Report of the Workshop on Assessing the Population Viability of Endangered Marine Mammals in U.S. Waters

13–15 September 2005
Savannah, Georgia

Prepared by the Marine Mammal Commission

2007

This is one of five reports prepared in response to a directive from Congress to the Marine Mammal Commission to assess the cost-effectiveness of protection programs for the most endangered marine mammals in U.S. waters

TABLE OF CONTENTS

LIST OF TABLES

EXECUTIVE SUMMARY

Congress, in its 2004 Omnibus Appropriations Act, directed the Marine Mammal Commission to "review the biological viability of the most endangered marine mammal populations and make recommendations regarding the cost-effectiveness of current protection programs." As part of its response to the directive, the Commission convened a workshop to examine the state of science regarding population viability analysis (PVA) for marine mammal populations. The workshop was held 13–15 September 2005 in Savannah, Georgia. Its goals were to (1) review estimates of the viability of the most endangered marine mammals, (2) review the status of ongoing modeling efforts, particularly PVA, for endangered marine mammals, and (3) develop recommendations to improve listing and management decisions based on explicit consideration and improved estimation of population extinction risk.

The workshop focused on the 22 marine mammal taxa occurring in U.S. waters and either listed as threatened or endangered under the Endangered Species Act or designated as depleted under the Marine Mammal Protection Act. Participants agreed that, with two or possibly three exceptions, all those taxa—with appropriate management—appear to have the potential for persisting into the future. The first exception is the Caribbean monk seal, which has not been observed and documented since 1952 and is presumed to be extinct. The second is the AT1 stock of killer whales, which numbers fewer than 10 individuals, has not produced a single surviving calf for more than 20 years, and is highly unlikely to persist beyond the lifetimes of existing individuals. The possible third exception is the eastern population of North Pacific right whales, which has been a matter of concern based largely on the rarity of sightings, lack of information on the population, and its history of commercial exploitation. That concern has been tempered somewhat by recent evidence of successful reproduction (observations of cow-calf pairs). In addition to these obvious exceptions, the stock structure of many marine mammal species is poorly known, and participants noted that some additional stocks, yet to be identified, may be unlikely to persist into the future. For the taxa that were considered to be potentially viable, the available published analyses at the time of the workshop were not sufficient for a systematic and consistent quantification of their respective viabilities. The workshop identified methodological issues that need to be addressed to allow meaningful quantitative comparisons among viability estimates.

Participants reviewed the current state of PVA for marine mammals and other wildlife. For candidate species or species that are already listed, the growing trend is to use PVA to support listing and management decisions. PVA provides a mechanism for integrating all relevant data into a quantitative assessment to produce an estimate of extinction risk over a defined period of time. Such analyses are more objective than the qualitative listing approaches used to date, more amenable to explicit inclusion of all relevant data, more transparent with respect to assumptions and uncertainties, more easily standardized, and more conducive to the kind of structured decision-making that is needed to improve listing and management of endangered, threatened, and depleted taxa.

Participants also highlighted several common impediments to quantitative analyses for many marine mammal taxa. These include poor understanding of stock structure, insufficient biological data for recognized taxa, insufficient data for characterizing potential relationships between specific threats or management actions and population responses, and difficulty in predicting future threats and management actions that may affect their risk of extinction. Further, participants expressed a need for caution when using commercially available analytical software without proper documentation and without an understanding of the structure, function, and assumptions incorporated into the software.

Participants generally agreed with the findings of the Quantitative Working Group convened by the National Marine Fisheries Service and the U.S. Fish and Wildlife Service (DeMaster et al. 2004) to develop quantitative criteria for listing decisions. Among other things, those findings included the need for (1) performance testing of models used in the listing process to determine the likelihood of correct vs. incorrect decisions, given the life histories of species involved and uncertainties in the data or analytical results, and (2) clear links between policy decisions and quantitative models, particularly with regard to interpreting the uncertainty reflected in model results.

Finally, participants in the Savannah workshop considered and expressed support for a proposed decision tree incorporating simple PVAs to assess the status of poorly known taxa and guide listing decisions. The decision tree would use available data on the species of interest; default values where data are lacking; a structured, standardized, and simple analytical framework; and explicit guidelines for interpreting results. With appropriate development, the decision tree might be used to structure listing decisions in much the same manner as the potential biological removal concept is used to identify strategic stocks in the management of marine mammal/fishery interactions under the Marine Mammal Protection Act.

I. INTRODUCTION

In its 2004 Omnibus Appropriations Act, Congress directed the Marine Mammal Commission to "review the biological viability of the most endangered marine mammal populations and make recommendations regarding the cost-effectiveness of current protection programs." In response to the directive, the Commission reviewed systems for classifying species (Lowry et al. 2007), reviewed current protection programs (Weber and Laist 2007), conducted a case study of the North Atlantic right whale recovery program to examine the issue of cost-effectiveness in depth (Reeves et al. 2007), and held a workshop on the state and utility of population viability analysis (PVA) in the management of marine mammal populations.

The population viability workshop was held 13–15 September 2005 in Savannah, Georgia. Its goals were to (1) review estimates of the viability of the most endangered marine mammals, (2) review the status of ongoing modeling efforts, particularly PVAs, for endangered marine mammals, and (3) develop recommendations to improve listing and management decisions based on explicit consideration and improved estimation of extinction risk. This report summarizes the discussions and findings of the workshop.

II. BIOLOGICAL VIABILITY

For the purposes of this report, we define *biological viability* (or simply *viability*) to mean the potential to persist far into the future with appropriate management of human-related threats. Although species are often characterized as either viable or not viable (implying a high or low potential for such persistence), there is a meaningful intermediate area between these two extremes. The transition from viable to not viable has been the subject of extensive research aimed at identifying the "minimum viable population." This term was based on the idea that a declining population would reach a predictable point at which factors driving it toward extinction would dominate and recovery would be impossible or highly unlikely. This concept has been largely abandoned in the face of a growing body of contrary, empirical data illustrating that viability is generally a function of multiple natural and anthropogenic influences, the transition from viable to nonviable is dependant on species and circumstances, and a threshold for such transition that can be applied generally across taxa is not readily and reliably predicted.

Practically speaking, the viability of marine mammal taxa can be categorized as follows:

- Taxa that are extinct. These are taxa that have zero potential of persisting, as exemplified by the Steller's sea cow.
- Taxa that are almost certain to become extinct in the near future. The persistence of such taxa is highly improbable and there is little or no hope that they will continue to persist or can be saved, irrespective of human efforts. The AT1 stock of killer whales appears to fall into this category.

- Taxa with the potential to persist far into the future, but that may require the extra protections provided for threatened or endangered species on an ongoing basis. The Hawaiian monk seal may be one such species.
- Taxa with the potential to recover fully but that require extra protections until they have done so. Most listed species fit this category.
- Taxa that have recovered. The eastern North Pacific population of gray whales is the best example of this category.

The primary distinguishing elements of these categories are (1) a taxon's inherent potential for recovery and persistence, and (2) its dependence on human intervention (e.g., management actions and policy decisions) to address threats. These two elements are becoming more entwined as the effects of human activities become more nearly ubiquitous (e.g., climate change) and the boundary between anthropogenic and natural risk factors becomes less distinguishable (e.g., climate change). As a rule, scientists conducting PVAs have not made this distinction but rather have estimated population viability as a function of the list of known risk factors, both natural and anthropogenic, that may influence a population's persistence. The primary outcome of a PVA is a measure of the population's probability of extinction over a set period[1] based on the projected effects of such risk factors. Thus, a PVA is an approach to risk analysis that attempts to predict the probability of extinction based on available data. Except in rare situations that are usually apparent even without modeling, its results generally do not provide definitive answers as to when a species is no longer viable. For example, results of a PVA indicating that a population is declining toward extinction may be due to anthropogenic factors unrelated to its intrinsic ability to reproduce and grow. If those factors can be identified and addressed by effective management action, the population's decline could be reversed to allow recovery. Thus, a predicted declining trend may simply underscore the urgent need for management attention. A review of the taxa considered in this report indicates that they have all been influenced in a significant manner by human activities.

Factors affecting population viability generally fall into three categories. The first includes features of the population itself (e.g., abundance, age/sex structure, distribution, life history characteristics such as reproductive and survival rates). The viability of a marine mammal population may be a particular concern if, for example, it contains only a limited number of reproductive females (for example, the AT1 stock of killer whales includes only two or three females of reproductive age). The second category includes factors that are a function of the population's environment or ecology (e.g., availability of prey, abundance of predators, exposure to disease, variation in the physical parameters of the environment). Small populations

[1] PVAs often indicate the risk that a population will reach some specific, small number other than zero, called a "quasi-extinction" threshold. Such thresholds are generally based on the assumption that extinction is virtually certain at or below the chosen level. They are required for demographic models that describe animal abundance using real numbers (rather than integers) that may approach zero exponentially but never actually reach that level. Quasi-extinction thresholds are often used when modeling marine mammal populations because actual extinction may be delayed beyond the time period modeled, even for rapidly declining populations, if individual animals are relatively long-lived. Some authors have interpreted the quasi-extinction threshold to represent the lip of an extinction vortex (an interaction of risk factors creating adverse feedback loops that hasten a population toward extinction), but this interpretation is often difficult to demonstrate and justify.

occupying limited areas may be especially vulnerable to catastrophes. For example, Florida manatees, which tend to have clustered distributions in shallow waters, are vulnerable to toxic effects of periodic red tides. The third category includes factors that are a function of human activities (e.g., habitat destruction, competition for prey, disturbance, contaminants). Human activities, particularly exploitation, likely played a decisive role in the extinction of the Steller sea cow, Caribbean monk seal, Japanese sea lion, and North Atlantic gray whale, and have placed many other marine mammal taxa at risk of extinction (e.g., right whales).
Small populations also are especially vulnerable to the following sources of variability (also called stochasticity).

