to carrying capacity for sea otters. However, such sampling programs are labor-intensive and must be maintained in a consistent manner over many years in order to provide meaningful information. Ideally, direct sampling of prey abundance should be a part of a larger program that also measures one or more indices of sea otter population status, as described in the following section.

Demographic, physiological, and behavioral indices used to detect food limitation: Because of their small size, elevated metabolic rate, and limited capacity for storing energy reserves, sea otters must maintain a consistently high rate of food intake (Costa and Kooyman 1982), and consequently they are vulnerable to even small fluctuations in prey abundance. Individuals that are thin or emaciated are seen in any sea otter population, even one that is well below carrying capacity. The importance of food limitation at the population level depends on its overall effects on one or more demographic processes, birth or death rates, age of first reproduction, immigration, or emigration. Research suggests that food limitation acts primarily via changes in per-capita death rates (Estes et al. 1996). Moreover, the increased mortality associated with food limitation is usually not distributed equally among sex and age classes, but disproportionately affects very young and very old age classes and may also affect males more than females (Monson et al. 2000), especially when males are concentrated spatially in areas of poorer habitat quality (Bodkin et al. 2000). This age-specific pattern of mortality can be useful in diagnosing the status of different sea otter populations. One would expect that a population that has ceased to grow or started to decline because of resource limitation should exhibit increased mortality among juveniles and old adults, while prime-age adults (particularly females) should be buffered from such changes. Such a pattern can be tested for by monitoring the survival of individually marked animals over time, or by evaluating the age and sex structure of the living population or the death assemblage (e.g., Monson and DeGange 1995, Monson et al. 2000, Laidre et al. 2006, Tinker et al. 2006a). In either case, it is the change in survival or age structure over time that is most informative, rather than estimates at single points in time or space that are often difficult to interpret. Data sets on age-specific survival and age-frequency distributions from multiple populations over a range of relative densities (Laidre et al. 2006; USGS, unpublished data) provide a useful baseline for comparative studies.

In addition to its effects on demographic rates, resource limitation also may cause physiological and behavioral changes in sea otters, and those changes can provide powerful and robust indices of population status (Bodkin and Ballachey 1996). Body condition is one such parameter because a reduced rate of food intake leads to a decrease in fat reserves and muscle tissue, with the result that the body weight of individuals at a given length is significantly lower in food-limited populations (Monson et al. 2000, Laidre et al. 2006). Because of the relatively high degree of individual variation in body condition, as well as the impact of measurement error (especially in total body length, which can be affected by immobilization drugs) and by undetected pregnancy in females (mass-to-length ratio is obviously greater for pregnant females irrespective of food abundance), the utility of body condition as an index of population status depends on obtaining sufficient sample sizes, which can be both costly and difficult when sea otter densities are low. Nonetheless, a review of data on mass and length obtained from a wide range of sea otter populations suggests that the mass-to-length ratio is a reliable indicator of relative body condition and population status (Monson 2009).

A number of behavioral traits are also indicative of sea otter population status with respect to food resources. In particular, sea otters in food-limited populations tend to spend a high proportion of their time feeding (Estes et al. 1986, Garshelis et al. 1986, Ralls and Siniff 1990, Gelatt et al. 2002), experience a low rate of food acquisition while foraging (Estes et al. 1981, Garshelis et al. 1986, Dean et al. 2002), and exhibit a diverse diet at the population level (Estes et al. 1981). The size of some prey items consumed by otters is generally smaller in food-limited populations (Estes and Duggins 1995). Recent evidence from California suggests that individual otters are more likely to be dietary specialists in food-limited populations, and that increased diet diversity at the population level occurs as a result of increased inter-individual variation in diet composition (Estes et al. 2003a, Bentall 2005, Tinker et al. 2007). Each of these behavioral traits can be readily measured from most sea otter populations using simple observational techniques, although reliable estimates of percent time feeding and individual diet composition necessitate the radio-tagging of individual animals and collection of longitudinal data using telemetric techniques (Garshelis et al. 1986, Ralls and Siniff 1990, Gelatt et al. 2002, Tinker et al. 2006b). Recent technical advances allow for remote measurement of activity budgets, and possibly even diet specialization, using archival time-depth recorders (TDRs; Bodkin et al. 2007a, Tinker et al. 2007). In the case of some prey species such as sea urchins, changes in the size-frequency distribution of consumed prey can be measured from shell fragments collected from scats found at haul-out sites.

It should be emphasized that for all of the indices mentioned above - demographic, physiological, and behavioral parameters - a comparative approach is most effective for assessing population status. Comparisons can be made between different locations where food abundance is known unambiguously for at least one of the populations (Bentall 2005), or they can be temporal comparisons for one location at multiple points in time, corresponding to differing sea otter population densities. In either case, it is the dynamic patterns of change in these indices, rather than their absolute values, that are most informative.

The role of food limitation in the southwest Alaska population decline: existing evidence

Sampling programs for sea otter demography, body condition, activity budgets, diet, and prey abundance were ongoing in the central/western Aleutians during the early stages of the population decline. Data from telemetry-based studies of sea otter ecology at Amchitka Island (1992-94) and Adak Island (1995-96), when paired with direct measures of sea urchin abundance made before and during the decline, allow for a comparative analysis of most of the indices of food abundance described above. Although this comparison is limited spatially to one portion of the region affected by the decline (i.e., the central Aleutians), it nonetheless represents the best available means of evaluating the support for food limitation as a causal factor in the current decline.

For each parameter for which data exist, there have been significant changes over the course of the decline, but in each case these changes have been in the opposite direction from what would be expected if food were limiting (see Estes et al. 2004 for a detailed summary of these changes). Put another way, the consistent pattern that emerges is one of increased food availability over the course of the decline. For example, sub-tidal surveys in 1997 showed a five-fold increase in urchin biomass as compared to similar surveys in 1989 (Estes et al. 1998, Estes et al. 2004), a substantial increase in food resources considering that urchins comprise over 50% of sea otter diets in the Aleutian Islands (Estes 1990, Watt et al. 2000). The size (and thus energy content) of urchins consumed by otters also increased, a change that was already evident by 1995 at Adak Island based on observational data and collected scats (Tinker and Estes 1996). In addition to the increased urchin abundance, episodic spawning events of Pacific smooth lumpsuckers in the early 1990s resulted in subsidized winter diets at both Amchitka and Adak Islands (Watt et al. 2000). This sudden increase in prey availability was reflected by improved body condition, evident over the course of the 1992-94 study at Amchitka Island (Monson et al. 2000). By 1997, otters throughout the central Aleutians exhibited greater mass-length ratios than otters at Amchitka and Adak Islands in the 1960s and 1970s (populations suspected to be at or near carrying capacity), and by 2004 the average body condition had improved even further (Laidre et al. 2006). The age structure of the living population also differed from that of the 1970s, suggesting a release from the type of age-specific mortality associated with food limitation (i.e., higher death rates in very young and very old animals) and a shift toward age-independent mortality (Laidre et al. 2006). In contrast to previously documented sea otter declines associated with food limitation, where late-winter starvation resulted in many moribund and dead animals on the beach (Bodkin et al. 2000), beachcast carcasses were almost completely absent at Adak and Amchitka Islands (Estes et al. 2004). Data on activity budgets collected at Amchitka in the early 1990s were equivocal with regard to whether percent time foraging had decreased as compared to 1980s estimates (Estes et al. 1982, Gelatt et al. 2002), although the percent time feeding at Adak Island in 1995 was significantly lower (Tinker and Estes 1996). Although telemetry-based data on activity budgets are not available from later in the decline, observational data on feeding otters at Adak Island show that the average duration of a typical feeding bout had declined from 180 minutes in the early 1990s to only 55 minutes in 2005, and daytime observational data indicate that the total percent time feeding had decreased by an equivalent amount (USGS, unpublished data). Diet diversity at Adak Island also decreased between 1995 and 2005, with urchins dominating the diet (USGS, unpublished data).

Conclusions and future directions

Based on all available data, it seems that the cause of the population decline in the central and western Aleutian Islands was unrelated to the abundance of prey resources. In fact, it appears that as a by-product of the sea otter decline, there is a virtually limitless food supply for the remaining animals. It would be of interest to determine whether this same pattern holds more broadly throughout the region affected by the decline, for example, in the eastern Aleutian Islands and the Alaska Peninsula where relevant data are lacking. However, even in the absence of such data, certain geographic characteristics of the decline – in particular, the degree of temporal synchrony across a wide range of different habitats – seem to be at odds with almost any conceivable scenario of food limitation. To invoke food limitation as a causal factor in other areas of southwest Alaska, one would need to explain simultaneous declines of prey species at many different islands, in both rocky and soft-sediment habitats, independent of initial otter density at the start of the decline. Given the documented examples of prey increases in the central part of the affected region, such a scenario seems extraordinarily unlikely.

Although food abundance likely has not been a factor in the recent decline, there are some conceivable scenarios in which prey abundance could act to limit future recovery. In particular, to the degree that predation by killer whales has altered sea otter behavior and habitat use patterns (see section 3.A.), those populations of sea otters that persist into the future may use only small fragments of their original range (i.e., those habitat areas offering some degree of refuge from killer whale predation). In that case, the effective carrying capacity at a given location would be considerably reduced. For example, if sea otter feeding habitat at a particular island were effectively reduced to 10% of its initial size as a result of altered sea otter behavior (e.g., sea otters only residing in shallow, protected coves), then the total population size that could be sustained would decrease by an equivalent amount. Under that scenario, food limitation would become a significant factor at relatively low overall sea otter densities. Although such a scenario is entirely speculative at present, it could become a significant issue for recovery and delisting in the future, and so it warrants investigation. Fortunately, it should be relatively straightforward to detect such a pattern through a monitoring program that makes use of a combination of the direct and indirect indices described above. It is recommended that such a monitoring program be incorporated into future recovery activities.

G. Disturbance

Since sea otters are slow swimmers relative to other marine mammals and spend much of their time at the surface resting, grooming, and nursing their young, they would appear to be highly vulnerable to disturbance by boats. However, there have been few systematic studies of behavioral responses to disturbance in sea otters, and there have been no studies of the effects of disturbance on stress levels, energy expenditures, foraging efficiency, or reproductive success. In the only published study of its type, Curland (1997) reported that sea otters in areas of disturbance by power boats, divers, and kayaks engaged in significantly greater amounts of travel than they did in areas without disturbance. It appears clear from anecdotal reports that the reaction of sea otters to disturbance: 1) is highly variable between seasons, sexes, and populations; and 2) may be modified by experience (reactions often decline in intensity with habituation, and may increase where populations are harassed or hunted). While the described responses to disturbance typically include diving and/or traveling away from the source of disturbance, in areas where sea otters rest, groom, and/or nurse in "rafts," disturbance frequently also causes the animals to disperse and the raft to break up and not reform for many hours (J. Watson, personal communication, 2008).

There is very little boat traffic in southwest Alaska, hence the impact of disturbance is likely to be very small. Similarly, injury by boat strikes is likely to be very rare. Indeed, even in British Columbia where sea otters experience boat traffic much more frequently than in southwest Alaska, the Canadian Sea Otter Recovery Team concluded that disturbance is "unlikely to be significant at this time" (L. Barrett-Lennard, personal communication).

H. Bycatch and entanglement in debris

Sea otters may encounter a variety of fixed and mobile fishing gear throughout their range. The potential exists for them to entangle in the webbing of nets or become trapped inside submerged pots or traps that they enter to consume the catch or bait. This is called bycatch, and it has been monitored by State and Federal observers in some, but not all, of the fisheries that might take sea otters. U.S. commercial fishermen are required to report their takes of sea otters and other marine mammals to the National Marine Fisheries Service (NMFS) as a condition of their Marine Mammal Exemption Permit required by the MMPA. Data from these programs suggest the frequency of lethal entanglement of sea otters depends on the gear involved, nature of the entanglement, experience of the otter, and the intervention (or not) of attending fishermen. Incidental mortality in commercial fisheries is currently infrequent; the NMFS 2006 List of Fisheries cites only two U.S. fisheries with lethal bycatch of sea otters. However, within the range of this DPS, there are a number of fisheries that have the potential to take otters but have had little or no observer coverage. Those fisheries include coastal gillnet fisheries along the southern Alaska Peninsula and in Bristol Bay and pot fisheries for shellfish (Angliss and Outlaw 2008).

The majority of documented sea otter bycatch in U.S. fisheries has occurred in fixed gear set in nearshore waters. In California, hundreds of sea otters entangled and drowned between 1976 and 1984 in halibut gill and trammel nets set on the ocean bottom in shallow coastal waters (Wendell et al. 1985). In an effort to reduce bycatch of sea otters and other species, a series of depth restrictions were placed on that fishery, and after 1991 set gillnets were restricted to waters deeper than 55 m (Forney et al. 2001). Although this action was followed by an initial decline in sea otter mortality (Estes 1990), subsequent changes in fishing effort and sea otter distribution again led to an estimated 17-125 sea otter deaths from 1995-1998 in the Monterey Bay area (Forney et al. 2001). Subsequently in the early 2000s, this fishery was moved further offshore (to depths of 110 meters or more) along much of the area encompassing the southern sea otter range.

Sea otters are also known to enter and drown in submerged fish and shellfish traps ("pots"). Four California sea otters were found dead in rock crab, lobster, and experimental pots set in coastal California waters between 1987 and 1991 (B. Hatfield, personal communication). Estes et al. (2003b) noted that increased California sea otter mortality from 1995-99 coincided with increased use of pots in a shallow-water live-fish fishery. Tests have shown that captive otters are capable of entering those traps (Estes et al. 2003), but later experimental results have shown that modifications to reduce the size of the trap entrances could nearly eliminate sea otter bycatch without reducing target species catch (Hatfield et al. in review).

Potential for bycatch of southwest Alaska sea otters

<u>Groundfish gear:</u> In Alaska, NMFS observers are placed on a portion of groundfish vessels using pots, trawls, and longlines to fish in federal

waters within the Gulf of Alaska, Bering Sea, and Aleutian Islands. In 1992, an observer documented the bycatch of eight sea otters in blackcod pots set illegally in nearshore Aleutian Islands waters closed to pot fishing (Perez 2003). Pots used in the groundfish fishery are converted tanner crab pots in which fingerlike obstructions are placed on the inside of the entrance to prevent fish escape. Blackcod pots are generally fished in deeper waters and fishing is illegal in shallow waters that include sea otter habitat. No other sea otter bycatch was documented by observers in this or other Alaskan groundfish fisheries through 2003 (Perez in prep). In 1997, a fisherman's self-report documented the retrieval of a dead otter in a Bering Sea/Aleutian Island groundfish trawl but it was unclear whether the otter was dead or alive prior to entering the net.

Salmon Gillnets: Since 1988, observer programs have documented marine mammal interactions in six commercial salmon gillnet fisheries operating in coastal Alaskan waters: PWS setnet (1990), PWS driftnet (1988-1991), South Alaska Peninsula driftnet (1990), Cook Inlet setnet and driftnet (2000), and Kodiak Island setnet (2002, 2005) fisheries. Set gillnets ("setnets") are anchored in place and extend seaward from their attachment points on shore. Drift gillnets ("driftnets") are fished while attached to and tended by a drifting vessel. These salmon gillnet fisheries are managed by the Alaska Department of Fish and Game as limited-entry fisheries (i.e., the maximum number of participants is limited) and the amount, timing, duration, and distribution of seasonal fishing effort is regionally managed on a real-time basis. Alaskan salmon gillnets are made with multifilament line, set at and suspended from the water's surface and limited to 272-364 m in length.

Four of the observed salmon fisheries occur outside the range (to the east) of the southwest Alaska DPS. In PWS, sea otters swam within 10 m of 2.0-6.3% of observed set and drift gillnets but became entangled in <0.25% of 9,428 sets observed from 1988-1991 (Wynne 1990, Wynne et al. 1991, 1992). The otters that became entangled were either able to free themselves or were extricated by the attending fisherman; none of the observed entanglements resulted in sea otter injury or death. However, beachcast sea otter carcasses were recovered on the adjacent Copper River Delta during those years with bullet holes and skull fractures, suggesting that the fate of live entangled sea otters depends on the actions of the fishermen who may or may not make efforts to release them unharmed (Wynne 1990). No sea otter entanglements were observed in Cook Inlet set or drift gillnets observed during the Alaska Marine Mammal Observer Program in 2000 (B. Mansfield, personal communication).

Two observer programs have monitored sea otter interactions with salmon gillnet fisheries conducted within the range of the southwest Alaska DPS. Observers in the South Alaska Peninsula (South Unimak) drift gillnet fishery operating near False Pass documented sea otters swimming within 10 m of 2.1% of 373 observed sets but none became entangled (Wynne et al.1991). In Kodiak, sea otters were frequently seen in the vicinity of set gillnets but were observed to entangle in only 4 sets in 2002 (Manly et al. 2003) and 1 set in 2005 (B. Mansfield, personal communication.). Two of the sea otter entanglements in 2002 and the sole entanglement in 2005 involved "momentary snags" from which the otters were able to escape unassisted and unharmed. The two entangled otters observed in 2002 were released unharmed with human assistance. However, one unobserved sea otter death was reported by a Kodiak setnet fisherman through a NMFS self-report in 2002. The Kodiak setnet fishery is now the only Alaskan fishery listed on the NMFS 2006 List of Fisheries as having lethal sea otter bycatch.

These observer data suggest that the potential exists for sea otters from the southwest Alaska DPS to entangle in other commercial salmon gillnet fisheries that have not been observed, i.e., Bristol Bay set and driftnets and Alaska Peninsula setnets. In addition, an unknown number of gillnets are set by personal-use, or subsistence, salmon fishermen in coastal waters throughout the range of this DPS. Because these fisheries all use surface-hanging gillnets, the aforementioned observer data (Wynne 1990, Wynne et al. 1992) suggest that sea otters that become entangled in these nets are often able to extricate themselves or otherwise remain afloat and available for release by attending fishermen. Based on the geographic scope of these activities, it is unlikely that unobserved commercial and personal-use gillnet fisheries currently pose a significant threat of incidental injury or mortality to sea otters in southwest Alaska.

Herring gillnets: Limited-entry herring gillnet fisheries occur in the Kodiak, Alaska Peninsula-Aleutian Islands, and Bristol Bay Management Areas. Gillnets may be either anchored or allowed to drift but must be constantly tended. Herring gillnets use a mesh size of 2¼ to 2½ inches (5.4- 6.3 cm) and range in length from 50-150 fathoms (91-273 m). They are suspended from a floatline but may be fished with the floatline and floats below the surface of the water.

Unlike the submerged gill and trammel nets used in the California halibut fishery, Alaskan gillnets are short and left untended only briefly, if at all. Most importantly, they are set at or near the water surface, allowing sea otters to remain at the surface until released. The mandated use of floating nets and the prohibition on sink gillnets in Alaskan fisheries limit the likelihood of incidental mortality of sea otters in Alaskan gillnets.

Crab pots: King, tanner, and snow crabs are commercially harvested in southwest Alaska, but most of these crab species inhabit waters that are deeper and/or further offshore than sea otters typically forage (Newby 1975) and the potential for sea otter entanglement is low. Accordingly, reports of sea otter bycatch in these fisheries are rare: one otter reportedly drowned in a king crab pot set in 100 m of water in the Aleutian Islands in 1975 (Funk 2003). Even if sea otters encounter pots used to harvest these species, the gear's design may limit lethal entrapment somewhat. Tanner crab pots must have a tunnel opening diameter of ≤ 5 inches (12.7 cm), which would allow entry by juvenile sea otters but would likely exclude adults (Hatfield et al. in review). Conversely, red king crab pots are required to have an unobstructed tunnel opening diameter of ≥ 5 inches (12.7 cm); based on observations of captive otters, they would likely enter such traps and some might escape while others might not (Hatfield et al. in review).

Dungeness crabs inhabit coastal waters of southwest Alaska, primarily in depths of less than 50 m (Funk 2003). They are harvested with pots by both commercial and personal-use fishermen in Kodiak, North and South Alaska Peninsula, and Aleutian Islands districts. Dungeness pots must have rigid tunnel openings that can be up to 9.5 inches (24.1 cm) in diameter, which is large enough for a sea otter of any age to enter (Hatfield et al. in review). There have been no observer programs for the dungeness fishery, and therefore any interactions would have to have been self-reported. Funk (2003) reported no incidental take of sea otters in this fishery, but subsequently there has been one report of a sea otter drowned in a dungeness pot in southeast Alaska (Hatfield et al. in review).

Potential for debris entanglement of southwest Alaska sea otters

Entanglement in marine debris has been identified as a significant contributor to mortality of northern fur seals in the Pribilof Islands (Fowler 1985, Swartzman et al. 1990, Fowler et al. 1994, Zavadil et al. 2003). Debris materials most commonly found to entangle fur seals include rubber packing bands and trawl net fragments. Despite the potential for encountering similar ensnaring debris throughout their range, sea otters are rarely seen entangled in marine debris, perhaps due to behavioral or anatomical differences between them and fur seals. Entanglement in marine debris is not currently considered a threat to the southwest Alaska DPS. However, there has been no directed effort to look for entangled otters so this assessment should be verified.

I. Subsistence harvest

Harvest of marine mammals by Alaska Natives is authorized under Section 101(b) of the MMPA, provided the taking is for subsistence purposes or for the purpose of creating and selling authentic native articles of handicrafts and clothing, and is not wasteful. Such taking cannot be further restricted unless the stock has been designated as depleted. A similar exemption for species listed as threatened or endangered exists under Section 10(e) of the ESA. The Secretary of the Interior may prescribe regulations to limit the taking of species or stocks listed as depleted, threatened, or endangered if such taking has been determined to have a material and negative effect.

The subsistence harvest of sea otters, polar bears (*Ursus maritimus*), and Pacific walruses (*Odobenus rosmarus divergens*) is monitored by FWS through a marine mammal marking, tagging, and reporting program (MTRP). The MTRP was established in 1989 in accordance with Section 109(i) of the MMPA. Implementing regulations for the MTRP require hunters to present hides and skulls of sea otters to an authorized FWS representative, known as a "tagger," within 30 days of harvest. The MTRP currently has sea otter taggers in 48 communities statewide, with 16 in southwest Alaska.

In addition to attaching tags to the hide and skull, taggers collect biological information about the otter (age class and sex) as well as about the harvest itself (date and location) and attach uniquely numbered plastic tags to the hide and skull. The information is entered into a computer database.

As there are no other assessments of sea otter harvest levels, compliance with the MTRP cannot be evaluated, but it likely varies over time and location. Because some animals may not be tagged, harvest estimates from MTRP data should be considered minimum values. For the remainder of this section, results derived from the MTRP are referred to as "reported" subsistence harvest.

In addition to non-compliance (i.e., tags not being applied to hides and/or skulls), erroneous information may be recorded inadvertently on the tagging certificate. For example, molecular genetic analysis indicates that 12% of sea otters (from a sample of 138) were reported as the wrong sex (Scribner et al. 2005). Most of the errors (13 of 17) consisted of males being identified incorrectly as females, resulting in a net 7% overestimate of the number of females harvested in that sample.

The reported subsistence harvest from the southwest Alaska DPS is the lowest of the three stocks of sea otters in Alaska, averaging 89 per year (range 23-180) during 1989-2008 (Table 7). This result is not surprising since there are few Alaska Native villages within the range of the DPS. The harvest has primarily been composed of adults (91%), most of which were males (Table 8).

Within the southwest Alaska DPS, the majority (82%) of the subsistence harvest comes from the Kodiak, Kamishak, Alaska Peninsula MU (Table 9). With the exception of 42 otters taken from the Chignik area, virtually all of the 1,775 animals taken from this unit during 1989-2008 were harvested in the Kodiak archipelago. As there is no permanent human habitation along the Alaska Peninsula east of Chignik, or in Kamishak Bay, it is not surprising that the majority of the subsistence harvest from this MU occurs around Kodiak. Subsistence harvest from the Western Aleutian and Eastern Aleutian MUs are very small; Adak is the only location within the Western Aleutian MU where sea otters have been taken.

To put these harvest levels in context, the most recent estimates of sea otter abundance from the Aleutian Islands range from 3,311 to 8,742 (Estes et al. 2005, Doroff et al. 2003). The annual reported harvest for both Aleutian MUs combined averages 1.7 otters, which amounts to 0.02–0.05% of the population. The annual reported harvest from the Bristol Bay MU averages 6.4 otters, or 0.06% of the estimated population of 11,303. In the southern Alaska Peninsula MU, the reported average annual harvest of 9.6 otters is 0.21% of the estimated population of 4,682 (Burn and Doroff 2005, FWS unpublished data). As expected, the reported average annual harvest of 82.3 otters from the Kodiak, Kamishak, Alaska Peninsula MU is proportionally the largest, accounting for 0.36% of the estimated population size of 22,957 (Burn and Doroff 2005, FWS unpublished data, USGS unpublished data). As noted above, the majority of this harvest comes from the Kodiak archipelago, where it accounts for 0.67% of the regional population.

The geographical scale at which sea otter populations are managed remains an important, although largely unexplored, issue. As noted by Gorbics and Bodkin (2001), the annual harvest rate during the commercial fur trade of the 18th and 19th centuries averaged only about 1.5% of the global sea otter population per year. Yet over time, range-wide reductions and extirpations occurred not because of excessive harvest, but because the harvest was not allocated proportionally to the abundance and distribution of sea otters. This resulted in the serial depletion of otters, beginning in the western Pacific and systematically expanding eastward across the Aleutian archipelago and southward along North America, as harvested populations became either reduced to unprofitable densities or locally extinct. The process of serial depletion is facilitated by the relatively sedentary nature of sea otters. Annual home range sizes of adult sea otters are relatively small, with male territories ranging from 4-11 km^2 and adult female home ranges from a few to 24 km^2 (Garshelis and Garshelis 1984, Ralls et al. 1988, Jameson 1989, Ballachey and Bodkin 2006). When mortality is spatially concentrated in areas equal to or smaller than the cumulative home range of the managed population, local depletion, potentially leading to serial depletion, may occur. Therefore, it is essential to consider the spatial scales at which sea otter subsitence harvest is managed.

Although the reported harvests represent relatively small proportions of the current estimated population sizes, it will be prudent to evaluate the potential consequences of future harvests quantitatively, particularly in the event that populations continue to decline and harvests do not. Age- and sex-specific population matrix models that incorporate present and projected population sizes, trajectories, and harvests should be constructed to consider how changes in abundance and harvests may interact over time and influence recovery. Because population abundance, trend, and harvest levels vary within the DPS, independent models should be developed that incorporate geographic variation and can guide management at appropriate geographic scales.

J. Habitat concerns

Sea otter habitat within the range of the southwest Alaska DPS appears, at least superficially, to be nearly pristine, with little obvious anthropogenic degradation. The resident human population is small and located mostly in the eastern part of the range in small towns and villages. Historically, the military had a significant presence at a number of sites throughout southwest Alaska but it currently has only a minor presence.

Developments that physically modify sea otter habitat are limited to nearshore waters immediately adjacent to towns, villages, and military bases, and are usually in the form of docks, piers, and boat harbors. Sea otters continue to use these sites. In fact, currently some of the highest densities of animals in the Aleutian Islands occur in the vicinity of Unalaska and Adak Islands, both of which are inhabited. The shoreline and nearshore waters throughout most the sea otter range should remain relatively free of such development, as much of these areas are within Federal and State refuges, parks, preserves, and sanctuaries.

At present, sea otter abundance within the range of the southwest Alaska DPS appears to be far below the carrying capacity of the habitat (Burn et al. 2003) and preferred prey, such as sea urchins, are abundant (Watt et al. 2000, Estes et al. 2004). The loss of extensive kelp beds due to intensive urchin grazing (Estes et al. 2004) has reduced sea otter resting habitat, but this is not known to have affected sea otter population dynamics. Recent estimates of sea otter growth rates, asymptotic (maximum) values of body mass and length, body condition, and age composition were all indicative of a population below nutritional carrying capacity (Laidre et al. 2006).

Current or foreseen commercial fishing practices, with the exception of the dungeness crab and sea urchin fisheries in the Kodiak Island area, do not target important sea otter prey and are not known to affect sea otter habitat in any obvious way. Some fishing for yellowfin sole (Limanda aspera)

Table 7. Reported sea otter subsistence harvest in Alaska, 1989-2008.
Data are from the U.S. Fish and Wildlife Service Marine Mammal Marking, Tagging, and Reporting Program.

Year	Southwest Alaska	Southcentral Alaska	Southeast Alaska	Unknown	Total
1989	61	61	153	5	280
1990	55	26	84	1	166
1991	23	92	114	1	230
1992	57	138	416	3	614
1993	180	200	832	4	1,216
1994	64	425	315	0	804
1995	56	374	198	2	630
1996	156	321	125	3	605
1997	149	274	332	0	755
1998	68	450	359	2	879
1999	74	219	328	1	622
2000	110	305	396	1	812
2001	80	221	350	0	651
2002	94	236	333	2	665
2003	126	264	382	2	774
2004	87	429	293	0	809
2005	111	443	350	2	906
2006	67	359	275	0	701
2007	64	192	446	2	704
2008	93	305	234	0	632
Total	1,775	5,334	6,315	31	13,455
Mean	88.8	266.7	315.8	1.6	672.8
Percent of statewide total	13.2	39.6	46.9	0.2	100.0

Table 8. Age and sex composition of the reported sea otter subsistence harvest from the southwest Alaska DPS of the northern sea otter, 1989-2008.

Data are from the U.S. Fish and Wildlife Service Marine Mammal Marking, Tagging, and Reporting Program. F=Female, M=Male, U=Unknown sex. Results are not corrected for possible errors in sex identification.

	Adult			Subadult			Pup			Unknown			All Ages		
Year	F	M	U	F	M	U	F	M	U	F	M	U	F	M	U
1989	6	21	19	0	0	0	1	1	1	1	0	11	8	22	31
1990	2	42	1	2	8	0	0	0	0	0	0	0	4	50	1
1991	0	19	2	1	1	0	0	0	0	0	0	0	1	20	2
1992	8	38	7	0	3	0	0	1	0	0	0	0	8	42	7
1993	83	65	16	4	2	2	0	0	0	0	0	8	87	67	26
1994	6	23	19	4	8	4	0	0	0	0	0	0	10	31	23
1995	5	48	1	1	1	0	0	0	0	0	0	0	6	49	1
1996	29	94	18	6	6	3	0	0	0	0	0	0	35	100	21
1997	24	84	21	5	7	2	1	1	4	0	0	0	30	92	27
1998	5	36	20	5	0	1	0	0	1	0	0	0	10	36	22
1999	5	49	4	5	5	1	1	3	1	0	0	0	11	57	6
2000	23	26	44	2	6	6	0	1	2	0	0	0	25	33	52
2001	19	43	1	4	9	2	0	0	2	0	0	0	23	52	5
2002	20	62	4	1	4	0	1	2	0	0	0	0	22	68	4
2003	24	76	4	8	6	2	1	3	2	0	0	0	33	85	8
2004	8	68	2	1	2	3	1	2	0	0	0	0	10	72	5
2005	20	69	2	5	9	0	2	3	1	0	0	0	27	81	3
2006	12	51	1	0	2	1	0	0	0	0	0	0	12	53	2
2007	12	44	0	1	3	1	0	1	2	0	0	0	13	48	3
2008	13	70	1	2	1	0	0	0	0	2	4	0	17	75	1
Total	324	1,028	187	57	83	28	8	18	16	3	4	19	392	1,133	250
		1,539			168			42			21			1,775	

Table 9. Reported subsistence harvest of sea otters from the southwest Alaska DPS of the northern sea otter by MU, 1989-2008.
Data are from the U.S. Fish and Wildlife Service Marine Mammal Marking, Tagging, and Reporting Program. One harvested sea otter could not be assigned to a MU.

