Alaska
Fisheries Science
Center

National Marine
Fisheries Service

U.S DEPARTMENT OF COMMERCE

AFSC PROCESSED REPORT 2008-08

2008 Supplemental Status Review and Extinction Assessment of Cook Inlet Belugas (*Delphinapterus leucas*)

October 2008

This document should be cited as follows:

2008 SUPPLEMENTAL STATUS REVIEW AND EXTINCTION ASSESSMENT OF COOK INLET BELUGAS (*DELPHINAPTERUS LEUCAS*)

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October 2008

Cover photo: Belugas (*Delphinapterus leucas*) in the Susitna River delta, Cook Inlet, Alaska, June 1994.
Photographer: J. Waite (AFSC-NMML).

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EXECUTIVE SUMMARY

Background

After completion (in 2005) of the Draft Conservation Plan for Cook Inlet belugas (*Delphinapterus leucas*) under the Marine Mammal Protection Act (MMPA); the National Marine Fisheries Service (NMFS) recommended that a Status Review be conducted to incorporate new scientific findings available since the publication of a scientific review in 2000 in the journal *Marine Fisheries Review* 62 (3). NMFS formally initiated this Status Review on March 29, 2006 to determine if Cook Inlet belugas should be listed under the U.S. Endangered Species Act (ESA). On April 20, 2006, NMFS received a petition from Trustees for Alaska to list Cook Inlet belugas as endangered under the ESA. After reviewing the information contained in the petition, as well as other scientific information readily available, NMFS determined the petitioned action may be warranted. Within 12 months of the date of the petition, NMFS was required to make one of the following findings:

1) the petitioned action is not warranted;
2) the petitioned action is warranted and the Secretary of Commerce will publish in the *Federal Register* (FR) a proposed regulation to implement the action pursuant to 50 CFR 424.16; or
3) the petitioned action is warranted, but

      A) the immediate proposal and timely promulgation of a regulation to implement the petitioned action is precluded because of other pending proposals to list, delist, or reclassify species; and

      B) expeditious progress is being made to list, delist, or reclassify qualified species, in which case such findings shall be promptly published in the FR.

The Status Review published in November 2006 provided a summary of the best available science to aid NMFS managers in this process. Based on the findings from the Status Review and consideration of the factors affecting this species, NMFS concluded Cook Inlet belugas constituted a distinct population segment (DPS) that was in danger of extinction throughout its range. NMFS issued a proposed rule to list the Cook Inlet beluga DPS as an endangered species

on April 20, 2007. Since completion of the 2006 Status Review and extinction assessment of Cook Inlet belugas, NMFS completed analyses of the 2006 and 2007 aerial survey data and generated abundance estimates for those years. In November 2007, NMFS initiated a review of the science presented in the 2006 Status Review by a panel of independent experts through the Center for Independent Experts (CIE). This review led to the publication of a new report in April 2008. The report provided an update of the best available science obtained since publication of the 2006 Status Review in response to the CIE review and public comments. After completion of the 2008 aerial survey and abundance estimate, a supplemental Status Review was completed in October 2008. This document provides the abundance estimate from 2008 and revised trend and extinction risk analyses.

Status of Cook Inlet Belugas

Temporal Changes in Distribution

Since the mid-1990s, 96% to 100% of the observed Cook Inlet belugas have congregated in the upper Inlet in shallow areas near river mouths—they were only occasionally found in the central or southern portions of the inlet during the summer months. It is unknown if this contracted distribution is a result of changing habitat, prey concentration, predator avoidance, or a more acute reduction of the population into all but a small number of preferred habitat areas. This concentration of belugas in the northernmost portion of Cook Inlet appears to be a fairly consistent pattern from June to October. Data from tagged whales (14 tags between July and March 2000-03) show that belugas use the upper inlet intensively between summer and late autumn, but during winter months they also disperse to mid-inlet offshore waters. Tagged whales and extensive surveys both within Cook Inlet and in the Gulf of Alaska indicate that belugas do not have a seasonal migration in and out of the inlet. Yakutat Bay is the only location outside of Cook Inlet where there is a known, persistent population of several belugas. It is not clear from available data whether this group is isolated from Cook Inlet.

Population Size and Trend

NMFS began comprehensive, systematic aerial surveys of the beluga population in Cook Inlet in 1993. Unlike previous efforts, these surveys included the upper, middle, and lower sections of the inlet. These surveys documented a decline in abundance of nearly 50% between 1994 and 1998, from an estimate of 653 whales to 347 whales. Although this rapid decline stopped after the Native subsistence harvest was regulated in 1998, beluga numbers have not increased.