Genetic Variability—Two major concerns for small populations are (1) whether they contain sufficient genetic diversity to persist in the face of changing environmental conditions, and (2) whether mating between a limited number of breeding individuals will lead to the expression of deleterious genes affecting reproduction or survival (hence, population growth rate) because of inbreeding or genetic drift (i.e., random fluctuations in gene frequency). Although the former concern is often related to a species' ability to adapt to change over long periods of time (i.e., evolutionary time frame), the rapid pace of climate change illustrates that this concern also is relevant to short-term changes (i.e., ecological time frame). As a general rule, genetic diversity decreases with decreasing population size. Smaller populations are therefore less likely to contain sufficient genetic variation to persist in the face of selection imposed by significant environmental change. With regard to inbreeding, matings in small populations are more likely to be between related individuals, which increases the probability that deleterious recessive genes will be expressed in their offspring. The probability of such events also is influenced by the species' mating pattern. For example, inbreeding effects are more likely in polygamous than in monogamous species. Even in the absence of inbreeding, genetic drift in small populations can result in the expression of genetic defects. The effects of genetic drift and inbreeding depression may not be expressed for several to many generations, with the time frame depending on population size, the initial genetic diversity, the frequency of deleterious genes, and the reproductive strategy. Thus, their adverse consequences are more likely to manifest themselves in populations that are held at low abundance for long periods of time or that are repeatedly reduced to small size.

Demographic Variability—Demographic variability is the result of random variations in biological processes or parameters, such as survival, reproduction, or sex ratio at birth. If the fate of each individual in a population is subject to the same probability, the resulting variation will be a function of population size. That is, as the size of a population shrinks, the likelihood that it will deviate from the expected norm for that process or parameter will increase. For example, if the expected sex ratio at birth is 50 percent females and 50 percent males, a substantial deviation from that sex ratio is more likely to occur in a small population than in a larger one[2]. For depleted marine mammal populations, having more females generally is beneficial because of their role in reproduction, whereas a relative increase in males generally is detrimental. The

[2] Similarly, significant deviation from the expected 50:50 heads to tails ratio is more likely to occur if you flip a coin a few times versus many times.

actual consequences of such deviations depend on the population's composition and social structure, including its reproductive strategy, but they become more likely to affect the long-term persistence of a population as its abundance declines.

Environmental Variability—Environmental variability can alter population demographics by changing prey abundance, weather or oceanographic conditions, or abundances of predators. All individuals in a population may be affected, whether the population is large or small. Populations that occupy large areas may be exposed to a range of environmental conditions and thus may experience a degree of buffering from poor conditions in portions of the range. Small populations are more likely to occupy smaller areas where environmental conditions tend to be more homogeneous and, therefore, such populations may be more vulnerable to unfavorable circumstances. Metapopulations (collections of related subpopulations) may be buffered against such circumstances if animals are dispersed among subpopulations experiencing different environmental conditions.

Catastrophic Variability—Catastrophic variability has been considered by some to be an extreme form of environmental variability and by others to be a separate type of risk factor because its nature and spatial-temporal patterns (e.g., hurricanes, tsunamis) are inconsistent with those of environmental variability. Here too, the problem for small populations is that all or some large percentage of the individuals in the population may be exposed to the effects of the same catastrophic event if their distribution is limited relative to the distribution of the event. Metapopulation structure and broad distribution enhance resilience to catastrophic variation.

Allee Effects—Population parameters also vary as a function of animal density. As the number of individuals in a population declines, the potential for population growth may increase because of reduced competition for prey, habitat, or other resources. However, at very low levels, populations also may experience accelerating declines in reproductive or survival rates due to so-called Allee effects. For example, if animals are sparsely distributed, adult females may be unable to find mating partners. Similarly, strategies for foraging, predator avoidance, and rearing young may be impaired in very small populations if those behaviors depend on the cooperation or participation of multiple individuals. Allee effects and other risk factors may combine either additively or synergistically to create cumulative effects that, ultimately, determine the population's risk of extinction. Such interactions may create negative feedback loops that hasten decline toward extinction, a phenomenon referred to as an extinction vortex.

Population Structure—Finally, assessment of the risk factors discussed here is confounded by insufficient information on population (or stock) structure. The conservation and management frameworks established by the Endangered Species Act, the Marine Mammal Protection Act, and related statutes are predicated on scientists being able to identify appropriate units to conserve. As indicated earlier, stock structure is poorly understood for many marine mammal species despite recent progress using molecular genetics techniques. The failure to recognize distinct population segments[3] or population stocks increases risk if, for any number of reasons, some are more vulnerable than others to the above-described risk factors and they are managed in a manner that does not recognize and adjust for that vulnerability. The identification and characterization of population structure are essential for accurate assessment of population viability. Therefore, this subject warrants continued scientific investigation.

III. POPULATION VIABILITY ANALYSIS

To support and improve listing and management decisions for taxa that are either candidate species or are already listed as endangered or threatened, the growing trend is toward more structured, objective decision-making using the best available quantitative tools to determine risk of extinction. All of the early marine mammal listings involved species that were reduced to low levels by human exploitation (usually commercial hunting) and ineffective or non-existent management. Those listing decisions required expert judgment and a degree of qualitative assessment. Even the best information available at that time was subject to important limitations. Abundance, for example, is clearly an important consideration with regard to the risk of extinction. However, it has subsequently become clear that abundance is only one indicator of extinction risk, and an imperfect one at that. Some small populations may have a low probability of extinction because of favorable environmental conditions and an absence of significant threats, whereas some large populations may have a high probability of extinction because of poor environmental conditions and significant threats. As noted above, the risk of extinction also is a function of a potentially wide range of factors, including those related to the taxon itself, its environment, and the threats posed by human activities.

PVAs provide a means for integrating many kinds of information to produce robust indicators of extinction risk. Such analyses vary in form as a function of the population under consideration, its life history traits, the nature and amount of data available on its biology and population dynamics, the nature and amount of data available on factors that may affect its risk of extinction (e.g., threats), and the modeler's technical (i.e., mathematical) preferences. When feasible, such analyses incorporate the types of variation described earlier, usually by representing variables as distributions of possible values and running multiple analyses drawing randomly from those values to estimate the range of possible outcomes and their probability. The results can be used for a number of purposes, including informing listing processes under the Endangered Species Act and Marine Mammal Protection Act, evaluating the effects of past management actions, and predicting the effects of proposed actions.

At the workshop, participants discussed existing PVA models or similar quantitative analyses for California sea otters, Cook Inlet beluga whales, Florida manatees, Hawaiian monk seals, North

[3] In 1996 the National Marine Fisheries Service and the Fish and Wildlife Service finalized a policy statement interpreting the Endangered Species Act term "distinct population segment" to mean a population that is (1) discrete from the remainder of the species (e.g., markedly separate), (2) significant to the species (e.g., its loss could cause a major gap in the range or includes unique genetic characteristics), and (3) threatened or endangered based on the Act's five listing factors. Use of the term in this report is intended to be consistent with that interpretation.

Atlantic right whales, southern resident killer whales, and the eastern and western stocks of Steller sea lions (Appendix 1). Several of the models had been used to inform listing decisions, but the others were not used for that purpose because they were created after listing had already occurred. These latter models were created for other purposes and varied in their objectives, model parameters, and complexity. In all cases the models were constrained to varying degrees by limited information pertaining to biology (e.g., reproductive and survival rates), cause-and-effect relationships between specific threats or management actions and population responses, and factors likely to determine the nature and extent of future threats. Given limitations in available data, scientists developing each of the models were required to make certain simplifying assumptions (e.g., homogeneity in vital rates over space or time, relevant risk factors). As is generally the case in science, the assumptions warrant further consideration and testing. However, an important characteristic of these models (and of PVA models in general) is that all such assumptions are made explicit, and therefore the potential implications of erroneous assumptions can be directly evaluated using sensitivity analyses. Moreover, as new information becomes available, the data used in the models can be updated, the model processes refined, and the validity of simplifying assumptions re-evaluated, all of which will result in more robust and reliable model results. Thus, PVA is an evolving process, open to review and subject to improvement as new information and insights become available.