Year	Western Aleutians	Eastern Aleutians	Bristol Bay	South Alaska Peninsula	Kodiak, Kamishak, Alaska Peninsula	Total
1989	0	1	8	17	35	61
1990	0	0	2	25	28	55
1991	0	0	1	0	22	23
1992	1	1	1	10	44	57
1993	0	15	5	5	155	180
1994	0	2	0	9	53	64
1995	0	1	11	1	43	56
1996	0	0	11	0	145	156
1997	0	1	6	18	124	149
1998	0	0	5	3	60	68
1999	0	0	4	10	60	74
2000	5	0	7	5	93	110
2001	0	0	0	14	66	80
2002	0	0	5	13	76	94
2003	0	0	18	12	96	126
2004	0	1	1	6	78	86
2005	1	0	13	0	97	111
2006	1	0	2	7	57	67
2007	0	0	14	7	43	64
2008	0	0	0	8	85	93
Total	8	22	114	170	1,460	1,774
Mean	0.5	1.3	6.7	10.0	85.9	104.4
Percent	0.5	1.2	6.4	9.6	82.3	

overlaps with the range of sea otters in the Bristol Bay MU, but the trawls used are designed to have minimal contact with the bottom and therefore are likely to have little impact on sea otter prey or their habitat (B. Wilson, personal communication)

Abnormally high levels of OC pesticides and PCBs were found in sea otter tissues from the Aleutian Islands (Estes et al. 1997, Bacon et al. 1999). Probable sources of PCBs in the Aleutian Islands are atmospheric and oceanic transport from Asia and leakage from dump sites at military bases (Bacon et al. 1999). More recent data indicate that high levels of PCBs are limited to a few local areas, including Unalaska, Adak, and Amchitka Islands, and presumably originated from military activities. It is not thought that the widespread declines in pinniped and sea otter populations are related to these contaminants as levels are higher in other regions with stable or increasing populations (Reese 1998).

Changes in climatic conditions, due to both "normal" climate variability (Hunt and Stabeno 2005) and global warming (Schumacher and Kruse 2005), are expected to modify both the physical environment and the biota within the range of the southwest Alaska DPS. It is difficult to predict how climate change will affect sea otter recovery. Therefore, it will be important to monitor relevant indices and to evaluate changes in regard to sea otter ecology and population dynamics. It is possible that global warming will have more of an impact on sea otters at the southern end of their range, but this expectation should be tempered by the realization that changes in climate and atmospheric conditions can influence ecosystems in many complex ways. For example, ccean acidification is a consequence of rising atmospheric CO_2 levels (Solomon et al. 2007), which reduces the concentration of carbonate ions. In turn, decreasing carbonate ions may affect the ability of sea otter prey species such as bivalves, snails, and crabs to form exoskeletons (Green et al. 2004).

K. Illegal take

In addition to the legal harvest of sea otters by Alaska Natives authorized by the MMPA, there may also be illegal take from the southwest Alaska DPS. Otters may be taken: 1) illegally by non-Natives for their valuable pelts; 2) wastefully by Alaska Natives; or 3) to reduce competition for sea otter prey resources such as crab and shellfish. As human habitation within the range of the DPS is sparse (and in many places non-existent), the extent to which sea otters may be taken illegally is unknown.

Investigations of a sea otter mortality event in the Kodiak archipelago in 1987 documented the carcass of at least one male sea otter that had been shot (DeGange and Vacca 1989). Intact carcasses of sea otters that have been shot are tacit evidence of illegal take, whether shot by a non-Native or an Alaska Native and not retrieved (and therefore a wasteful take in accordance with Federal regulations). Most of the otter carcasses investigated by DeGange and Vacca (1989) were in advanced stages of decomposition with evidence of scavenging, making conclusive determination of the cause of death difficult. In a study of sea otter reproduction and survival in the Kodiak archipelago from 1986-1990 (Monson and DeGange 1995), two instrumented animals were determined to have been killed by gunshot, one of which still had the pelt when discovered (D. Monson, personal communication).

The FWS's Office of Law Enforcement (OLE) has no records of unlawful harvest of sea otters within the range of the southwest Alaska DPS. The information it does have comes from other areas where otters occur in Alaska. That information is offered both as factual, based on current and past investigations, and anecdotal, based on interviews of witnesses. Whether or not similar unlawful activities are occurring with the listed population is impossible to say.

The OLE has investigated 14 cases of illegal take and/or illegal sale of sea otters in Alaska from 2000 to 2006. Seventy-six otters or parts of otters were seized during these investigations and 20 violations were documented. The defendants were both Alaska Natives and non-Natives. Although the OLE has not documented large-scale unlawful activity to date, sea otters have not received the investigative attention that many other species have. Therefore, the occurrence or extent of unlawful take in many areas is unknown (S. Oberholtzer, personal communication).

In addition to the above information provided by the OLE, the managers of each National Wildlife Refuge within the range of the DPS and the Alaska State Troopers Bureau of Wildlife Enforcement were also contacted. None of these additional sources had any records of illegal take of sea otters from the DPS. Although illegal take may have occurred and may occur in the future, it is likely that the numbers are small and any impact on the recovery of the DPS would be minimal.

4. Threats Analysis

The importance of potential threats to the existence and recovery of southwest Alaska sea otters was analyzed based on the description of threats in section 3, published literature, and the experience and professional judgement of Recovery Team members. Threats were organized according to the five ESA listing factors, and were evaluated based on the factors listed below.

- Potential impact—the amount of effect the threat could have on the population in an area where the threat occurs.
- Geographical scope—the geographical extent of the threat across the MU being considered.
- Likelihood—the likelihood that the threat will occur within the next 10 years.
- Level of confidence—the degree of confidence in the assessment of the previous three factors.
- Importance to recovery—an overall assessment of how much the threat could affect recovery.
- Management potential—an estimate of the likelihood that the threat could be managed to reduce or eliminate its impact.

Threats analyses for each MU are shown in Tables 10-14. Table 15 shows the importance to recovery and management potential of all threats for all management units, and these considerations are further discussed below.

Habitat Loss – low importance

Habitat loss as a threat to recovery was ranked low for all MUs. The physical habitat is largely unspoiled throughout the vast majority of the range of the southwest Alaska DPS. The human population in this area is small, and development has been limited to the few, widely scattered towns, villages, and military installations. The shoreline and near-shore waters throughout most of the region should remain relatively free of habitat modification in the future due to their protection status; much of the sea otter habitat is within Federal and State refuges, parks, preserves, and sanctuaries. The highest-density human population and highest degree of habitat modification are in the eastern portion of the DPS's range, where sea otter numbers do not appear to have declined.

While in general terms loss of habitat can have a severe impact on sea otter populations, development that could have an impact in this region is not likely except on a highly localized scale. Management potential is high because of a) the section 7 review process required under the ESA and b) state and federal land-use regulations.

Other factors, in addition to development, that could have an impact on habitat used by the southwest Alaska DPS are atmospheric change and climate warming, either through changes in physical habitat attributes, ocean chemistry, or the prey base. However, the Team concluded that within the 10-year time window used in this evaluation, atmospheric change and climate warming are not likely to exert an important influence on the amount or suitability of sea otter habitat.

Oil Spills – low to moderate importance

Sea otters are particularly vulnerable to oil spills because of the risks from: 1) fouling of their highly insulative fur; 2) toxicity of oil ingested during grooming; 3) ingestion of toxic hydrocarbons stored in benthic invertebrate prey; and 4) the fact that they often form large aggregations, creating the potential for large numbers of animals to be exposed to the effects of a spill. Currently, large, bulk oil tanker traffic within the range of the southwest Alaska DPS is infrequent (22 voyages/yr) so most of the potential harm from oil spills comes from fuel supplies aboard freighters and fishing vessels. Many of those vessels use diesel fuel, which is less toxic and disperses and evaporates much more rapidly than crude oil. Most spills would probably be relatively small and have limited local impacts. Fuel storage facilities in ports throughout the range of the DPS are also a potential source of localized spills. Due to the large linear extent of the DPS, even a large spill from a crude oil tanker would be unlikely to affect a substantial proportion of the overall sea otter population. This relationship between the likelihood and geographic extent of oil spills (i.e., small spills with local impacts are more likely than large spills with widespread impacts) is reflected in the threats analysis tables (Tables 10-14).

Oil spills were rated as a moderate threat to the recovery of two MUs: Eastern Aleutian Islands

Table 10. Threats analysis for the Western Aleutian Islands MU of the southwest Alaska DPS of the northern sea otter.

Listing Factor	Threat	Potential Impact	Geographic Scope	Likelihood	Level of Confidence	Importance to Recovery	Management Potential
A. Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range	Habitat Loss	High	Local	Not Likely	High	Low	High
	Oil Spills	High	Local/ Widespread	Very/Not Likely	High	Low	High
	Prey Base	High	Local	Not Likely	High	Low	High
B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes	Subsistence Harvest	Moderate	Local	Somewhat Likely	High	Low	High
C. Disease or Predation	Infectious Disease	Moderate	Local/ Widespread	Not Likely	Low	Low	Low
	Predation	High	Entire Unit	Very Likely	High	High	Low
D. Inadequacy of Existing Regulatory Mechanisms	N/A						
E. Other Natural or Anthropogenic Factors Affecting its Continued Existence	Fishery Bycatch	Moderate	Local	Not Likely	Moderate	Low	High
	Illegal Take	High	Local	Not Likely	Moderate	Low	Low
	Disturbance	Low	Local	Not Likely	High	Low	Moderate
	Biotoxins	Moderate	Widespread	Not Likely	Low	Low	Low
	Point-Source Contaminants	Low	Local	Very Likely	High	Low	Low
	Non-Point-Source Contaminants	Low	Entire Unit	Very Likely	High	Low	Low

Table 11. Threats analysis for the Eastern Aleutian Islands MU of the southwest Alaska DPS of the northern sea otter.

Listing Factor		Threat	Potential Impact	Geographic Scope	Likelihood	Level of Confidence	Importance to Recovery	Management Potential
A.	Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range	Habitat Loss	High	Localized	Not Likely	High	Low	High
		Oil Spills	High	Local/ Widespread	Very/Somewhat Likely	High	Moderate	Moderate
		Prey Base	High	Local	Not Likely	High	Low	High
B.	Overutilization for Commercial, Recreational, Scientific, or Educational Purposes	Subsistence Harvest	Moderate	Local	Somewhat Likely	High	Low	High
C.	Disease or Predation	Infectious Disease	Moderate	Local/ Widespread	Not Likely	Low	Low	Low
		Predation	High	Entire Unit	Very Likely	Moderate	High	Low
D.	Inadequacy of Existing Regulatory Mechanisms	N/A						
E.	Other Natural or Anthropogenic Factors Affecting its Continued Existence	Fishery Bycatch	Moderate	Local	Not Likely	Moderate	Low	High
		Illegal Take	High	Local	Not Likely	Moderate	Low	Low
		Disturbance	Low	Local	Not Likely	High	Low	Moderate
		Biotoxins	Moderate	Widespread	Not Likely	Low	Low	Low
		Point-Source Contaminants	Low	Local	Very Likely	High	Low	Low
		Non-Point-Source Contaminants	Low	Entire Unit	Very Likely	High	Low	Low

Table 12. Threats analysis for the Bristol Bay MU of the southwest Alaska DPS of the northern sea otter.

Listing Factor	Threat	Potential Impact	Geographic Scope	Likelihood	Level of Confidence	Importance to Recovery	Management Potential
A. Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range	Habitat Loss	High	Local	Not Likely	High	Low	High
	Oil Spills	High	Local/ Widespread	Very/Not Likely	High	Low	Moderate
	Prey Base	High	Local	Not Likely	High	Low	High
B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes	Subsistence Harvest	Moderate	Local	Somewhat Likely	High	Low	High
C. Disease or Predation	Infectious Disease	Moderate	Local/ Widespread	Not Likely	Low	Low	Low
	Predation	High	Entire Unit	Somewhat Likely	Moderate	Moderate	Low
D. Inadequacy of Existing Regulatory Mechanisms	N/A						
E. Other Natural or Manmade Factors Affecting its Continued Existence	Fishery Bycatch	Moderate	Widespread	Somewhat Likely	Moderate	Low	High
	Illegal Take	High	Local	Not Likely	Moderate	Low	Low
	Disturbance	Low	Local	Not Likely	High	Low	Moderate
	Biotoxins	Moderate	Widespread	Not Likely	Low	Low	Low
	Point-Source Contaminants	Low	Local	Very Likely	High	Low	Low
	Non-Point-Source Contaminants	Low	Entire Unit	Very Likely	High	Low	Low

Table 13. Threats analysis for the South Alaska Peninsula MU of the southwest Alaska DPS of the northern sea otter.

Listing Factor	Threat	Potential Impact	Geographic Scope	Likelihood	Level of Confidence	Importance to Recovery	Management Potential
A. Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range	Habitat Loss	High	Local	Not Likely	High	Low	High
	Oil Spills	High	Local/ Widespread	Very/Not Likely	High	Low	Moderate
	Prey Base	High	Local	Not Likely	High	Low	High
B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes	Subsistence Harvest	Moderate	Local	Somewhat Likely	High	Low	High
C. Disease or Predation	Infectious Disease	Moderate	Local/ Widespread	Not Likely	Low	Low	Low
	Predation	High	Entire Unit	Very Likely	Moderate	High	Low
D. Inadequacy of Existing Regulatory Mechanisms	N/A						
E. Other Natural or Anthropogenic Factors Affecting its Continued Existence	Fishery Bycatch	Moderate	Widespread	Somewhat Likely	Moderate	Low	High
	Illegal Take	High	Local	Not Likely	Moderate	Low	Low
	Disturbance	Low	Local	Not Likely	High	Low	Moderate
	Biotoxins	Moderate	Widespread	Not Likely	Low	Low	Low
	Point-Source Contaminants	Low	Local	Very Likely	High	Low	Low
	Non-Point-Source Contaminants	Low	Entire Unit	Very Likely	High	Low	Low

Table 14. Threats analysis for the Kodiak, Kamishak, Alaska Peninsula MU of the southwest Alaska DPS of the northern sea otter.

Listing Factor	Threat	Potential Impact	Geographic Scope	Likelihood	Level of Confidence	Importance to Recovery	Management Potential
A. Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range	Habitat Loss	High	Local	Not Likely	High	Low	High
	Oil Spills	High	Local/ Widespread	Very/Somewhat Likely	High	Moderate	Moderate
	Prey Base	High	Local	Not Likely	High	Low	High
B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes	Subsistence Harvest	Moderate	Local	Very Likely	High	Moderate	High
C. Disease or Predation	Infectious Disease	Moderate	Local/ Widespread	Somewhat Likely	Low	Moderate	Low
	Predation	High	Widespread	Somewhat Likely	Low	Moderate	Low
D. Inadequacy of Existing Regulatory Mechanisms	N/A						
E. Other Natural or Anthropogenic Factors Affecting its Continued Existence	Fishery Bycatch	Moderate	Widespread	Somewhat Likely	Moderate	Low	High
	Illegal Take	High	Local	Somewhat Likely	Moderate	Moderate	Low
	Disturbance	Low	Local	Not Likely	High	Low	Moderate
	Biotoxins	Moderate	Widespread	Not Likely	Low	Low	Low
	Point-Source Contaminants	Low	Widespread	Very Likely	High	Low	Low
	Non-Point-Source Contaminants	Low	Entire Unit	Very Likely	High	Low	Low

Table 15. Summary of importance of threats to recovery of the southwest Alaska DPS of the northern sea otter by management unit.

		Management Unit					
Listing Factor	Threat	Western Aleutians	Eastern Aleutians	Bristol Bay	South Alaska Peninsula	Kodiak, Kamishak, Alaska Peninsula	Management Potential
A. Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range	Habitat Loss	Low	Low	Low	Low	Low	High
	Oil Spills	Low	Moderate	Low	Low	Moderate	Moderate
	Prey Base	Low	Low	Low	Low	Low	High
B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes	Subsistence Harvest	Low	Low	Low	Low	Moderate	High
C. Disease or Predation	Infectious Disease	Low	Low	Low	Low	Moderate	Low
	Predation	High	High	Moderate	High	Moderate	Low
D. Inadequacy of Existing Regulatory Mechanisms	N/A						
E. Other Natural or Manmade Factors Affecting its Continued Existence	Fishery Bycatch	Low	Low	Low	Low	Low	High
	Illegal Take	Low	Low	Low	Low	Moderate	Low
	Disturbance	Low	Low	Low	Low	Low	Moderate
	Biotoxins	Low	Low	Low	Low	Low	Low
	Point-Source Contaminants	Low	Low	Low	Low	Low	Low
	Non-Point-Source Contaminants	Low	Low	Low	Low	Low	Low

and Kodiak, Kamishak, Alaska Peninsula. Major ocean passes in the eastern Aleutian Islands are transited by ships using the great circle route between Asia and the west coast of North America, thus increasing the risk of a large spill. Kodiak Island has the highest human population within the range of the DPS and is adjacent to a major shipping route into Cook Inlet, thus justifying the moderate ranking for that MU. For the other three management units, this threat was rated as low. Should oil and gas exploration and production take place in the North Aleutian Basin, threat classifications for the Eastern Aleutian Islands, Bristol Bay, and South Alaska Peninsula MUs should be re-evaluated.

The management potential for prevention and containment of small oil spills is thought to be high, but the potential for containing or cleaning up large spills is low. Overall, the management potential for oil spills is considered moderate.

Prey Base – low importance

The potential for effects on the sea otter's prey base to become a threat to recovery was rated low for all MUs. While it is well established that high sea otter densities can dramatically reduce the abundances and sizes of many benthic invertebrate prey, the general result of such changes is a reduction in carrying capacity and an "equilibrium" sea otter population below peak size. In the western portion of the range of the DPS, where the decline has been most pronounced, indices of prey abundance, as well as sea otter diet, activity budgets, condition, and demography, all suggest a population well below carrying capacity.

In the eastern MUs, where sea otter densities are much higher, populations would be expected to level off, or already would have leveled off, at sizes that can be supported by the prey base over the long term. That situation should not be viewed as a "threat" to recovery because at such a point, the population would be fully recovered.

The primary factor affecting abundance of most sea otter prey is sea otter predation. Currently, there is little, if any, competition between humans and sea otters for prey within the range of the southwest Alaska DPS. Should new fisheries develop that create significant competition, this could be resolved through the State and Federal fishery regulatory processes, although such processes do not necessarily guarantee adequate protection of the sea otter prey base. Because human activities are not a factor currently affecting the prey base, the management potential for this threat is considered high.

Subsistence Harvest – low to moderate importance

Subsistence harvest was rated as a low threat to recovery in all MUs except Kodiak, Kamishak, Alaska Peninsula, for which it was rated as a moderate threat. From east to west, the human population becomes progressively sparser. Kodiak Island has the largest human population and largest reported subsistence harvest. Between 1989 and 2005, 1,857 otters were reportedly harvested from the southwest Alaska DPS, with the Kodiak archipelago accounting for 80% of that total. Annual harvest rates as a percentage of the estimated population size were low in all MUs and progressively increased from west to east, with a maximum of 0.32% in the Kodiak, Kamishak, Alaska Peninsula unit. In addition to being low, the harvest consists largely of males (73%), which means that it has less of an impact on population growth than if more females were taken. The current level of harvest is not excessive in relation to the population size, and is not thought to be a population-regulating factor. The threat classification of moderate was based on the fact that the human population within the range of the Kodiak, Kamishak, Alaska Peninsula MU is relatively large, and the fact that in much of that range the sea otter population is readily accessible to subsistence hunters using small boats.

In the future, should subsistence hunting be determined to be a threat to recovery it could be regulated. Since this DPS is listed as threatened under the ESA, it is automatically considered to be depleted under the MMPA, and Alaska Native take may be subject to regulation under provisions of that Act. Under provisions of the ESA, Native take may be regulated if it is shown to be "materially and negatively" affecting the species. Therefore the management potential for subsistence hunting is considered high.

Infectious Disease – low to moderate importance

Infectious disease was rated as a low threat to recovery in all MUs except Kodiak, Kamishak, Alaska Peninsula, for which it was rated as a moderate threat. There is no evidence that disease was a factor in the sea otter decline that occurred in the western portion of the range of the DPS, but the fact that very little disease work took place during the peak of the decline hinders our ability to rule disease out completely as either a primary or a contributory factor. Disease is recognized to be a natural process in all wildlife populations, and thus the mere presence of disease should not automatically be viewed as detrimental or pathological. There have been numerous documented cases of disease in sea otters in California, Washington, and Alaska. Generally, those populations have increased, although not always at expected rates. Situations in which disease could threaten recovery include those involving: 1) exposure to a novel disease in an immunologically naïve population, 2) widespread immunosuppression (due to population stressors, immunosuppressive disease agents, or reduced genetic variability), 3) maintenance of a pathogenic disease agent in a biotic or abiotic reservoir, and 4) extremely small population size. Introduction of a new epizootic disease agent into small remnant populations could be a very significant threat to recovery, and this seems to be what has occurred in Alaska with PDV. Because of these possibilities and uncertainties, the level of confidence in our assessment of the importance of disease to recovery is ranked as low in all MUs.

Disease is ranked as a moderate threat to the Kodiak, Kamishak, Alaska Peninsula MU because of the relatively large number of carcasses recovered in recent years in Kachemak Bay in the adjacent southcentral Alaska stock. Valvular endocarditis caused by *Streptococcus bovis* complex was the primary cause of death in 41% of the fresh carcasses examined during 2002-2006. While Kachemak Bay apparently was the center of the outbreak, cases were documented in the region from Umnak Island to PWS. This spike in mortality did not appear to have population-level effects, as the number of otters around the Kenai Peninsula increased at close to the maximum expected growth rate during the period of the outbreak. The management potential for disease is considered low, as the widespread distribution of animals would make capture, handling, and treatment extremely difficult.

Predation – moderate to high importance

Predation, specifically by killer whales but also potentially by other predators, was ranked as a high threat to recovery for the two Aleutian Islands MUs and the South Alaska Peninsula MU. Predation was rated as a moderate threat to recovery for the Bristol Bay and Kodiak, Kamishak, Alaska Peninsula MUs. The rationale for the different ratings was the relative extent of documented population declines. The populations of MUs that were rated high have declined greatly while those of the other two units either have not declined (Kodiak, Kamishak, Alaska Peninsula) or appear to have declined to a lesser extent (Bristol Bay).

While predation has not been determined with certainty to be the main cause of the sea otter decline, this hypothesis is strongly supported through a weight-of-evidence analysis. Observations supporting the hypothesis include: 1) increases in observed predation events involving killer whales attacking sea otters; 2) persistence of sea otters in areas that appear to provide refuge from predation by killer whales; 3) observed changes in otter behavioral responses that are consistent with avoidance of killer whales; 4) analyses demonstrating that the observed numbers of attacks on sea otters by killer whales is similar to that which would be expected if killer whale predation were solely responsible for the decline; 5) energetic analyses demonstrating that a small number of killer whales would be theoretically capable of causing the observed decline; 6) the rapid disappearance of large numbers of sea otters of all age classes over a broad area; 7) a scarcity of beachcast carcasses compared to what would be expected if the decline were driven by starvation, disease, or toxins; and 8) a high rate of disappearance of radio-tagged otters.

Several of the observations listed above would apply equally to predation by sharks, or to predation by both sharks and killer whales. There is no evidence for shark predation on sea otters in Alaska as there is for killer whale predation. White sharks are a significant cause of mortality of sea otters in California, but it is very unlikely that they occur more than rarely in the Aleutian Islands. However, the fact that sleeper sharks that are known to prey on marine mammals elsewhere are present in the range of the southwest Alaska DPS means that shark predation could be a contributing factor in the population decline.

Some have questioned the killer whale predation hypothesis because they don't believe that a predator that apparently has co-existed with a prey species for millennia could cause such a drastic decline in that species. However, in other systems there are documented cases where predation has driven prey populations to low levels or even extinction. In many of those instances, anthropogenic factors such as the introduction of predators, modification of habitat, or disruption of the food web were involved. Other instances of drastic fluctuations in abundance, such as the lynx–snowshoe hare multiyear cycle, appear to be within the normal range of non-equilibrium population dynamics.

Even if one accepts the hypothesis that predation caused the otter decline, there are great uncertainties about what to expect in the future. One scenario is that killer whales will continue to exert enough predatory pressure on otters to prevent recovery or cause even further decline. In this scenario, the killer whales would have to obtain the majority of their energetic requirements from alternative prey. An alternate scenario is that predation on sea otters becomes energetically unprofitable that killer whales shift their diet completely away from otters, to a point at which they are no longer a factor in the otters' population dynamics for some period of time. Poorly understood factors, such as killer whale prey preferences, pod home ranges and migratory patterns, and learned forms of predatory behavior, make projections into the future extremely uncertain. The management potential for predation as a threat is ranked low.

Fishery Bycatch – low importance

There is a history of California sea otters becoming entangled and dying in sunken gillnets (a gear type not currently used in Alaska) and fish and shellfish pots set in shallow coastal waters. In areas where this has occurred, the issues have largely been resolved by limiting the use of sunken gillnets to waters deeper than those where sea otters usually feed and by modifying pot design.

Available data suggest that sea otter mortality due to fishery bycatch within the range of the southwest Alaska DPS has been very low. Nearly all pot fisheries, with the exception of the dungeness crab fishery, occur in waters deeper than sea otters usually dive. Salmon and herring gillnet fisheries occur within the eastern portions of the range of the DPS. However, because the nets have floats at the top and are closely attended, few otters are entangled and most of those that are caught are released alive. For these reasons, fishery bycatch is rated as a low threat to recovery in all MUs. Because it is possible to modify fishing gear and fishing practices to avoid entangling otters, management potential for fishery bycatch is considered high.

Illegal Take – low to moderate importance

The illegal take of sea otters within the range of the southwest Alaska DPS is thought to be low, and all known occurrences have been in the Kodiak, Kamishak, Alaska Peninsula MU where the largest human population resides. Few people reside in or visit the western portion of the range, so opportunities for illegal take are very limited there. For this reason, illegal take as a potential threat to recovery was ranked as low for all MUs except for the Kodiak, Kamishak, Alaska Peninsula unit, for which it was ranked as moderate. Illegal take was considered to have a low management potential due to the vast range of the DPS.

Disturbance – low importance

Disturbance of sea otters results primarily from boat traffic. Boat traffic is light throughout most of the range of the southwest Alaska DPS. Traffic is highest in the eastern portion, where sea otter populations are at the highest levels. Otter populations have thrived in areas with much greater volumes of boat traffic, such as southeast Alaska and British Columbia. For these reasons, disturbance was ranked low as a threat to recovery, and management potential was ranked as moderate due to the vast range of the DPS

Biotoxins – low importance

Biotoxins produced by HABs have been implicated in mortality events in marine mammals and marine birds. The frequency and severity of these events worldwide may be increasing as a function of both increased nitrification of nearshore waters, increased ocean temperatures, and the anthropogenic alteration of ocean food webs. Paralytic shellfish poisoning (PSP), a biotoxin that accumulates in filter-feeding bivalves, has caused illness and death in humans in Alaska and may have caused a minor mortality event in sea otters at Kodiak Island in 1987. There is experimental evidence that sea otters avoid prey with PSP toxins. At present there is no indication that biotoxins are a threat to recovery of southwest Alaska sea otters and they are therefore ranked as low importance. Due to the nature of this threat, the management potential for biotoxins is low.

Point-Source Contaminants – low importance

High levels of PCBs have been found in sea otters and mussels in localized areas within the range of the southwest Alaska DPS, notably around Dutch Harbor, Adak Island, and Amchitka Island. However, overall PCB levels within the range of this DPS are lower than in areas such as southeast Alaska where otter populations are thriving. Potential adverse effects of exposure to high levels of PCBs include impaired reproduction and immunosuppression. Overall, throughout the range of the DPS there is no evidence that point-source contaminants, in this instance PCBs, are a threat to the recovery of the sea otter population. Therefore, this threat is rated as low. Due to te geographic scope of this threat, management potential for point-source contaminants is low.

Non-Point-Source Contaminants – low importance

Some contaminants, particularly DDTs, are transported in the atmosphere from Asia to the northeastern Pacific and Bering Sea, where they precipitate out into the ocean and enter the food chain through phytoplankton. To date, there are no data indicating adverse effects of such contaminants on southwest Alaska sea otters. It would be expected that higher trophic-level predators are affected before otters, but there may be substantial inter-specific differences in responses to contaminants. This threat to recovery was rated as low, with low management potential due to the geographic scope of this threat.

5. Recovery Strategy

Clearly the southwest Alaska DPS of sea otters has shown a major decline. The decline has been of such magnitude that the population is now at a perilously low level in much of the Aleutian Islands, with some island groups currently having less than one-tenth the number of otters they had in 1990. This is a region where just 20 years ago the population was large and probably at or near the environmental carrying capacity.

Changes in sea otter behavior and habitat use have occurred in this area concurrent with the decline. Shifts in otter distribution toward very shallow and nearshore areas and away from deeper and offshore areas are evident. These changes in habitat use may limit the ability of the population to recover, and require improved understanding.

The range of the DPS is large, with regionally variable environmental characteristics. The pattern of the decline, and perhaps the cause(s) of it, also varies regionally. To design and manage an effective recovery program, the DPS should treated as five "management units" (see section 2.I).

The only identified threat factor that is judged to have a high importance to recovery is predation. The weight of evidence suggests that killer whale predation is the most likely cause of the sea otter's decline in the Western Aleutian Islands management unit, with the role of predation being either less clear or unimportant in other units.

High priority should be given to studies that will help understand and assess the importance and implications of predation. There may be few actions that can be taken to mitigate predation as a threat, but the sea otter recovery program should search for solutions and be open to novel ideas.

Factors unrelated to predation are known to affect sea otter population trends. However, nearly all of these other factors are judged to have a low importance to recovery of the southwest Alaska DPS. Nonetheless, additional research into other current and future threats to recovery, particularly disease, is warranted. Actions should be taken wherever possible to mitigate threats from any source, and thereby minimize mortality and maximize productivity.

For each MU, a comprehensive system is needed to monitor the population abundance and trend of sea otters (see Section 7.A.1) and the status of associated ecosystems (the habitat monitoring plan described in Appendix A). Information from the monitoring program needs to be incorporated on an ongoing basis into a quantitative analytical framework (the "population viability analysis (PVA) model" described in Appendix B). The resulting analyses can be used to reevaluate the current population status with respect to uplisting or downlisting criteria, and when appropriate, any relevant quantitative criteria can be re-set to reflect the new information.

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6. Recovery Goals, Objectives, and Criteria

A. Recovery Goal

The goal of this recovery program is to control, reduce, or eliminate threats to the southwest Alaska DPS of the northern sea otter to the extent that this DPS no longer requires the protections afforded by the ESA and therefore warrants delisting.

B. Recovery Objectives

The Recovery Team identified three objectives for the southwest Alaska sea otter recovery program, as follows:

1. Achieve and maintain a self-sustaining population of sea otters in each MU.
2. Maintain enough sea otters to ensure that they are playing a functional role in their nearshore ecosystem.
3. Mitigate threats sufficiently to ensure persistence of sea otters.

C. Criteria for delisting

Demographic criteria

To meet recovery objective 1, managers must understand how the demographic characteristics of southwest Alaska sea otters (e.g., birth rates and mortality rates) interact to cause the population to increase, decrease, or remain stable. Population modeling is commonly used to integrate demographic data, and a growing trend in endangered species conservation is to use such models as the basis for PVAs that can project risk of extinction over time (U.S. Marine Mammal Commission 2007). These PVA models have several advantages over qualitative assessments for informing ESA listing and other management decisions. Some of those are: 1) they are more objective than qualitative assessments; 2) they can incorporate and use all relevant data; and 3) assumptions and uncertainties in data can be clearly identified and incorporated into results.

At the time that the southwest Alaska sea otter DPS was listed as threatened under the ESA, models to estimate extinction risks had not been developed for this population. In the process of preparing this recovery plan, such a model was created by members of the Recovery Team, and it is described in detail in Appendix B. This model was developed using data collected in the Western and Eastern Aleutian Islands MUs. Data from the other three MUs were not robust enough to support the creation of unit-specific PVAs.