Data analyses indicated that the documented decline in beluga abundance from 1994 to 1998 is adequately explained by the estimated mortalities from the subsistence hunt for the same period. With the very limited hunt between 1999 and 2008 (a total of 5 whales), NMFS anticipated that the population would begin to recover at a growth rate of 2% to 6% per year. However, a Bayesian analysis including the 2008 estimate of abundance indicates that there is a probability of only 5% that the growth rate is above 2%, and a probability of 62% or more that the population will decline further. The best available data at this time indicate that the Cook Inlet beluga population is not growing as expected despite the limits on the subsistence take.

Determination of Distinct Population Segment

NMFS established Cook Inlet belugas as a distinct population segment (DPS) and, therefore, a species as defined under Section 3(15) of the ESA on June 22, 2000. At the time, the Cook Inlet stock had been designated as depleted under the MMPA (May 31, 2000) and included all belugas in waters of the Gulf of Alaska north of 58° N latitude (including, but not limited to, Cook Inlet, Kamishak Bay, Chinitna Bay, Tuxedni Bay, Prince William Sound, Yakutat Bay, Shelikof Strait, and off Kodiak Island and freshwater tributaries to these waters). The population of belugas in Cook Inlet is discrete from other Alaskan and Russian beluga populations in the Arctic. Physically, these whales are isolated from other populations by the Alaska Peninsula. Despite extensive, dedicated marine mammal survey effort, the lack of sightings along the southern side of the Alaska Peninsula and Aleutian Islands chain suggests that the Cook Inlet population does not disperse into the Bering Sea. Behaviorally, belugas show strong maternally-driven site-fidelity to summering areas, suggesting opportunity for intermixing may only occur during

winter migrations. However, the available data suggest that belugas remain in Cook Inlet year-round and do not undertake extensive migrations. Furthermore, the genetic characteristics of this population differ markedly from the other four beluga populations that occur off western and northern Alaska. Given the site-fidelity of beluga populations, it is unlikely that immigrants from other Arctic beluga populations would repopulate Cook Inlet in the foreseeable future if the Cook Inlet beluga population goes extinct. As it is the only population found in subarctic waters east of the Alaska Peninsula, its loss would result in a significant reduction in the range of the taxon.

Risk Assessment

Risk Factors

The ESA defines an endangered species as any species in danger of extinction throughout all or a significant portion of its range, and a threatened species as any species likely to become endangered within the foreseeable future. Section 4(b)(1)(a) of the ESA requires that determinations of whether a species is threatened or endangered be based solely on the best scientific and commercial data available, after taking into account those efforts, if any, being made to protect the species. The Secretary shall determine whether any species is endangered or threatened because of any of the following factors listed under Section 4(a)(1) of the ESA:

A) The present or threatened destruction, modification, or curtailment of habitat or range;
B) Overutilization for commercial, recreational, scientific, or educational purposes;
C) Disease or predation;
D) The inadequacy of existing regulatory mechanisms; or
E) Other natural or manmade factors affecting its continued existence.

There are a number of behavioral and ecological characteristics that put Cook Inlet belugas at considerable risk of extinction. These include but are not limited to the following: 1) life history characteristics such as slow population growth rate; 2) distorted age, size or stage structure of the population, and reduced reproductive success; 3) strong depensatory or Allee effects; 4) habitat

specificity or site fidelity; and 5) habitat sensitivity. The genetic and spatial isolation of the Cook Inlet beluga population and its strong site-fidelity greatly increases the risk of inbreeding and expression of deleterious genes should this population decline continue. At a reduced abundance and with a contraction of their range, this population is far more vulnerable to losses due to stranding, predation, or disease. Cook Inlet belugas rely heavily on several fish prey species that are available only seasonally and are also of considerable commercial interest. Disturbances that cause belugas to temporarily or permanently abandon summer feeding areas could reduce their ability to survive through the winter months.

Population Viability Analysis

A detailed population viability analysis (PVA) model, including immature and mature stages of both sexes, developed for the Cook Inlet beluga PVA in the 2006 Status Review was rerun and expanded to include new data from 2008 and address issues raised during the CIE review process, in particular the possibility that small, gray calves and juveniles are undercounted in aerial surveys. This model focused on the behavior of a declining population with less than 500 belugas. Small population effects, demographic stochasticity, Allee effects, predation mortality, and unusual mortality events were modeled explicitly. The modeled Allee effect and predation mortality produced thresholds of population size below which the population could not recover; extinction occurred more or less rapidly depending on the height of the population size threshold. This threshold was particularly pronounced when predation (C) was set at two mortalities or greater per year causing a visible break point below which there was little likelihood of the population avoiding extinction. The probability of extinction within 100 years ranged from 1% to 27%, and within 300 years ranged from 29% to 70% in the models that were considered to have parameters most representative of the Cook Inlet beluga population (ES-Fig.1, models a, c-e, g-h). What was thought to be the most realistic model (ES-Fig. 1, model h), with an average of one predation mortality per year and a 5% annual probability of an unusual mortality event killing 20% of the population, resulted in a 1% probability of extinction in 50 years, 26% probability of extinction in 100 years and 70% probability of extinction in 300 years, and an 80% probability that the population was declining. The model with five predation mortalities per year (ES-Fig. 1, model f) showed that the extinction probability was sensitive to changes or