IV. VIABILITY OF THE MOST ENDANGERED MARINE MAMMALS

To assess the biological viability of the most endangered marine mammals, the workshop focused on the 22 species and stocks occurring in U.S. waters and listed as endangered or threatened under the Endangered Species Act or designated as depleted under the Marine Mammal Protection Act (Table 1). Some other marine mammal species (e.g., the Yangtze River dolphin or baiji[4]) may be equally or more endangered but were not discussed in detail at the workshop or included in this report because they do not occur in U.S. waters or have not been listed. A wide variety of taxa, including marine mammals and other species, either have been rendered extinct or have been brought near to extinction before rebounding. Those cases also provide useful insights regarding the question of species viability.

Extinctions and recoveries

Human activities have caused the extinction of at least four marine mammal taxa. The northern (Steller's) sea cow inhabited kelp-forested coastlines of the Bering Sea until the second half of the 18th century, when it was driven extinct by commercial seal and sea otter hunters who hunted sea cows for food (Stejneger 1887, Forsten and Youngman 1982). The last reliable sighting of a Caribbean (West Indian) monk seal was in 1952 (Kenyon 1977); this species also was a victim of uncontrolled hunting, disturbance, and habitat destruction. The Japanese sea lion was last sighted and documented in 1951 and is listed as extinct by IUCN–The World

[4] During the preparation of this report, an international team of marine mammal scientists conducted an extensive survey of the Yangtze River and failed to detect a single baiji.

Conservation Union (IUCN). It was exploited for various purposes and persecuted because it was perceived as a competitor of fisheries. The North Atlantic population of gray whales vanished by the 18th century, and whaling almost certainly was a contributing, if not decisive, factor in its demise.

A number of other marine mammal taxa have been brought near to extinction and then recovered, at least partially, following protection from the main threat, which almost always has been deliberate exploitation (Table 2). Many populations of fur seals (*Arctocephalus* spp.), elephant seals (*Mirounga* spp.), sea otters, and baleen whales were harvested to the point where hunting for them was no longer profitable. In a few cases (e.g., Guadalupe fur seal), numbers were so low that a species or population was considered extinct, only to be discovered again and to recover under protection. These examples (and examples from other taxonomic groups, Table 2) demonstrate the potential resilience and viability of wild species and populations, even when reduced to low population size.

Assessment of biological viability

PVAs have not been completed for most of the 22 taxa considered at the workshop because of insufficient information or lack of a standard for conducting such analyses on data-poor species. The general opinion of workshop participants was that all but 2 or possibly 3 of the 22 listed marine mammal taxa are potentially viable if human-related threats are effectively managed. The Caribbean monk seal probably is extinct, and the AT1 stock of killer whales does not appear to be biologically viable, regardless of recovery efforts. The viability of the eastern population of North Pacific right whales has been a matter of concern based on its history of exploitation, rarity of sightings, and lack of biological information on the population.

The discussion of viability for the remaining taxa was based primarily on PVAs or a combination of expert opinion and varying amounts of quantitative information. Of the eight taxa for which quantitative biological analyses had been conducted, three (Florida manatee, eastern population of Steller sea lions, and southern sea otter) have experienced positive growth in recent years, although growth rates have been difficult to characterize (Florida manatee) or slower than expected (southern sea otter, eastern population of Steller sea lions). Thus, each of these three taxa is reasonably considered biologically viable although the persistence of at least two of them likely will continue to depend on rigorous, effective management of known threats (i.e., disease, contaminants, and fisheries for the southern sea otter and boat strikes and loss of warm-water refuges for the Florida manatee).

The Hawaiian monk seal is now declining at about 4 to 5 percent per year, and its total population size in 2005 was estimated at 1,250 to 1,300. The species consists of seven reproductively isolated subpopulations including one in the main Hawaiian Islands and six in the remote Northwestern Hawaiian Islands. The subpopulations are subject to a range of threats (e.g., reduced prey availability, entanglement, shark predation, male aggression, disease). Although monk seals benefit from a robust research program that helps direct a growing management effort and although some sweeping protection and conservation measures have been implemented recently, the species clearly is at risk of extinction. Rigorous, effective management of human-related threats is essential to its conservation. The species has persisted for 12 to 15 million years in approximately the same geographic range, and there is no reason to believe that it cannot continue to persist far into the future as long as human-related threats are managed effectively.

The western population of Steller sea lions has declined by about 80 percent in the past three decades, and much of the decline has yet to be explained. Contributing causes may include natural changes in environmental conditions leading to a reduction in prey, predation by killer whales that lost a preferred prey source because of commercial whaling, deliberate or incidental killing in connection with fisheries, and competition for prey with large-scale commercial fishing that rapidly expanded in the region in the 1960s and early 1970s. The most recent abundance estimate is about 38,000 animals, and recent counts suggest that the population may have stabilized. Although there is still cause for concern regarding the future of this population and much remains to be learned about the importance of various risk factors, there is no basis for concluding that the population is incapable of persisting if human-related threats are managed effectively.

The number of North Atlantic right whales was reduced by commercial whaling prior to the 1940s and currently numbers about 300 to 350 whales. This population's estimated rate of increase was positive in the 1980s but apparently declined in the 1990s due to mortality from ship strikes and entanglement in fishing gear. The population appears capable of maintaining a positive population growth rate if human-related threats are controlled, and therefore it appears to be viable.

The southern resident stock of killer whales occurring each summer in the Puget Sound area has been subject to a range of human-related threats. A relatively large portion of the population was captured in the 1960s and early 1970s to supply animal-display facilities. Prior to that, the animals were subject to unregulated shooting and harassment (Hoyt 1981). In recent years, the stock has been subject to three primary threats: loss of prey (primarily chinook salmon) secondary to loss of salmon habitat and fishing, exposure to contaminants, and noise and disturbance due to watercraft traffic, including whale-watching vessels. Abundance of the stock under pristine conditions is unknown although it is likely to have been in the low hundreds. Although the elevated risk of extinction for this stock is a matter of significant concern, there is no basis for concluding that its low numbers and recent decline are due to an inherent lack of biological viability, particularly in view of the multiple human-related threats to which the stock is exposed.

Abundance of the Cook Inlet beluga whale was reduced sharply in the 1990s by Alaska Native subsistence harvests. The harvest was restricted in the late 1990s, but trend analysis of abundance estimates from 1994 through 2006 indicates a statistically significant decline. Research is urgently needed to identify the causes for continued decline after the harvest was brought under control. A number of natural and anthropogenic factors could be contributing to the continued decline. There is no basis for concluding that the population has lost the potential to recover, but it is clear that its recovery potential is eroding rapidly and the population is in great need of rigorous, effective management to identify and address the factors perpetuating the decline.

PVAs have not been conducted or were not discussed at the workshop for the remaining 12 taxa considered. The bowhead whale, fin whale, humpback whale, sperm whale, mid-Atlantic stock of bottlenose dolphins, Guadalupe fur seal, northern fur seal, and southwest Alaska stock of northern sea otters all are either known to be increasing or number at least 10,000 animals, and there is no basis for concluding that any of them has lost the potential to recover and persist. Blue and sei whale population structure and abundance are not well known. The National Marine Fisheries Service recognizes eastern and western North Pacific blue whale stocks and a North Atlantic stock. The eastern North Pacific stock, the only one for which an abundance estimate is available, consists of about 3,000 animals and may be increasing. It is therefore reasonably considered to be viable. The status of the other two blue whale stocks that occur in U.S. waters is not known, although the general sense of workshop participants was that they are capable of persisting if human threats are effectively managed. There are no current, reliable estimates of sei whale abundance in the North Pacific and North Atlantic Oceans. The sei whale estimate of >133 in Table 1 represents a combined minimum estimate for the eastern North Pacific and Hawaii stocks, but likely it is strongly and negatively biased. Better estimates of sei whale abundance are clearly needed and will provide a better basis for judging the viability of sei whale stocks. The Antillean subspecies of West Indian manatee occurred historically over a relatively large range in coastal areas of Central and South America and around islands in the Caribbean Sea. Manatees are now rare in the U.S. Virgin Islands, and the most recent (1994) count in Puerto Rico suggested a minimum of 86 animals. As the Antillean manatee is threatened in U.S. waters mainly by boat strikes and entanglement in fishing nets, the primary recovery challenge appears to be controlling those human-related threats. Any further decline in numbers would erode the potential for recovery of the manatee populations in Puerto Rico and the Virgin Islands. Outside U.S. waters, deliberate killing also poses a significant risk and undermines the potential for recovery. Finally, the viability of the eastern population of North Pacific right whales is a significant concern. Since the mid-1990s only 23 individuals have been identified, including three cow-calf pairs. It is unlikely that the identified animals represent the entire population, but it also seems unlikely that there are a great many more than that number. Whether 23—or even 25 to 50—individuals would be sufficient for recovery is unclear. However, populations of other mammal species have recovered from such low numbers (e.g., northern elephant seals, southern sea otters) so there is a basis for hoping that this population is still capable of recovery.