Based on the definition of a threatened species in the ESA, it is generally recognized that a species or DPS should be considered for delisting when there is an acceptably low likelihood of it becoming endangered (endangered meaning likely to become extinct) in the foreseeable future. Therefore, an initial step is deciding what probability of becoming endangered is "acceptably low" and what timescale (i.e., how many years) constitutes the "foreseeable future." While the responsible management agencies (FWS and NMFS) have put significant effort into developing standards for these parameters (DeMaster et al. 2004, Regan et al. 2008), standard values that could be applied to species such as the sea otter have not been established. Based on expert opinion, the Recovery Team recommended using less than a 5% probability of becoming endangered within 25 years as the criterion for delisting, and more than a 5% probability of becoming extinct within 25 years as the definition of endangered.

The population simulations described in Appendix 2 were used to determine at what fraction of carrying capacity (K) the delisting criterion (<5% probability of becoming endangered within 25 years) would be satisfied for the Aleutian Islands MUs. Results indicated a threshold abundance of approximately 49% of K. Although the simulations were based only on Aleutian Islands data, the Recovery Team considered that this value could be applied to the other MUs on an interim basis while data are collected that will allow creation of unit-specific PVAs (see item 1.1.5 in section 7.A.).

Using the definition of "high density habitat" described by Bodkin and Udevitz (1999) and adopted by Burn et al. (2003) for the Aleutian Islands, available sea otter habitat was estimated for each MU using bathymetric data contained in the NOAA Southern Alaska Coastal Relief Model (Lim et al. 2009). Burn et al. (2003) showed that the density of sea otters in the Aleutian Islands at K is about 15 otters/km^2 of appropriate habitat. Estimates of equilibrium density at K do not exist for the Bristol Bay, South Alaska Peninsula, and Kodiak, Kamishak, Alaska Peninsula MUs. In surveyed areas within the Kodiak, Kamishak, Alaska Peninsula MU, sea otters occur at densities ranging from 5 otters/km^2 in the Katmai National Park area to 1.3 otters/km^2 in Kamishak Bay

(USGS unpublished data). The current status of this and the other areas relative to their carrying capacity is unknown, however. Pending acquisition of better data (see item 1.1.5 in section 7.A.), the Recovery Team recommended assigning an average value for equilibrium density of 3 otters/km^2 to the remaining three MUs. These values were used to produce estimates of K for each MU as shown in Table 16. A MU would meet the demographic criterion for delisting when its abundance is greater than 49% of K.

Ecosystem-based criteria

To meet recovery objective 2, managers need a metric that can be used to determine at what point sea otters are serving their normal functional role in southwest Alaska ecosystems. In an overall sense, the sea otter is closely linked to kelp forest ecosystems in the North Pacific Ocean. As a keystone species in those ecosystems, its presence or absence in a region can have a large effect on physical and biotic features. In rocky areas, there are two alternative phases—kelp-dominated when otters are present and urchin-dominated when they are absent. In soft-sediment areas, the relationships between otter abundance and ecosystem state are much less well known.

In the Aleutian Islands, sea otter habitat is largely rocky, and the extensive long-term datasets on sea otters and their ecosystem can be used to describe the relationship between sea otter abundance and the point at which an urchin-dominated ecosystem (that has resulted from reduced otter numbers) undergoes a phase shift to a kelp-dominated ecosystem (due to increased numbers of sea otters). This is described in detail in Estes et al. (2010) and Appendix A.

The proposed ecosystem-based recovery criterion, applicable only to the two Aleutian Islands MUs, is that sea otters must be sufficiently abundant to either maintain, or bring about, a phase shift to the kelp-dominated state. Attainment of this criterion will be evaluated by measuring kelp forest density at a number of sites at a number of islands as explained in Appendix A. A MU will be considered to meet the ecosystem-based recovery criterion when >50% of the islands sampled are judged to be in a kelp-dominated state.

The Recovery Team believes that it is not appropriate to propose ecosystem-based recovery criteria for other MUs at this time. Within the Bristol Bay MU, habitats used by sea otters are almost entirely soft-sediment, and while there may be a relationship between sea otter numbers and ecosystem state in these circumstances, that relationship has not been defined. In the South Alaska Peninsula and Kodiak, Kamishak, Alaska Peninsula MUs there is a mix of rocky and soft-sediment habitats, and for rocky areas preliminary results indicate that the relationship between otter density and kelp cover may be different from that described for the Aleutian Islands (see Appendix A). Potentially fruitful options for developing ecosystem-based recovery criteria for these other areas are described in Section 7.B, item 1.6.2.

Threats-based criteria

To meet recovery objective 3, managers must determine that all known threats are being adequately mitigated. Guidance for making such a determination is given below, organized by the five threat categories used in the ESA for listing determinations. Threats based criteria should be applied to each MU separately.

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

- Sea otter habitat within the range of the southwest Alaska DPS appears to be nearly pristine, with little obvious anthropogenic degradation. Although habitat destruction or modification likely did not play a role in the decline of this DPS, new threats to sea otter habitat should be minimized. Prior to delisting the DPS, threats to its habitat should be negligible as specified below.

- Aquatic habitat sufficient to support a recovered population of sea otters is protected through appropriate management measures. Conservation measures provide for: 1) adequate protection of the otters' prey base; and 2) the continued availability of habitat that affords refuge from marine predators.

- Safeguards are in place to prevent spills of oil and hazardous materials, and state-of-the-art response capabilities are available in the event of any spills.

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

- Prior to delisting the southwest Alaska DPS of northern sea otters, any overutilization for commercial, recreational, scientific, or educational purposes that threatens its continued existence should be managed as specified below.

- Subsistence harvest of sea otters is accurately monitored and managed to ensure that it does not compromise the recovered status of the DPS.

Factor C: Disease or predation

- Predation is believed to have played an important role in the decline of the southwest Alaska DPS of northern sea otters. Currently, disease is considered to have a relatively minor impact on the sea otter population. Prior to delisting the DPS,

Table 16. Estimates of available habitat, equilibrium density, and carrying capacity for the five management units in the southwest Alaska DPS of the northern sea otter.

Management Unit	Available Habitat (km²)	Equilibrium Density (otters/km²)	Carrying Capacity (K)
Western Aleutian	4,260	15	63,897
Eastern Aleutian	2,046	15	30,697
Bristol Bay	12,299	3	36,898
South Alaska Peninsula	11,159	3	33,476
Kodiak, Kamishak, Alaska Peninsula	15,362	3	46,086
Total			211,054

threats from disease or predation should be understood as specified below.

- Information is adequate to conclude that neither disease nor predation is currently compromising the recovered status of sea otters.

Factor D: The inadequacy of existing regulatory mechanisms

- Inadequacy of existing regulatory mechanisms, including the MMPA and the ESA, did not contribute to the sea otter decline. Prior to delisting the southwest Alaska DPS of northern sea otters, existing regulatory mechanisms should exist as specified below.
- Existing regulatory mechanisms that provide protections for sea otters should not have been weakened.

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

- Prior to delisting the southwest Alaska DPS of northern sea otters, natural and anthropogenic threats to its continued existence should be minimized as specified below.
- A mechanism (e.g., fishery management and monitoring) is in place and will remain in place that ensures fishery bycatch is less than the potential biological removal (as defined under the MMPA).
- Enforcement of MMPA and ESA regulations is adequate to restrict illegal take to negligible levels.
- All types of human disturbance are controlled sufficiently to make it unlikely that disturbance will jeopardize the recovered status of the DPS.
- Toxins, including point-source and non-point-source pollutants and biotoxins, do not pose significant health risks to the recovered sea otter population.
- Contingency plans and response capability for oil spills are in place and are capable of dealing with a major oil spill in important areas used by sea otters.

Summary of criteria for delisting

Each of the five MUs of the southwest Alaska sea otter DPS identified in this plan will be evaluated independently with respect to delisting criteria as shown in Table 17. The status of all MUs will be evaluated against demographic and threats-based criteria, while the Western Aleutian and Eastern Aleutian MUs will also be evaluated against the ecosystem-based criterion.

To delist a species listed as threatened, the responsible management agency must find that the DPS is no longer “likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.” For this criterion, “foreseeable future” has been fixed at 25 years as specified in the PVA model (Appendix B). The Recovery Team did not find a non-arbitrary way to define “significant portion of its range” for the southwest Alaska sea otter DPS. For this plan, it is recommended that delisting be considered when any three of the five MUs meet all of the recovery criteria specified for them in Table 17. Delisting should not be considered if any one of the MUs meets the criteria specified for uplisting to endangered as described in the following section.

Table 17. Summary of criteria that must be met prior to delisting the southwest Alaska DPS of the Northern sea otter.

	Management Unit				
De-listing Criterion	Western Aleutian	Eastern Aleutian	Bristol Bay	South Alaska Peninsula	Kodiak, Kamishak, Alaska Peninsula
Demographic	Probability of becoming endangered within 25 years is <5%	Probability of becoming endangered within 25 years is <5%	Probability of becoming endangered within 25 years is <5%	Probability of becoming endangered within 25 years is <5%	Probability of becoming endangered within 25 years is <5%
Ecosystem-based	>50% of islands are in the kelp-dominated state	>50% of islands are in the kelp-dominated state	None proposed	None proposed	None proposed
Threats-based	Threats are adequately mitigated	Threats are adequately mitigated	Threats are adequately mitigated	Threats are adequately mitigated	Threats are adequately mitigated

D. Criteria for reclassification to endangered (uplisting)

Demographic criteria

Based on the definition of an endangered species in the ESA, it is generally recognized that a species or DPS should be considered for reclassification from threatened to endangered when there is an unacceptably high likelihood of it becoming extinct within the foreseeable future. Decisions must be made as to what is an unacceptably high probability of becoming extinct and how many years constitute the foreseeable future. While the responsible management agencies (FWS and NMFS) have put significant effort into developing standards for these parameters (DeMaster et al. 2004, Regan et al. 2009), standard values that could be applied to species such as the sea otter have not been established. Based on expert opinion, the Recovery Team has recommended using more than a 5% probability of becoming extinct within 25 years as the criterion for uplisting to endangered. Also, for long-lived species equating extinction to the point where only one individual remains is not appropriate (Regan et al. 2009), and the Team therefore specified a "quasi-extinction" threshold, defined as the point at which all islands[1] within a MU have <5 female otters and no males.

The population simulations described in Appendix 2 were used to determine the threshold density at which the reclassification criterion (≥5% probability of becoming extinct within 25 years) would be satisfied. Results indicated that the threshold density is approximately 4% of K. While the simulations were based only on Aleutian Islands data, in the judgment of the Recovery Team this value can be applied to the other MUs on an interim basis while data are collected that will allow unit-specific PVAs to be created (see item 1.5.1 in section 7.A.). A MU would meet the demographic criterion for reclassification to endangered when its abundance is less than 4% of K as shown in Table 16.

Ecosystem-based criteria

If the southwest Alaska sea otter DPS is being considered for uplisting from threatened to endangered, it is unlikely that the population will have recovered its functional role in the ecosystem and more likely that its functional role will have diminished. One would expect that less of the rocky habitat normally used by otters would be in the kelp-dominated state and more would be urchin-dominated. While it might be possible to design an ecosystem-based uplisting criterion

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[1] In this context we use the term "island" to refer to a single island or island group that is geographically distinct from other such islands or island groups (i.e., separated by at least 10 km) and that has sufficient sub-tidal habitat to support an isolated sub-population of sea otters.

Table 18. Estimates of carrying capacity, delisting abundance, uplisting abundance, and current status relative to carrying capacity for the five management units in the southwest Alaska DPS of the northern sea otter.

Management Unit	Carrying Capacity (K)	Delisting Abundance Threshold	Uplisting Abundance Threshold	Most Recent Abundance Estimate	Status (%K)
Western Aleutian	63,897	31,309	2,556	6,451	0.10
Eastern Aleutian	30,697	15,042	1,228	2,291	0.07
Bristol Bay	36,898	18,080	1,476	11,253	0.30
South Alaska Peninsula	33,476	16,403	1,339	4,724	0.14
Kodiak, Kamishak, Alaska Peninsula	46,086	22,582	1,843	28,955	0.63
Total	211,054	103,417	8,442	53,674	0.25

based on these likely changes, such a criterion is not being proposed at this time.

Threats-based criteria
If the population status has deteriorated such that reclassification to endangered is being considered, then clearly some threat or threats have not been adequately mitigated. Any formal status review of the DPS should include a detailed analysis of the threats that may be affecting the population at that time, but threats-based criteria for uplisting are not being proposed at this time.

Summary of criteria for reclassification to endangered
Each of the five MUs of the southwest Alaska sea otter DPS identified in this plan will be evaluated independently with respect to reclassification criteria. To list a species as endangered, the responsible management agency must find that the DPS "is in danger of extinction throughout all or a significant portion of its range." The Recovery Team did not find a non-arbitrary way to define "significant portion of its range" for the southwest Alaska sea otter DPS. For this plan, it is recommended that reclassification to endangered be considered when at least three of the MUs meet the demographic criteria specified above.

E. Summary of demographic status in relation to delisting and uplisting criteria

The most recent estimates of sea otter abundance (Table 1) indicate that the four MUs that have shown clear evidence of population declines (Western and Eastern Aleutian Islands, Bristol Bay, and South Alaska Peninsula) are all at levels between their de-listing and up-listing thresholds (Table 18). The Kodiak, Kamishak, Alaska Peninsula unit is above the de-listing threshold. Pending the development of PVAs specific to the Bristol Bay, South Alaska Peninsula, and Kodiak, Kamishak, Alaska Peninsula MUs (see item 1.5.1 in section 7.A.), and better estimates of carrying capacity (see item 1.1.5 in section 7.A.), delisting and uplisting thresholds for these MUs are considered provisional.

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7. Recovery Program

The recovery program is organized with seven major categories, each with one or more specific actions that promote recovery of the southwest Alaska DPS of the northern sea otter. These actions are presented as a step-down outline in Section 7.A. below. Each discrete recovery action is assigned a priority and narrative description in Section 7.B.

A. Recovery Action Outline

1. **Population monitoring and research**
    - **1.1.** Continue to estimate sea otter population size and trends in southwest Alaska
        - **1.1.1.** Evaluate and improve methods for abundance surveys
        - **1.1.2.** Conduct surveys in each management unit and estimate abundance and population trends
        - **1.1.3.** Identify areas of consistently high sea otter abundance, relative to other areas, within each management unit
        - **1.1.4.** Evaluate potential biases in survey data and identify methods that may be useful for correcting such biases
        - **1.1.5.** Refine estimates of equilibrium density and carrying capacity for the Bristol Bay, South Alaska Peninsula, and Kamishak, Kodiak, Alaska Peninsula MUs
        - **1.1.6.** Evaluate the overall population monitoring program on an ongoing basis to assess sensitivity
    - **1.2.** Estimate vital rates
        - **1.2.1.** Estimate birth rates
        - **1.2.2.** Estimate age- and sex-specific survival rates
    - **1.3.** Monitor health and body condition
        - **1.3.1.** Monitor general body condition
        - **1.3.2.** Analyze data for evidence of food limitation
    - **1.4.** Monitor distribution and movements
    - **1.5.** Continue development of population viability analysis models
        - **1.5.1.** Investigate PVA models for the Bristol Bay, South Alaska Peninsula, and Kamishak, Kodiak, Alaska Peninsula MUs
        - **1.5.2.** Revise and re-run the PVA model(s) incorporating new data
    - **1.6.** Monitor the functional role of sea otters in ecosystems
        - **1.6.1.** Monitor status of the kelp forest ecosystem
        - **1.6.2.** Develop methods to monitor sea otter impacts on marine ecosystems with soft-sediment substrates
    - **1.7.** Develop new research tools
2. **Habitat needs and habitat protection**
    - **2.1.** Identify important habitats or areas of special biological significance
        - **2.1.1.** Identify characteristics of areas that are serving as refuges for remnant groups of sea otters
        - **2.1.2.** Explore the feasibility of creating more areas that can serve as refuges
        - **2.1.3.** Protect important habitats
    - **2.2.** Prepare a habitat conservation plan under Section 10 of the ESA for Alaska state-managed fisheries
    - **2.3.** Ensure adequate capabilities for response to oil spills

3. **Manage impacts of human uses**
    - **3.1.** Ensure that Alaska Native subsistence harvest does not affect recovery
        - **3.1.1.** Continue to estimate harvest levels through the FWS marine mammal marking, tagging, and reporting program
        - **3.1.2.** Quantitatively evaluate the impact of sea otter harvest on recovery
        - **3.1.3.** Develop sea otter harvest management plan(s) with Alaska Native tribes and tribally authorized organizations
        - **3.1.4.** Continue outreach to promote “males only” harvest
        - **3.1.5.** Use pelts from stranded otters to reduce demand for subsistence-harvested animals
    - **3.2.** Ensure that incidental take in fisheries does not affect recovery
        - **3.2.1.** Monitor the level of incidental take in fisheries within the sea otter’s range in southwest Alaska
        - **3.2.2.** Quantitatively evaluate the impact of incidental take on recovery in each MU
        - **3.2.3.** Develop programs to reduce incidental take in situations where it occurs
    - **3.3.** Ensure that entanglement in marine debris does not affect recovery
        - **3.3.1.** Encourage net recycling and environmentally responsible disposal
    - **3.4.** Eliminate intentional illegal take
        - **3.4.1.** Establish an outreach program on the ecological role of sea otters in the nearshore marine ecosystem
        - **3.4.2.** Increase enforcement efforts to investigate and prosecute cases of illegal killing
    - **3.5.** Evaluate potential impacts of scientific research activities
4. **Disease**
    - **4.1.** Evaluate the potential role of disease as a threat to recovery
        - **4.1.1.** Expand the marine mammal stranding network in southwest Alaska
        - **4.1.2.** Use carcasses and tissues from subsistence-harvested animals to investigate disease processes
        - **4.1.3.** Analyze carcasses and tissues and report results
        - **4.1.4.** Conduct live-capture studies to examine health status of live sea otters
        - **4.1.5.** Evaluate population-level effects of identified diseases
    - **4.2.** Develop disease management plans where practical
5. **Predation**
    - **5.1.** Continue to evaluate the role of predation as a threat to recovery
    - **5.2.** Develop predation management plans, where practical
6. **Protect from other natural or human-caused threats**
    - **6.1.** Ensure that oil spills do not impede recovery of sea otters and/or negatively affect the nearshore marine environment in southwest Alaska
    - **6.2.** Establish an outreach program to mariners on how to avoid striking sea otters
    - **6.3.** Continue to measure and monitor contaminant levels in sea otters
    - **6.4.** Monitor occurrence of biotoxins in sea otters and their prey
    - **6.5.** Evaluate the feasibility of translocating sea otters to enhance recovery
    - **6.6.** Evaluate potential impacts of recreational activities, tourism, and other forms of direct human disturbance
7. **Implement the recovery program for southwest Alaska sea otters**
    - **7.1.** Maintain the Southwest Alaska Sea Otter Coordinator position within FWS
    - **7.2.** Continue and enhance coordination of management efforts among FWS, Alaska Natives, and the State of Alaska
    - **7.3.** Continue and enhance coordination of research efforts among FWS, USGS, the State of Alaska, Alaska Natives, academic institutions, and others
    - **7.4.** Develop and continue a program of outreach to stakeholders
    - **7.5.** Secure adequate funding for southwest Alaska sea otter management and research needs

B. Recovery Action Narrative

The following action items are generally assigned priority independent of one another. However, many of the items are in fact not independent. For example, marking of individuals requires capture and handling, during which measurements and samples should be obtained that can be used to evaluate competing hypotheses related to population status, a low-priority objective. Reproductive rates, while not assigned a high priority, can, and should, be estimated through counts of dependent pups during surveys of abundance. Thus, assignment of a priority lower than high to a given action should not necessarily mean that no progress is going to be made on that action. In many instances, multiple types of information can be collected for little additional cost while conducting high-priority studies.

Priorities have been assigned to each action item according to the guidance provided in the FWS Recovery Handbook (FWS 1990), as follows:

- **Priority 1**: Actions that must be taken to prevent extinction or to prevent the species from declining irreversibly.
- **Priority 2**: Actions that must be taken to prevent a significant decline in species population/habitat quality or to prevent some other significant negative effect short of extinction
- **Priority 3**: All other actions necessary to provide for full recovery of the species.

1. Population monitoring and research

1.1. Continue to estimate sea otter population size and trends in southwest Alaska

1.1.1. Evaluate and improve methods for abundance surveys—**Priority 2**

A variety of survey methods have been used over the past several decades to estimate the number of sea otters that inhabit different areas within Alaska. In addition to statistical sampling considerations, the remote nature of many of these areas imposes logistical constraints on the type of survey platforms that can be used to conduct survey operations safely. Although survey data exist for each of the five MUs in the southwest Alaska DPS, survey methods should continue to be re-evaluated and re-designed as necessary to improve accuracy and precision.

In addition to statistical sampling methods, an important consideration in survey design is how to correct for otters that are missed by observers during aerial and/or skiff surveys. One commonly used aerial survey method includes procedures for developing a correction factor to account for undetected otters. This method employs intensive searches within strip transects to estimate detection. Thus far, it has been tested using a small, single-engine aircraft on floats, which may be impractical for MUs where the use of twin-engine aircraft is required for safety reasons. For those MUs, some other method should be developed, tested, and implemented to correct survey data for missed (non-detected) otters.

1.1.2. Conduct surveys in each management unit and estimate abundance and population trends—**Priority 1**

The decline of sea otters throughout much of southwest Alaska since the mid-1980s is well documented through aerial and skiff surveys. In order to monitor population trends and assess recovery, surveys of sea otters in each MU should be conducted at regular intervals using established methods that yield comparable results. As delisting criteria are based in large part on the population status within each MU, this is a priority 1 item. Specific actions that are needed in each MU are listed below.

- Western Aleutian MU:
    - Continue to conduct skiff survey counts at established index sites.
    - Establish an additional index site (aircraft-based) in the Islands of Four Mountains.
- Eastern Aleutian MU:
    - Continue to conduct skiff survey counts at established index sites.
- Bristol Bay MU:
    - Establish a system of index sites for monitoring trends, such that a representative subset of available habitat (stratified by geographic attributes) can be covered by aerial transects in 1-2 days.
- South Alaska Peninsula MU:
    - Continue to conduct aerial survey counts at established index sites.

- Kodiak, Kamishak, Alaska Peninsula MU:
  - Continue to conduct aerial survey counts at established index sites.

1.1.3. Identify areas of consistently high sea otter abundance, relative to other areas, within each management unit—**Priority 2**

Repeated surveys identified in item 1.1.2 have the potential to identify areas where sea otters occur in consistently high numbers over time. Knowledge of where such areas are will be very useful for evaluating the adequacy of habitat protection measures. Also, a detailed analysis of the characteristics of habitat in those areas could provide valuable insight into the importance of habitat in recovery.

1.1.4. Evaluate potential biases in survey data and identify methods that may be useful for correcting such biases—**Priority 2**

Both aerial and skiff surveys have biases. With the exception of double-counting otters, the vast majority of bias in survey data is negative (i.e., it causes counts and estimates to be lower than the actual number of animals). Factors that may contribute to bias in survey results include search intensity, observer experience and fatigue, environmental conditions, and habitat types. The overall effect of bias in survey results is a reduction in the accuracy of counts or estimates, which in turn may compromise assessment of population trends. Analyses of survey data may help identify the most important sources of bias and facilitate development of methods to correct for that bias.

1.1.5. Refine estimates of equilibrium density and carrying capacity for the Bristol Bay, South Alaska Peninsula, and Kodiak, Kamishak, Alaska Peninsula MUs—**Priority 1**

Estimates of sea otter equilibrium density and carrying capacity do not exist in the scientific literature for the Bristol Bay, South Alaska Peninsula, and Kodiak, Kamishak, Alaska Peninsula MUs. The estimates of equilibrium density developed by Burn et al. (2003) were based on survey-specific data from the Aleutian archipelago, and are not considered applicable to these other three MUs. Along with developing PVA models for each specific MU (item 1.5.1), valid estimates of equilibrium density and carrying capacity will be needed to update demographic de-listing and up-listing criteria, and this is a high priority item.

1.1.6. Evaluate the overall population monitoring program on an ongoing basis to assess sensitivity—**Priority 1**

The ability to detect population trends is largely dependent on the precision of the population estimates calculated from survey results. As new survey information is collected, the results should be analyzed to determine if sample size and survey frequency are sufficient to detect population trends with the desired level of confidence, or alternatively if adequate information might be obtained through reduced survey effort in some areas. Results of this analysis may also be used to guide survey design considerations in item 1.1.1. As this item relates to evaluating the use of survey information to detect population status relative to delisting criteria, it has high priority.

1.2. Estimate vital rates

Information on sea otter demography is needed to help understand the causes of changes in abundance, and to predict likely rates of population decline or recovery. Demographic studies are conducted in two main ways—by taking measurements from individuals and by making observations at the level of populations. Individual-based studies generally require that otters be tagged or marked so that they are individually recognizable and their fates can be followed over time to estimate survival and reproductive rates. Standardized, safe techniques for sea otter capture, immobilization, instrumentation, and marking have been developed and are readily available. Capture operations also provide opportunities to collect biological samples and information (e.g., teeth can be collected for age determination, blood and tissue samples can be collected for health screening and genetic studies, and body length and mass can be measured for assessments of nutritional status). Under some circumstances, sampling at the level of the population may be necessary or preferable to longitudinal study of individuals (one that follows the same animals over a period of time). For example, adequate sample sizes of individually marked animals may be difficult to obtain because densities of individuals are low or because particular habitats have certain characteristics. Population-level estimates of vital rates can be derived from the standing age structure of the living population, the age-at-death distribution, or a combination of the two.

1.2.1. Estimate birth rates—**Priority 3**

Information on sea otter reproductive status can be obtained from examination of reproductive tracts of dead animals (obtained from subsistence harvest or strandings), or potentially by analysis of hormones in blood taken from animals live-captured and released. However, it is unlikely that sufficient samples will be available from such sources to quantify birth rates in each of the various MUs of the DPS. During skiff surveys at index sites, pups and adults are recorded independently, and the ratio of pups to adults can provide a relative index of reproductive rates that can be monitored over time. Because a low rate of reproduction is not considered to be a causal factor in the decline, or an impediment to recovery, a low priority should be given to estimating birth rates.

1.2.2. Estimate age- and sex-specific survival rates—**Priority 1**

An excessively high mortality rate appears responsible for the decline in southwest Alaska sea otters, at least in the Aleutian Islands, and this factor will constrain recovery until survival improves. Therefore, monitoring survival in the current population will be essential for projecting recovery and understanding constraints to recovery, and such monitoring is a high priority. Although the elevated mortality during the Aleutian decline was apparently constant across all ages, demographic recovery models require information on survival rates for specific age and sex classes. The most direct method to measure such rates is to follow individually identified animals and determine at what age they die. Given the geographical and environmental conditions, and the low density of animals in much of southwest Alaska, an intensive effort will be required to obtain useful data on survival by such a method. An alternative is to locate carcasses of animals that have died and determine their age (from teeth) and sex (from observing reproductive organs or analyzing genetic samples). However, such samples are clearly biased against sources of mortality such as predation, for which carcasses will not be available for examination. In low-density populations, it may be impossible to obtain enough carcasses to determine survival rates reliably. Studies of survival rates might be possible in MUs where otter numbers are still relatively high (e.g., Kodiak, Kamishak, Alaska Peninsula MU or Bristol Bay MU), but inferences for areas where otters are not declining are unlikely to be applicable for areas where a decline is underway.

1.3. Monitor health and body condition

Although it is difficult to understand all of the specific causes of the recent sea otter population decline, useful inferences can sometimes be made based on the condition of living animals. Specifically, comparing a suite of health and morphometric variables between MUs that have declined greatly and units with relatively high abundance could provide useful indicators of the status of the depressed populations.

Methods to assess body condition in sea otter populations include capturing, handling, measuring, and sampling of tissues from live animals. Such methods are well developed and can be employed throughout most of the habitat occupied by sea otters. However, in areas where individuals occur at very low densities, a high level of effort by researchers will be required.

1.3.1. Monitor general body condition—**Priority 3**

Recent analyses of age- and sex-specific weights, total lengths, and other linear dimensions of sea otters show that such measurements are sensitive indicators of nutritional status. Contrasts in body condition between animals from areas with stable populations and animals from areas with declining populations thus may provide supplementary evidence to resolve competing hypotheses on the cause of the decline or the constraints to recovery. Similarly, contrasts based on the screening of blood, serum, and other tissues for evidence of disease, contaminants, and immune system function are likely to provide further insights into the cause of decline and the status of the populations relative to recovery. Because food limitation is not thought to be a cause of the sea otter decline, and data on body condition have already been collected in the area of decline, this action is a low priority at present. Nonetheless, such data should be collected and archived whenever animals are captured for other purposes, as baseline data will be important for interpreting any changes in status over time.

1.3.2. Analyze data for evidence of food limitation—**Priority 2**

Sea otters are unique among marine mammals in that they feed almost exclusively on large benthic invertebrates, usually in shallow waters close to shore, and bring their prey to the surface. Therefore, experienced observers can record dive attributes (e.g., dive times, surface times, and prey capture success rates) and sizes, numbers, and taxonomic classes (often to genus or species) of prey consumed. Those records can be used to estimate diet diversity, energy recovery rates, and activity-time budgets, which can indicate the current status of those units relative to prey availability. Because of the potential for additional factors to constrain recovery and the broad inferential capacity of these types of data, this action is a moderate priority.

1.4. Monitor distribution and movements—**Priority 1**

A fundamental shift in the fine-scale distribution of sea otters in the Aleutian Islands has occurred over the period of population decline, with otters now associated mostly with very shallow-water habitats, often less than 2 m deep, and rarely occurring in deeper or offshore waters. Given the clear change that has occurred with the population decline in the Aleutians, measuring and monitoring distribution and habitat use patterns in each MU is a high priority, as variations in these metrics might be used to monitor changes in the extent of the decline or in the factors that have caused it.

Fine-scale distribution and habitat use can be quantified by systematic visual observations of live sea otters, their locations, and their activities. Observations should be made across broad geographic areas, and sampling protocols should be developed with consideration for potential biases (e.g., distance bias in visual observations). Another more powerful and potentially less biased approach is to attach telemetry instruments to animals and follow their movements over time. This can provide detailed data on habitat use by individuals over extended periods.

Another important aspect of distribution and movements is the frequency and nature of large-scale movements and inter-island dispersal. This is important both for understanding and explaining apparent changes in local abundance and for understanding the likelihood and possible rates of recolonization of depleted areas. Data on large-scale movements and inter-island dispersal are also needed for parameterization of the PVA model. Methods for obtaining such information are tracking with satellite-linked tags and detailed examination of genetic characteristics of animals. Both of these techniques may require development of additional tools, and they will, in any event, be difficult to apply in areas with very low densities of otters. However, given the high value of such data, both efforts to collect data and the development of new techniques for doing so should be considered high priorities.

1.5. Continue development of population viability analysis models

1.5.1. Investigate PVA models for the Bristol Bay, South Alaska Peninsula, and Kamishak, Kodiak, Alaska Peninsula MUs—**Priority 1**

As part of the development of this recovery plan, a PVA has been created for southwest Alaska sea otters (Appendix 2) and used to describe demographically-based delisting and reclassification criteria in this recovery plan (sections V.C.1 and V.D.1). The PVA model was developed using demographic data collected in the Aleutian Islands. At the time this plan was developed, demographic data for other MUs were limited, so results from the Aleutians PVA were applied provisionally to all MUs. As additional data are collected (e.g., items 1.1, 1.2, and 1.4 of this section), additional modeling should be carried out to either verify that the existing PVA is appropriate for application to all MUs, or to create specific PVAs for the Bristol Bay, South Alaska Peninsula, and Kamishak, Kodiak, Alaska Peninsula MUs. Because the demographic status of each individual MU is important for evaluating listing status of the DPS as a whole this is a high priority.