underestimation of this parameter and that the population at its current size of 375 would be near the threshold population size (200-300 animals) for this model, even if the population was otherwise healthy but suffered occasional unusual mortality events. The model with no threshold effects (i.e., Allee or predation) resulted in a 68% probability of decline and 29% probability of extinction within 300 years (ES-Fig. 1, model a). Even with this most optimistic scenario, with no harvest after 2008, the probability that the population would be larger than 500 animals in 2108 (within 100 years) was only 24% (ES-Fig. 2, model a).
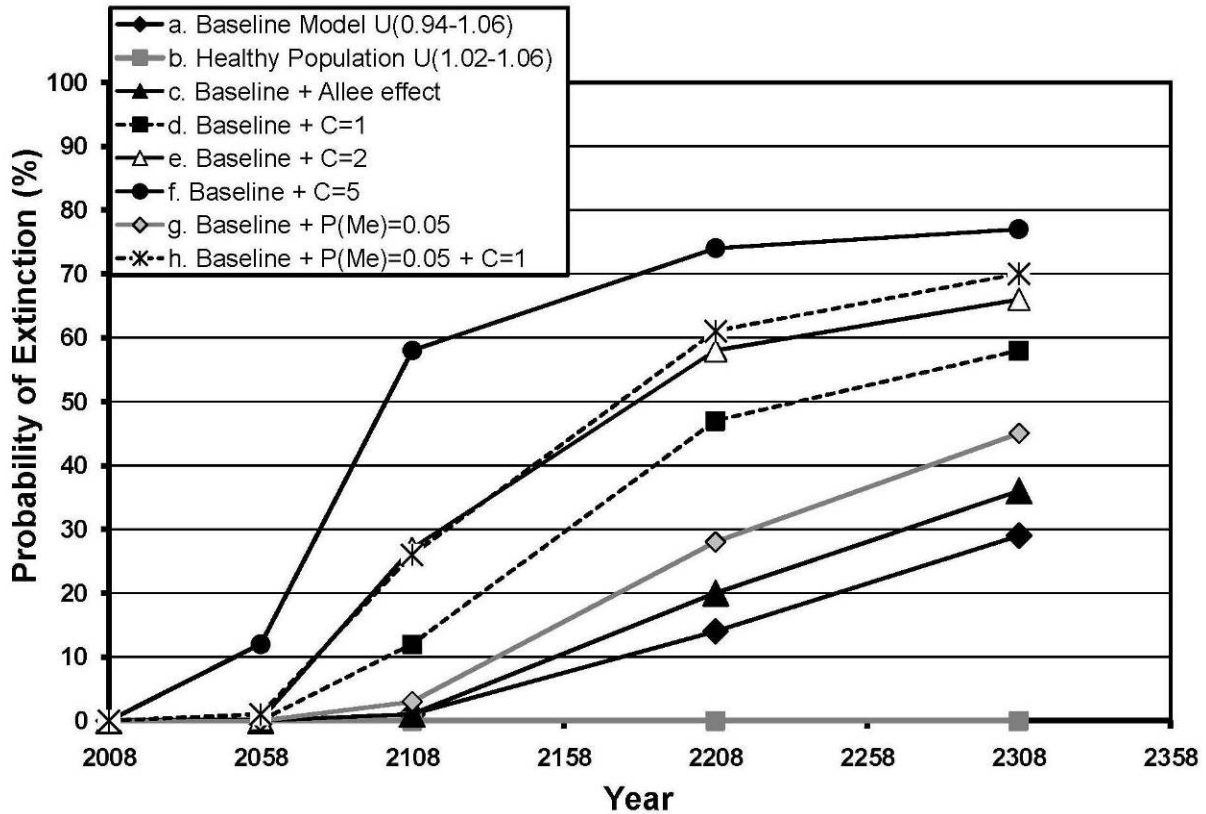
ES-Figure 1. Probability of extinction by year for the Cook Inlet beluga population resulting from each population viability analysis model. Models using the same parameters are the same line style, color, and symbol type with open symbols indicating the