V. IMPROVING LISTING DECISIONS

Two efforts to improve listing decisions were discussed at the workshop, one undertaken by the National Marine Fisheries Service and the Fish and Wildlife Service, and one proposed by Dr. Daniel Goodman. Those efforts are summarized briefly below.

Quantifying the listing process

In 2004 the National Marine Fisheries Service and Fish and Wildlife Service convened a Quantitative Working Group to evaluate listing decisions under the Endangered Species Act and develop procedures that would be "more transparent, consistent, and scientifically and legally defensible." The working group identified several conceptual models for listing purposes, all of which are directly or indirectly related to risk of extinction (DeMaster et al. 2004).

The working group noted that implementation of these approaches would require explicit policy input. Specifically, policy guidance must be provided regarding the degree to which errors in the listing process are acceptable if they result in over-protection (i.e., listing species that, in fact, are not endangered or likely to become so in the foreseeable future) versus under-protection (i.e., failing to list species that, in fact, are endangered or likely to become so). The extent to which the process should be precautionary (i.e., favor over-protection) also would require specification.

In many cases, the Fish and Wildlife Service and National Marine Fisheries Service must make listing decisions for species with very limited information on either population status or threats. The working group recommended that in those cases quantitative proxies, or "alternative decision metrics," be developed (e.g., a 95 percent decline in abundance could serve as a proxy for an unacceptable probability of extinction). Both listing standards and proxies should be tested to determine the likelihood of correct versus incorrect listing decisions, given the life history of a species and uncertainty in the data or analytical results. The working group has initiated such performance testing for a suite of potential listing standards and proxies. An additional option recommended for consideration by the working group is a threshold approach similar to that used by a number of other organizations including IUCN in its Red List of Threatened Species, parties to the Convention on International Trade in Endangered Species of Wild Fauna and Flora for listing species on its appendices, and the Committee on the Status of Endangered Wildlife in Canada for listing species under the Canadian Species at Risk Act.

A theoretical decision tree for listing under the Endangered Species Act

Participants at the Commission's PVA workshop also considered a theoretical decision-making framework for listing decisions proposed by Dr. Goodman. The framework was developed to simplify and standardize listing decisions using quantitative tools and criteria, better document the decision-making process, provide default assessment methods for data-poor taxa, and guide the use of limited resources to develop more reliable assessments. The suggested framework is designed to classify "at-risk" populations into four categories based on their population dynamics and then use category-specific quantitative modeling approaches to assess a population's viability and determine whether the population should be listed based on defined decision rules. The four suggested categories are (1) populations that are too small, (2) populations that are nearly too small, (3) populations that are large but declining, and (4) populations that are large but have volatile population dynamics (Table 3). For each category, standards would be established for classifying species as threatened or endangered.

Several technical and scientific issues would have to be addressed before such a framework could be used. The framework would have to include options for taxa with known threats but little or no population data. Also, the term "too small" would have to be defined, and quantitative assessment methods for each category would need to be developed and tested. Those methods would require some flexibility to take into account different life history types (e.g., long-lived versus short-lived species or species with low versus high reproductive rates). Although some at-risk marine mammals may fit less cleanly than others into these four categories, an explicit decision tree framework with specific categories and corresponding quantitative assessments would enhance the objectivity and consistency of the listing process for data-poor taxa. For that reason, workshop participants favored the development of such a framework. With appropriate development, the decision tree might help structure listing decisions in much the same manner as the potential biological removal concept has structured the management of incidental take of marine mammals in commercial fisheries under the Marine Mammal Protection Act.

VI. IMPROVING OTHER MANAGEMENT DECISIONS

In addition to listing decisions under the Endangered Species Act, decisions regarding a range of marine mammal management actions are based on qualitative assessments of limited quantitative data on population status, trends, and threats. The underlying analyses often are not explicit with regard to assumptions and uncertainties, and therefore they can seem subjective and arbitrary. This problem can be addressed, at least in part, by making the decision-making process more explicit, objective, and quantitative. For example, when deciding among management actions, quantitative models can be used to analyze or predict their alternative effects, thereby informing the decision-making process. Such models also can provide a mechanism to evaluate the significance of assumptions and uncertainty inevitably associated with management decisions. In addition, they may help identify factors likely to affect population recovery and help characterize the nature and significance of the likely effects. All of these benefits—if communicated effectively between scientists and managers—would result in a more structured and comprehensive process for making management decisions.

Recently both the Fish and Wildlife Service and the National Marine Fisheries Service have increased their use of explicit, quantitative models to inform management decisions. In part, this increase is related to requirements for "objective, measurable" recovery criteria in recovery plans prepared under the Endangered Species Act. Quantitative models can be and have been used to assess relative risks from various threats and relative benefits of alternative management strategies in the recovery planning process. They also have been used in section 7 consultations under the Endangered Species Act to assess the effects of incidental takes of individuals from species or populations listed as endangered or threatened.

Many PVA analyses have been conducted using VORTEX (Bob Lacy, Department of Conservation Biology, Chicago Zoological Society) or other standardized software. Although such software can be very useful for heuristic purposes (e.g., exploring the dynamics of different populations and the effects of different management actions), workshop participants expressed concern that it may be used without adequate understanding of the actual structure, function, and assumptions incorporated into the software model. Such uninformed use may result in misunderstanding of population status and risk of extinction. Although participants did not necessarily suggest that complex models should be created from first principles for each analysis, they did suggest that analyses using standardized software should be conducted with a thorough understanding of their limitations and assumptions. All the case studies presented at the workshop and reviewed here were customized models specifically tailored to the circumstances of the population and the nature of the available data.

VII. ACKNOWLEDGMENTS

The Marine Mammal Commission thanks all those who participated in, or provided support for, the workshop. The Commission is especially grateful to the presenters and panel members who led many of the discussions. A steering committee of experts directed the Commission's response to the Congressional directive, including this workshop. Its members were David Cottingham, Deborah Crouse, Daniel Goodman, Lloyd Lowry, Linda Manning, Randall Reeves, Michael Simpkins, and Phil Williams. David Laist managed various components of the Commission's response, and Michael Simpkins and Timothy Ragen collaborated on preparation of the workshop report. SRA International, particularly Linda Manning and Regan Maund, provided expert assistance in planning and facilitating the workshop and also helped with drafting of this report. Elizabeth Taylor, a 2005 Knauss Sea Grant Fellow at the Commission, also contributed substantially in organizing the workshop and drafting early versions of this report. Jeff Benoit from SRA International and Suzanne Montgomery provided editorial assistance for later revisions. Katherine Ralls helped with the development of Table 2.

VIII. LITERATURE CITED

Baker, S. C., and P. J. Clapham. 2004. Modeling the past and future of whales and whaling. Trends in Ecology and Evolution 19(7):365–371.

Bartholomew, G. A., and C. L. Hubbs. 1960. Population growth and seasonal movements of the northern elephant seal, *Mirounga angustirostris*. Mammalia 24:313–324.

Canadian Wildlife Service and U.S. Fish and Wildlife Service (CWS and FWS). 2005. Draft international recovery plan for the whooping crane. Recovery of Nationally Endangered Wildlife (RENEW), Ottawa, and U.S. Fish and Wildlife Service, Albuquerque, NM.

DeMaster, D., R. Angliss, J. Cochrane, P. Mace, R. Merrick, M. Miller, S. Rumsey, B. Taylor, G. Thompson, and R. Waples. 2004. Recommendations to NOAA Fisheries: ESA listing criteria by the Quantitative Working Group, 10 June 2004. U.S. Dep. Commerce, NOAA Tech. Memo. NMFS-F/SPO-67.

Forsten, A., and P. M. Youngman. 1982. *Hydrodamalis gigas*. Mammalian Species. No. 165. American Society of Mammalogists. 3 pp.

Gallo, J. P. 1994. Factors affecting the population status of Guadalupe fur seal, *Arctocephalus townsendi* (Merriam, 1897), at Isla de Guadalupe, Baja California, Mexico. Ph.D. Thesis, University of California, Santa Cruz.

Hoyt, E. 1981. Orca the whale called killer. Firefly Books, Ltd. Altona, Manitoba, Canada. 291 pp.

Hubbs, C. L. 1956. Back from oblivion, Guadalupe fur seal: still a living species. Pacific Discovery 9(6):14–21.

Hubbs, C. L., and K. S. Norris. 1971. Original teeming abundance, supposed extinction, and survival of the Juan Fernández fur seal. Antarctic Research Series 18:35–51.

Kenyon, K. W. 1977. Caribbean monk seal extinct. Journal of Mammalogy 58(1):97–98.

Lowry, L., D. Laist, and E. Taylor. 2007. Endangered, threatened, and depleted marine mammals in U.S. waters: a review of species classification systems and listed species. Marine Mammal Commission, Bethesda, MD.

McCullough, D. R., J. K. Fischer, and J. D. Ballou. 1996. From bottleneck to metapopulation: recovery of the Tule elk in California. Pages 375–403 *in* Metapopulations and Wildlife Conservation (D. R. McCullough, ed.). Island Press.

National Park Service (NPS). 1998. Tule elk, the return of a species. Point Reyes National Seashore, CA.