1.5.2. Revise and re-run the PVA model(s) incorporating new data—**Priority 1**

The existing PVA model uses demographic data collected in the Aleutian Islands through the year 2008. It is expected that substantial new data will be collected in the Aleutian Islands and other MUs as the action items described in this recovery plan are undertaken. At appropriate intervals (e.g., every five years) those new data should be incorporated into the PVA(s) and the model(s) should be re-run. After that, target demographic thresholds for delisting and reclassification should be revised, if necessary. Because results from the PVA are an essential part of determining whether or not the southwest Alaska DPS is recovering this is a high priority.

1.6. Monitor the functional role of sea otters in ecosystems

1.6.1. Monitor status of the kelp forest ecosystem—**Priority 1**

Sea otter predation is an essential process in maintaining nearshore reef habitats as kelp forest ecosystems, and in rocky areas the status of a regional sea otter population can be inferred from the state of kelp forests. The recent collapse of the sea otter population in southwest Alaska caused a shift to the deforested phase state throughout all or most of the Aleutian archipelago. This change was documented through SCUBA-based monitoring at various islands across the archipelago. Further analysis of these data has shown that a large-scale phase shift, from the currently deforested phase state to the kelp forest phase state, could be detected with a high level of confidence by sampling as few as 5 to 10 sites at several islands (Appendix 1). A sampling effort of this sort could easily be accomplished with four divers (two dive teams) in 1-2 days at each island, less than the time required to complete the skiff surveys.

Kelp forest monitoring should be carried out in conjunction with the USFWS skiff-based sea otter monitoring program, using the established USGS kelp forest monitoring protocol (as described by Estes and Duggins [1995] and in Appendix 1). The reassessment of ecosystem status will require samples from at least 4 of the following 5 specific islands: Attu (Near Islands), Amchitka (Rat Islands), Adak (Andreanoff Islands), Chiginadak (Islands of Four Mountains), and Akutan (Fox Islands). A minimum of 10 sites should be sampled per island.

When monitoring the status of the kelp forest ecosystem in southwest Alaska, researchers should be aware of, and pay attention to, ecological factors other than sea otters that might be affecting the abundance of urchins (and other major kelp predators) and of kelp.

1.6.2. Develop methods to monitor sea otter impacts on marine ecosystems with soft-sediment substrates —**Priority 2**

The direct effects of persistent sea otter foraging on their prey species generally include reductions in abundance and size distribution, and such reductions have been documented for a variety of species, including abalones, crabs, mussels, urchins, and clams, over a broad geographic area. Because sea otters appear to prey preferentially on larger individuals, there is a consistent shift toward smaller size classes and reduced mean prey size as sea otters colonize previously unoccupied habitat. It is possible that these direct effects of sea otter predation on prey could be used in ways similar to the kelp forest/urchin barren phase state as a tool to evaluate the status of sea otter populations (Appendix 1).

For example, if otters at equilibrium density effectively maintain truncated prey size distributions by consuming most individuals above some particular size, then one could reasonably hypothesize that reduced otter density will result in a shift in the distribution toward larger individuals and an increase in the mean size of preferred prey species. Such an approach for evaluating sea otter populations is appealing for several reasons, including its relatively low cost and high statistical power, as well as the fact that it provides a way to assess the role of food limitation in influencing otter populations in the nearshore ecosystem. However, in contrast to the breadth of data supporting the idea that the sea otter functions as a “keystone species” in the nearshore, maintaining kelp forests through predation on herbivorous sea urchins, the data on direct effects of otter predation on prey size are both geographically and taxonomically limited.

Because of the abundance of soft-sediment habitats across much of the range of this DPS, and given the importance of clams in the otter’s diet in such habitats, clams may be appropriate for testing such hypotheses in these habitats. Also, given the presence and consumption of intertidal mussels throughout the sea otter’s range, and the availability of historical data, mussels may provide similar opportunities. Initially, research could explore the functional relationship between otter density and prey population attributes (e.g., mean size, size distribution, and biomass) and the potential generality of findings across broad geographic scales.

1.7. Develop new research tools —**Priority not assigned**

There is a great deal of uncertainty about the factors that may act to influence population recovery. While marine mammals are inherently difficult to study, a variety of recently developed technologies could be used to obtain data relevant to sea otter recovery actions in southwest Alaska. The continued development of such tools is important for marine mammal science in general, and is not assigned a priority or cost in this recovery plan specific to southwest Alaska sea otters. Nonetheless, some potentially promising areas for investigation are identified below. Given the broad scope of these research tools, priority and costs could not be assigned to this item.

- TDRs—TDRs can provide long-term, high-resolution data on sea otter dive depths, habitat use, dietary specialization, activity budgets, female reproductive events, and causes of death. These instruments should be considered for deployment in any project involving live capture and marking. Because the instruments are archival, they must be recovered in order to obtain the data, and this requires recapture of the marked animals. However, pop-off TDR instruments that do not necessitate recapture could be developed for sea otters.

- Satellite transmitters—Conventional radiotelemetry techniques, while greatly extending our knowledge of sea otter biology and ecology, only allow for tracking animals within a limited range. Satellite-linked transmitters have been used on a number of wide-ranging species and could be developed for use with sea otters. Because spatial and temporal resolution of locations from the ARGOS doppler system are relatively poor, incorporation of GPS technology will be essential for studies addressing fine-scale habitat use.

- Survival pop-up tags—Estimating sea otter survival with visual observations and conventional radiotelemetry requires intensive and costly monitoring. Even then, some deaths go undetected, meaning that untestable assumptions need to be made about the fates of missing animals. Satellite-linked tags that transmit a specific signal when an animal dies would provide better information on the timing, location, and perhaps causes of mortality.

- Stable isotope and fatty acid analyses—Stable isotopes in tissues can provide information on trophic status of species on various time scales. Analyses of stable carbon and nitrogen isotopes in the teeth of sea otters may show patterns of variation that are characteristic of growing and food-limited populations, and analyses of killer whale teeth could help clarify the role of killer whale predation in the sea otter decline. Analyses of fatty acid signatures in sea otters and their prey and in potential predators may provide similar information.

- Molecular genetics—Recent advances in analytical approaches to molecular genetic data have potential application to questions of dispersal distances and recolonization probabilities of sea otters. Tissue samples from sea otters in many locations in southcentral and southwest Alaska are available and could be used to evaluate, for example, the likelihood of dispersal across Aleutian Island groups separated by various distances. Advanced molecular genetic methods could also be applied to historical samples to examine the frequency and duration of population declines in the past, for example, following human occupation of North America.

2. Habitat needs and habitat protection

2.1. Identify important habitats or areas of special biological significance

There is virtually no evidence to suggest that the quality or quantity of sea otter habitat, as traditionally defined, has been reduced or degraded within the range of the southwest Alaska DPS. However, reasonably good evidence exists to suggest that use of that habitat, at least in the Aleutian Islands MUs, has shifted significantly. Prior to the decline, sea otters were widely distributed and most prevalent in canopy-forming kelp forests, generally in water depths from 5-20m and often hundreds of meters from shore, depending on bathymetry. Following the decline, the otters have become highly aggregated in very shallow water a few meters or less in depth and most often less than tens of meters from shore. This shift in use of habitat may be explained as a response to increased predation pressure. The observed shift in habitat use suggests that these very nearshore areas are important to the conservation of sea otters. Consequently, improved understanding of the relationships between current sea otter distribution and the features that define this shift in habitat use is a high priority.

2.1.1. Identify characteristics of areas that are serving as refuges for remnant groups of sea otters—**Priority 1**

Locations of sea otters can be accurately determined through visual observations and plotted in a geographic information system. Efforts to date have largely consisted of plotting locations of otters encountered during population surveys. Those data provide little information on how animals are using the habitat for specific behaviors (e.g., foraging, resting, socializing), and on what features of that habitat are making it particularly suitable. Additional insights could be gained through following individuals over time and identifying specific areas and specific types of habitat used for specific kinds of behavior. Such work could employ visual scan sampling or telemetry. A variety of techniques (e.g., SCUBA sampling, remote sensing) should then be applied to describe the physical and biological attributes of those areas.

2.1.2. Explore the feasibility of creating more areas that can serve as refuges—**Priority 2**

Certain features appear to define those habitats where sea otters are now most commonly found. These include very shallow constrictions leading to protected waters (e.g., Clam and Shagak lagoons at Adak Island), highly convoluted shallow nearshore habitats with close access to the supratidal, and kelp beds associated with shallow-water benthic features. Such features can be interpreted as providing refuge from predation by limiting access of killer whales. However, the relationships between habitat features and sea otter density are not well understood. Efforts to improve this understanding could include analyses of existing survey information on the current spatial distribution of sea otters and information on the physical attributes associated with that distribution. Such analyses would likely benefit from the types of work identified under 2.1.1. It may prove feasible to modify some existing habitats and thus enhance the features responsible for creating what is considered refuge from predation.

2.1.3. Protect important habitats—**Priority 2**

Specific relationships between habitats and use (or avoidance) by sea otters in the Aleutian Islands following the decline are not well understood. The actions identified under 2.1.1 should improve understanding of habitat features and sea otter use. If it is determined that certain habitat features provide sea otters with refuge from predation, this could lead to the identification of habitats and habitat features that merit additional assessment and protection. Habitat protection could be accomplished through educational efforts, consultation, and regulation. It should be noted, however, that the identification and protection of habitats where sea otters presently persist may not be adequate, by itself, to achieve recovery criteria, as the available evidence suggests those specific habitats are rare and constitute a relatively small proportion of what is considered typical sea otter habitat.

2.2. Prepare a habitat conservation plan under Section 10 of the ESA for Alaska state-managed fisheries—**Priority 3**

The majority of sea otter habitat in southwest Alaska is located within State of Alaska waters, i.e., from the mean high tide line seaward to a distance of three miles. Habitat conservation plans provide private landowners, corporations, and state or local governments with a means to obtain permits for activities that might incidentally take ESA-listed species. The development of a habitat conservation plan for sea otters would allow FWS and the State of Alaska to work together to minimize the impact of commercial fisheries and other human activities on the recovery of sea otters in southwest Alaska. Direct impacts from commercial fisheries and other human activities are not believed to be a significant threat to recovery of the southwest Alaska DPS, therefore this item is a low priority.

2.3. Ensure adequate capabilities for response to oil spills—**Priority 1**

The U.S. Coast Guard and the Alaska Department of Conservation are responsible for oil spill prevention and response in Alaska. The FWS should work with those agencies to ensure that adequate resources are positioned in southwest Alaska to prevent spilled oil from reaching sea otters and their habitat, as well as to clean and care for any otters that become oiled by a spill. Given the extensive ship traffic and potential for oil exploration within the range of the DPS, and the known potential for sea otter mortality from oil spills, this item is a high priority.

3. Manage impacts of human uses

3.1. Ensure that Alaska Native subsistence harvest does not affect recovery

3.1.1. 3.1.1. Continue to estimate harvest levels through the FWS marine mammal marking, tagging, and reporting program—**Priority 3**

Section 10 of the ESA includes an exemption for subsistence harvest of a listed entity by Alaska Natives, provided that such taking does not materially and negatively affect the listed entity. In order to evaluate the impact of the subsistence harvest, it is necessary to monitor the number, age, sex, and geographic distribution of harvested sea otters. This is an ongoing responsibility of FWS, which operates the marine mammal MTRP. This program was initiated in December 1988 and operates using FWS base funding. As subsistence harvests of sea otters occur over a relatively small proportion of the range of the southwest Alaska DPS, this item is a low priority.

3.1.2. Quantitatively evaluate the impact of sea otter harvest on recovery—**Priority 2**

In areas where subsistence harvest occurs regularly and at levels likely to have more than negligible impacts (i.e., more than one or two animals every few years), quantitative analyses should be conducted to evaluate the possible population-level consequences of harvest mortality. Such analyses are of relatively low priority in all MUs west of Kodiak, where subsistence harvest is considered to be of low importance to recovery, but they are of moderate priority in the Kodiak, Kamishak, Alaska Peninsula MU, where it is considered to be of moderate importance to recovery and highly amenable to management. In this MU, the potential consequences of a given level of harvest should be evaluated regularly, and the expected impacts gauged over a range of spatial scales (from the local population up to the scale of the MU) and time frames. Analyses should use a formal demographically-based population model such as an age- or stage-structured projection matrix, parameterized with the best available estimates of vital rates for the specific area or habitat in question. Note that the spatially structured matrix model developed for the PVA (Appendix 2) constitutes just such an analytical framework. Use of spatially explicit demographic models is important because results can often be non-intuitive, and may not be well predicted simply by considering total mortality level as a fraction of existing population size.

The procedural approach to such an analysis is fairly straightforward. Briefly, a population vector is created by multiplying the best estimate of the current population size (for the area in question) by the stable stage distribution vector associated with the projection matrix (see Appendix B for a further description), and then using standard matrix multiplication techniques to project population dynamics (over a 10-25 year period) with and without incorporating the additional harvest mortality. Harvest mortality should be apportioned to age and sex classes according to actual reported ratios, and multiple iterations of the simulation should be run to account for environmental and demographic stochasticity (randomness), as well as the uncertainty associated with harvest levels, vital rate estimates, and initial population sizes. By comparing expected population growth trajectories and final population sizes with and without the harvest mortality incorporated, managers can evaluate the likely impact of harvest mortality on population persistence and/or probability of recovery. In particular, the relative importance of a given set of results can be evaluated by answering three questions: 1) what is the magnitude of the effect (i.e., the percentage decrease in expected growth rates associated with the harvest); 2) does the effect lead to a qualitative change in the expected outcome for the population (i.e., the population would have positive growth without harvest but negative growth with the harvest); and 3) what is the scale over which the effect occurs (i.e., what proportion of the MU would experience the effect in question)? Clearly, a harvest that results in a substantial decrease in population growth over a very large area would constitute the greatest cause for concern. However, in certain cases a significant impact that occurs only at a local scale – for example, at a single island within an island group – might still be important to the recovery of the entire management unit if that island happens to have a key “source population” (i.e., it is an area of net positive growth) and is surrounded by other islands with either declining or depleted otter populations.

3.1.3. Develop sea otter harvest management plan(s) with Alaska Native tribes and tribally authorized organizations—**Priority 3**

Although Section 10 of the ESA does not allow regulation of subsistence harvest prior to a determination that such harvest is materially and negatively affecting the species or population, FWS should work with individual Alaska Native tribes and tribally authorized organizations to develop local and regional sea otter management plans that minimize the impact of subsistence harvests on recovery. The current levels and geographic distribution of the harvest do not suggest that it is a significant threat to recovery of the DPS; therefore, this is a low-priority item.

3.1.4. Continue outreach to promote "males only" harvest—**Priority 3**

FWS has been working with MTRP taggers in southwest Alaska since 2006 to distribute handouts that request sea otter hunters to avoid harvesting female sea otters, especially ones with pups. The FWS should take advantage of other outreach mechanisms and opportunities to distribute this message to sea otter hunters.

3.1.5. Use pelts from stranded otters to reduce demand for subsistence-harvested animals—**Priority 3**

The sea otter stranding program is a potential source of pelts that are currently used for educational purposes. Over the past several years, the program has taken in far more pelts than have been distributed to educational institutions. One possible use of surplus pelts would be to offset the harvest of sea otters in southwest Alaska by making these pelts available to artisans for handicraft purposes. Current levels of subsistence harvest are not considered to represent a significant threat to recovery; therefore, this item is a low priority.

3.2. Ensure that incidental take in fisheries does not affect recovery

3.2.1. Monitor the level of incidental take in fisheries within the sea otter's range in southwest Alaska—**Priority 3**

The NMFS fishery observers have collected valuable data on the frequency of sea otter bycatch in numerous Alaskan fisheries. Their data show that trawl, pot, and longline fisheries conducted within the range of the southwest Alaska sea otter DPS operate in deep waters and rarely interact with sea otters. Data on bycatch collected by NMFS observers in southern Alaska Peninsula, Kodiak, and Cook Inlet waters have shown that sea otters encounter coastal salmon set and drift gillnets but that entanglement is infrequent and rarely lethal. Bycatch in Alaskan fisheries is currently considered a low threat to southwest Alaska sea otters and the monitoring of incidental mortality in fisheries by observers is therefore a low priority. However, rates may vary over time, and therefore FWS should support the continued collection of sea otter bycatch data in existing NMFS observer programs and develop programs for coastal fisheries in those parts of the southwest Alaska sea otter range that have had little or no prior observer coverage (e.g., northern Alaska Peninsula, Bristol Bay).

3.2.2. Quantitatively evaluate the impact of incidental take on recovery in each management unit—**Priority 3**

The number, age, sex, and reproductive status of animals killed incidentally in fisheries will dictate the overall impact of bycatch on the population. Also, because the potential exposure of sea otters to entanglement in Alaskan fisheries varies regionally, FWS should evaluate the frequency and demographic characteristics of lethal sea otter bycatch within each management unit. The data can then be used in a model as described under 3.1.2 to quantitatively evaluate the impact of incidental take on recovery. Existing observer data suggest that mortality rates are low; therefore, quantitative assessment of the impact of incidental take on recovery in each MU should be considered a low priority.

3.2.3. Develop programs to reduce incidental take in situations where it occurs—**Priority 2**

Alaskan salmon fisheries may operate in shallow coastal waters where sea otters aggregate. Once entangled, sea otters are often able to free themselves from surface-hung nets but they sometimes require human intervention. In those situations, the sea otter's fate is dependent on the fisherman's response. Therefore, FWS should develop outreach material for distribution to Alaskan salmon fishermen that 1) discourages them from setting gear in areas of high sea otter concentration and 2) encourages rapid response and proper techniques for safely releasing entangled otters. Because this represents an inexpensive means of reducing the mortality rate for entangled otters, it should be considered a moderate priority.

3.3. Ensure that entanglement in marine debris does not affect recovery

3.3.1. Encourage net recycling and environmentally responsible disposal—**Priority 3**

While there are no data to suggest that entanglement in debris has had an impact on the southwest Alaska sea otter DPS, it is well known that marine debris frequently causes deaths of other wildlife. In some other regions (e.g., Hawaii), public-private partnerships have formed that encourage and facilitate the proper disposal of nets and other potentially entangling materials. A similar program should be established in southwest Alaska.

3.4. Eliminate intentional illegal take

There is little evidence that illegal take of sea otters is a widespread threat in southwest Alaska. However, efforts to minimize and eventually eliminate this source of sea otter mortality should be pursued.

3.4.1. Establish an outreach program on the ecological role of sea otters in the nearshore marine ecosystem—**Priority 3**

Sea otters are often viewed as competitors for crab and shellfish resources that are taken for commercial and/or subsistence purposes. The role of sea otters in structuring the nearshore marine ecosystem is not well-known to coastal human communities in Alaska. The FWS should develop outreach materials for distribution throughout southwest Alaska that emphasize the importance of sea otters to overall health of the ecosystem.

3.4.2. Increase enforcement efforts to investigate and prosecute cases of illegal killing—**Priority 3**

The FWS OLE currently relies on citizens to report incidents of sea otters being killed illegally. FWS should increase law enforcement activity in southwest Alaska to investigate and prosecute cases of illegal killing of sea otters, in an effort to minimize and eventually eliminate this source of mortality. Illegal killing is not believed to be a significant threat to recovery of the DPS, so this is a low-priority item.

3.5. Evaluate potential impacts of scientific research activities—**Priority 3**

Permits for scientific research are processed by the FWS Division of Management Authority based in Arlington, Virginia. The potential impacts of research activities on sea otters in southwest Alaska should continue to be considered in the context of other human activities that have the potential to affect the DPS.

4. Disease

4.1. Evaluate the potential role of disease as a threat to recovery

To understand the possible role of disease in the decline, both living and dead sea otters should be examined. Examination of carcasses provides information on the demographic characteristics of the mortality event, and insights into the significance of different disease processes, anthropogenic contaminants, biotoxins, human interaction, predation, and nutritional factors (although some factors such as predation may be underestimated due to carcass consumption and loss). Concurrent study of live-captured animals provides critical information on temporal and spatial exposure to infectious disease, potential for isolation and characterization of infectious agents, measurement of physiological parameters of stress or disease, general health and body condition, contaminant exposure, and diet. Ideally, results of live-animal investigations should be integrated with the necropsy data to determine the relevance of the described disease agents and possible effects at the population level. Modeling efforts should be undertaken to both better understand and predict the influences of pathogens on sea otter population decline and recovery.

4.1.1. Expand the marine mammal stranding network in southwest Alaska—**Priority 2**

Examination of carcasses of dead sea otters can provide information on the causes of their deaths, and help to assess the importance of potential threats such as predation (action 5.1) as well as disease. Currently, most of the sea otter carcasses found in Alaska are from the Kachemak Bay area where there is a strong and well-supported stranding program and a relatively large human population. Other parts of Alaska, particularly the southwest, lack the human population base; also, the value of carcasses and mechanisms to collect, store, and transport them may have not been adequately communicated in such areas. The FWS should make sure that local residents and researchers working in southwest Alaska are aware of the interest in carcasses, and if transportation of carcasses is not practical, FWS should train local people on necropsy protocols. This could include not only sea otter researchers, but also other marine mammal scientists, FWS refuge staff, Alaska Natives, and others. Fresh carcasses could be frozen for later, more extensive, necropsies.

4.1.2. Use carcasses and tissues from subsistence-harvested animals to investigate disease processes—**Priority 3**

Currently, there is some collaborative work with Native hunters in the Kachemak and PWS areas to collect carcasses and tissues. Since very few sea otters from the southwest Alaska DPS are taken by Alaska Natives, effort in this area might not be very productive. Nonetheless, Native people who are on the water or the shoreline engaged in other subsistence activities could be very helpful in finding and collecting carcasses, and their participation in such efforts should be encouraged.

4.1.3. Analyze carcasses and tissues and report results—**Priority 2**

Full necropsies of sea otter carcasses, with subsequent analyses for diseases and contaminants, are time-consuming and expensive. Base support to continue this work should be provided through FWS and the USGS.

4.1.4. Conduct live-capture studies to examine health status of live sea otters—**Priority 2**

Morphometric and health screening information can be collected in conjunction with behavioral and demographic studies or whenever sea otters are captured and sedated. During capture and handling, researchers should adhere to a detailed and rigorous protocol for collecting morphological and health data. Such data are available from several sea otter populations over various time scales, and from populations of varying status relative to resource availability. A comprehensive analysis of existing datasets should be conducted, and the results should be used to aid in developing procedures for future data collection and interpretation. Additionally, specific efforts should be initiated to acquire morphological and health data from all Alaska sea otter stocks. Because health issues are evidently not a major factor in the southwest Alaska population decline but may impede recovery, this action is considered a moderate priority.

4.1.5. Evaluate population-level effects of identified diseases—**Priority 2**

Determining whether disease has an effect at a population level is extremely important and will require close collaboration between biologists working on population demography and pathologists/veterinarians working on health and disease topics. For example, information on what segment of the population appears to contribute most to a decline, either through mortality or reduced reproductive success, can help identify which potential disease processes to investigate and which can be ruled out. Comparisons of a disease agent across populations that are in states of decline and increase, along with retrospective and prospective studies, can help determine whether the disease agent is most likely endemic or epidemic.

4.2. Develop disease management plans where practical—**Priority 3**

In most cases, very little can be done to alter a naturally occurring disease situation. However, understanding the epidemiology of a disease agent can sometimes point to potential management options. For example, restricting access of sea otters to fish waste was effective in reducing the incidence of gastric perforations due to Pseudoterranova decipiens. Other strategies might be available for other disease agents if they were better understood.

Disease management will be particularly important if translocations of otters are to occur. Precautions would have to be taken to avoid introducing disease agents to naïve groups of animals, particularly if those groups are small and concentrated in small areas. Major considerations in such disease management plans should include understanding what disease agents are endemic in the southwest Alaska DPS as well as thoroughly screening any introduced animals to avoid introducing a new disease agent to the area. Populations that are reduced to small numbers in restricted areas are particularly susceptible to effects from the introduction of a new disease agent. If morbilliviruses are determined to be a significant risk in these cases, vaccination of the remnant populations should be considered.

5. Predation

5.1. Continue to evaluate the role of predation as a threat to recovery—**Priority 1**

Because elevated mortality from predation has been identified as the most likely cause of the decline in southwest Alaska sea otters, and because predation has been ranked high as a threat to recovery for three of the five MUs (it is ranked as a moderate threat to the Bristol Bay and Kodiak, Kamishak, Alaska Peninsula MUs – see section IV), continued research aimed at understanding patterns, causes, and consequences of predation on sea otters in southwest Alaska is a high priority. While predation by killer whales is thought to be the primary driver of the decline in the Western and Eastern Aleutian MUs, it is important to note that the persistence and eventual recovery of these already-depleted stocks could be affected by predation mortality from species other than killer whales. In particular, the potential impacts of predation by sharks (on all age classes) and eagles (on pups) should be evaluated.

Some basic information on the occurrence of predation may be gathered by examination of sea otter carcasses (item 4.1.1). A more complete understanding of the patterns, causes, and consequences of predation will require two types of dedicated research: studies focused on sea otter populations themselves and studies focused on the predators. In the case of sea otter-focused studies, research on behavior, habitat use, dispersal characteristics, and population biology/demography (including measurements of survival and reproductive parameters) will be important for answering five general questions: 1) when and where predation mortality has occurred or is occurring, and the per-capita rate of predation mortality in those areas; 2) how sea otter behavior and habitat use patterns have changed (or are changing) in response to predation; 3) whether refuge from predation mortality is afforded by low population density or occupation of certain types of habitat, and thus a predictable functional relationship exists between density and per-capita predation rate; 4) whether Allee (underpopulation) effects occur at very small population sizes, and the nature of such effects; and 5) whether small or low-density populations occupying refuge habitats are sustainable, and at what rate dispersal occurs between such remnant populations.

The second focal area for further research involves studies aimed at better understanding the predators (or potential predators) of sea otters, especially killer whales. In particular, data are needed on the population structure of the predators, as well as their range use patterns, diet composition, and prey consumption rates. It has been demonstrated that only a small number of killer whales (one or a few pods) could account for the majority of the sea otter mortality needed to explain the decline. If that is the case, it may be difficult to identify those individuals from the total population of transient (mammal-eating) killer whales in the North Pacific Ocean. Diet studies of killer whale pods may be informative here, although given the currently low numbers of sea otters remaining in the area of decline, the potential for failing to detect sea otter consumption would be high, even if it were still occurring at a high enough frequency to be demographically significant for the greatly reduced population. Perhaps more importantly, a clear understanding is needed of how the population of transient killer whales is structured spatially and of the typical movement patterns of transient pods. Such information will be important for interpreting the geographic scope of the decline and predicting likely recovery patterns.

5.2. Develop predation management plans, where practical—**Priority 1**

For the most part, the number of feasible management actions that could be undertaken to mitigate predation mortality on sea otters is quite limited. In the unlikely event that it could be determined exactly which killer whales or other predators were responsible for the majority of the sea otter predation, and if the decline continued to the point of uplisting the DPS to endangered or spread to other DPSs, then the option of lethal or non-lethal removal of the specific individuals might be considered. However, this option is likely to be inadvisable under almost any imaginable scenario, chiefly due to the likelihood of misidentifying the responsible individuals, or failing to detect all the responsible individuals . Another management approach would involve natural or man-made “predation refuges,” defined as locations where sea otters are either free from predation risk or at much lower predation risk due to the limited ability of killer whales (and/or other predators) to gain access or to hunt there effectively. Potential refuges include shallow lagoons or rocky-reef areas, as well as embayments with limited or only shallow-water access to the open ocean. Survey data suggest that some such areas within the area of decline already function as refuge habitats; sea otter numbers have either not declined or have declined more slowly in those areas. If sea otter numbers continue to drop in southwest Alaska, management plans could include the construction of more refuge habitats (for instance, by constructing "predator exclusion devices" across narrow channels, or otherwise altering certain areas of habitat to restrict access by killer whales and/or other sea otter predators).

6. Protect from other natural or human-caused threats

6.1. Ensure that oil spills do not impede recovery of sea otters and/or negatively affect the nearshore marine environment in southwest Alaska—**Priority 2**

The risk posed to sea otters by exposure to spilled oil is well known: surface contact and ingestion of oil from grooming and consuming contaminated prey will have both immediate and chronic impacts on otters. Thousands of sea otters died due to the *Exxon Valdez* oil spill in PWS, and at least a few from the southwest Alaska DPS died from the Selendang Ayu spill in the Aleutian Islands. Ensuring that oil spills do not have an impact on southwest Alaska sea otters or their habitat is a high priority. Oil spill planning and response are primarily the responsibility of the U.S. Coast Guard and the State of Alaska Department of Environmental Conservation. The FWS should work with those agencies to: 1) minimize risks to sea otters by routing tankers and other vessel traffic away from nearshore waters used by sea otters; 2) provide relevant agencies with information on sea otter distribution and essential sea otter habitat for their use in oil spill response plans; and 3) ensure that adequate resources are positioned in southwest Alaska to prevent spilled oil from reaching otter habitat and to clean and care for otters that become contaminated. Additional measures to protect sea otters and the nearshore marine environment from oil spills may also be developed.

6.2. Establish an outreach program to mariners on how to avoid striking sea otters—**Priority 3**

Available data suggest that otters are seldom struck by boats, and such instances would be expected to be rare in most of southwest Alaska because of the low level of vessel traffic. Therefore, actions to avoid such strikes are a low priority. However, there is some potential for strikes near boat harbors, popular fishing areas, etc., and actions should be taken to prevent them. The FWS should develop outreach materials to inform mariners about the potential for hitting sea otters, and provide suggestions for ways to minimize the likelihood of strikes (e.g., traveling at slow speed in areas where otters are sighted). This information should be made available to the public by posting at boat harbors, in trade journals, in fishing regulation books, etc.

6.3. Continue to measure and monitor contaminant levels in sea otters—**Priority 3**

There is no evidence that contaminants are having an impact on sea otters in southwest Alaska at the present time. Measuring contaminant levels in otter tissues is therefore a low priority. However, tissue samples should be collected whenever possible and archived for possible future contaminant analysis. It is essential that proper protocols be used for sample collection and storage.

6.4. Monitor occurrence of biotoxins in sea otters and their prey—**Priority 3**

Biotoxins are not known to cause mortality in southwest Alaska sea otters but very little testing has been conducted. Biotoxins such as domoic acid from phytoplankton blooms occur off the west coast of the U.S., where they have caused mortality in both sea otters and pinnipeds. Monitoring for potential biotoxin impacts on sea otters in the southwest Alaska DPS is a low priority. However, those responsible for managing Alaska's coastal marine resources and ecosystems should be vigilant for signs of biotoxin impacts on sea otters or their prey. The FWS should consult with appropriate researchers and produce a summary of likely symptoms of biotoxins in sea otters, and make that information available to people who are in a position to observe otters regularly in the wild (e.g., biologists, stranding network participants, and Alaska Native hunters). People should be prepared to collect and submit samples to evaluate whether an algal bloom is occurring where wildlife mortality is observed.

6.5. Evaluate the feasibility of translocating sea otters to enhance recovery—**Priority 2**

If the decline of sea otters in southwest Alaska continues, it is possible that densities in some areas will become low enough to depress reproductive success, and that gaps in the range will form that are difficult to fill in by natural immigration. Methods for capturing and translocating sea otters have been developed, and translocation has been used to re-establish populations in areas where sea otter were extirpated. Evaluation of the feasibility of translocating sea otters to enhance recovery of the southwest Alaska population is a medium priority. The first step in such an evaluation would be to develop a decision matrix to help determine when circumstances are such that translocation should be considered. The decision matrix should be used in combination with a geographic and demographic model of the population decline to project when and where translocation would be a useful recovery tool. If a decision to translocate is made, additional feasibility factors will need to be considered, such as availability of potential source populations, potential genetic consequences of moving animals, and whether threats to otters in the potential translocation area have been mitigated sufficiently.

6.6. Evaluate potential impacts of recreational activities, tourism, and other forms of direct human disturbance—**Priority 3**

Human activities that occur in nearshore marine waters throughout southwest Alaska have the potential to take sea otters through disturbance. The nature and level of activities are not well known, and therefore the potential impacts are unclear. Comprehensive studies of boating, tourism, and other human activities that could cause disturbance would help identify areas where mitigation measures may be necessary. At present, however, such studies are a low priority.