inclusion of the unusual mortality event parameter P_{Me} set at a 5% annual probability of a 20% mortality. The constant mortality effect parameter (C) was set at 1, 2 or 5 whales per year. U = uniform distribution (of the annual growth multiplier). The Baseline model allowed declining and increasing annual growth while the Healthy Population model allowed only increasing annual growth.
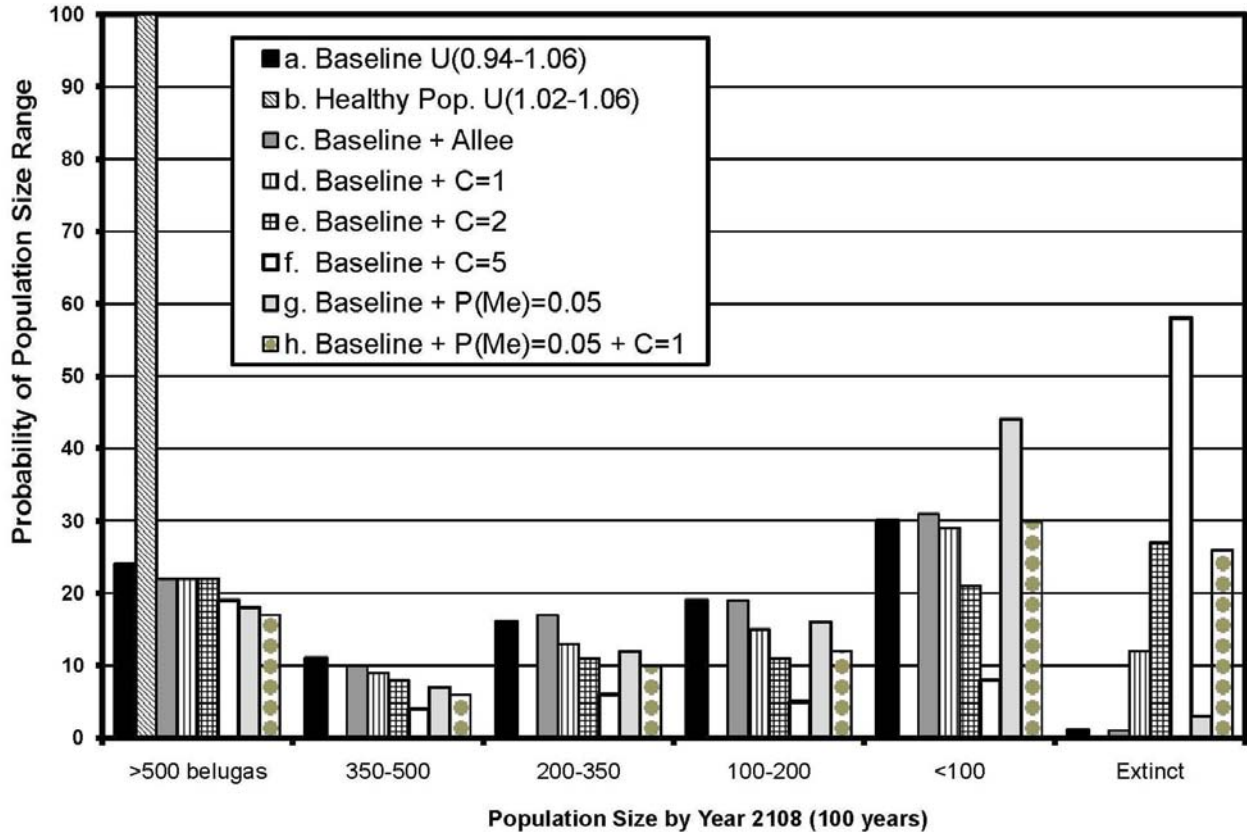
ES-Figure 2. Probability of the Cook Inlet beluga population size resulting from population viability analysis outcomes after 100 years. The Baseline model allowed declining and increasing annual growth while the Healthy Population model allowed only increasing annual growth. U = uniform distribution (of the growth multiplier), C = constant mortality effect parameter (e.g., predation) set at 1, 2, or 5 belugas, P_{Me} = unusual mortality event parameter set at 5% annual probability of 20% mortality.