New Zealand Department of Conservation (NZ DOC). 2001. Black robin recovery plan, 2001–2011. Department of Conservation, Wellington, New Zealand.

Pucek, Z. 2004. Status survey and conservation action plan, European bison. IUCN–The World Conservation Union.

Reeves, R. R., A. Read, D. Boness, L. Lowry, and S. K. Katona. 2007. Report of the North Atlantic right whale program review, 13–17 March 2006, Woods Hole, Massachusetts. Marine Mammal Commission. Bethesda, MD.

Riedman, M. L., and J. A. Estes. 1990. The sea otter (*Enhydra lutris*): behavior, ecology, and natural history. Biological Report 14. U.S. Fish and Wildlife Service, Department of the Interior. Washington, D.C. 126 pp.

Soule, M. E. 1986. Conservation Biology, the Science of Scarcity and Diversity. Sinauer Associates, Inc. Sunderland, MA.

Stejneger, L. 1887. How the great northern sea-cow (*Rytina*) became exterminated. American Naturalist 21:1047–1054.

Stewart, B. S., P. K. Yochem, H. R. Huber, R. L. DeLong, R. J. Jameson, W. J. Sydeman, S. G. Allen, and B. J. LeBoeuf. 1994. History and present status of the northern elephant seal population. Pages 29–48 *In* (B. J. LeBoeuf and R. M. Laws, eds.) Elephant Seals: Population Ecology, Behaviour and Physiology. University of California Press, Berkeley.

Wakefield, S., J. Knowles, W. Zimmermann, and M. van Dierendonck. 2003. Status and action plan for the Przewalski's horse (*Equus ferus przewalskii*). Pages 82–92 *in* Status Survey and Conservation Action Plan, Equids: zebras, asses, and horses (P. D. Moehlman, ed.). IUCN–The World Conservation Union.

Weber, M. L., and D. W. Laist. 2007. The status of protection programs for endangered, threatened, and depleted marine mammals in U.S. waters. Marine Mammal Commission, Bethesda, MD.

Table 1. Summary of the biological status of 22 marine mammal species and stocks currently listed as endangered or threatened under the Endangered Species Act or designated as depleted under the Marine Mammal Protection Act (from Lowry et al. 2007). Question marks indicate apparent population trends that have not been confirmed.

Taxon	Current population size	Current population trend	Population size relative to historical level
Endangered species			
West Indian manatee, Florida	>3,300	Increasing?	Unknown
West Indian manatee, Antillean	Unknown	Declining?	Reduced?
Caribbean monk seal	0	N/A	Extinct
Hawaiian monk seal	1,252	Declining 4.9 percent a year	Reduced 60 percent from 1958
Steller sea lion, western population	38,513	Stable	Reduced 81 percent from the 1970s
Blue whale[1]	>2,994	Increasing?	Reduced
Bowhead whale, western Arctic population	10,545	Increasing 3.4 percent a year	Reduced 54 percent from the 1800s
Fin whale	>11,970	Unknown	Reduced
Humpback whale	>6,692	Increasing	Reduced
North Atlantic right whale	299	Declining?	Reduced
North Pacific right whale, eastern population	>23	Unknown	Reduced
Sei whale[2]	>133	Unknown	Reduced
Sperm whale[3]	>14,468	Unknown	Reduced
Killer whale, southern resident population	84	Unknown	Reduced
Threatened species			
Southern sea otter	2,825	Increasing	Reduced
Northern sea otter, southwest Alaska population	41,865	Declining	Reduced 55 to 67 percent from 1976
Guadalupe fur seal	7,408	Increasing	Reduced
Steller sea lion, eastern population	44,996	Increasing	Unknown
Depleted (only) species			
Northern fur seal, eastern population	688,028	Declining	Reduced 65 percent from the 1950s
Beluga whale, Cook Inlet population	278	Declining?	Reduced 57 percent from 1994
Bottlenose dolphin, mid-Atlantic coastal population	33,000	Unknown	Reduced
Killer whale, AT1 group	8	Declining	Reduced 64 percent from 1988

[1]Data are not available for the North Atlantic and western North Pacific stocks.
[2]Data are not available for the Nova Scotia stock.
[3]Data are not available for the North Pacific stock.

Table 2. Examples of wild populations that have recovered from extremely low population sizes, with (*) or without the assistance of captive/assisted breeding programs; (+) indicates that additional animals survive in captivity.

Species, stock, or population	Estimate of minimum population size (approximate date)	Estimate of current wild population size	Source(s) of information
Marine mammals			
Northern elephant seal (*Mirounga angustirostris*)	20–100 (1890)	>175,000	Bartholomew and Hubbs 1960, Stewart et al. 1994
Southern sea otter (*Enhydra lutris nereis*)	50 (1938)	>2,500	Riedman and Estes 1990
Guadalupe fur seal[5] (*Arctocephalus townsendi*)	70–75 (1955)	>7,000	Hubbs 1956, Gallo 1994
Southern right whale (*Eubalaena australis*)	<300 (1920)	>7500	Baker and Clapham 2004
Juan Fernandez fur seal (*Arctocephalus philippii*)	700–750 (1970)	>12,000	Hubbs and Norris 1971, UNEP[6]
Terrestrial mammals			
Black-footed ferret* (*Mustela nigripes*)	18 (1987)	650+	Black-footed ferret recovery team[7]
Tule elk (*Cervus elaphus nannodes*)	28 (1895)	3,200	McCullough et al. 1996, NPS 1998
Przewalski horse* (*Equus ferus przewalskii*)	31 (1945)	175+	Wakefield et al. 2003
European bison* (*Bison bonasus*)	54 (1918)	1700+	Pucek 2004
Golden lion tamarin* (*Leontopithecus rosalia*)	<200 (1970s)	1,500+	Smithsonian Natl. Zoological Park[8]
Birds			
Mauritius kestrel* (*Falco punctatus*)	4 (1974)	800–1,000	Birdlife International[9]
Chatham island black robin* (*Petroica traversi*)	5 – one breeding pair (1979)	250	NZ DOC 2001[10]

[5] Considered extinct in the 1930s and early 1940s
[6] United Nations Environment Programme; http://www.unep-wcmc.org/species/data/species_sheets/juanfern.htm
[7] http://www.blackfootedferret.org/
[8] http://nationalzoo.si.edu/ConservationAndScience/EndangeredSpecies/GLTProgram/default.cfm
[9] http://www.birdlife.org/datazone/index.html
[10] also see: http://www.doc.govt.nz/Conservation/001~Plants-and-Animals/001~Native-Animals/Black-Robin.asp

Table 2, continued.

Species, stock, or population	Estimate of minimum population size (approximate date)	Estimate of current wild population size	Source(s) of information
Whooping crane* (*Grus americana*)	21 (1944)	>300^{+}	CWS and FWS 2005
California condor* (*Gymnogyps californianus)*	25–35 (1979)	127^{+}	California Dept. of Fish and Game[11]
Seychelles warbler (*Acrocephalus sechellensis*)	50 (1965)	>2,000	Birdlife International[5]
Guam rail* (*Gallirallus owstoni*)	100 (1983)	400^{+}	Smithsonian Natl. Zoological Park[12]

[11] http://www.dfg.ca.gov/hcpb/species/t_e_spp/tebird/condor.shtml
[12] http://nationalzoo.si.edu/Support/AdoptSpecies/AnimalInfo/Guamrail/default.cfm

Table 3. A theoretical decision tree and analytical recommendations for listings under the Endangered Species Act

Decision Tree Categories	PVA Model Recommendation
Populations that are too small	When a species or population is already known to be too small (through obvious proxy measures), then no PVA is necessary; the population should be listed.
Populations that are nearly too small	When a population is nearly too small, the important question is whether the population is likely to become too small. In that case, a simple PVA model could be designed to test whether the current or foreseeable population trend is, or is likely to be, negative.
Populations that are large, but declining	When a population is large but declining, the important question is whether the population could decline too much. In that case, a population viability model could be designed to test whether the current or foreseeable trend is likely to cause the population to become too small. The model would need to be slightly more complex because it must evaluate the possibility that the population decline could halt before the population became too small.
Populations that are large, but have volatile population dynamics	When a population is large but volatile (highly variable), the important question is whether the volatility in population dynamics is large enough to cause the population to become too small. For populations with a high degree of variability, a model could be designed to test whether the population could reach a critical threshold size as a result of random fluctuations. Such models must accurately represent the variability in various population parameters.

APPENDIX I

Examples of Existing Marine Mammal Population Models

North Atlantic Right Whales (Presented by Hal Caswell, Woods Hole Oceanographic Institution)

Background and purpose: The North Atlantic right whale is the least abundant species of large whale in the world. It occurs primarily along the East Coast of the United States and Canada. It was reduced to levels approaching extinction by centuries of commercial whaling and is listed as endangered under the Endangered Species Act. Current abundance is about 300 to 350 whales. Entanglement in fishing gear and collisions with ships are the major factors impeding recovery. A model was developed to estimate the population's reproductive, survival, and population growth rates; predict extinction risks; and evaluate hypotheses about factors influencing population trends, including potential effects of variable oceanographic conditions.