7. Implement the recovery program for southwest Alaska sea otters

Implementation of the recovery actions in this plan will require coordination between multiple agencies and organizations, including Federal and State agencies, private groups, Alaska Native tribes, and tribally authorized organizations.

7.1. Maintain the Southwest Alaska Sea Otter Coordinator position within FWS—**Priority 2**

As the federal agency with management responsibility under the ESA and MMPA, FWS should designate a Recovery Coordinator and ensure that this position is adequately funded. This item is a medium priority.

7.2. Continue and enhance coordination of management efforts among FWS, other Federal agencies, Alaska Natives, and the State of Alaska—**Priority 2**

The Recovery Coordinator should work closely with various divisions within FWS, other federal agencies, the State of Alaska, and Alaska Natives to implement management actions for the recovery of the southwest Alaska DPS. This includes implementation of Section 7 (Interagency Cooperation) and Section 6 (Cooperation with the States) of the ESA and Section 119 of the MMPA (Marine Mammal Cooperative Agreements in Alaska). This item is a medium priority.

7.3. Continue and enhance coordination of research efforts among FWS, USGS, other Federal agencies, the State of Alaska, Alaska Natives, academic institutions, and others—**Priority 2**

Additional research into threats to the recovery of the southwest Alaska DPS is needed to help guide future management actions. The Recovery Coordinator should facilitate communication and cooperation among the various research agencies and institutions conducting studies of sea otters in southwest Alaska. This item is a medium priority.

7.4. Develop and continue a program of outreach to stakeholders—**Priority 3**

The results of research studies and management actions should be made available to agencies and organizations that are not directly involved in these activities. Management and policy decisions made by FWS relative to this DPS should be regularly communicated to stakeholders. The Recovery Coordinator should take a lead role to develop outreach and educational materials for distribution to stakeholders and other interested parties. Potential outreach opportunities include the use of internet web pages and list servers, presentations at public meetings, and FWS Refuge Information Technicians, to name a few. Because at this time there are few detrimental interactions between people and sea otters, this item is a low priority.

7.5. Secure adequate funding for southwest Alaska sea otter management and research needs—**Priority 1**

This recovery plan identifies research and management actions with annual total costs ranging from $2.665M -$3.650M over the first five years. One of the greatest challenges will be to secure funding needed to implement the plan. The principal Federal agencies involved (FWS and USGS) should commit adequate funding to support their in-house management and research activities as identified in this plan. The Recovery Coordinator should work with a broad range of partners to develop proposals for research and management activities identified in the plan and ensure that the highest-priority activities receive funding. As the ultimate success or failure of the recovery program depends on the ability to implement the plan, this item has high priority.

8. Implementation Schedule

A. Key to Responsible Agencies

ADEC	Alaska Department of Environmental Conservation
ADFG	Alaska Department of Fish and Game
ANO	Alaska Native Organizations
ASLC	Alaska SeaLife Center
AVPS	Alaska Veterinary Pathology Services
FWS	U.S. Fish and Wildlife Service
NMFS	National Marine Fisheries Service
USGS	U.S. Geological Survey

B. Recovery Implementation Schedule

Plan Action		Priority	Responsible Agency	Action Duration	Fiscal Year Costs ($K)					Comments
					FY 1	FY 2	FY 3	FY 4	FY 5	
1.	**Population Monitoring**									
1.1.	Continue to estimate sea otter population size and trends in southwest Alaska									
1.1.1.	Evaluate and improve methods for abundance surveys	2	USGS	2 years	125	125				Estimate detection
1.1.2.	Conduct surveys in each management unit and estimate abundance and population trends	1	FWS USGS	annually	200	200	200	200	200	Surveys will be conducted in different management units
1.1.3.	Identify areas of consistently high sea otter abundance, relative to other areas, within each management unit	2	FWS USGS	annually	5	5	5	5	5	This action will rely on survey data collected under 1.1.2
1.1.4.	Evaluate potential biases in survey data and identify methods that may be useful for correcting such biases	2	USGS	2 years	10	10				
1.1.5.	Refine estimates of equilibrium density and carrying capacity for the Bristol Bay, South Alaska Peninsula, and Kamishak, Kodiak, Alaska Peninsula MUs	1	FWS USGS							Timing is dependent on collection of additional survey data for these MUs
1.1.6.	Evaluate overall monitoring program on an ongoing basis to assess sensitivity	1	FWS USGS	annually	5	5	5	5	5	This action will rely on survey data collected under 1.1.2

B. Recovery Implementation Schedule

Plan Action		Priority	Responsible Agency	Action Duration	Fiscal Year Costs ($K)					Comments
					FY 1	FY 2	FY 3	FY 4	FY 5	
1.2.	Estimate vital rates									
1.2.1.	Estimate birth rates	3	USGS	4 years						Obtained in 1.2.2 at no additional cost
1.2.2.	Estimate age- and sex-specific survival rates	1	USGS	4 years	375	375	375	175		Includes $100K charter costs in years 1-3. Year 4 is for analysis and reporting
1.3.	Monitor health and body condition									
1.3.1.	Monitor general body condition	3	USGS FWS							Obtained in 1.2.2 at no additional cost
1.3.2.	Analyze data for evidence of food limitation	2	USGS	3 years	60	60	60			
1.4.	Monitor distribution and movements	1	USGS FWS	4 years						Obtained in 1.2.2 at no additional cost
1.5.	Continue development of population viability analysis models									
1.5.1.	Investigate PVA models for the Bristol Bay, South Alaska Peninsula, and Kamishak, Kodiak, Alaska Peninsula MUs	1	USGS							Timing is dependent on collection of additional survey data for these MUs
1.5.2.	Revise and re-run the population viability analysis incorporating new data	1	USGS						5	

Plan Action		Priority	Responsible Agency	Action Duration	Fiscal Year Costs ($K)					Comments
					FY 1	FY 2	FY 3	FY 4	FY 5	
1.6.	Monitor the functional role of sea otters in ecosystems									
1.6.1.	Monitor status of the kelp forest ecosystem	1	USGS	3 years	125		125		125	
1.6.2.	Develop methods to monitor sea otter impacts on marine ecosystems with soft-sediment substrates	2	USGS	2 years	125	125				
1.7.	Develop new research tools									Priority and costs not assigned
2.	**Habitat needs and habitat protection**									
2.1.	Identify important habitats or areas of special biological significance									
2.1.1.	Identify characteristics of areas that are serving as refuges for remnant groups of sea otters	1	USGS FWS	3 mo.	30					
2.1.2.	Explore feasibility of creating more areas that can serve as refuges	2	USGS FWS	6 mo.		50				
2.1.3.	Protect important habitats	2	FWS ADFG	annually	20	20	20	20	20	
2.2.	Prepare a habitat conservation plan under Section 10 of the ESA for Alaska State-managed fisheries	3	FWS ADFG	2 yr.		50	50			
2.3.	Ensure adequate capabilities for response to oil spills	1	FWS USCG ADEC	annually	20	20	20	20	20	Partial salary for USFWS Oil Spill Coordinator

B. Recovery Implementation Schedule

Plan Action		Priority	Responsible Agency	Action Duration	Fiscal Year Costs ($K)					Comments
					FY 1	FY 2	FY 3	FY 4	FY 5	
3.	**Manage impacts of human uses**									
3.1.	Ensure that Alaska Native subsistence harvest does not affect recovery									
3.1.1.	Continue to estimate harvest levels	3	FWS	annually	100	100	100	100	100	
3.1.2.	Quantitatively evaluate the impact of sea otter harvest on recovery	2	FWS USGS	6 mo.		50				
3.1.3.	Develop sea otter harvest management plan(s)	3	FWS ANO	annually	50	50	50	50	50	
3.1.4.	Continue outreach to promote “males only” harvest	3	FWS ANO	annually	5	5	5	5	5	
3.1.5.	Use pelts from stranded otters to reduce demand for subsistence-harvested animals	3	FWS	annually	10	10	10	10	10	
3.2.	Ensure that incidental take in fisheries does not affect recovery									
3.2.1.	Monitor the level of incidental take in fisheries within the sea otter’s range in southwest Alaska	3	NMFS	annually	1,200	1,200	1,200	1,200	1,200	Cost to conduct a fishery observer project
3.2.2.	Quantitatively evaluate the impact of incidental take on recovery in each management unit	3	FWS	1 mo.			10			
3.2.3.	Develop programs to reduce incidental take in situations where it occurs	2	NMFS FWS	3 mo.				25		

B. Recovery Implementation Schedule

Plan Action		Priority	Responsible Agency	Action Duration	Fiscal Year Costs ($K)					Comments
					FY 1	FY 2	FY 3	FY 4	FY 5	
3.3.	Ensure that entanglement in marine debris does not affect recovery									
3.3.1.	Encourage net recycling and environmentally responsible disposal	3	NMFS	annually	20	20	20	20	20	
3.4.	Eliminate intentional illegal take									
3.4.1.	Establish an outreach program on the ecological role of sea otters in the nearshore marine ecosystem	3	FWS ASLC	1 year			25	10	10	
3.4.2.	Increase enforcement efforts to investigate and prosecute cases of illegal killing	3	FWS	annually	75	75	75	75	75	
3.5.	Evaluate potential impacts of scientific research activities	3	FWS	annually	10	10	10	10	10	
4.	**Disease**									
4.1.	Evaluate the potential role of disease as a threat to recovery									
4.1.1.	Expand the marine mammal stranding network in southwest Alaska	2	FWS ASLC	annually	10	10	10	10	10	
4.1.2.	Use carcasses and tissues from subsistence-harvested animals to investigate disease processes	3	FWS AVPS	annually	30	30	30	30	50	Increased costs in FY5 for analysis and reporting
4.1.3.	Analyze carcasses and tissues and report results	2	FWS	annually	20	20	20	20	70	Increased costs in FY5 for analysis and reporting

B. Recovery Implementation Schedule

Plan Action		Priority	Responsible Agency	Action Duration	Fiscal Year Costs ($K)					Comments
					FY 1	FY 2	FY 3	FY 4	FY 5	
4.1.4.	Conduct live-capture studies to examine health status of live sea otters	2	FWS USGS ASLC	6 mo. Per year		400	400	400	500	Increased costs in FY5 for analysis and reporting
4.1.5.	Evaluate population-level effects of identified diseases	2	FWS						20	Postponed due to limited sample size at present
4.2.	Develop disease management plans where practical	3	FWS						25	Postponed due to limited sample size at present
5.	**Predation**									
5.1.	Continue to evaluate the role of predation as a threat to recovery	1	USGS ASLC	2 years		150	150			
5.2.	Develop predation management plans, where practical	1	FWS NMFS	3 mo.	20					
6.	**Protect from other natural or human-caused threats**									
6.1.	Ensure that oil spills do not impede recovery of sea otters and/or negatively affect the nearshore marine environment in southwest Alaska	2	FWS ADEC	annually	10	30	10	10	10	Costs in FY2 include incorporation of sea otter critical habitat into spill response plans
6.2.	Establish an outreach program to mariners on how to avoid striking sea otters	3	FWS	annually	10	10	10	10	10	

B. Recovery Implementation Schedule

Plan Action		Priority	Responsible Agency	Action Duration	Fiscal Year Costs ($K)					Comments
					FY 1	FY 2	FY 3	FY 4	FY 5	
6.3.	Continue to measure and monitor contaminant levels in sea otters	3	FWS	1 yr.					100	Postponed due to limited sample size at present
6.4.	Monitor occurrence of biotoxins in sea otters and their prey	3	FWS	3 yr.		100	100	100		
6.5.	Evaluate the feasibility of translocating sea otters to enhance recovery	2	FWS USGS ASLC	6 mo.		40				
6.6.	Evaluate potential impacts of recreational activities, tourism, and other forms of direct human disturbance	3	FWS	annually	5	5	5	5	5	
7.	**Implement the recovery program for southwest Alaska sea otters**									
7.1.	Maintain the Southwest Alaska Sea Otter Coordinator position within FWS	2	FWS	annually	100	100	100	100	100	
7.2.	Continue and enhance coordination of management efforts among FWS, other federal agencies, Alaska Natives, and the State of Alaska	2	FWS	annually	15	15	15	15	15	
7.3.	Continue and enhance coordination of research efforts among FWS, USGS, other federal agencies, the State of Alaska, Alaska Natives, academic institutions, and others	2	FWS USGS ADFG ANO	annually	15	15	15	15	15	

B. Recovery Implementation Schedule

Plan Action	Priority	Responsible Agency	Action Duration	Fiscal Year Costs ($K)					Comments
				FY 1	FY 2	FY 3	FY 4	FY 5	
7.4. Develop and continue a program of outreach to stakeholders	3	FWS	annually	10	10	10	10	10	
7.5. Secure adequate funding for southwest Alaska sea otter management and research needs	1	FWS USGS	annually	10	10	10	10	10	
SUBTOTAL	1			785	760	885	410	365	
	2			515	1075	655	620	765	
	3			1,525	1,675	1,710	1,635	1,680	
TOTAL				2,825	3,510	3,250	2,665	2,810	

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Appendix A. Ecosystem-based Recovery Criteria

Background

Recovery criteria for ESA-listed species are commonly based on estimates or measures of population size, status, and viability. These criteria can be as simple as the minimum number of individuals required to achieve a low probability of extinction over some sufficiently long period of time. Typically though, delisting criteria are more complex than this, including considerations of distribution and range, population age and sex structure, meta-population structure and dynamics, unforeseen or anticipated future risks, and trends in these various metrics and concerns over time. Regardless of the details, the key determinants for delisting are nearly always fundamentally demographic in nature.

Suitable habitat is also a necessary condition for the recovery of any species (Noss et al. 1997). This unarguable fact has been codified in the ESA through the requirement to designate critical habitat in the ESA listing process. The essential underlying notion for a critical habitat designation, however, is for the protection of physical and biological features that are essential to the conservation of the listed entity. Although habitat considerations are necessary components of any ecosystem-based recovery criterion, they are insufficient in and of themselves. This is because while the designation of critical habitat makes allowance for effects of the environment on a listed species, it makes no allowance for the species' effects on its environment.

There is growing evidence for the importance of individual species in ecosystems (Karieva and Levin 2004). Each species can exert a variety of effects on the wider biological community, not only by providing material resources and energy to the next-higher trophic level, but also through such well-known processes as competition, predation, mutualism, and the alteration of physical habitat. In some cases, a species' ecological role may be so strong that it influences entire landscapes and associated patterns of biodiversity. Such species are known as "strong interactors" (Paine 1992, Power et al. 1996) and have been variously referred to as "keystone species" (Paine 1969), "foundation species" (Dayton 1972), and "ecosystem engineers" (Jones et al. 1994), and in various other terms that imply functional importance (Soulé et al. 2003).

Not all species are strong interactors. However, strong interactors are often large vertebrate predators (Bascompte et al. 2005). These tend to be the same species that are most vulnerable to extinction because of their intrinsically low population densities, extensive habitat requirements, and generally low reproductive potential, and that occur in disproportionately large numbers on the list of threatened and endangered species (Ray et al. 2005). The so-called minimum viable population size is an inadequate recovery criterion for such species because in many cases, such a population size is likely insufficient to restore the species' ecological functionality. What is needed instead is recovery to an "ecologically effective population size" (Soulé et al., 2003, 2004).

Establishing delisting criteria based on ecological effectiveness presents additional challenges to the recovery planning process. At a minimum, the functional effects of the species on the ecosystem must be understood, and the relationship between the species' abundance and its functional role must be both known and measurable.

Sea otters have well-documented and widely known effects on coastal ecosystems. Lowry and Pearse (1973) were the first to demonstrate significant impacts of sea otter predation on the distribution and abundance of sea urchins and abalones. These direct effects were subsequently found to extend across successively lower trophic levels, indirectly influencing the distribution and abundance of kelp (Estes and Palmisano 1974). This process of interaction occurs eastward along the North Pacific rim from at least the Commander Islands through the Aleutian archipelago and across southeast Alaska and British Columbia (Estes and Duggins 1995, Watson 1993). Top-down forcing processes of this general kind, known as "trophic cascades" (Paine 1980, Carpenter and Kitchell 1993), occur widely in nature (Pace et al. 1999).

The presence or absence of sea otter predation (and the resulting trophic cascade) drives rocky reef ecosystems to two distinctive "phase states" (sensu Hughes 1994, Steneck et al. 2003)—kelp forests where sea otters are present in sufficient abundance to limit urchin biomass, and deforested sea urchin barrens where they are not. The discrete and dichotomous nature of these phase states on shallow rocky reefs is maintained by the fact that the endpoints in the potential continuum of variation between a kelp forest and a sea urchin barren are stable, whereas intermediate community configurations are not. Urchin barrens tend to be self-sustaining because urchins switch their foraging behavior from being sessile, "sit-and-wait" consumers of detrital fallout from the kelp canopy when a kelp forest is present, to being mobile hunters when it is not (Harrold and Reed 1985, Konar and Estes 2003). Additionally, individual kelp plants, once established, physically repel marauding urchins via a "whiplash effect" that occurs through the interplay of wave surge and the plants' flexible morphology (Konar 2000, Konar and Estes 2003).

The sea otter-induced trophic cascade has numerous effects on other species and ecosystem processes that permeate coastal food webs in diverse ways. For example, the surface tension exerted by kelp on the water column attenuates waves and coastal currents, thus influencing sediment deposition and rate of coastal erosion (Jackson 1984, 1997). Kelp forests are more productive than urchin barrens, fixing an estimated 3-4 times more inorganic carbon per unit area through photosynthesis (Duggins et al. 1989). This process in turn fuels elevated secondary production, resulting in increased growth rates and elevated population levels of various consumer species (Duggins et al. 1989, Estes 1996, Estes et al. 2004). For instance, the rock greenling, a numerically dominant kelp forest fish in the central and western Aleutian Islands, occurs at 10-fold higher population densities in sea otter-dominated kelp forest systems than in sea otter-impoverished urchin barrens (Reisewitz et al. 2005). The indirect effects of sea otter predation affect the behavior and foraging ecology of other coastal marine wildlife, for example glaucous-winged gulls (Irons et al. 1986) and bald eagles (Anthony et al. 2008). Many additional indirect effects may be unrecognized or are undocumented.

Sea otters also forage in soft-sediment (sand, gravel, and mud) habitats. Their known effects in these systems are mainly reductions in infaunal prey populations (especially bivalve mollusks) and sediment bioturbation. Such effects have been reported from central California (Kvitek et al. 1988), southeast Alaska (Kvitek and Oliver 1992, Bodkin et al. 2007) and the Kodiak archipelago (Kvitek et al. 1992). Indirect effects of sea otter predation in soft-sediment systems, while likely important, are largely unstudied.

The questions

Development of recovery criteria based on a species' impact on its ecosystem requires the ability to 1) identify the process or processes of primary interest, 2) define a functional relationship between the outcomes of this/these process/processes and the focal species' population size, 3) establish a state or range of states of the ecosystem that constitutes/constitute ecosystem recovery, and 4) develop quantitative measures for classifying a given region as "recovered" or "not recovered". Even for sea otters and kelp forests, this process is fraught with complexity. For example, should the delisting criterion be based on rocky reef systems, soft-sediment systems, or some combination of the two? Should the delisting criterion be founded on a simple ecosystem metric (e.g., prey abundance) or a more complex one involving all or some combination of the known indirect effects of sea otter predation? For those cases in which the response is a graded function of sea otter density, how does one choose a specific point along this response function as the delisting criterion? And finally, how does one account for the spatial and temporal variation in the chosen ecosystem state variable that is independent of sea otter predation?

Many of these difficulties can be resolved by focusing the ecosystem-based delisting criterion on the sea otter-urchin-kelp trophic cascade, or more specifically on the abundance of kelp and urchins in rocky reef habitats. There are a number of reasons for this. One is that kelps and urchins are broadly distributed, abundant, and easily measured and monitored. Another is that while the abundance of kelps and urchins vary substantially within and among sites (Estes and Duggins 1995) and even at specific sites through time (Watson and Estes, unpubl. manuscript), the configuration of the ecosystem is defined by two distinct phase states—kelp forests or urchin barrens. This latter aspect of temperate reef ecosystem behavior simplifies the task of defining a delisting criterion. One simply needs to answer two straightforward, qualitative questions, as follows. For downlisting, has the ecosystem recovered to a kelp-dominated state?. For uplisting, has the ecosystem further degraded in terms of an increase in the extent of urchin barrens?

These questions are pertinent for several reasons. The predominant phase state in rocky reef habitats varies as a function of sea otter predation throughout and even well beyond the geographic range of the population decline in southwest Alaska. A database is under development on kelp and urchin abundance from the Aleutian Islands and elsewhere within the range of the decline. This database could easily be used to establish a monitoring program to define ecosystem phase states. Finally, rocky reef habitat is sufficiently common in shallow coastal waters from lower Cook Inlet through the western Aleutian Islands to permit analyses based on this habitat type throughout the current range of the sea otter population decline in southwest Alaska.

Approach

Information on the abundance of kelp and urchins in rocky reef systems of the central and western Aleutian Islands has been obtained from numerous islands and at various times since the mid-1980s. The sampling protocol for all of these data followed well-established procedures (Duggins and Estes 1995, Estes et al. 2004) as follows. A 1 km-square grid was superimposed on a map of each island,

and the intersections between the grid lines and the island perimeter were identified as potential sampling sites. A random subset of locations was then selected from this set of potential locations for field sampling. Sand- or mud-bottom habitats, which are relatively uncommon in the Aleutians, were not sampled and sampling was restricted to waters 6-8m deep. Although samples have been obtained from a range of depths, the patterns across depths are generally similar (except at depths < 1-2 m, where intense wave surge favors kelp assemblages, and at depths > 10-15m, where light attenuation limits the settlement and growth of kelps). Three to 31 sites were sampled per island, depending on weather and time restrictions.

Samples of the benthic community were taken at each site by placing a 0.5 x 0.5 m square quadrat at randomly selected intervals (range—1-10 diver kicks) along a transect parallel to the bathymetric contour on the seafloor. Attached kelps within the quadrat were counted and identified to species. Although size and morphology vary among kelp species, for simplicity the data were pooled and species or age differences in kelp biomass were not considered in the present analysis. Twenty quadrats were sampled per site, as determined by SCUBA-diving bottom-time constraints. All urchins within the quadrats were counted and collected, and their individual test diameters were later measured. Urchin sampling at each site continued until 200 individuals were obtained or 20 plots were sampled, at which point the collections from that particular quadrat were considered complete and further sampling was terminated. An allometric function relating urchin mass to test diameter was developed so that urchin biomass density could be estimated from animal density and population size structure (see Estes and Duggins 1995 for details).

The intent of these procedures was to obtain a representative view of reef communities at islands with varying sea otter densities in the Aleutian archipelago. Early data from islands with and without sea otters have been analyzed and are reported by Estes and Duggins (1995). Most of the data collected after 1990 are still unpublished, except for those reported by Estes et al. (1998) and Estes et al. (2004). The data in this Appendix were obtained between 1987 and 2006, during which time 34 island–by-year combinations were sampled (Table A-1). Average urchin biomass and kelp densities were computed for each of these island-year combinations. K-means Cluster Analysis (Jain and Dubes 1988) was used to analyze intrinsic groupings of the data, and to assign each island to one of the identified groupings. Linear Discriminant Analysis was then used to assess the effectiveness of the classification scheme, and to create a canonical discriminant function that could be used to assign new data points to one cluster (ecosystem state) or the other.

A logistic regression was used to analyze the relationship between ecosystem state, a discrete response variable categorized as 0 (urchin-dominated) or 1 (kelp-dominated), and sea otter density (number of animals counted per km surveyed). This analysis was restricted to islands and years for which there were both benthic survey data and sea otter skiff counts (N = 17).

A bootstrapped re-sampling analysis was then applied to the full database (N = 463 sites at 34 island/year combinations, 10,000 iterations per sampling scenario) to determine the probability of correctly assigning islands to their ecosystem phase states with increased numbers of sample sites per island (beginning with 1). Because sea otter collapse or recovery may occur asynchronously among islands, the re-sampling analysis was expanded to determine the number of islands that must be sampled to be assured, with less than 5% probability of error, that the majority of islands (i.e., > 50%) were either kelp-dominated or urchin-dominated. We assume a goal of being able to estimate the true proportion of recovered islands to ±10% with >95% accuracy. A threshold of 50% was selected as the only non-arbitrary criterion for ecosystem recovery short of requiring recovery at every island (in which case the necessary sampling effort would be less).

Results

Information on kelp density and urchin biomass collected in 1987 from five islands in the Aleutian archipelago at which sea otters were either at or near carrying capacity (Amchitka 1987; Adak 1987) or completely absent (Shemya 1987, Nizki 1987, Alaid 1987) sorted into essentially non-overlapping areas of the two-dimensional phase space (Figure A- 1). Islands with abundant sea otters consistently had low urchin biomass densities and high but variable kelp densities. Islands lacking sea otters, in contrast, consistently had low kelp densities and high but variable urchin biomass densities. These findings demonstrate that the rocky reef ecosystem structure (i.e., phase state) in the Aleutian Islands is predictable, depending on the presence or absence of sea otters (Figure A-1).

Similar information is now available for 33 additional island/year combinations across the Aleutian archipelago, bringing the total to 38. This database was reduced to 34 (Table A-1) by excluding the 4 island/time combinations with fewer than 3 sample sites. Some of these data were obtained from the original study sites at Shemya, Nizki, Alaid, Amchitka, and Adak islands; the remaining data were obtained from 12 new islands that have been sampled one or more times during the past two decades. These latter data are from islands at which the sea otter populations were in various stages of recovery or decline when they were sampled.

A notable pattern in the data is that the phase space distribution is very similar for all of the datasets, thus indicating the rarity or absence of intermediate community configurations associated with phase state transformations. The pattern holds regardless of whether the data are plotted as averages within sites (Figure A-2) or by averages within islands (Figure A-3).

These results lend further support to the conclusion that kelp forest phase state can be determined relatively easily with a high degree of confidence. Using K-means cluster analysis, each island/year combination was classified into one of two distinct clusters, corresponding to alternate phase states (Figure A-3). Discriminant analysis indicated that this classification was highly robust (jackknife classification accuracy = 100%, $F_{2,31}$ = 44.644, $P<0.001$), suggesting that the canonical discriminate function (DF = -1.8391+ 0.0087 × Urchin biomass density - 0.0593 × Kelp density) provides a reliable algorithm for assigning any future data to the appropriate ecosystem phase state (urchin-dominated vs. kelp-dominated)

An initial estimate of the population density of sea otters needed to bring about a phase shift from the kelp-dominated to the urchin-dominated phase state was determined by fitting a logistic regression to phase state (urchin barren = 0, kelp-dominated = 1) vs. sea otter density for the 17 island/year combinations for which all the required information was available. This analysis provided a threshold estimate of 6.3 sea otters km^{-1} of coastline (Figure A-5).

The sample sizes (number of sites per island and number of islands) required to detect the ecosystem phase state was determined by bootstrap re-sampling, drawing sites randomly with replacement from the entire dataset while incrementing both the number of islands sampled and the number of sites sampled per island. The resample was iterated 10,000 times for each island by site combination (beginning with 1 site at 1 island). Under the simplest scenario--that the entire archipelago is in one phase state or the other (approximately the current situation)--only 6 sites from any given island (or combination of islands) would be sufficient to provide a 95% probability of correct classification (Figure A-6, left panel). Additional islands would have to be sampled under the more complex scenario of different islands recovering at different rates (as might be expected, for example, if recovery occurred through outgrowth from one or several points of origin as opposed to uniformly across the archipelago). Under this latter scenario, the optimal sampling strategy will vary depending on the geographic pattern of recovery and the proportion of islands in each of the two phase states. The most conservative assumption (in terms of statistical power) would be that islands recover randomly and asynchronously, and that we wish to determine the point at which half of the islands have recovered: in this case, 23 islands would have to be sampled to provide a 95% probability of correctly estimating the true proportion within ±10% (Figure A-6, right panel).

Recommendations

The proposed ecosystem-based recovery criterion is that sea otters must be sufficiently abundant to either maintain or bring about a phase shift to the kelp-dominated state. The important question is “what sort of field monitoring effort would be required to establish the community phase state with a sufficiently high level of statistical confidence to serve as a recovery criterion?” The cluster analysis indicates little or no ambiguity in assigning islands to phase states with as few as three sites sampled per island (Fig 3). An independent resampling analysis supports this view by establishing that only 6 sample sites from a given island would be sufficient to determine the ecosystem phase state with 95% certainty for any given island (Figure 5).

The question of how many islands to sample is more complicated. Under a scenario of phase state uniformity throughout the archipelago, samples from just one island would be adequate to characterize the entire region. This is probably more or less the situation that presently occurs due to the collapse of sea otters across the Aleutian archipelago. However, spatial (i.e., inter-island) variability in phase state, as might occur if otter populations declined or recovered in a temporally asynchronous manner, would require the sampling of multiple islands. Here the question is how many islands to sample to determine the proportion that are kelp-dominated. If the true proportion is close to zero or one, the precision of the estimate is unimportant and thus a small number of sample islands would suffice. If, on the other hand, the true proportion is near 0.5, then a relatively large number of islands would need to be sampled to determine, with a reasonable degree of confidence, whether that proportion is greater than or less than 0.5. The resample analysis for this latter scenario indicates that 23 islands would have to be sampled to estimate the true proportion within ± 10%. Given that 6 sites can easily be sampled by 2 or 3 dive teams in a single day, this would require 23 days of ship time. However, with proper planning, this estimate could probably be halved (to 12 days) by sampling two or more nearby islands in one day. Furthermore, this entire hypothetical scenario is founded on the assumption that islands with different phase states occur randomly across the archipelago. This is a highly unlikely situation. It is much more likely that any such variation in island phase state would occur in large spatial aggregates, in which case many fewer islands would have to be sampled. A monitoring program in which 6 sites were sampled from 5-7 islands distributed more or less uniformly across the archipelago will probably be more than sufficient for future decision-making.

It is also likely that sites in close proximity to localized refuges with high sea otter densities will not be representative of larger areas. Such areas should be avoided in order to obtain a representative view of ecosystem recovery. In addition, the development of a sea otter delisting criterion based on the kelp forest/urchin barrens phase state would be most easily done as a time series of measurements from established monitoring sites. Such an approach would provide information on intrinsic spatial and temporal variation in rocky reef phase state structure and on temporal trends of the system toward becoming either more extensively urchin-dominated or more extensively kelp-dominated. It would also provide a spatially explicit understanding of ecosystem recovery.

Implications for monitoring and population assessment

The determination of sea otter population status based on counts or surveys is an expensive and time-consuming endeavor, producing data that are often difficult to interpret. Long time series of counts are required to establish population trends with reasonable levels of statistical confidence, and the criteria or criterion chosen for delisting will always be arbitrary to a certain degree because key demographic variables—distribution, abundance, and age structure—are continuous in nature. Although some direct measures of sea otter abundance are needed to confirm the status and trends of populations, the characterization of kelp forest phase state provides an easier, less costly, less arbitrary, more sensitive, and more reliable means of assessing recovery. This is due in part to the discrete nature of the phase states and in part to the fact that only a modest field-sampling effort is required to define the phase state in any given area with a high level of confidence.

Additional benefits of an ecosystem-based recovery criterion

Using the kelp forest phase state as a delisting criterion offers two further benefits. There are uncertainties in estimating the density of otters required to have a population that is demographically viable over time, and therefore incorporating an ecosystem-based recovery criterion provides an independent measure of how well the population is doing and serves as a further safeguard against extinction. In addition, kelp forests likely have been the "natural" state of coastal reef ecosystems in Alaska over recent evolutionary time (Steinberg et al. 1995, Estes et al. 2005). Of the two phase states, the kelp-dominated state should benefit the greater number of species and ecosystem processes.