Conclusions of the Status Review

- The contraction of the range of this population northward and westward into the upper inlet makes it far more vulnerable to catastrophic events which have the potential to kill a significant fraction of the population.

- The population is not growing at 2% to 6% per year as had been anticipated since the cessation of unregulated hunting.

- The population is discrete and unique with respect to the species, and if it should fail to survive, it is highly unlikely that Cook Inlet would be repopulated with belugas. This would result in a permanent loss of a significant portion of their range.

- The importance of seasonal anadromous fish runs in Cook Inlet to belugas is evident. The bulk of their annual nutrition is acquired during the summer months.

- Belugas in Cook Inlet are unique in Alaska given their summer habitat is in close proximity to the largest urban area in the state.

- While the impact of disease and parasitism on this population has not been quantified, this population is at greater risk because of its small size and limited range such that a novel disease would spread easily through this population.

- The PVA shows a 26% probability of extinction in 100 years and 70% probability of extinction in 300 years (for the model assuming one predation mortality per year and a 5% annual probability of an unusual mortality event killing 20% of the population). It is likely that the Cook Inlet beluga population will continue to decline or go extinct over the next 300 years unless factors determining its growth and survival are altered in its favor.

1. INTRODUCTION

1.1. Scope and Intent of the Status Review Update

Following the 2006 review of the status of the beluga (*Delphinapterus leucas*) population in Cook Inlet, Alaska (Hobbs et al. 2006), the National Marine Fisheries Service (NMFS) proposed listing the Cook Inlet beluga population as an endangered distinct population segment (DPS) under the U.S. Endangered Species Act (ESA) on April 20, 2007 (72 FR 19854). NMFS continued to gather available scientific data and complete the 2006 and 2007 abundance estimates. In November 2007, NMFS hosted an independent review of the science presented in the 2006 Status Review supplemented with information available before October 2007 by a panel of experts through the Center for Independent Experts (CIE). The 2008 Status Review (Hobbs et al. 2008) included a review of data that have become available since the 2006 review, the findings of the CIE review, and updated models of extinction scenarios for the Cook Inlet beluga DPS using the 2006 and 2007 abundance estimates. After completing the 2008 aerial survey and generating an abundance estimate, new trend and extinction risk analyses were run. This *AFSC Processed Report* provides these revised analyses. The Introduction is reproduced here as it appears in the 2008 assessment (Hobbs et al. 2008), with the addition of actions that occurred after April 2008.

1.2. History of the Status of Cook Inlet Belugas

1.2.1. Candidate Species Listing—1988

Status Reviews are prepared by NMFS for marine species that are being considered for listing as a "Species of Concern" (69 FR 19975, April 15, 2004), Candidate Species (50 CFR 424.02), or that are already listed as endangered or threatened under the ESA (5 USC 1533). On August 31, 1988, NMFS announced the creation of a list of Candidate Species being considered by the Secretary of Commerce (NMFS is an agency within the U.S. Department of Commerce) for listing as threatened or endangered species under the ESA. A Candidate Species is a species that

the U.S. Fish and Wildlife Service (USFWS) or NMFS is considering listing as endangered or threatened but which has not yet been the subject of a proposed rule. Candidate Species are afforded no protection under the ESA, but § 4(b)(3)(C)(iii) of the Act requires the agencies to monitor the status of certain candidate taxa "to prevent their extinction while awaiting listing" (58 FR 51146, September 30, 1993).

Belugas found in Cook Inlet, Alaska, and infrequently in waters east of the Alaska Peninsula (Laidre et al. 2000), were included on the 1988 List of Candidate Vertebrate and Invertebrate Marine Species (53 FR 33516, August 31, 1988). The decision to list Cook Inlet belugas as a Candidate Species was based on information summarized in a species account (Hazard 1988) that was part of a larger compendium on selected marine mammal species in Alaska published in early 1988 (Lentfer 1988). This compendium was distributed to the NMFS and USFWS where it was used to develop or update research and management plans for species under their jurisdiction (MMC 1989:173). At the time, the limited available research suggested belugas in Cook Inlet made up a small population numbering less than 500 animals that was isolated from all other beluga populations in Alaska waters. On September 15, 1988, the NMFS office in Anchorage, Alaska (NMFS Alaska Region Office) prepared a review (Morris 1988) of all available information on Cook Inlet belugas including priorities and recommendations for research that would be needed to sustain the population at a stable level.

Cook Inlet belugas remained on the Candidate List when it was revised on June 11, 1991 (56 FR 26797). Aerial surveys were conducted on 8 and 10 June (Shelden 1994) and 18-21 June (NMFS 1992) in 1991 to determine the size of the population. The highest uncorrected count for these surveys was less than 250 animals. The status report prepared by the NMFS Alaska Region Office (NMFS 1992) again included recommendations for research to determine trends, genetic status, winter distribution, and life history parameters. Abundance surveys and tissue sampling began in 1992, while other studies such as ship-based oceanographic sampling, tagging studies, and acoustic monitoring have occurred when funding allowed since 1994. When the candidate list was revised on July 14, 1997 (62 FR 37560, December 18, 1997), it was noted that Cook Inlet belugas continued to be listed and that research had been initiated as a result of the 1991 listing.