Approach: The matrix population model is based on biologically defined life history stages. It incorporates environmental and demographic variability by using observed variations in vital rates over the past 20 years. Parameters (e.g., stage-specific reproductive and survival rates) are estimated using mark-recapture methods (Caswell 2001, Fujiwara and Caswell 2002, Caswell and Fujiwara 2004). The estimation procedure automatically incorporates effects of uncertainty in the data and provides confidence intervals around parameter estimates.

Data: The model uses data from 1980 through 1998 on age, sex, and reproduction for individual whales documented in a photo-identification catalog of the population. The catalog is believed to include at least some records for most of the individuals in the population, although the number of resighting records varies greatly among individual whales. The data are subjected to a multi-stage mark-recapture analysis to estimate parameters under a variety of statistical hypotheses that consider variation over time and the effects of environmental variables. Information-theoretic criteria are used to select the statistical hypotheses that are most highly supported by the available data.

Results: The analysis suggests that the population was increasing at about 4 percent per year in the early 1980s, but that the growth rate declined until it became negative in the mid-1990s. The decline appears to be due to a declining trend in birth rates and survival of mothers and calves between 1980 and 1998. The rate of decline has varied with the North Atlantic Oscillation, suggesting that the population's dynamics are affected by changing atmospheric and oceanographic conditions. By the late 1990s the estimated population growth rate was below replacement level, indicating that the population would not persist without mitigation of human impacts. As for all marine mammals, population growth is most sensitive to the survival of mature females. All other things equal, a reduction of human-related mortality by two adult females per year could return the population growth rate to replacement level. Further reduction would allow recovery, albeit at a slow rate.

<u>*Recent publications using the model*</u>:

Caswell, H., M. Fujiwara, and S. Brault. 1999. Declining survival probability threatens the North Atlantic right whale. Proceedings of the National Academy of Science 96:3308–3313.
Fugiwara, M., and H. Caswell. 2001. Demography of the North Atlantic right whale. Nature 414:537–541.
Caswell, H., and M. Fujiwara. 2004. Beyond survival estimation: mark-recapture, matrix population models, and population dynamics. Animal Biodiversity and Conservation 27:471–488.
Kraus, S. D., M. W. Brown, H. Caswell, C. W. Clark, M. Fujiwara, P. K. Hamilton, R. D. Kenney, A. R. Knowlton, S. Landry, C. A. Mayo, W. A. McLellan, M. J. Moore, D. P. Nowacek, D. A. Pabst, A. J. Read, and R. M. Rolland. 2005. North Atlantic right whales in crisis. Science 309:561–562.
Caswell, H. 2006. Applications of Markov chains in demography. Pages 319–334 *in* (A. N. Langville and W. J. Stewart, eds.) MAM2006: Markov Anniversary Meeting. Boson Books, Raleigh, NC, USA.
Caswell, H. 2007. Sensitivity analysis of transient population dynamics. Ecology Letters 10:1–15.

Southern Resident Killer Whales (Presented by Paul Wade, National Marine Mammal Laboratory, National Marine Fisheries Service)

<u>*Background and purpose*</u>*:* Southern resident killer whales comprise three distinct pods, identified as J, K, and L pods, that occur principally in Washington's Puget Sound and southern British Columbia, Canada. Historically, the population may have included more than 200 individuals. In the mid-1960s the stock was thought to number at least 100 animals, but it then declined sharply in the late 1960s and early 1970s as a result of live captures for aquaria. By 1995 abundance had recovered to 98 animals. Since then, the stock first declined to 81 animals in 2001 and then increased to 88 animals in 2004. The declining trend seems to have been driven primarily by changes in the largest pod, L pod. The recent increase, however, has been driven primarily by an increase in J and K pods. Pod-specific trends are important because males rarely mate with females from their own pod (and resident killer whales in the North Pacific mate only within their ecotype). As a result, the reproductive success of a resident pod is determined not only by the fecundity of the females within that pod but also by the availability of fertile males from other resident pods. Three potential factors may be impeding recovery of southern resident killer whales: high contaminant loads; declines in available prey, particularly salmon; and disturbance by whale-watching ventures and other vessel activity. The purpose of this model was to estimate extinction risks for southern resident killer whales as part of a status review to inform decision-makers regarding listing of the stock as threatened or endangered under the Endangered Species Act.

Approach: The analysis used a sex-, age- and pod-structured model that allowed for (1) demographic variability, (2) environmental variability, (3) potential catastrophes, (4) Allee effects, and (5) variation in carrying capacity. Variation in survival rates was based on observed rates over the past 30 years, with an apparent 6-year cycle. Allee effects were imposed using social constraints on reproduction: females in a pod could not become pregnant unless another pod included an adult male. Carrying capacity was allowed to vary with values ranging from 100 to 400 animals.

Data: Demographic parameters for the model were estimated using censuses of southern resident killer whales conducted annually since 1974. Parameters included age and sex composition, survival, fecundity, and reproduction of each of the population's three pods. Initial parameters and ranges for environmental variability, catastrophes, and carrying capacity were based on a literature review and expert judgment. The model was initialized with the known 2003 age, sex, and pod composition and was projected into the future for 100, 200, and 300 years. Because survival rates varied throughout the 29-year census record, the model was run using survival estimates from three subsets of the data: the most recent 10 years (with the lowest survival rates), the most recent 14 years, and all 29 years.

Results: Scenarios using survival estimates from the preceding 29 years of data resulted in 0.1 to 3 percent likelihood of extinction in 100 years and 2 to 42 percent in 300 years. Scenarios using survival estimates from the most recent 10 years resulted in 1 to 19 percent likelihood of extinction in 100 years and 68 to 94 percent in 300 years. Survival rates in all three pods followed a similar pattern; the largest pod, which occurs farthest from shore, was the most severely affected by changes in survival. In addition, changes were evident in all age and sex classes, but old males appeared to be most affected. The results suggest that the patterns in survival may be caused by environmental factors (e.g., changes in prey availability through time). Results of the analysis supported a 2006 decision to list southern resident killer whales as endangered.

Recent publications using the model:

Krahn, M. M., M. J. Ford, W. F. Perrin, P. R. Wade, R. P. Angliss, M. B. Hanson, B. L. Taylor, G. M. Ylitalo, M. E. Dahlheim, J. E. Stein, and R. S. Waples. 2004. Status review of Southern Resident killer whales *(Orcinus orca)* under the Endangered Species Act. U.S. Dept. of Commerce, NOAA Tech. Memo. NMFS-NWFSC-62. 73 pp.

Cook Inlet Beluga Whales (Presented by Daniel Goodman, Montana State University)

Background and purpose: Data from National Marine Fisheries Service aerial surveys indicate that the Cook Inlet beluga whale stock declined from an estimated 653 individuals in 1994 to 347 in 1998, and then declined at a lower rate to 278 in 2005. The decline from 1994 to 1998 was due primarily to subsistence harvesting by Alaska Natives. After the harvest was brought under management control in 1998 (only three whales were harvested between 1999 and 2004), the stock was expected to recover at a rate of 2 to 6 percent annually. Its failure to recover suggests that other factors also are affecting this stock. Its summer range contracted concurrently with the decline, and beluga whales now are rarely seen in offshore waters or the lower reaches of the inlet. Because their remaining habitat is near Anchorage, the largest urban area in Alaska, the whales are exposed to a range of human activities. An analysis was conducted to characterize the trend to date and project whether the population is likely to decline further.

Approach: The analysis fits an exponential population growth model to past survey estimates and produces a distribution of possible growth rates, including negative values consistent with a decline. The distribution of growth rates is then used to predict the trend in the near future. The model does not include the effects of subsistence harvest (which may be negligible at present) or environmental variability.

Data: The analysis used annual abundance estimates (and variances of those estimates) from aerial surveys conducted between 1994 and 2004.

Results: The results suggest that even without environmental variation and harvests, the likelihood of continuing decline is 45 percent.[13] The National Marine Fisheries Service currently is reviewing a petition to list the Cook Inlet beluga whale stock as endangered under the Endangered Species Act.

Recent publications using the model:

Lowry, L., G. O'Corry-Crowe, and D. Goodman. 2006. *Delphinapterus leucas* (Cook Inlet population). *In* 2006 IUCN Red List of Threatened Species. IUCN–The World Conservation Union.

Florida Manatees (Presented by Michael Runge, Patuxent Wildlife Research Center, U.S. Geological Survey)

Background and purpose: The West Indian manatee, listed as endangered under the Endangered Species Act, is comprised of Antillean and Florida subspecies. The Florida subspecies occurs in rivers and coastal waters of the southeastern United States. Because of their limited tolerance for cold temperatures, most Florida manatees winter near warm-water discharges from natural springs or power plant outfalls located in the southern two-thirds of the Florida peninsula. The anticipated loss of discharge sites as a result of power plant closures represents a long-term threat that would reduce available habitat and lower the effective environmental carrying capacity for manatees. At present, the largest source of direct human-related manatee mortality—and

[13] A more recent analysis using the same model and data from 1994 to 2006 suggests an 81 percent probability that the population is declining (i.e., the growth rate is negative).

probably the most significant factor impeding population recovery—is collisions with boats. Watercraft-related deaths typically account for a quarter to a third of all manatee deaths annually.