Caveats

In June 2008 researchers extensively resampled long-term rocky reef monitoring sites at islands across the Aleutian archipelago from the Near Islands in the west to the Islands of Four Mountains in the east. Although the data from this effort have not yet been analyzed, all of the islands visited were extensively urchin-dominated, as expected from the low sea otter densities and the results reported herein. In August 2008 eight additional locations were sampled (using the same sampling protocol) from the Fox Islands in the eastern Aleutians to Katmai National Park on the eastern end of the Alaska Peninsula. Although these data also have not yet been analyzed, the sample sites east of the Fox Islands were almost entirely kelp-dominated, despite low sea otter densities comparable to the Aleutian Islands. These observations indicate that the relationship between sea otter density and the phase state varies considerably between the Aleutian Islands and the southern Alaska Peninsula, thereby suggesting that the proposed ecosystem-based delisting criterion might be achieved along the Alaska Peninsula at a sea otter density below that required for demographic viability (in contrast with the Aleutian Islands situation).

Most of the information used to define the relationship between sea otter density and ecosystem phase state (i.e., the logistic regression in Figure 5) is from areas in which the otters were either extinct, at or near K, or in decline. This function could very well differ for systems in which the sea otter populations were recovering.

Forces other than sea otter predation also cause phase state shifts. Such factors as storm surge (Ebeling et al. 1985), other predators (Steneck et al. 2003), fisheries (Steneck 1997), and urchin wasting disease (Lafferty 2004) have been shown to drive kelp forest phase shifts in other regions. Despite decades of research, none of these factors has been seen to operate in southwest Alaska, whereas a large body of research attributes phase shifts to changes in the intensity of sea otter predation. In the seemingly unlikely event that a phase shift did occur for reasons other than sea otter predation, this would probably be evident from the associated evidence. For example, urchin wasting disease is not size-selective; storm effects are restricted to exposed shorelines; effects of fisheries are evident from landings and other records; and any new urchin predator capable of driving a phase shift would likely be observed. In conclusion, the likelihood of failure to meet this ecosystem-based recovery criterion when the sea otter population had in fact recovered is minimal.

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Table A-1. Summary of the datasets used to examine the relationship between sea otter density and sea urchin biomass in the Aleutian Islands.

Island/Site	Year	N_Sites	Urchin biomass	Kelp density
Adak	1987	28	29.99	9.09
	1997	27	258.34	0.97
	1999	30	269.12	1.49
	2000	28	310.6	1.59
Agattu	1993	3	70.11	2.93
Akutan	1994	12	53.51	11.85
Alaid	1987	9	371.07	2.24
	1994	8	324.85	1.59
Amchitka	1987	29	31.19	5.33
	1999	31	454.81	0.36
	2000	11	361.53	0.24
Anangula	1994	4	1.15	17.01
Attu	1987	9	208.8	1.83
	1990	14	127.31	2.08
	1993	15	59.83	4.75
	2000	15	35.14	3
Bering	2006	19	31.65	3.76
Chuginadak	1994	12	450.01	0.79
Kiska	1993	11	30.57	18.51
	2000	20	581.89	1.61
Nizki	1987	7	561.93	1.19
	1994	7	249.99	3.34
	1997	7	245.33	4.36
Ogliuga	1994	4	211.89	1.01
Rat	1994	10	611.9	1.06
Seguam	1994	10	514.66	0.24
Shemya	1987	16	413.81	0.79
	1994	12	250.48	4.28
	1997	16	426.44	3.3
Tanaga Bay	1994	7	118.79	3.74
Tanaga, Hot Sp	1994	6	45.91	8.88
Umnak	1994	4	3.77	9.08
Unalaska	1994	12	4.73	25.25
Yunaska	1994	10	515.27	0.77

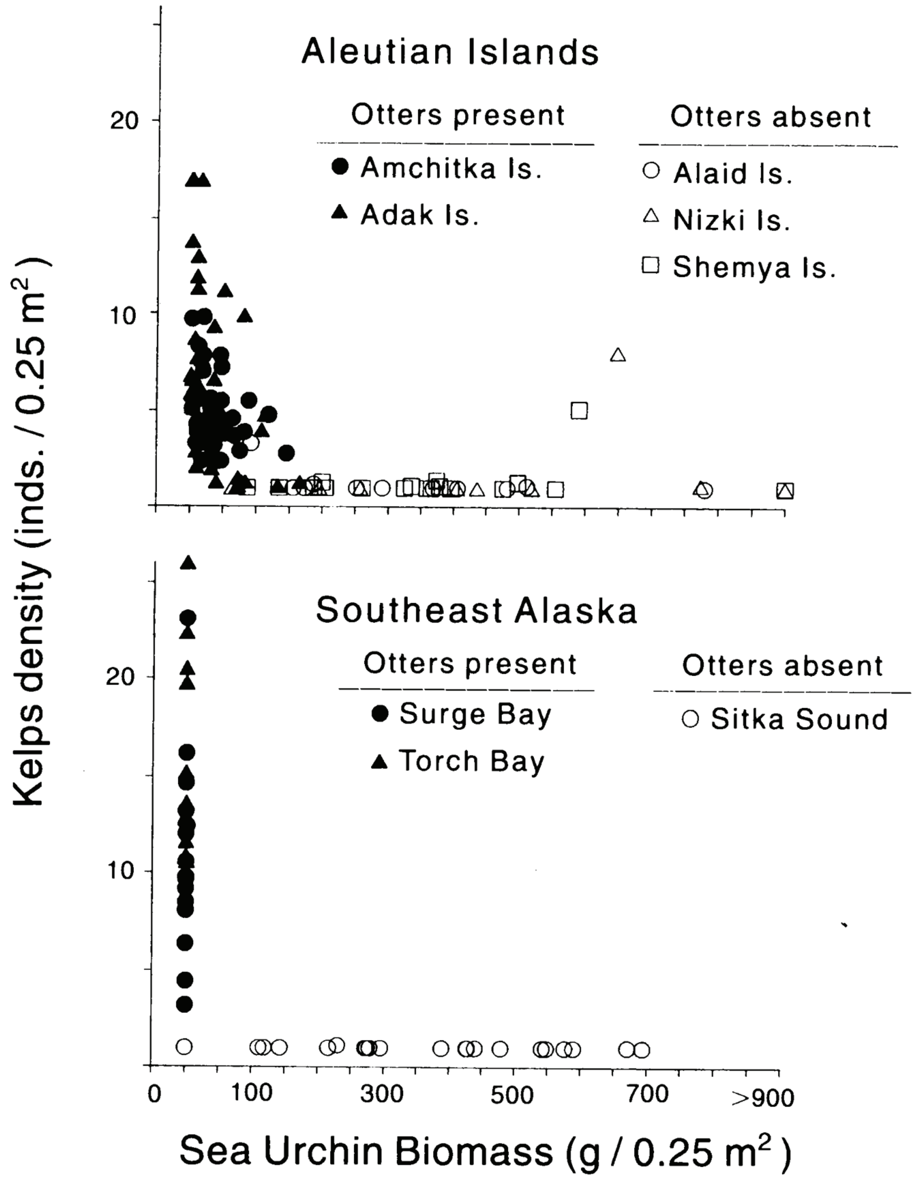

Figure A-1. Published data on kelp density vs. sea urchin biomass from islands in the Aleutian archipelago at which otters were abundant (Adak and Amchitka) or absent (Alaid, Nizki, Shemya).
This is a reproduction of Figure 7 from Estes and Duggins (1995). Lower panel provides comparable data from southeast Alaska.

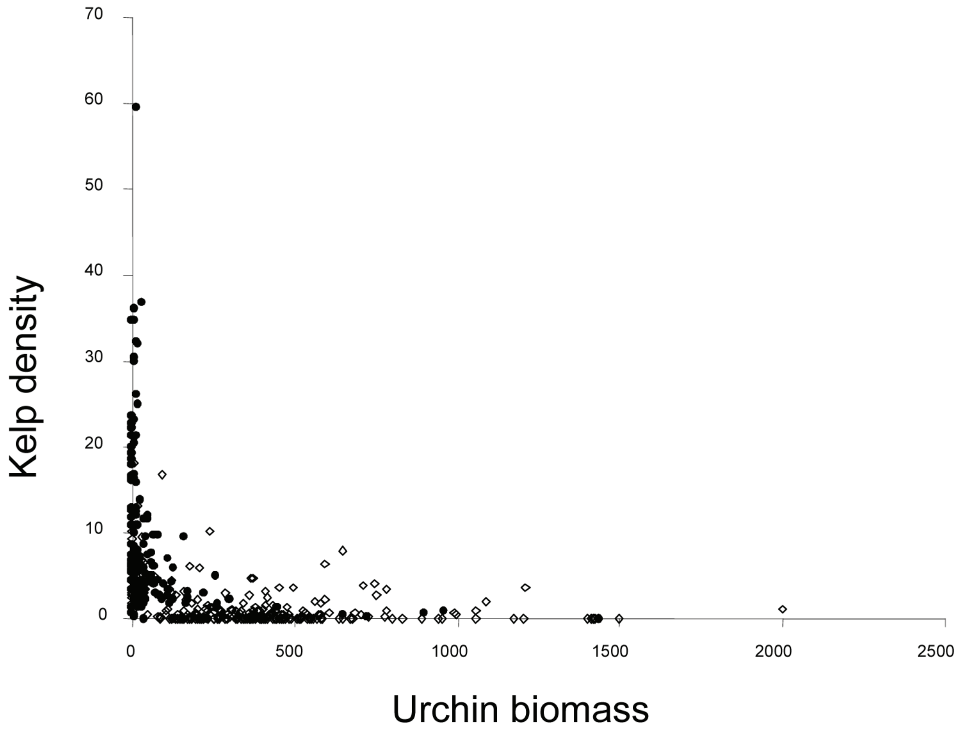

Figure A-2. Available data on kelp density vs. urchin biomass from the Aleutian archipelago, obtained from 19 islands sampled at various times between 1987 and 2006.
Symbol shape and shading represent different islands and times; open symbols are from lower otter density systems (<5 km); closed symbols are from higher otter density systems (>5/km).

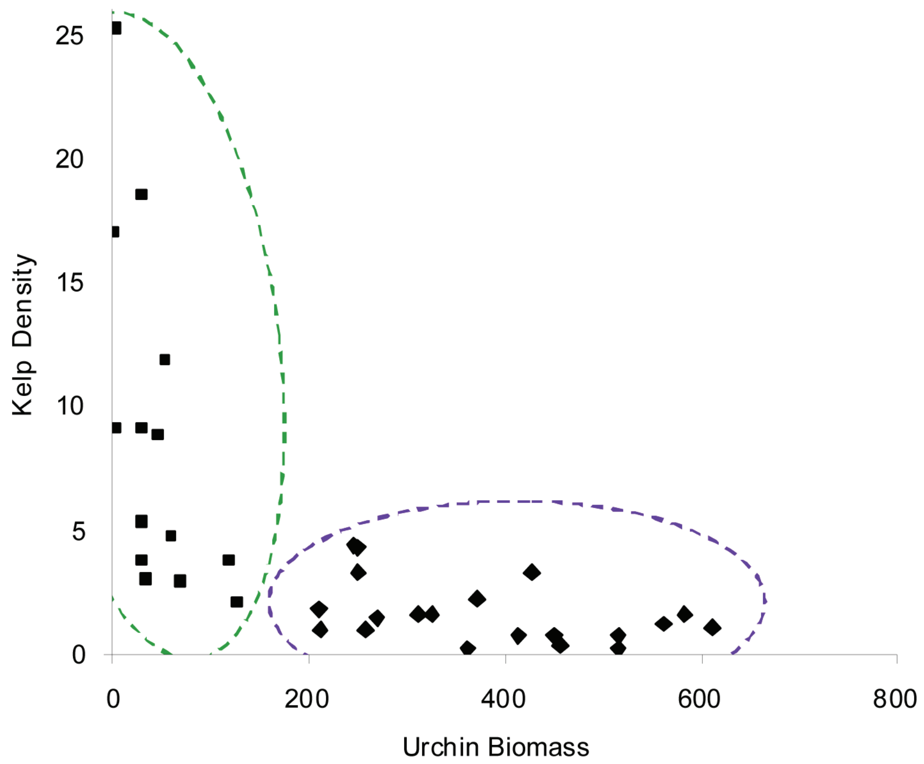

Figure A-3. Data from Figure A-2, averaged by island/year combination.
Dashed lines represent 90% confidence ellipses around the two data aggregates identified by K-means cluster analysis, which correspond with kelp-dominated (green, square symbols) and urchin barren (purple, diamond symbols) phase states. Note that each data point represents the average of all sites sampled at one island during one year. N = 34 island/year combinations representing 463 sites in total.

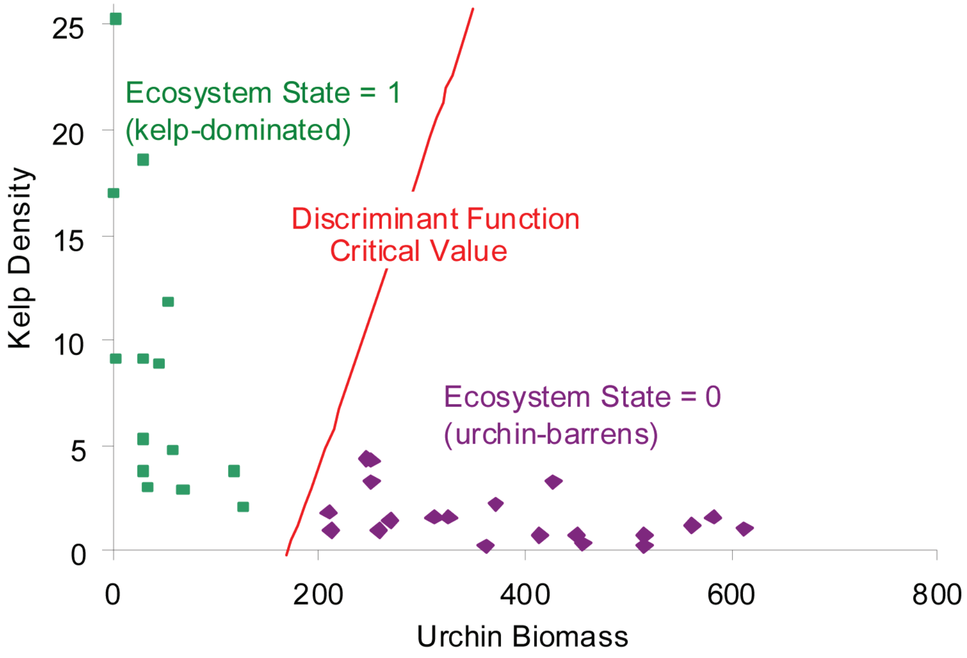

Figure A-4. Canonical discriminant function fit to database of kelp density vs. sea urchin biomass from the 34 island/year combinations currently available from the Aleutian archipelago (see Table A-1).
The function has a jackknife classification accuracy of 100% ($F_{2,31} = 44.644$, $P<0.001$). Thus, if DF > -0.295, the ecosystem state should be classified as 0 (urchin-dominated), otherwise the ecosystem state should be classified as 1 (kelp-dominated).

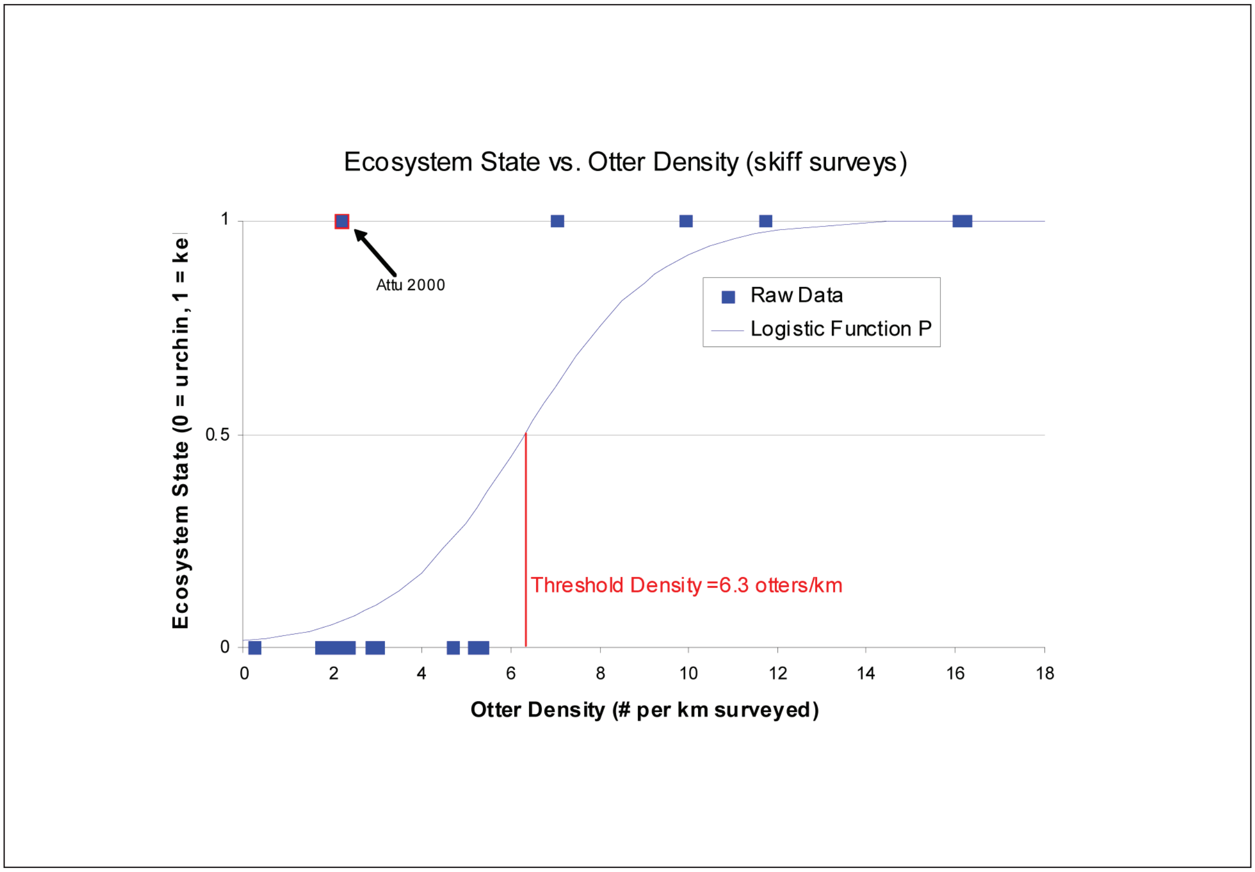

Figure A-5. Logistic regression of ecosystem phase state vs. sea otter density (using skiff surveys) as determined by data from 17 island/year combinations.
The one anomalous data point (Attu Island, 2000) is explained by the fact that all of the diving surveys that year were carried out in Massacre Bay, due to inclement weather and the short time window available for sampling. Massacre Bay receives an inordinately large input of fresh water from the Peaceful and Henderson river drainages, in turn apparently inhibiting sea urchin recruitment (and thus the phase shift from the kelp- to the urchin-dominated phase state) at the more protected inner-bay sites that were accessible to sampling.

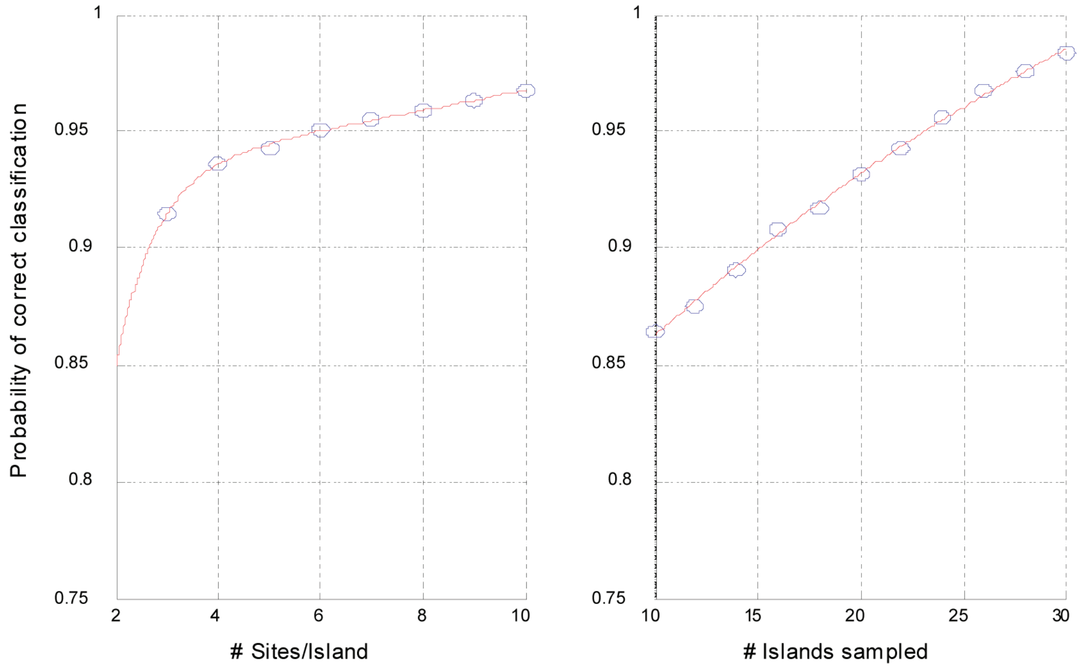

Figure A-6. Sampling intensity vs. statistical power in correctly classifying ecosystem phase state. The left panel shows the number of sites per island. The right panel shows the number of islands that would have to be sampled to estimate the phase state proportion within ± 10%, assuming the actual proportion was 50%.

Appendix B. Population Viability Analysis

Introduction

Population viability analysis (PVA) consists of a wide range of quantitative methods used to predict the probable future status of a population or populations of conservation concern (McCullough and Beissinger 2002, Morris and Doak 2002). PVA can be used to assess extinction risk within any specified time frame which can help guide management decisions. Since its inception in the late 1980s (Gilpin and Soulé 1986), PVA has developed rapidly and is a continually evolving process, incorporating new analytical methods as they become available. PVA is typically based on a computer model that simulates annual population processes such as births and deaths, incorporates various forms of uncertainty, and provides insights into the likelihood of attaining a specific goal, such as a specified population size, by some future date.

PVA can help meet many common objectives of endangered species management such as synthesizing all available information on the population or species of interest, ranking risks, and assessing impacts of habitat loss (Ralls et al. 2002). The interagency Quantitative Working Group has explored the possibility of using standardized PVA models as a guide to listing decisions (Regan et al. 2009). The Marine Mammal Commission has recommended the use of PVA for several purposes including the determination of species status under the Endangered Species Act and the Marine Mammal Protection Act, evaluating the effectiveness of past management actions, and predicting effects of proposed management actions (Marine Mammal Commission 2008).

Although PVAs can be performed by incorporating species-specific data into a generic PVA model such as VORTEX (Miller and Lacy 2005), it is generally preferable to develop a custom, species-specific model when there are sufficient data on a species or population (Marine Mammal Commission 2007). Sea otters in SW Alaska are an excellent candidate for developing a custom PVA because of the long history of study by academic groups and state and federal agencies, the existence of multiple time series on population trends, and information on abundance, demographic parameters, and density-dependent variation in vital rates.

The Recovery Team decided that a custom PVA for SW Alaska sea otters would be a useful tool both for developing delisting and uplisting criteria and for making future management decisions. Importantly, once a PVA framework has been designed, the model can be re-run as new survey data and other information become available, which could result in re-classification criteria that are more up-to-date and relevant. In this way the target numbers for delisting can be updated to reflect new information, while the underlying delisting criteria (e.g., less than 5% chance of becoming endangered within 25 years) is maintained. Because of the way in which a simulation-based PVA incorporates uncertainty, critical threshold values tend to be more conservative (higher) when there is more uncertainty in population parameters. Thus, the PVA is inherently precautionary, and also provides a strong incentive for regular data collection on the population and an adaptive management approach whereby the model is re-run as more information and better data become available. A recent example of a similar approach is the PVA developed for the draft Island Fox Recovery Plan (Bakker et al. 2009). Finally, a PVA provides a quantitative framework within which specific new threats can be evaluated with respect to their effect on overall population persistence, and specific management actions can be evaluated with respect to their efficacy. An excellent example of such uses is the recently completed PVA model for the Steller sea lion (Goodman 2008), which allows for a probabilistic evaluation of recovery scenarios and explicitly considers the role of uncertainty in making recommendations.

Methods

A variety of methods have been used in the design of PVA models, depending on the causes and patterns of variation in the focal population, the nature of threats to its persistence, and the availability of data (Beissinger and McCullough 2002). The simplest approach consists of the evaluation of variation in population abundance, and implications of this variation for viability, without any reference to population structure (variation in age, sex, size or life-history stage): this is sometimes referred to as a “count-based” PVA. A second approach involves the incorporation of some level of population structure (for example, a matrix model that incorporates stage-specific transition probabilities); this approach can be referred to as a “demographically-structured” PVA (Morris and Doak 2002). The benefit of the former type of model is that it involves very few parameters, requires little or no information about underlying population structure, and can be developed with a minimum of information (e.g., a time series of survey results). The benefit of the latter approach is that it can provide more robust and illuminating results in certain instances, assuming that there are sufficient data available with which to parameterize a structured population model. In particular, a demographically-structured PVA may be more appropriate in cases where population structure itself plays a role in determining the probability of persistence, as is the case when vital rates vary as a function of age or sex, and/or when demographic stochasticity becomes prominent at very small population sizes.

In the case of the sea otter in SW Alaska, demographic stochasticity is a very real concern due to the fragmented nature of small, remnant populations scattered throughout the Aleutian archipelago (Estes et al. 2005). Fortunately there are multiple sources of information about sea otter demography, age and sex-specific vital rates, and the ways in which these rates tend to vary as a function of population density (Eberhardt and Siniff 1988, Siniff and Ralls 1991, Eberhardt 1995, Monson 1995, Monson and Degange 1995, Monson et al. 2000, Tinker et al. 2006). Moreover, there is additional information on the nature of the elevated mortality that caused the recent population decline (Laidre et al. 2006); specifically, the decline was apparently driven by a dramatic increase in age-independent mortality, representing a fundamental and rapid switch from the age-varying mortality (primarily focused on juveniles and aged-adults) more typically associated with density-dependent population regulation in sea otters (Monson et al. 2000). In light of these facts, it was concluded that developing a demographically-structured PVA was both feasible and appropriate for this population.

Our overall approach to developing the SW Alaska sea otter PVA model can be broken down into five steps:

1. We develop a matrix-based population model to simulate dynamics in a population that is spatially structured into demographically distinct sub-populations, linked by dispersal.

2. Using time-series of population surveys available for various islands throughout the Aleutian archipelago, we use maximum-likelihood techniques to estimate the annual per-capita death rate associated with an age-independent source of mortality (e.g., predation) responsible for the recent decline, as well as variation in this parameter.

3. For each management unit, we simulate 25 years of population dynamics using model projections that incorporate a) density-dependent variation in vital rates and inter-island dispersal, b) variation in age-independent mortality (δ) and c) environmental and demographic stochasticity. To account for various sources of uncertainty, we run a large number of simulations and allow all parameters to vary according to the appropriate variance distributions.

4. The suite of simulations are re-run with 10 different starting densities, and for each initial density we record the proportion of simulations that go to extinction within a 25 year period (Ω). We then solve for the density at which $\Omega = 0.05$, the point at which there is $\geq 5\%$ probability of extinction within a 25 year period. This density represents the up-listing threshold, D^U.

5. The suite of simulations are again re-run with 10 different starting densities, and for each initial density we record ω, the proportion of simulations in which population abundance drops to D^U, the up-listing threshold. We then solve for the density at which $\omega = 0.05$; above this density there is <5% probability of the population becoming endangered (dropping below D^U) within 25 years. This density represents the de-listing threshold, D^D.

Step 1. Developing a Population Model
A spatially-structured demographic model has been recently developed for the southern sea otter, and provides a template for developing a PVA model for SW Alaska. The California model is based around a two-sex, age-structured Leslie matrix model with a yearly time-step (Tinker et al. 2006). A modification of this basic model incorporates spatial complexity by allowing for dispersal between a series of semi-discreet sub-populations (Tinker et al. 2008); below we describe how we have adapted this model for the SW Alaska sea otter DPS.

The model tracks annual demographic transitions for males and females, with two types of transitions possible: reproduction (R) and survival (s) – we refer to female survival as s_f and male survival as s_m. Reproduction in sea otters, while occurring year-round (Jameson and Johnson 1993), tends to be somewhat seasonal in SW Alaska (Monson 1995, Tinker and Estes 1996, Monson et al. 2000), with most adult females annually producing a single pup in late spring or early summer. Because of this seasonality, we are able to simplify the birth-flow life history to a birth-pulse model that tracks reproduction and survival from winter of one year to winter of the next. Specifically, we assume that reproductive females begin gestation at the start of the year, give birth to pups half way through the year (after a 6-month gestation period), rear their pups for the next 6 months, and, assuming the pup survives, wean at the end of the year. Following Tinker et al. (2006), the reproduction term in our model, R, represents the probability that a female of age i produces a single viable juvenile of either sex by the end of the year, and thus incorporates the age-specific annual birth rate (b_i), the probability that the pup is male or female (we assume a 1:1 sex ratio at birth), the likelihood the pup survives until weaning (given by the age-specific weaning success rate, w_i), and the probability that the mother survives the entire year (i.e., 6 months gestation plus 6 months pup dependency). Accordingly, we calculate reproduction for females of age i as:

$$R_i = s_{f,i} \cdot \frac{1}{2} b_i \cdot w_i \tag{1}$$

where $s_{f,i}$ is the age-specific survival rate of the mother. Note that equation 1 implies that juveniles actually begin their first year of independence at 0.5 years old; however, for simplicity of presentation we refer to this juvenile class as age 0 (thus $s_{f,0}$ represents survival from 0.5 to 1.5 years-of-age for recently-weaned juvenile females, $s_{f,1}$ represents survival of females from 1.5 to 2.5 years-of-age, and so on). Because there are few data available with which to parameterize vital rates for older animals in Alaska, we simplified our model to track just 12 age classes, with the 12th age class accounting for aged-adults ≥ 11.5 years-of-age (collapsing older age classes has little effect on the resulting projections of population dynamics because of the relatively small proportion of the population occurring in these age classes). The resulting survival and reproduction rates are combined into a population projection matrix, $P_{j,i}$, used to model demographic transitions to age class j from age class i for each sex (Table B-1).

In the Aleutian archipelago, each island or island group is assumed to represent a semi-discreet sub-population. The number of animals of each sex and age-class that are alive at the start of year t in sub-population x are tracked in a 24×1 population vector, $n_{x(t)}$, the first 12 cells of which tabulate individuals in each female age class, while the last 12 cells tabulate individuals in each male age class. The total abundance of the sub-population can be calculated as the sum of this vector, $N_{x(t)} = \Sigma n_{x(t)}$. When the total abundance of a given sub-population is large ($N_{x(t)} \geq 100$), annual population dynamics are calculated using standard matrix multiplication (Caswell 2001):

$$n_{x(t+1)} = n_{x(t)} \cdot P_{j,i} \tag{2}$$

where $P_{j,i}$ is the demographic transition matrix (Table B-1). When population sizes drop below 100, demographic stochasticity can result in a lower population growth rate than that predicted by matrix multiplication: accordingly, for small population sizes (<100) we use methods described by Morris and Doak (2002) to make calculations that explicitly incorporate demographic stochasticity in all vital rates.

To parameterize the above-described population projection model, we used published estimates of age- and sex-specific vital rates reported from radio-telemetry studies of sea otters in SW Alaska; in the few cases where parameter estimates were not available from Alaskan sea otter studies, we used comparable data collected from California sea otters. As with many species, sea otter vital rates exhibit negative density-dependence, with the result that population growth can be described approximately as a theta-logistic function of density (Figure B-1; Siniff and Ralls 1991, Eberhardt 1995, Tinker et al. 2006). Certain life-history stages are particularly labile: for example, weaning success and survival rates of juvenile age-classes tend to decrease substantially at high population density, while survival rates of prime-age adults show much less variation (Monson et al. 2000). Birth rates (b_i) also show little or no variation with density: females generally reach reproductive maturity between 2 and 4 years of age, and thereafter maintain a relatively constant rate of pup production irrespective of density. Most previous studies of reproduction in sea otters report a birth rate of 0.88 - 0.98 pups per female per year, with a slightly lower rate for younger females (Jameson and Johnson 1993, Eberhardt and Schneider 1994, Riedman et al. 1994, Monson et al. 2000, Tinker et al. 2006). For the current model we assume the age of first reproduction is 2.5 years ($i = 2$), and we set birth rate to 0.4 pups per year for $i = 2$, 0.7 pups per year for $i = 3$, 0.9 pups per year for $3 > i < 11$, and 0.8 pups per year for the final adult age class.