1.2.2. Status Review—1998-2002

Prompted by a sharp decline in the estimated abundance of Cook Inlet belugas between 1994 (653 animals) and 1998 (347 animals), a reduction of nearly 50% (Hobbs et al. 2000a), NMFS initiated a Status Review of the population on November 19, 1998 (63 FR 64228). The comment period on the Status Review, which began at the same time that workshops were convened to review beluga populations throughout Alaska, extended from November 19, 1998 through January 19, 1999. The workshops were held by the Alaska Beluga Whale Committee (November 16-17, 1998) and the Alaska Scientific Review Group (November 18-20, 1998), a body established under the MMPA to provide scientific advice regarding marine mammals to NMFS and the USFWS.

NMFS received two petitions in March 1999 to list Cook Inlet belugas as endangered under the ESA. One petition (brought by Joel Blatchford, a Native Alaskan beluga hunter; the Alaska Center for the Environment, the Alaska Community Action on Toxics, the Alaska Wildlife Alliance, the Center of Biological Diversity, the Center for Marine Conservation, the National Audubon Society, and the Trustees for Alaska) requested an emergency listing under Section 4(b)(7) of the ESA and the designation of critical habitat. Both petitions (the second brought by the Animal Welfare Institute) requested immediate promulgation of regulations to govern the subsistence hunt. NMFS determined that the petitions contained substantial scientific or commercial information indicating that the petitioned actions may be warranted (64 FR 17347, April 9, 1999). To ensure that the Status Review was comprehensive and based on the best available scientific information, NMFS sponsored a workshop on March 8-9, 1999 in Anchorage that reviewed relevant scientific information on this population. At this workshop, NMFS received additional public comments and recommendations. The abstracts of presentations from this workshop (Moore et al. 1999) were subsequently published in a special issue of *Marine Fisheries Review* 62(3).

1.2.3. MMPA Subsistence Hunt Management—1999-2008

In 1999, a temporary legislative moratorium on subsistence hunting Cook Inlet belugas by Native Americans was enacted (Pub. L. No. 106-31, Section 3022, 113 Stat. 57, 100, May 21, 1999). This legislation resulted in no hunt in 1999 and 2000, though hunters voluntarily suspended the hunt in spring 1999. Following the "depleted" determination under the MMPA, NMFS proposed regulations limiting the hunt of belugas in Cook Inlet, Alaska, on October 4, 2000 (65 FR 59164). While these regulations were undergoing public comment, the moratorium was made permanent in December 2000 (Pub. L. No. 106-553). The only exclusion to the moratorium is through a co-management agreement between NMFS and Alaska Native organizations (ANO). NMFS has since promulgated regulations for the taking of Cook Inlet belugas by Alaska Natives for the years 2001-2004 (69 FR 17973, April 6, 2004). A Final Environmental Impact Statement (EIS) was released with the final proposed regulations in July 2003 (68 FR 55604, September 26, 2003). Proposed long-term harvest regulations through a period which should see population recovery are currently under review (71 FR 8268, February 16, 2006) and discussed in this document (see Section 3.5). Preparation of a Supplemental EIS reviewing these long-term harvest regulations was completed in June 2008 (73 FR 35133, June 20, 2008).

1.2.4. NMFS MMPA Depleted Decision and ESA Not Warranted Decision—1999-2000

Following these reviews and taking into account the best information available at that time, NMFS proposed designating the Cook Inlet population of belugas as "depleted" under the MMPA on October 19, 1999 (64 FR 56298) and conducted a public hearing on November 22, 1999. NMFS issued a final rule on May 31, 2000 (65 FR 34590) designating these belugas as Depleted based on its determination that the abundance estimate was below the Optimum Sustainable Population (OSP) level. At the time, the Cook Inlet stock included all belugas in waters of the Gulf of Alaska north of 58º N latitude (including, but not limited to, Cook Inlet, Kamishak Bay, Chinitna Bay, Tuxedni Bay, Prince William Sound, Yakutat Bay, Shelikof Strait, and off Kodiak Island and freshwater tributaries to these waters). On June 22, 2000, NMFS also determined that Cook Inlet belugas were not in danger of extinction nor likely to become so in

the foreseeable future. Therefore, NMFS determined that listing this population under the ESA was not warranted at the time (65 FR 38778). However, NMFS remained concerned about the status of the Cook Inlet beluga population and continued to include the population on the list of Candidate Species under the ESA. During this petition review, NMFS established Cook Inlet belugas as a DPS and therefore, a species as defined under Section 3(15) of the ESA (65 FR 121, June 22, 2000).