Since 1985 population models have been used to assess trends in manatee abundance. More recently, modeling objectives have focused on evaluation of negative and positive effects of specific threats and management actions. At the Savannah workshop, the model's utility for predicting population trends and estimating the effects of changes in carrying capacity were described. The model will be used to inform reclassification decisions under the Endangered Species Act and Florida state statutes.

Approach: The model projects population trends based on reproduction and survival probabilities for each of several life history stages, such as calves, juveniles, adult males, and adult females. The model accounts for variability in demographic parameters and the largely independent dynamics of four relatively discrete manatee subpopulations in Florida. The model incorporates catastrophes (e.g., red tide bloom, disease epidemic), density dependence, and changes in habitat availability (e.g., availability of winter warm-water refuges). Additional factors can and will be added to address specific questions that arise, such as the effects of specific management actions, hurricanes, and climate change.

Data: Extensive data were used to develop model parameters (e.g., survival rates, reproductive rates, carrying capacity), including 10 to 25 years (depending on subpopulation) of photo-identification mark-recapture data on more than 1,000 animals. Estimates for model parameters such as carrying capacity and density dependence were based on the advice of an expert panel.

Results: The model has been used to estimate population trends for each of four regional subpopulations in Florida. When based on data from the last 10 years, the model indicates that manatees are increasing in three regions: the northwest (growth rate [λ] = 1.037), upper St. Johns River (λ = 1.062), and Atlantic (λ = 1.010). Manatees in the southwest region appear to be declining (λ = 0.989), based on the most recent 10 years of data. Further analysis suggests that management actions should focus on increasing adult survival rates and that improved monitoring of those rates would reduce the overall uncertainty in model results.

Recent publications using the model:

Runge, M. C. 2003. A model for assessing incidental take of manatees due to watercraft-related activities. In U.S. Fish and Wildlife Service Environmental Impact Statement: Rulemaking for the incidental take of small number of Florida manatees (*Trichechus manatus latirostris*) resulting from government programs related to watercraft access and watercraft operation in the state of Florida, Appendix I (March 2003) U.S. Fish and Wildlife Service. Jacksonville, FL.

Runge, M. C., C. A. Langtimm, and W. L. Kendall. 2004. A stage-based model of manatee population dynamics. Marine Mammal Science 20:361–385.

Haubold, E. M., C. Deutch, and C. Fonnesbeck. 2006. Final biological status review of the Florida manatee, *Trichechus manatus latirostris*. Florida Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission. St. Petersburg, FL.

Hawaiian Monk Seals (Presented by Albert Harting, Harting Biological Consulting)

Background and purpose: The Hawaiian monk seal is the most endangered seal in U.S. waters. Hawaiian monk seals occur almost entirely in the Hawaiian archipelago, where about 90 percent of all animals live on and around the remote Northwestern Hawaiian Islands. The majority of pups are born at six relatively discrete breeding subpopulations located at French Frigate Shoals, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Atoll, and Kure Atoll. Since the late 1950s when Hawaiian monk seals were first studied, beach counts at these six major pupping colonies have declined by more than 60 percent. The total population is currently estimated to number fewer than 1,300 animals and is declining at a rate of about 4 to 5 percent per year.

Threats to monk seal recovery include both human-related and natural factors that have varied over time and by colony. The human-related factors include disturbance and displacement of hauled-out seals by people and animals, entanglement in derelict fishing gear, depletion of prey resources by commercial fishing, interactions with recreational and commercial fishing gear, and oil spills. Natural factors include shark predation; naturally occurring biotoxins; disease; aggressive behavior by some adult male seals toward pups, juveniles, and adult females; the effects of oceanographic changes on prey resources; and the loss of pupping beaches to erosion. In general, the small, isolated nature of the Northwestern Hawaiian Islands makes their local ecosystems exceedingly vulnerable to both natural and human impacts.

A metapopulation model was developed to predict population abundance over relatively short time horizons (5 to 20 years), assess the sensitivity of the population to natural or management-induced perturbations, and perform long-range projections for risk assessments and population viability analyses.

Approach: The metapopulation model represents the species as a group of spatially distinct and largely independent breeding subpopulations. The primary events during each simulation year are births and deaths, both of which could be affected by simulated catastrophes. The simulation also can include specific natural perturbations (e.g., adult male aggression and shark predation), migration between subpopulations, and management actions (e.g., captive rearing or translocation of pups and removal of aggressive adult males). Density-dependent regulation of reproductive and survival rates also can be incorporated in simulations.

Data: The model uses the life history and demographic data collected through an intensive program of cohort tagging and replicate seasonal counts of each subpopulation. These efforts, conducted since the early 1980s, provide detailed data for estimating age-specific survival and reproductive rates, current age/sex structure, inter-site movement rates, and the effects of specific natural perturbations (e.g., shark predation rates).

Results: This model has been used for analyzing the impact of shark predation on monk seal recovery at French Frigate Shoals, conducting National Environmental Policy Act assessments, analyzing potential impacts of an epizootic outbreak in the Northwestern Hawaiian Islands, and assessing the likely benefits from proposed management interventions. Further development of this model will include linking the model to oceanographic data, analyzing historical data for evidence of density-dependent regulation to refine the density-dependence formulation now used in the model, and adding the main Hawaiian Islands monk seal subpopulation to the model.

Due primarily to poor juvenile survival rates, long-term projections using survival rates derived from observations in recent years (2001–2005) indicate a decline at all subpopulations except Laysan Island, with French Frigate Shoals declining most precipitously. Projections utilizing survival rates (and variances) from all data years (1985–2005) again predict a marked decline at French Frigate Shoals, with a gradual decline at Kure Atoll (remaining subpopulations stable or increasing).

Recent publications using the model:

Harting, A. L. 2002. Stochastic simulation model for the Hawaiian monk seal. Ph.D. thesis, Montana State University, Bozeman. 328 pp.
National Marine Fisheries Service. 2006. Draft recovery plan for the Hawaiian monk seal, *Monachus schauinslandi*. National Marine Fisheries Service. Silver Spring, MD. 148 pp.

Steller Sea Lions — Western Stock (Presented by Daniel Goodman, Montana State University)

Background and purpose: Over the past three or four decades, Steller sea lions have declined precipitously throughout portions of their range, and the species was listed as threatened under the Endangered Species Act in 1990. In 1997 the species was determined to consist of at least two distinct population segments. The western population segment, occurring from the central Gulf of Alaska and westward, was listed as endangered, while the eastern segment, occurring from California through Southeast Alaska, remained listed as threatened. The decline of the western stock slowed during the 1990s, and since 2000 its abundance may have stabilized. The cause or causes of decline have been a matter of extensive debate and controversy. Leading hypotheses include nutritional stress as a result of competition with commercial fisheries and/or shifting environmental conditions, and predation by killer whales. The marked changes observed in western Steller sea lion abundance in recent decades suggest that the population may face significant extinction risks even at relatively high abundance. A population viability analysis was undertaken to estimate the population's extinction risk.

Approach: The PVA for western Steller sea lions simulated a random population growth regime based on observed, unexplained variation over recent decades, with the assumption that such variation would continue into the future. The results were summarized in terms of probability of extinction over time and were used to inform efforts by the Steller Sea Lion Recovery Team to develop recovery criteria (National Marine Fisheries Service 2006).

Data: Available data consist of 45 years of episodic census figures. Additional information on environmental variation also was used in the model.

Results: The analysis concluded that in the next 100 years the population has a 37 percent chance of declining to an effective population size of 1,000 animals, which equates to a total population size of 4,743 animals.

Recent publications using the model:

National Marine Fisheries Service. 2006. Draft Revised Recovery Plan for the Steller sea lion (*Eumetopias jubatus*). National Marine Fisheries Service, Silver Spring, MD.

Steller Sea Lions — Eastern Stock (Presented by Daniel Goodman, Montana State University)

Background and purpose: The eastern stock of Steller sea lions, a distinct population segment from California through Southeast Alaska, has increased steadily in size since the 1980s. It is recovering from human-caused mortality from the late 1800s to the 1970s. A population growth model was developed to determine whether the population is still growing.

Approach: The analysis fits an exponential growth model to past survey estimates to produce a distribution of possible growth rates. The distribution indicates the likelihood of continued increase. The residuals of the growth model are random and relatively small, suggesting that neither external factors (e.g., environmental variation) nor density dependence is altering the population dynamics appreciably, and those factors need not be incorporated into the model.

Data: The available data were drawn from annual censuses conducted from 1976 to 2002.

Results: The exponential growth model fits the population data well and indicates that the population is still growing.

Recent publications using the model:

National Marine Fisheries Service. 2006. Draft Revised Recovery Plan for the Steller sea lion (*Eumetopias jubatus*). National Marine Fisheries Service, Silver Spring, MD.