We incorporated density-dependent variation in weaning success and survival into our model by using published data sets on sea otter survival and reproduction to parameterize vital rates at low density and high density populations, and interpolating between these extremes to obtain appropriate values for intermediate densities. We used vital rate estimates from Kodiak Island in the late 1980s to characterize a low density, rapidly growing population (Monson and Degange 1995), while estimates from Amchitka Island in the early 1990s were used to characterize a high density population near carrying capacity, K (Monson 1995, Monson et al. 2000). Because the vital rate estimates for these populations were available for only certain year-classes, and were reported by life-history stage – juveniles, adults, aged-adults – rather than by annual year-classes, we first generated smoothed age- and sex-specific survival rate estimates by fitting 3-parameter logistic functions to all available data from each location:

$$s_{f,i} = \mathrm{e}^{\alpha_1+\alpha_2 \cdot i+\alpha_3 \cdot i^2} \; / \, (1+\mathrm{e}^{\alpha_1+\alpha_2 \cdot i+\alpha_3 \cdot i^2}) \tag{3}$$

where i is female age, and a_1, a_2 and a_3 are parameters fit by maximum likelihood (similar functions were used to re-calculate male survival and weaning success). Fitting equation 3 resulted in "inverted U" survivorship schedules, typical of sea otters and other large carnivores (Caughley 1966), with highest survival among prime age adults and density-dependent mortality concentrated in juvenile and aged-adult animals. The resulting age-specific vital rate estimates for a low density population (designated by superscript "L") and a high density population (designated by a superscript "H") are illustrated in Figure

2. When incorporated into the matrix model, the low-density parameter estimates values produced an algebraically-derived expected rate of growth (λ) of slightly less than 1.15, while the high-density values produced a rate of growth of slightly less than 1.00. By adjusting all weaning success values upwards slightly (by 1.1% for Kodiak data and by 1.5% for Amchitka data) we obtained $\lambda^L = 1.15$ and $\lambda^H = 1.00$: these two sets of vital rates were then used as bounding values for low density and high density populations (Table B-2). At intermediate densities we used a multiplier, v, to interpolate between the two sets of vital rates. Specifically, we describe the density of each sub-population, D_x, as a ratio of carrying capacity ($D_x=N_x/K_x$), and for a given value of D we interpolate female survival as

(4)
$$s_{f,i}(D) = s^H_{f,i} \cdot \nu(D) + s^L_{f,i} \cdot (1-\nu(D))$$

where:

(5)
$$\nu(D) = 1.005 \cdot D^{1.933}$$

Male survival and weaning success were interpolated in an analogous fashion. Equation 5 was parameterized such that variation in λ describes a theta-logistic function of density, with $\lambda_{max} = 1.15$, K = 15 sea otters per km^2 of shallow sub-tidal habitat (< 40m depth) (Burn et al. 2003) and theta = 2 (Figure B-1).

In addition to tracking within-population dynamics, the model must also account for age- and sex-specific dispersal that can demographically link sub-populations. Specifically, the numbers of animals of each sex and age-class that move from sub-population x to sub-population y in year t is tracked in a 24×1 vector, $m_{y,x(t)}$, and total dispersal is given by $M_{y,x(t)} = \Sigma m_{y,x(t)}$. Equation 2 is modified to account for dispersal, such that my,x(t) is subtracted from $n_{x(t)}$ (and added to $n_{y(t)}$) prior to matrix multiplication, thereby assuming (for computational simplicity) that dispersal occurs at the beginning of the year, followed by reproduction and survival at the new location. While this approach is computationally simple, a much greater challenge was parameterization of $m_{y,x(t)}$. It is known that sea otters are capable of making movements of hundreds of kilometers, although dispersal distance appears to follow a leptokurtic distribution such that very long-distance movements are rare (Tinker et al. 2008). Unfortunately, there is little information on the rate of long distance dispersal between sea otter populations in SW Alaska. No inter-island movements were observed over the course of 2-3 year radio telemetry studies at Adak and Amchitka Islands (Monson 1995, Tinker and Estes 1996). This observation, together with the spatially staggered pattern of population recovery in the Aleutian archipelago following cessation of the fur trade (Kenyon 1969), suggest that inter-island movements are relatively rare, stochastic events. Furthermore, it can be inferred from the pattern of sea otter re-colonization of the Aleutian archipelago (Kenyon 1969) and the Commander Islands (Bodkin et al. 2000) that the likelihood of inter-island movements is a decreasing function of the distance between islands and an increasing function of the relative density of the source population, D_x. Given the above observations, we concluded that a reasonable approach to modeling sea otter dispersal was to represent the total number of animals dispersing from island x to island y in year t ($M_{y,x(t)}$) as a random integer drawn from a Poisson distribution with parameter $\gamma_{y,x}$, where $\gamma_{y,x}$ is a function of inter-island distance ($\Delta_{y,x}$) and relative population density (D) at island x:

(6)
$$\gamma_{y,x} = \beta_1 \cdot D_x^{\beta_2} \Big/ \left[(1-\beta_3) + \beta_3 \cdot e^{(\Delta_{y,x})}\right]$$

Parameters β_1, β_2 and β_3 together determine the shape and slope of the relationship between dispersal rate, population density and inter-island distance. Equation 6 is flexible enough to produce a wide range of functional forms that generally decrease with distance and increase with density, and can be parameterized such that the total number of animals moving from island x to y in any given year will usually be 0 unless the islands are relatively close together and/or the population density at island x is near K. To estimate values for β_1, β_2 and β_3, we polled the recovery team to obtain expert opinion on the range of "biologically reasonable" dispersal rates under various combinations of inter-island distance and density (Table B-3). We then fit equation 6 to the upper and lower extremes of these ranges to obtain parameter estimates for high dispersal and low dispersal scenarios (Figure B-3).

When dispersal is non-zero ($M_{y,x(t)} > 0$), it is necessary to specify the age/sex distribution of the dispersing animals: the simplest way to achieve this is to scale $m_{y,x(t)}$ proportionally to $n_{x(t)}$, thereby assuming that each individual in the population is equally likely to disperse. However, it is well established that sea otter dispersal rates vary based on age and sex (Ralls et al. 1996, Tinker et al. 2008), and therefore we adjusted my,x(t) such that sub-adult males were two times more likely to disperse than adult males or sub-adult females, and five times more likely to disperse than adult females (sub-adults are defined as ages $0 \leq i \leq 2$, and adults are defined as $i \geq 3$). These ratios correspond roughly to published age/sex-specific dispersal kernels estimated for southern sea otters (Ralls et al. 1996, Tinker et al. 2008).

Step 2. Estimating age-independent mortality, δ
Time series of sea otter abundance were available for seven islands in the central/western Aleutian Islands, with each time series consisting of four or more skiff surveys conducted between 1991 and 2007 (Table B-4). A number of lines of evidence suggest that the sea otter populations at all these islands declined at approximately equal rates over this period (Estes et al. 2005), and that this decline was caused by the introduction of a new

source of age-independent mortality (Laidre et al. 2006). We reasoned that the per-capita death rate (δ) associated with this new mortality source could be quantified using the population model developed in Step 1. Specifically, we simulated population dynamics at each island from 1990-2007, with age-specific survival rates adjusted to reflect δ, and used maximum likelihood methods to minimize the deviation between expected and observed population trends by adjusting δ. This maximum likelihood approach allowed us to evaluate alternate scenarios of variation in δ within an information theoretic framework.

The skiff counts of otter abundance provided us with estimates of $N_{x(t)}^{obs}$, the observed abundance of population x at time t. For simplicity we designated 1990 as t=0, since the first population count was conducted in 1991. We initialized population vectors for each island population ($n_{x(0)}$) by multiplying the expected abundance in 1990 ($N_{x(0)}^{exp}$, itself a free parameter to be fitted) by the algebraically-derived stable stage distribution at K (thereby assuming that most of these populations were still near equilibrium densities in 1990). For each year after 1990 (t = 1, 2, …17) we generated expected values for population abundance, $N_{x(t)}^{exp}$, using matrix model projections (described in Step 1) and adjusting age-specific survival rates as follows:

(7)
$$s_{f,i}(D,t) = s_{f,i} \cdot (1 - \delta(D,t))$$

where δ(D, t) represents per-capita age-independent mortality, calculated as a logit-function of population density (D) and time (t). Equation 7 represents the joint probability of surviving density-dependent mortality factors (disease, starvation, etc.) and the new age-independent mortality source (hypothesized to be predation), and these two types of mortality are assumed to act independently. We evaluated nine different functional forms for δ(D, t), allowing for scenarios of constant mortality, time-varying mortality (both linear and non-linear functions), density-varying mortality (both linear and non-linear functions), and time/density interactions (Table B-5). For each functional form we solved for parameter values by minimizing deviation between expected and observed counts using maximum likelihood. Specifically, for each year at which a given island was surveyed, we calculated the negative log likelihood as:

(8)
$$L_{x,t} = \log(\sigma_t) + \frac{1}{2}\log(2\pi) + \frac{\left(N_{x(t)}^{obs} - N_{x(t)}^{exp}\right)^2}{\sigma_t^2}$$

where σ^2_t represents the variance due to observer error at time t (Hilborn and Mangel 1997), solved for iteratively following Pascual et al. (1997). We summed equation 8 across islands and survey years, and solved for the parameter values that minimized the total negative log likelihood. For this analysis we made the simplifying assumption that the primary source of variance in population estimates was observer-based error (sampling error), rather than process error (Tinker et al. 2006). Finally, we used Akaike's Information Criterion (AIC) to evaluate the relative level of support for each functional form (Burnham and Anderson 2002). The functional form with the lowest associated AIC was assumed to provide the best fit to the available survey data, but to account for model uncertainty we calculated AIC weights (Burnham and Anderson 2002) as measure of relative support for each functional form (higher AIC weights indicate greater support for a given functional for

Step 3: Population Simulations
Sea otter population dynamics for the Eastern and Western Aleutian Island management units were simulated for 25-year periods using the above-described population model. Simulations were iterated 1000 times for each parameter set, incorporating demographic stochasticity and dispersal stochasticity as described in Step 1, and with low-dispersal and high-dispersal scenarios (Table B-3) equally represented. The principal source of variance in simulation results was age-independent mortality (δ), which typically accounted for the majority of sea otter deaths in any given simulation. For this analysis we assumed that the range of past variation in δ provided the best available estimate of future variation; however, our estimates of δ included both sampling uncertainty and model-selection uncertainty. To account for these sources of uncertainty we used a boot-strap re-sampling technique, whereby 1,000 sets of 7 time series were drawn randomly with replacement (i.e., such that the same time series could be drawn multiple times) from the full set of 7 time series, and the model fitting described in Step 2 was repeated. The "best model" (the functional form with the highest AIC weight) differed between boot-strap iterations, as did the maximum likelihood parameter estimates, and the results from each iteration were used to generate estimates of δ for one 25-year simulation. The relative support for each functional form (averaged across all simulations) was calculated as the mean AIC weight. In those cases where the optimal functional form included one or more time-varying terms (Table B- 5), the value of "t" in δ(D, t) was represented as a random value between 1 and 17 (the total number of survey years), weighted such that more recent years had greater representation. Specifically, for each year of a given simulation, we drew a random value from a left-skewed beta-distribution, and re-scaled this value to obtain the number of years before present: the two parameters of the beta-distribution (φ_1 and φ_2) were themselves drawn randomly for each simulation (such that $0.5 < \varphi_1 < 1.0$ and $1.5 < \varphi_2 < 3.0$) to allow for a range of different weighting functions (Figure B-4). The value of D in δ(D, t) was itself an output of the population simulation, and thus varied between islands depending on local population dynamics. To allow for stochastic differences in annual per-capita mortality among

islands (i.e., environmental stochasticity), we added a random error term, $\varepsilon_{x(t)}$, to our estimate of $\delta(D, t)$ at each island, where $\varepsilon_{x(t)}$ had a mean of 0 and standard deviation of 0.02 (corresponding to the variance in estimates of $\delta(D, t)$obtained when Step 2 was repeated for each island separately).

Step 4: Determining the Up-listing Threshold
Up-listing from threatened to endangered status is recommended when there is an unacceptably high likelihood of extinction (defined by the Recovery Team as ≥ 5% probability) within the foreseeable future (defined by the Team as 25 years). We used population simulations to determine the threshold density (D^U) at which this condition is satisfied for a given management unit. Density was calculated as the number of otters per area of sea otter habitat, scaled relative to *K*, where sea otter habitat is defined as sub-tidal area between the coastline and the 40m isobath, and *K* was assumed equal to 15 otters per km^2 (as defined by Burn et al. 2003). The full suite of 1000 simulations (described in Step 2) was re-run 10 times, with starting density (D^0) ranging between 1% and 50% of K. Note that D^0 was specified as the average density for the management unit, but the actual distribution of otters between islands was randomized for each simulation. For each of the 10 values of D^0 we recorded Ω, defined as the proportion of simulations that went to quasi-extinction within the 25 year period. We define quasi-extinction as the point at which there are no islands remaining[1] with ≥5 females and ≥1 males. We then plotted Ω versus D^0, and fit an exponential function from which we calculated DU, the value D^0 at which $\Omega = 0.05$ and thus the density at which there is ≥ 5% probability of extinction within a 25 year period.

Step 5: Determining the De-listing Threshold
De-listing from threatened status is recommended when there is an acceptably low likelihood of becoming endangered (< 5% probability) within the foreseeable future (which we have defined as 25 years). We used population simulations to determine the threshold density at which this condition is satisfied for a given management unit. The full suite of 1,000 simulations (described in Step 2) was re-run 10 times, with starting density (D^0) ranging between D^U and 80% of *K* (as defined by Burn et al. 2003). For each of the 10 values of D^0 we recorded ω, defined as the proportion of simulations in which final density was less than or equal to DU. Using the same approach described in Step 4, we solved for the critical density, DD, at which $\omega = 0.05$; above this density there is < 5% probability of the population becoming endangered within 25 years.

Results

A per-capita annual rate of age-independent mortality (δ) varying between 0.15 and 0.35 resulted in expected population dynamics that closely fit observed time series of skiff surveys (Figure B-5). Note that age-independent mortality was estimated in addition to age-dependent mortality from all other sources. Of the nine functional forms evaluated for $\delta(D,t)$, the three models with most empirical support all included terms for both density and time (Figure B-6). The best-supported models consistently showed a decrease in δ over time, and an inverse relationship between δ and density (Figure B-7). However, the two functional forms with greatest support (Models 5, 7 and 9) also included a 2nd order density term that resulted in decreased values of δ at very low densities (Figure B-7,A and B). This dip in mortality at low densities may represent a "refuge effect", potentially resulting from localized habitat areas at some islands that provide a degree of refuge from predators (and/or other mortality sources) for small, local populations.

Simulations of future population dynamics generally indicated continued decreases in abundance, although stochastic differences between islands, as well as temporal variation in δ and density-dependent refuge effects, allowed for predictions of population stability or even increases at some islands (Figure B-8). Because of the prevailing negative trends in abundance, however, simulations with low starting densities frequently resulted in quasi-extinction of sea otters within a management unit during the 25-year simulation period. The relationship between Ω (the proportion of simulations going to quasi-extinction within 25 years) and D^0 (population density at the start of the simulations) was well described by a negative exponential trend (Figure B-9 A). The value of D^U, the threshold density at which $\Omega=0.05$, did not differ significantly between the Eastern Aleutian Islands management unit ($D^U = 3.65\%$ of *K*, 95% confidence limits = 3.58-3.95) and the Western Aleutian Islands management unit ($D^U = 3.84\%$ of *K*, 95% confidence limits = 3.57-4.12). Using the average upper 95% confidence limit as a conservative estimate of D^U, these results support an up-listing threshold density of approximately 4% of *K*, or 0.6 otters per km^2 of habitat.

We conducted a second set of simulations to determine an appropriate de-listing threshold density. As was the case for the up-listing simulations, the relationship between ω (the proportion of simulations in which population density for a management unit dropped below D^U) and D^0 (population density at the start of the simulations) was described by a negative exponential trend (Figure B-9 B). The value of D^D, the threshold density at which $\omega=0.05$, did not differ significantly between the Eastern Aleutian Islands ($D^D = 48.69\%$ of *K*, 95% confidence limits = 47.87-49.51) and the Western Aleutian Islands (DD = 47.36% of *K*, 95% confidence limits = 46.06-48.64).

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[1] In this context we use the term "island" to refer to a single island or island group that is geographically distinct from other such islands or island groups (i.e., separated by at least 10 km) and that has sufficient sub-tidal habitat to support an isolated sub-population of sea otters.

Using the average upper 95% confidence limit as a conservative estimate of D^D, our results support a de-listing density of 49% of *K*, or 7.35 otters per km^2 of habitat.

Discussion

The results of the SW Alaska PVA model described here can be used as a basis for recommending conservative, demographically-based up-listing and de-listing thresholds. The recommendations are conservative for two main reasons: first, because the many sources of uncertainty that have been explicitly incorporated into the model lead to substantial variation in simulation results (e.g., see Figure 8), and when combined with stringent probability requirements (e.g., less than 5% of simulations go to extinction in 25 years) this tends to elevate the up-listing and de-listing thresholds. The second main reason that the results are likely conservative is that the survey time-series used to parameterize mortality rates were collected for the most part during a period of population decline (although there have been slightly positive growth rates for the last few years at some islands). It is therefore not surprising that a fairly high proportion of future population projections also exhibit negative population growth, a fact that acts to elevate the up-listing and de-listing thresholds. It is worth emphasizing that the conservative nature of the PVA-based recommendations is entirely appropriate at this time, given the high degree of uncertainty and the prevailing negative population trends over the past 15-20 years. Both of these factors may change over the next 5-10 years. That is to say, the collection and analysis of more demographic and dispersal data may reduce the level of uncertainty in many parameters, and the incorporation of more years of survey data into model parameterization will potentially lead to a higher proportion of simulations with positive population growth. A key advantage of the PVA model structure developed and presented here is that it is both possible and advisable to update parameters and re-run the model as new data or information become available, thereby avoiding the inevitable obsolescence of fixed listing threshold recommendations. We recommend that this process be conducted at 5-year intervals, and the up-listing and de-listing thresholds be updated accordingly. It is worth noting that the probabilistic definitions of the thresholds will remain the same, it is only the numerical estimates that will change to reflect the most recent data.

The model results highlight a number of interesting aspects of the recent population decline that may have important implications for recovery. One of these aspects is the high degree of temporal and spatial consistency in the pattern of decline and (by extension) the source of mortality driving the decline (e.g., Figure B-5). Such patterns can be informative when evaluating various potential mortality sources in terms of their likely contribution to past declines and future population recovery. A second important pattern to consider is that the per-capita rate of mortality appears to have been greater at lower densities, but at very low densities there is indication of reduced mortality (Figure B-7). Again, this pattern can provide insight into potential sources of mortality, but also must be considered with respect to implications for recovery. Given the relative degree of support for predation as a causal factor, the density-dependent nature of the per-capita rate of mortality may be used to provide insight into the nature of predator functional responses. Moreover, if there is indeed a "refuge effect" at very low densities, population abundance should be expected to stabilize at many islands, particularly those with abundant protected habitat, with encouraging implications for the likelihood of population persistence. Survey results over the next 5-10 years will be crucial in evaluating whether this pattern continues to hold.

There are a number of additional benefits to the demographically-based PVA model presented here that have not been fully realized at this time. One of the benefits is that a comprehensive sensitivity analysis can be performed to explore the relative importance of various parameters in driving model results. We recommend that this exercise be conducted, however this will require additional time for programming, computation, and analysis. A second important benefit of a demographically structured, spatially-explicit model is that it is very amenable for use in exploring the population-level impacts of specific management actions or newly-emerging threats. For example, if there were a concern about an emerging threat (e.g., a new nearshore fishery or mineral resource extraction project), the model could be re-run under various scenarios of direct mortality at the location in question, as well as scenarios of habitat loss, and the effects on probabilities of population persistence at the level of the management unit could be evaluated. Incorporating such modifications into the existing model structure can be accomplished with relatively minor investment of time in additional programming and computer simulations, and will provide a means of objective evaluation of the importance of threats or the benefits of alternative management actions.

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Table B-1. Projection matrix used in population model to calculate demographic transitions (survival and reproduction) occurring between year t and year t+1.

		Female Age Classes												Male Age Classes											
		i = age at time t												i = age at time t											
Sex	j = age at time $t+1$	0	1	2	3	4	5	6	7	8	9	10	≥11	0	1	2	3	4	5	6	7	8	9	10	≥11
F	0	0	0	R_2	R_3	R_4	R_5	R_6	R_7	R_8	R_9	R_{10}	R_{11}	0	0	0	0	0	0	0	0	0	0	0	0
F	1	$s_{f,1}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	2	0	$s_{f,2}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
0	3	0	0	$s_{f,3}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	4	0	0	0	$s_{f,4}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	5	0	0	0	0	$s_{f,5}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	6	0	0	0	0	0	$s_{f,6}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	7	0	0	0	0	0	0	$s_{f,7}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	8	0	0	0	0	0	0	0	$s_{f,8}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	9	0	0	0	0	0	0	0	0	$s_{f,9}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	10	0	0	0	0	0	0	0	0	0	$s_{f,10}$	0	0	0	0	0	0	0	0	0	0	0	0	0	0
F	≥11	0	0	0	0	0	0	0	0	0	0	$s_{f,11}$	$s_{f,12}$	0	0	0	0	0	0	0	0	0	0	0	0
M	0	0	0	R_2	R_3	R_4	R_5	R_6	R_7	R_8	R_9	R_{10}	R_{11}	0	0	0	0	0	0	0	0	0	0	0	0
M	1	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,1}$	0	0	0	0	0	0	0	0	0	0	0
M	2	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,2}$	0	0	0	0	0	0	0	0	0	0
M	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,3}$	0	0	0	0	0	0	0	0	0
M	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,4}$	0	0	0	0	0	0	0	0
M	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,5}$	0	0	0	0	0	0	0
M	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,6}$	0	0	0	0	0	0
M	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,7}$	0	0	0	0	0
M	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,8}$	0	0	0	0
M	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,9}$	0	0	0
M	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,10}$	0	0
M	≥11	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	$s_{m,11}$	$s_{m,12}$

Table B-2. Vital rate estimates used to parameterize the projection matrix (Table B-1) for sea otter populations at low density and high density ("at K") situations.
The low density rates (estimated from sea otters at Kodiak Island in the late 1980s) result in an expected annual growth rate (λ) of 1.15, while the high density rates (estimated from sea otters at Amchitka Island in the early 1990s) result in an expected λ of 1.00. For populations at intermediate densities, vital rate parameter estimates are obtained by interpolation between these extreme values, such that population growth rates exhibit negative density-dependence as described by a theta-logistic model of population dynamics (see text for further details).

		Low-Density Population			High-Density Population		
Age Class	Birth Rate (b_i)	Wean Rate (w^L_i)	Female Survival (s^L_i)	Male Survival (s^L_i)	Wean Rate (w^H_i)	Female Survival (s^H_i)	Male Survival (s^H_i)
0	0.000	0.000	0.912	0.904	0.000	0.742	0.736
1	0.000	0.000	0.937	0.910	0.000	0.827	0.803
2	0.400	0.588	0.953	0.917	0.201	0.878	0.845
3	0.700	0.727	0.963	0.924	0.314	0.908	0.871
4	0.900	0.817	0.968	0.928	0.432	0.925	0.887
5	0.900	0.871	0.970	0.930	0.534	0.933	0.895
6	0.900	0.902	0.969	0.929	0.611	0.934	0.895
7	0.900	0.918	0.965	0.923	0.659	0.930	0.889
8	0.900	0.923	0.957	0.913	0.678	0.918	0.875
9	0.900	0.919	0.944	0.897	0.671	0.896	0.850
10	0.900	0.905	0.923	0.874	0.636	0.856	0.811
11	0.900	0.878	0.889	0.847	0.573	0.790	0.753
12	0.900	0.829	0.835	0.818	0.482	0.684	0.670

Table B-3. Estimates (based on expert opinion of the Recovery Team) of the number of sea otters dispersing from a source population on one island to a remote population on another island, as a function of the density relative to carrying capacity at the source population, and the distance between the two islands.
In all cases, the source population is assumed to consist of exactly 100 animals (before dispersal), so that the number of animals dispersing corresponds to the % of the source population. The two sets of estimates were used to fit functions corresponding to a low dispersal scenario and a high dispersal scenario, as described in the Methods section.

	Inter-Island Distance			
	25 km		50 km	
Percent of K	Lower	Upper	Lower	Upper
50	0.5	5	0.01	0.5
100	2	15	0.2	2.5

Table B-4. Time series of skiff survey counts of sea otters (excluding dependent pups) for seven islands in the Aleutian archipelago between 1991 and 2007.

These counts were conducted through the period of decline in this region, and were used in the current analysis to fit age-independent mortality rates.

	Island						
Year	Adak	Amchitka	Attu	Kiska	Little Kiska	Semichi Islands	Kagalaska
1991	3,494	-	-	-	-	-	-
1992	-	-	-	-	-	-	-
1993	2,200	1,308	-	949	146		191
1994	-	-	2,078	-	-	178	-
1995	898	-	-	-	-	-	-
1996	733	-	-	-	-	-	-
1997	688	-	-	237	38	-	26
1998	525	-	-	-	-	-	-
1999	635	246	-	-	-	-	-
2000	713	340	436	-	-	62	43
2001	515	-	-	-	-	-	-
2002	461	-	-	-	-	-	-
2003	306	235	106	170	13	19	11
2004	277	167	-	-	-	-	-
2005	209	74	118	114	14	5	-
2006	272	-	-	-	-	-	11
2007	231	155	262	114	26	18	-

Table B-5. Alternate functional forms evaluated for modeling δ, the per-capita rate of age-independent mortality.
The simplest function evaluated (Model 1) corresponds to a constant rate of mortality, while the remaining eight models represent linear or non-linear functions of density (*D*), time (*t*), or density and time. The symbols $\psi_1, \psi_2 \ldots \psi_n$ represent parameters fit by maximum-likelihood (see text for further details).

Model ID	Functional Form for *f(D,t)*
1	$f(D,t) = \psi_1$
2	$f(D,t) = \psi_1 + \psi_2 \times t$
3	$f(D,t) = \psi_1 + \psi_2 \times t + \psi_3 \times t^2$
4	$f(D,t) = \psi_1 + \psi_2 \times D$
5	$f(D,t) = \psi_1 + \psi_2 \times D + \psi_3 \times D^2$
6	$f(D,t) = \psi_1 + \psi_2 \times D + \psi_3 \times t$
7	$f(D,t) = \psi_1 + \psi_2 \times D + \psi_3 \times D^2 + \psi_4 \times t$
8	$f(D,t) = \psi_1 + \psi_2 \times D + \psi_3 \times t + \psi_4 \times t^2$
9	$f(D,t) = \psi_1 + \psi_2 \times D + \psi_3 \times D^2 + \psi_4 \times t + \psi_5 \times t^2$

In each case δ was calculated as a logit of f, the function listed in column 2 using the equation:

$$\delta(D,t) = {e^{f(D,t)}}/{1 + e^{f(D,t)}}$$

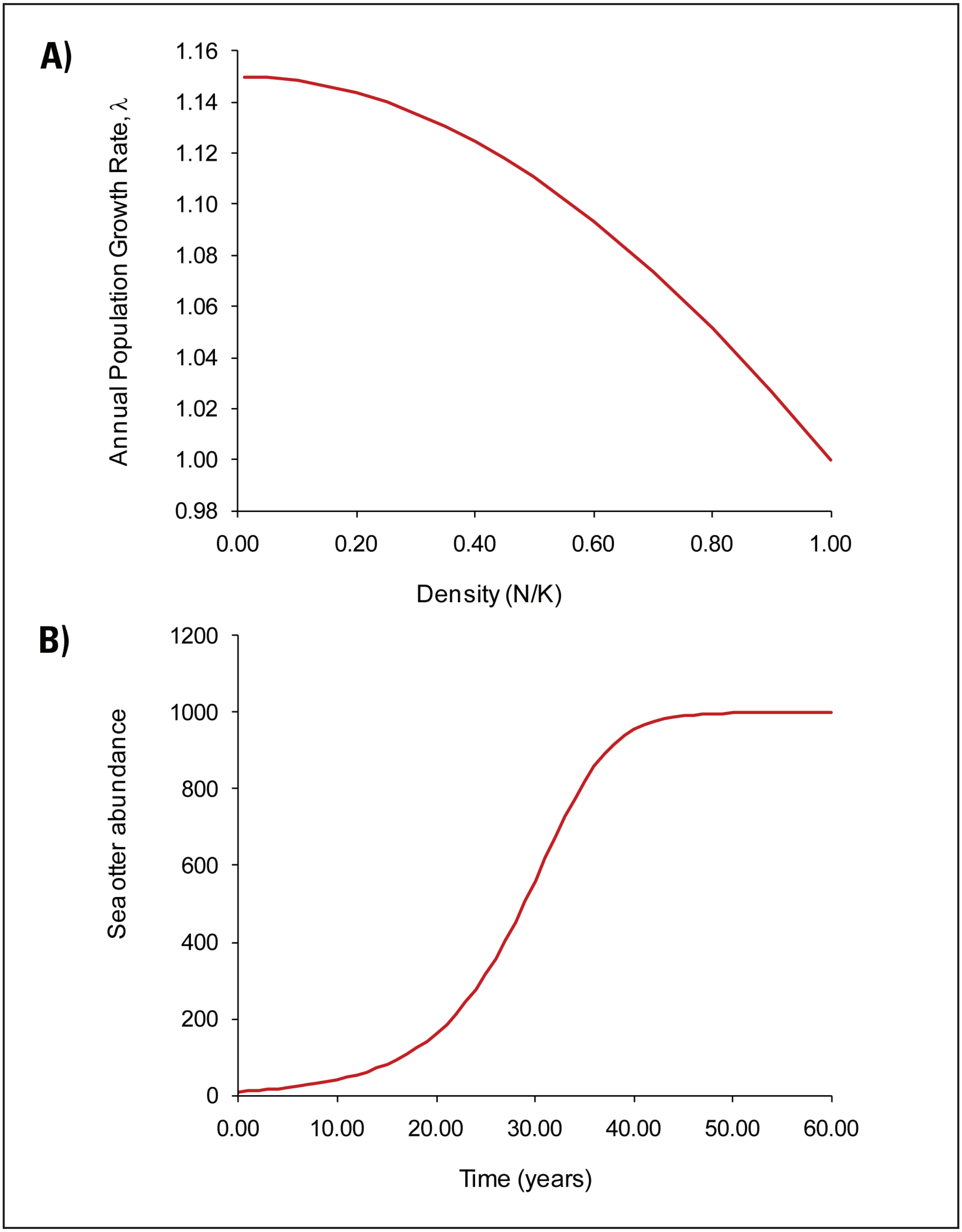

Figure B-1. Theta-logistic population growth typical of sea otter populations.
A) Annual rate of population growth, λ, plotted as a function of local population density, D (where D is calculated as a fraction of carrying capacity, K). The curve can be described by a theta-logistic function, $\lambda(D) = \exp[(r_{max})*(1-D^{\theta})]$, where r_{max} represents the maximum instantaneous growth rate ($ln(r_{max})$=1.15, the value of λ at low population densities) and θ controls the shape of the negative density-dependent function (θ=2). B) Trends in population abundance over time for a sea otter population where growth follows the theta-logistic pattern shown in A).