1.2.5. Court Challenge to ESA Not Warranted Decision—2001

The decision not to list can be challenged in court under the citizen suit provision of the ESA (16 U.S.C. § 1540(g)). In their suit (Cook Inlet Beluga, et al. v. Daley, No. 00-1017 D.C.), the petitioners argued that NMFS had acted in an "arbitrary and capricious" manner by not listing Cook Inlet belugas under the ESA. On August 20, 2001, U.S. District Court Judge James Robertson ruled that the Agency had acted within the scope of its legal authority, adequately explained its decision, based its decision on facts in the record, and considered the relevant factors and, therefore, upheld the decision not to list.

1.2.6. Species of Concern—2004

On April 15, 2004, NMFS moved Cook Inlet belugas from the Candidate Species list to the newly created Species of Concern list (64 FR 19975). This list is limited to species under NMFS jurisdiction and does not apply to the regulatory practices of the USFWS. NMFS uses the term "Species of Concern" to identify species about which NMFS has some concerns regarding status and threats but for which insufficient information is available to indicate a need to list the species under the ESA. This may include species for which NMFS has determined, following a biological Status Review, that listing under the ESA is "not warranted," pursuant to ESA Section 4(b)(3)(B)(i) but for which significant concerns or uncertainties remain regarding their status and/or threats, as is the case for Cook Inlet belugas. NMFS may conduct ESA Status Reviews on each Species of Concern as agency resources permit.

1.2.7. Conservation Plan—2005

On March 16, 2005, NMFS completed a draft Conservation Plan for Cook Inlet belugas as required under the MMPA. The comment period for the plan closed June 27, 2005 (70 FR 30697). A final version of the plan is currently under review at the NMFS Alaska Regional Office (NMFS 2005).

1.2.8. Status Review—2006

NMFS formally initiated a Status Review on March 29, 2006 (71 FR 14836) to aid NMFS managers in determining if Cook Inlet belugas should be listed under the ESA. This review, published in November 2006 (Hobbs et al. 2006) concluded: 1) The contraction of the range of the population northward into the upper inlet made it far more vulnerable to catastrophic events with the potential to kill a significant fraction of the population; 2) The population was not growing at 2% to 6% per year as had been anticipated since the cessation of unregulated hunting; 3) The population was discrete and unique with respect to the species, and if it should fail to survive, it was highly unlikely that Cook Inlet would be repopulated with belugas; resulting in a permanent loss of a significant portion of the range for the beluga species; 4) The importance of seasonal anadromous fish runs in Cook Inlet to belugas was evident and that the bulk of their annual nutrition was acquired during the summer months; and 5) The PVA model showed a 26% probability of extinction in 100 years and 68% probability of extinction in 300 years (for the model assuming one predation mortality per year and a 5% annual probability of an unusual mortality event killing 20% of the population). Based on the best available science at the time, the Cook Inlet beluga population was likely to continue to decline or go extinct over the next 300 years unless factors determining its growth and survival were altered in its favor.

1.2.9. Proposed Rule to List—2007

On April 20, 2006, NMFS received a petition from Trustees for Alaska to list Cook Inlet belugas as endangered under the ESA. After reviewing the information contained in the petition as well as other scientific information readily available, NMFS determined that the petition presented

substantial scientific information indicating that the petitioned action may be warranted (71 FR 44614, August 7, 2006). Within 12 months of the date of the petition, NMFS was required to make one of the following findings: 1) The petitioned action is not warranted; 2) the petitioned action is warranted, in which case the Secretary shall promptly publish in the *Federal Register* a proposed regulation to implement the action pursuant to 50 CFR 424.16; or 3) the petitioned action is warranted, but A) the immediate proposal and timely promulgation of a regulation to implement the petitioned action is precluded because of other pending proposals to list, delist, or reclassify species, and B) expeditious progress is being made to list, delist, or reclassify qualified species, in which case such findings shall be promptly published in the *Federal Register* (71 FR 44614). Based on the findings from the Status Review and consideration of the factors affecting this species, NMFS concluded Cook Inlet belugas constituted a DPS that was in danger of extinction throughout its range. NMFS issued a proposed rule to list the Cook Inlet beluga DPS as an endangered species on April 20, 2007 (72 FR 19854). Public hearings and public comments on the proposed listing were held through August 3, 2007.