California Sea Otters (Presented by Martin Tinker, Dept. of Ecology and Evolutionary Biology, University of California, Santa Cruz)

Background and purpose: Over-exploitation for the fur trade in the 18th and 19th centuries brought sea otters in the North Pacific to near-extinction by the early 1900s. When commercial harvesting was banned by an international treaty in 1911, only a small remnant population of what is now recognized as the southern sea otter subspecies survived in California along the coast between Monterey and Big Sur. Recovery of the subspecies after 1911 was extraordinarily slow, and in 1977 the U.S. Fish and Wildlife Service listed the subspecies as threatened under the Endangered Species Act. In the late 1980s a translocation program was implemented to establish southern sea otters at San Nicolas Island off southern California and thereby provide another colony that could serve as a source for recovery efforts if the still-small mainland population were to be affected by a large catastrophe, such as an oil spill. At the same time, the Service established a management zone south of Point Conception where sea otters were to be excluded to prevent a southward expansion of the sea otter range and thereby protect commercial shellfish fisheries in southern California.

Recent surveys indicate that the population is increasing but still at a slower rate than expected. The population also has been expanding its range northward as well as southward into the management zone. In the late 1990s the number of otters in the management zone increased, and efforts to remove them were limited and largely unsuccessful. In addition, the translocated population failed to increase as expected and currently numbers just a few tens of animals. As a result, the Fish and Wildlife Service is considering steps to declare the translocation program a failure and to allow the mainland population to continue its expansion southward.
A population model was developed at the request of the Fish and Wildlife Service to project the rate of population recovery and range expansion over time. It provides a general framework for evaluating management options and conservation strategies for the southern sea otter.

Approach: The model is stage-based and incorporates spatial structure, movement rates, and demographic information. The model begins by estimating survival rates by location and time for each age/sex class (stage). The survival rates are then combined with stage-specific dispersal kernels to parameterize a multi-state matrix model for the entire population (structured by spatial region and allowing for inter-regional movements), which utilizes stage-structured difference equations to predict annual rates of range expansion into unoccupied habitat. Uncertainty is explicitly incorporated by using Monte Carlo simulations that allow all parameters to vary within the full range of previously observed values, and simulations include variance due to measurement and process error.

Data: The model uses survey data from the past 20 years (including total counts and counts of mature animals and dependent pups), age-at-death data from beach-cast carcasses, and radio telemetry data from free-ranging sea otters. From these data sources, estimates are derived (using maximum likelihood methods) for stage-specific reproduction, survival, and annual individual movement distances for identified age and sex components of regional subpopulations, as well as associated variance estimates for each parameter.

Results: Survival rates vary regionally, with lowest survival in the north and highest in the south. Female survival decreased in the mid-1990s. Sea otters are expanding their range southward at a rate of about 6 km/year.

Recent publications using the model:

Tinker, M. T., D. F. Doak, J. A. Estes, B. B. Hatfield, M. M. Staedler, and L. Bodkin James. 2006. Incorporating diverse data and realistic complexity into demographic estimation procedures for sea otters. Ecological Applications 16:2293–2312.

Tinker, M. T., J. A. Estes, K. Ralls, T. M. Williams, D. Jessup, and D. P. Costa. 2006. Population Dynamics and Biology of the California Sea Otter (*Enhydra lutris nereis*) at the Southern End of its Range. MMS OCS Study 2006–2007. Coastal Research Center, Marine Science Institute, University of California, Santa Barbara, California. MMS Cooperative Agreement Number 14-35-0001-31063.

U.S. Fish and Wildlife Service. 2005. Draft Supplemental Environmental Impact Statement on the Translocation of Southern Sea Otters. Ventura, California. 242 pp + appendices.

APPENDIX II

Workshop Attendees

Presenters	
Hal Caswell	Wood Hole Oceanographic Institution
Jean Cochrane	U.S. Fish and Wildlife Service
Deborah Crouse	Endangered Species Office, U.S. Fish and Wildlife Service
Daniel Goodman	Montana State University, Bozeman
Albert Harting	Harting Biological Consulting
Lloyd Lowry	Marine Mammal Commission, Committee of Scientific Advisors
Marta Nammack	National Marine Fisheries Service
Michael Runge	Patuxent Wildlife Research Center, U.S. Geological Survey,
Barbara Taylor	Southwest Fisheries Science Center, National Marine Fisheries Service
Martin (Tim) Tinker	Dept. of Ecology and Evolutionary Biology, University of California, Santa Cruz
Paul Wade	National Marine Mammal Laboratory, National Marine Fisheries Service
Michael Weber	Consultant
Attendees	
Steven Beissinger	University of California, Berkeley
Daryl Boness	Marine Mammal Commission, Committee of Scientific Advisors
Douglas Burn	U.S. Fish and Wildlife Service
Therese Conant	National Marine Fisheries Service, Office of Protected Resources
Paul Dayton	Scripps Institution of Oceanography, University of California, San Diego
Daniel Doak	Dept. of Ecology and Evolutionary Biology, University of California, Santa Cruz
Thomas Eagle	National Marine Fisheries Service
David Gouveia	National Marine Fisheries Service
Dawn Jennings	U.S. Fish and Wildlife Service
Craig Johnson	National Marine Fisheries Service
Rosa Meehan	U.S. Fish and Wildlife Service
Katherine Ralls	Smithsonian National Zoological Park, Conservation & Research Center
Randall Reeves	Okapi Wildlife Associates
Gregory Silber	National Marine Fisheries Service
Robert Small	Alaska Department of Fish and Game
James Valade	U.S. Fish and Wildlife Service
Marine Mammal Commission and SRA Staff	
David Cottingham	Marine Mammal Commission
David Laist	Marine Mammal Commission
Linda Manning	SRA International
Regan Maund	SRA International
Timothy Ragen	Marine Mammal Commission
Michael Simpkins	Marine Mammal Commission
Elizabeth Taylor	Marine Mammal Commission

APPENDIX III

Workshop Agenda

Workshop Objectives

- Review efforts to use PVA and other types of population models for endangered, threatened and depleted marine mammals occurring substantially in U.S. waters;
- Evaluate the extent to which PVA or other types of models can be relied upon for determining population status and predicting population trends for ESA classification listing decisions; and
- Evaluate the ability of PVA or other types of models to improve the decision-making process with regard to developing and selecting potential management actions other than ESA classification listing decisions.

Day One ***Review existing efforts to use PVA and other types of population models for endangered, threatened and depleted marine mammals***

8:30 – 9:00 Opening Remarks, Introductions, and Agenda Review *(David Cottingham)*

9:00 – 10:30 Current Approaches to the Assessment and Management of Listed Species under U.S. Law

- Overview of U.S. Fish and Wildlife Decision-making Processes from Listing through Delisting — Current Approaches (*Deborah Crouse)*
- Overview of National Marine Fisheries Service Decision-making Processes from Listing through Delisting — Current Approaches (*Marta Nammack)*
- Cases of U.S. Fish and Wildlife Service Use of Decision-Making Tools Including PVA *(Jean Cochrane)*
- Current Tools and New Approaches at the National Marine Fisheries Service (*Barbara Taylor and Paul Wade*

10:30 – 10:45 Break

10:45 – 5:00 Reviewing Existing Marine Mammal Population Models

- Florida Manatees (*Mike Runge)*
- Hawaiian Monk Seals (*Albert Harting)*
- North Atlantic Right Whales (*Hal Caswell)*
- Southern Resident Killer Whales (*Paul Wade)*
- California Sea Otters (*Tim Tinker)*
- Steller Sea Lions and Cook Inlet Beluga Whales (*Daniel Goodman)*

Day Two	***Evaluate the extent to which PVA or other types of models can be relied upon for determining population status and predicting population trends for ESA classification listing decisions***
8:30 – 9:00	**Opening Remarks/Agenda Review**
9:00 – 9:45	**Overview of Available Information on the Status of Listed Marine Mammals (***Lloyd Lowry)*
9:45 – 10:00	**Break**
10:00 – 12:30	**Panel Session IA:** What steps could or should be taken to better use or develop PVA models or other types of models to improve ESA classification decisions for "data rich" species?
12:30 – 2:00	**Lunch break**
2:00 – 5:00	**Panel Session IB:** What steps could or should be taken to better use or develop PVA models or other types of models to improve ESA classification decisions for "data poor" species?
5:00 - 5:30	**Closing Remarks**

Day Three	***Evaluate the ability of PVA methods to improve decision-making processes for developing and selecting potential management actions other than ESA classification decisions***
8:30 – 9:30	**Overview of the Status of Protection Programs for Listed Marine Mammals** *(Michael Weber)*
9:30 – 11:00	**Panel Session II –** What steps could or should be taken to develop or improve population models for use in making routine/operational management decisions for the endangered, threatened, and depleted species?
11:00 – 11:15	**Break**
11:15 – 1:00	**Panel Session II, continued**
1:00 – 2:00	**Closing Remarks and Next Steps**