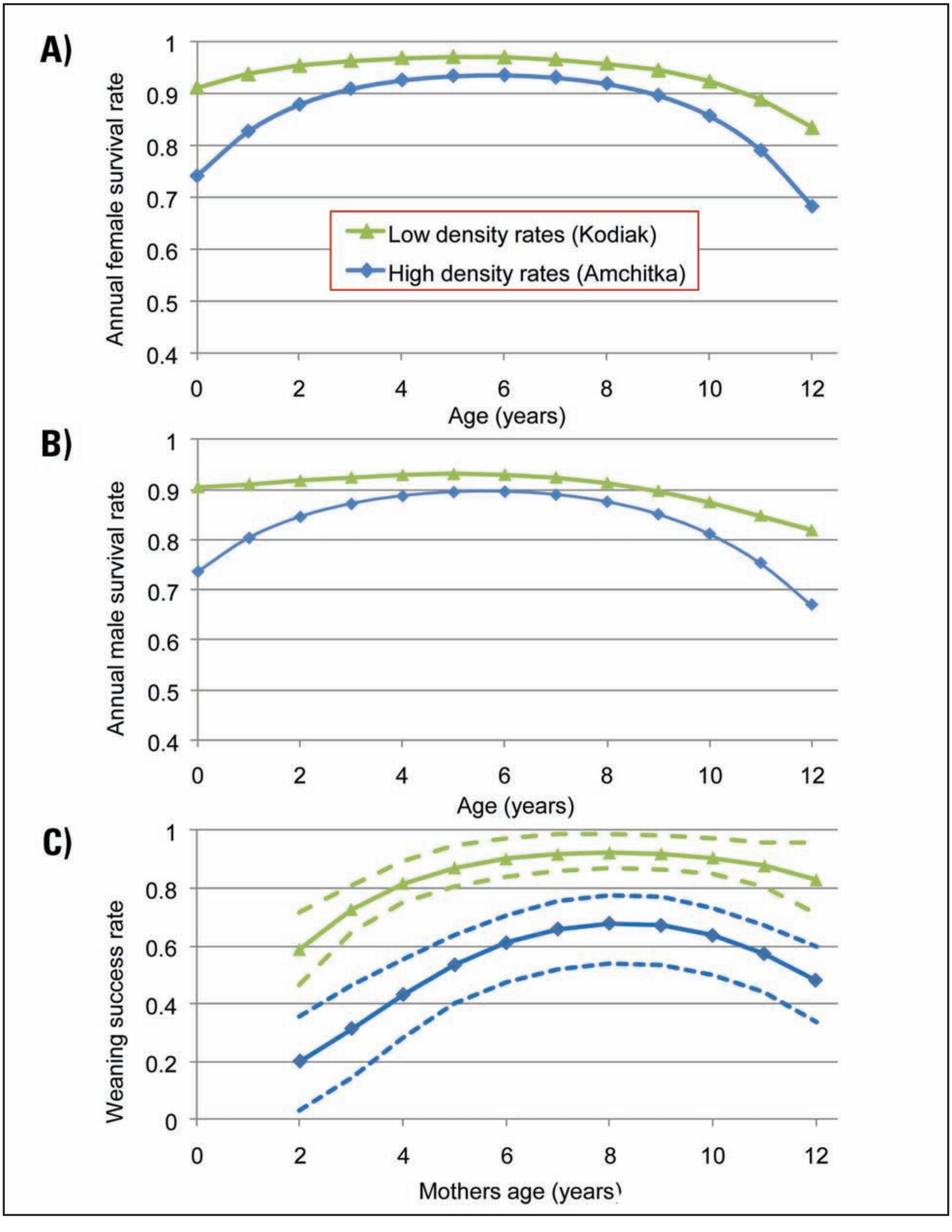

Figure B-2. Smoothed, age-specific vital rate estimates calculated for two sea otter populations, one at low density (parameterized from sea otters at Kodiak Island) and one at high density, near K (parameterized from sea otters at Amchitka Island).

A) Annual rate of survival for females. B) Annual rate of survival for males. C) Annual rate of weaning success for females of age i. The 95% confidence limits are also shown for each weaning success function (dashed lines): low density and high density weaning success rates were adjusted upwards slightly (i.e., within the plotted confidence limits) in order to obtain predicted λ values of 1.15 and 1.00, respectively (Table B-2).

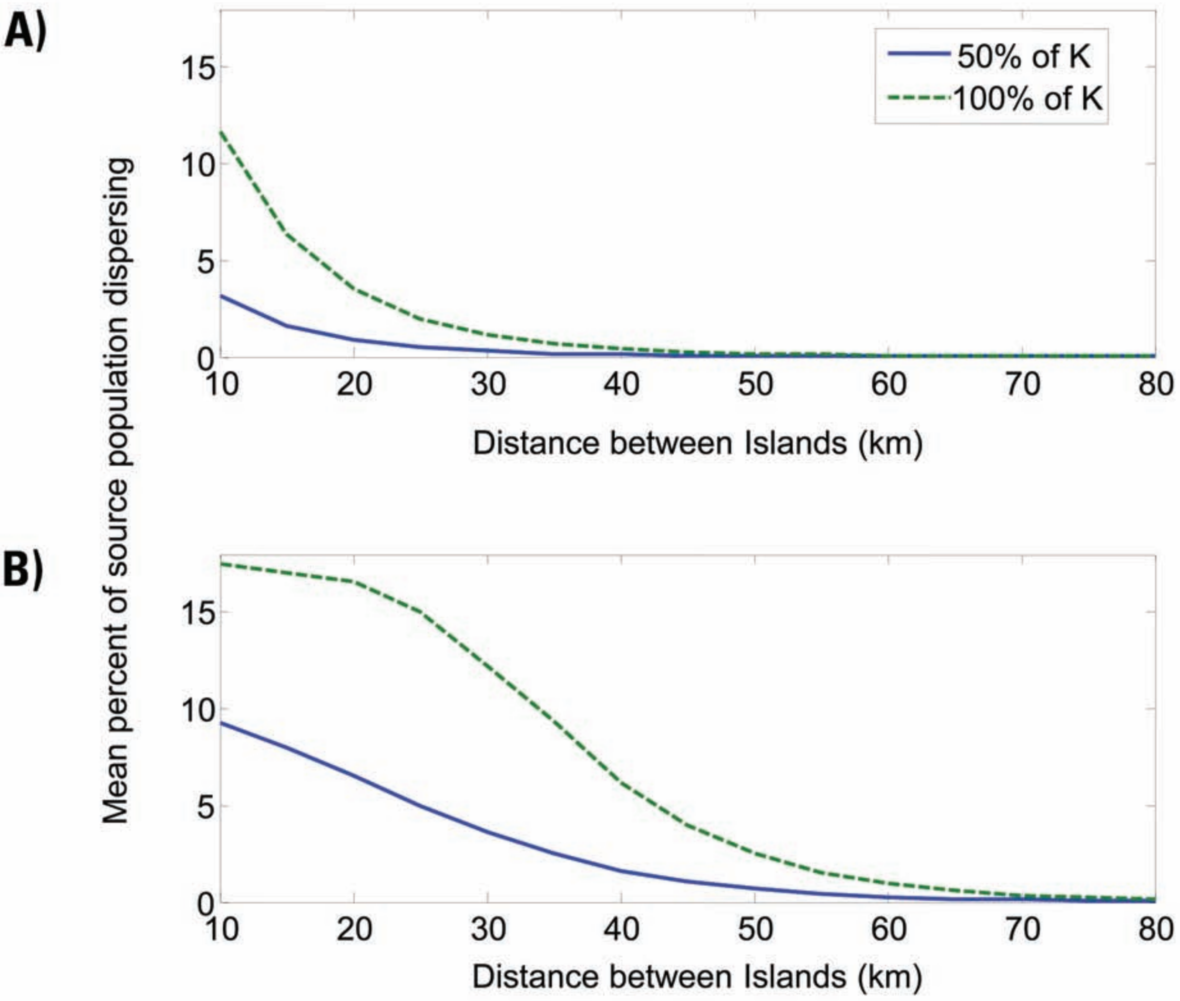

Figure B-3. Estimated average rates of inter-island dispersal (expressed as percentage of the source population dispersing) under two scenarios: A) a low dispersal scenario, and B) a high dispersal scenario. The plotted curves represent the mean values predicted by Poisson distributions having parameter γ, where γ was calculated using equation 6, which was parameterized by fitting to the values shown in Table B-3. In both cases, the parameterized function leads to reduced dispersal with increases in the distance between Islands and higher dispersal with higher density of the source population relative to carrying capacity, *K*.

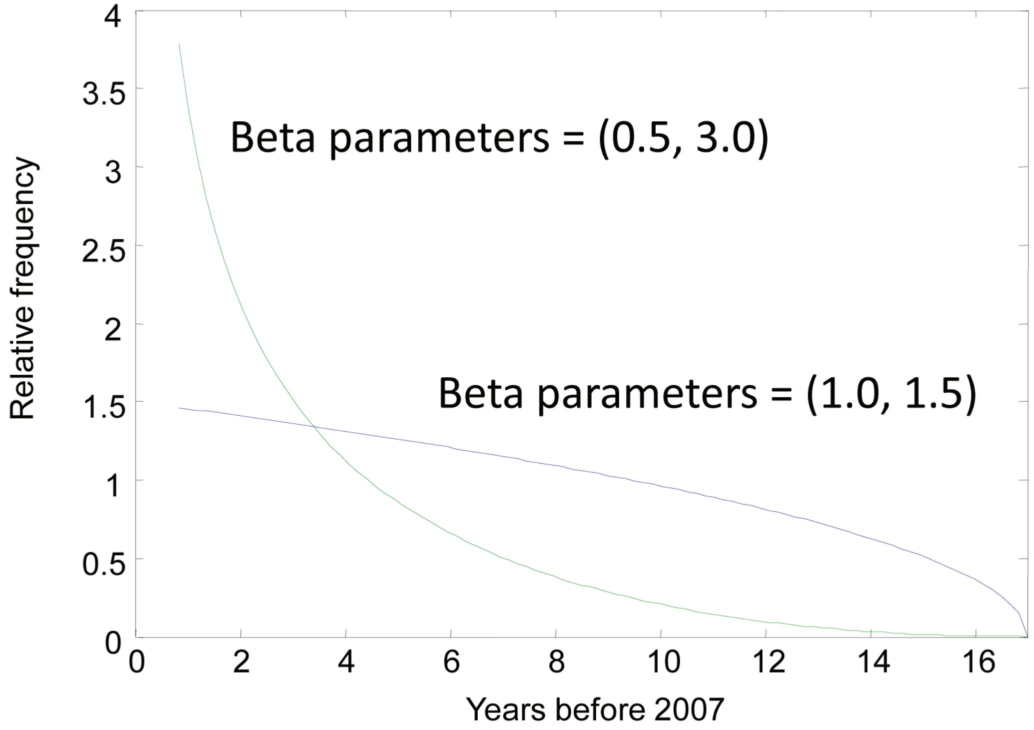

Figure B-4. Examples of weighting functions used to randomly select values of t, which were then used to generate values of age-independent mortality, δ(D,t), in simulations of future population dynamics in southwest Alaska sea otters.

A different beta distribution was used for each simulation (see Methods for further details).

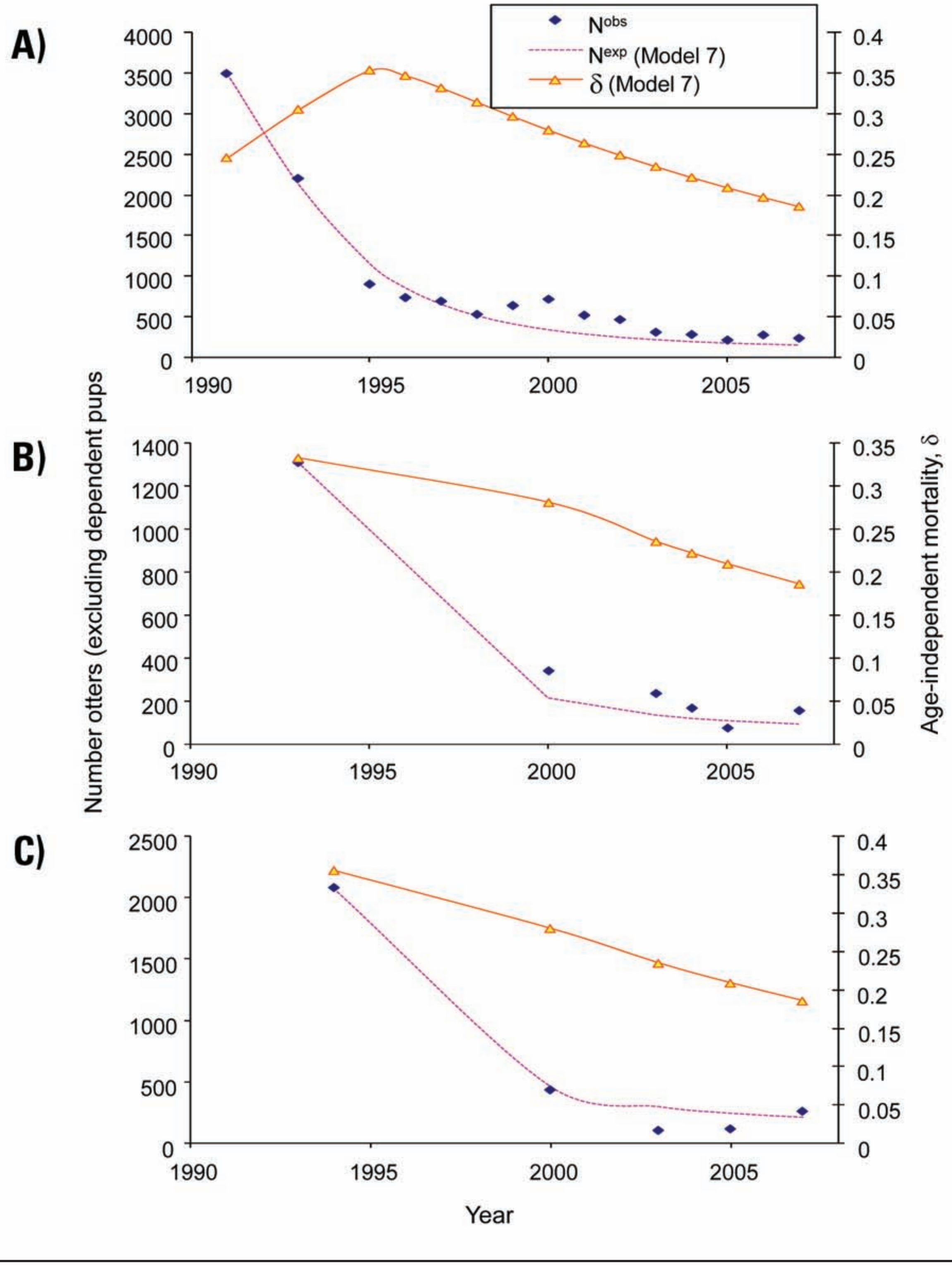

Figure B-5. Sample results from the maximum-likelihood fitting of age-independent mortality (δ) to observed skiff counts (Nobs).

The best-fit values for δ, along with resulting expected trends in sea otter abundance (N^{exp}), are shown for three sites in the central-west Aleutians: A) Adak Island, B) the north-east half of Amchitka Island, and C) Attu Island.

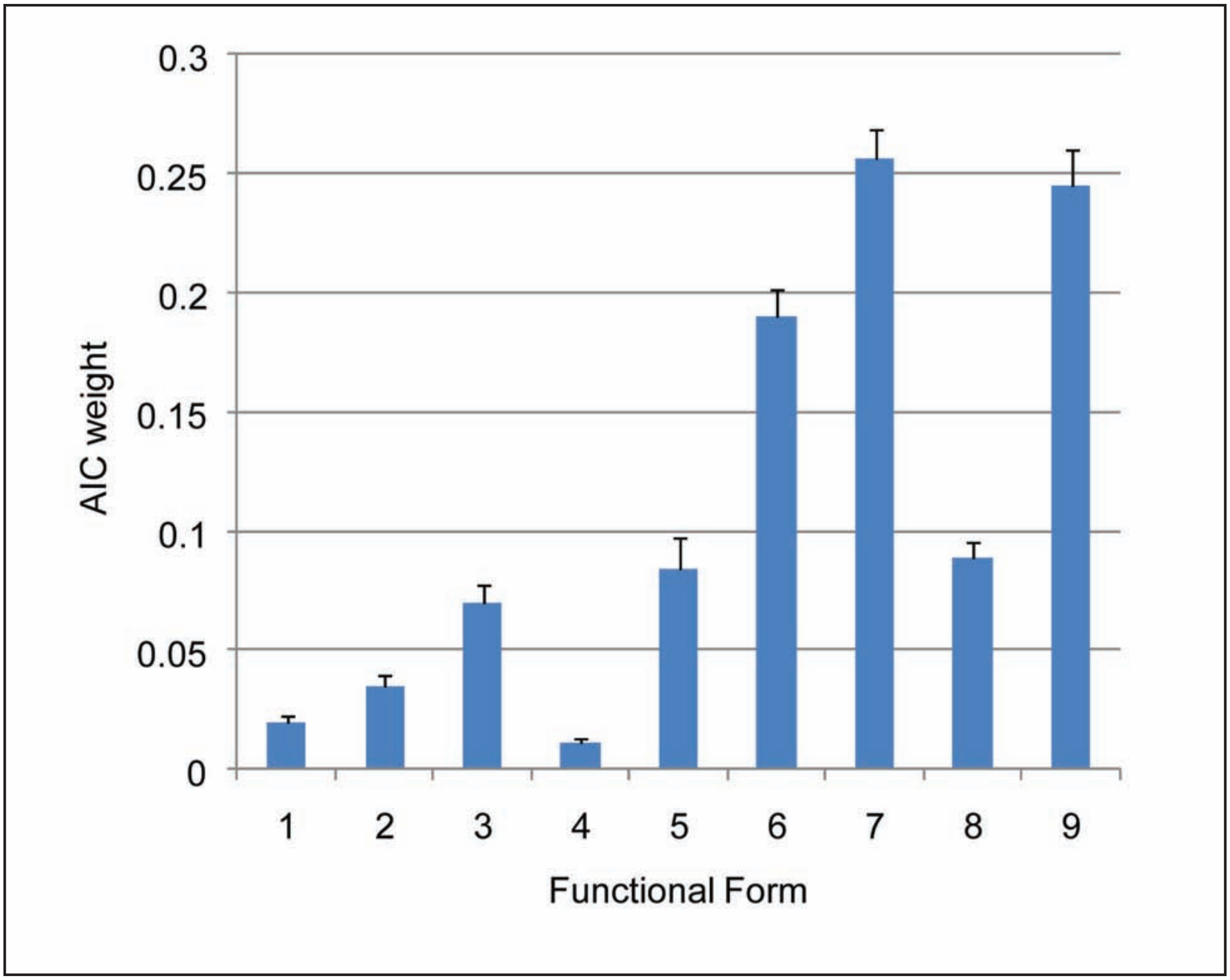

Figure B-6. Relative degree of support for nine different functional forms used to model δ, the per-capita rate of age-independent mortality, as measured by maximum-likelihood AIC weights (higher AIC weights indicate greater support for a given functional form).
Details on each functional form are summarized in Table B-5.

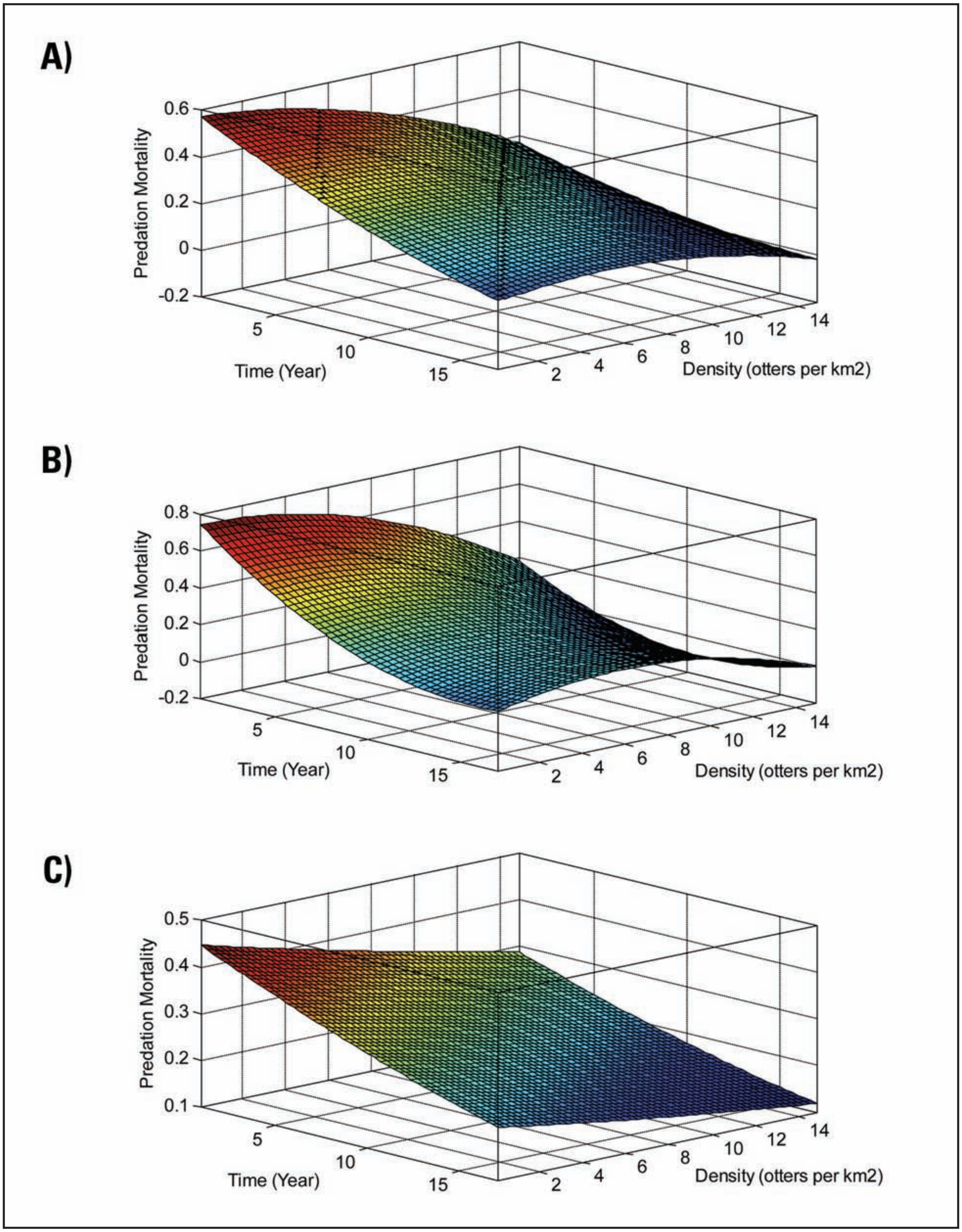

Figure B-7. Best-fit values of δ, the per-capita rate of age-independent mortality, plotted as a function of time (t) and relative population density (D).
Maximum-likelihood results are shown for three alternate model fits, corresponding to the three function forms of δ (Table B-5) with the greatest empirical support (Figure B-6). A) Model 7, B) Model 9, C) Model 6.

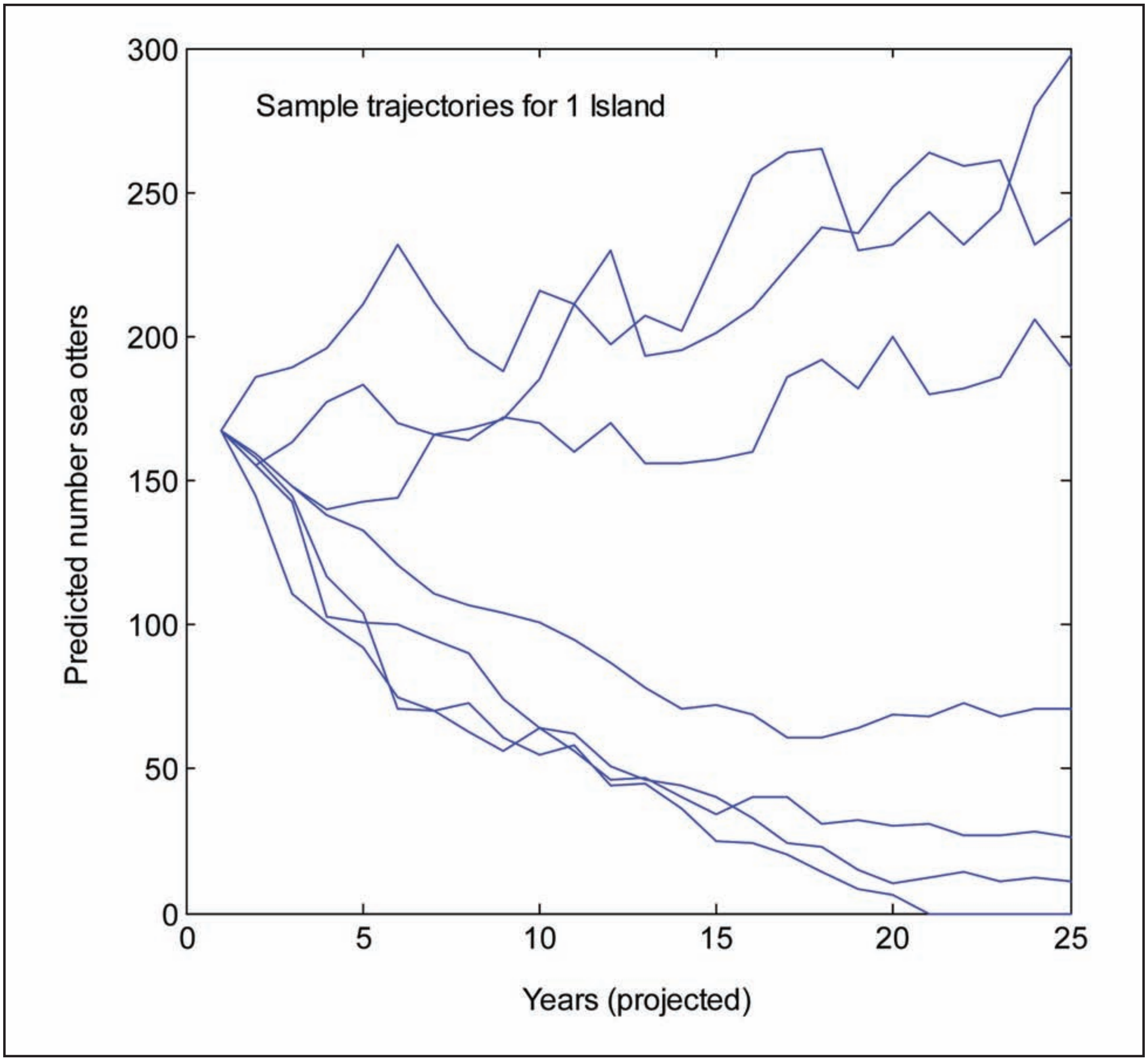

Figure B-8. Sample simulation results from the PVA model projections.
Each of the seven plotted lines represents the projected abundance over time for one island in the western Aleutian management unit, as recorded from a different iteration of the model. Because of demographic and environmental stochasticity, as well as uncertainty in model parameters (including the rate of age-independent mortality), possible population trends for this single island range from slow growth, to stability, to rapid decline.

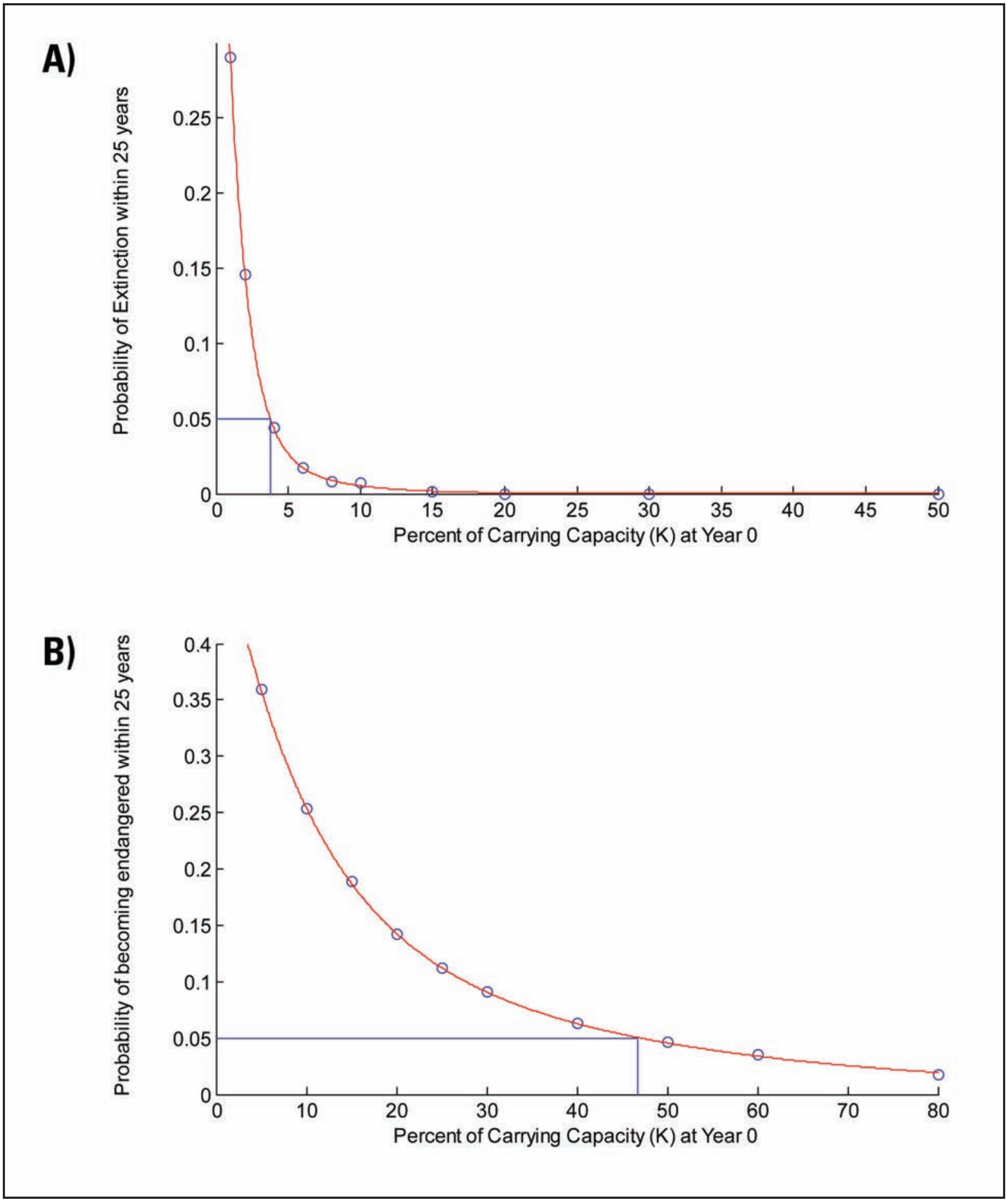

Figure B-9. PVA model results for the western Aleutian management unit.
A) The proportion of simulations in which total population abundance declined to quasi-extinction within the 25-year simulation period, plotted as a function of starting population density (D^0). The average density at which 5% or more of the simulations declined to quasi extinction for the western Aleutian Islands was 3.8% of *K*, however the upper 95% Confidence Limit for both western and eastern Aleutians was 4% of *K*, which we recommend as an appropriate uplisting criteria.
B) The proportion of simulations in which total population abundance declined to D^U, the up-listing threshold, by the end of the 25-year simulation period, plotted as a function of starting population density (D). The average density at which 5% or fewer of the simulations declined to D^U for the western Aleutian Islands was 47% of *K*, however the upper 95% Confidence Limit for both western and eastern Aleutians was 49% of *K*, which we recommend as an appropriate delisting criteria.

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