1.2.10. Center for Independent Experts Review—2007

In August 2007, NMFS scientists at the National Marine Mammal Laboratory (NMML) of the Alaska Fisheries Science Center (AFSC) requested an independent review of scientific documents, analysis and the resulting conclusions which supported the proposed listing of Cook Inlet belugas as endangered under the U.S. ESA. This included a review of the background biological data, population data, model structure and assumptions, the analysis methods applied to the extinction risk assessment, and the conclusions resulting from that assessment. The review panel was composed of four appointed reviewers from the Center for Independent Experts (CIE), with one selected as the chair by the CIE. The panel convened at the NMML in Seattle, Washington, from November 13-16, 2007 to review the extinction risk assessment for Cook Inlet belugas. Each reviewer was provided with a set of documents for review in the days prior to meeting in Seattle. The three independent CIE reviewers and CIE chair met during the specified meeting dates to discuss and compile the draft peer-review reports. The authors of the primary review documents were available during the review meeting to address questions from the CIE

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reviewers. NMML received the reviews from the experts and a summary document from the panel chairman on January 10, 2008.

Overall, the CIE review panel agreed that the assessment represented the best available science and that the conclusions were supported by the scientific findings presented in the Status Review. The panel went on to recommend that the following information be included in a subsequent Status Review (see Hobbs et al. 2008):

- A thorough explanation of the abundance survey technique and analyses.
- Using video records to provide information on population structure.
- A discussion of the published vital rates and the key parameters used in the model.
- More information on the progressive reduction of the area in Cook Inlet used by belugas.
- A separate analysis of the survey data since 1999.
- A review of beluga populations once depleted and now recovering.
- The influence of variability in the data series on the estimation of turnover rates in the models.
- A clear statement of the assumption that environmental conditions will remain unchanged.

1.2.11. Status Review—2008

The 2008 Status Review addressed scientific issues raised during the public comment period (that closed on August 3, 2007) and updated the November 2006 Status Review to account for scientific data and other information that has become available in the interim including abundance estimates from 2006 and 2007. The CIE review panel comments on the November 2006 Status Review and updated and auxiliary analysis were addressed in the Status Review published in April 2008 (Hobbs et al. 2008). The area in Cook Inlet used by belugas in early summer decreased significantly since surveys conducted in the 1970s and after the harvest moratorium. With the addition of the 2006 and 2007 abundance estimates, the PVA showed a 39% probability of extinction in 100 years and 79% probability of extinction in 300 years (for the model assuming one predation mortality per year and a 5% annual probability of an unusual mortality event killing 20% of the population).

1.2.12. Extension of Listing Decision—2008

A final determination whether to list Cook Inlet belugas as endangered under the ESA was to occur within one year of the announcement of the proposed rule on April 20, 2007 (72 FR 19854). On April 22, 2008, NMFS extended the deadline for the final listing determination to October 20, 2008 (73 FR 21578). As per the *Federal Register* notice, the State of Alaska questioned the interpretation of the trend analysis noting "that the June 2007 count of belugas was the largest since 2001, indicating, in their estimation, that the population is beginning to recover from the unsustainable harvests of the 1990s." Stating that "substantial disagreement exists regarding the population trend," NMFS allowed a 6-month extension so results from the annual aerial survey to be conducted in June 2008 could be included "to better inform [NMFS's] final decision and potentially resolve the disagreement over the scientific information upon which it will be based." This supplemental Status Review includes the abundance estimate from 2008 and revised trend, range contraction, and extinction risk analyses for the Cook Inlet beluga DPS.

1.3. Key Questions in ESA Evaluations

1.3.1. The 'Species' Question

For the purpose of the ESA, Congress has defined a species as "any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature." As amended in 1978, the ESA allows listing of "distinct population segments" of vertebrates, as well as named species and subspecies. Guidance on what constitutes a DPS is provided by the joint NMFS-USFWS interagency policy on vertebrate populations (61 FR 4722, February 7, 1996). To be considered "distinct," a population, or group of populations, must be "discrete" from other populations and "significant" to the taxon (species or subspecies) to which it belongs. During the 1999 Status Review, it was concluded that Cook Inlet belugas are discrete from other Alaska beluga populations. In particular, all available data, including morphology, core and summer ranges, as well as genetics, indicated that the Cook Inlet

belugas are an independent population that is distinct from other populations (65 FR 121, June 22, 2000). In addition, the loss of the population would result in a significant gap in the range of the taxon. Therefore, Cook Inlet belugas were considered significant with respect to the Alaska taxon and were designated a DPS on June 22, 2000 (65 FR 121). Additional information is presented in Hobbs et al. (2006).

1.3.2. The 'Extinction Risk' Question

The ESA defines the term endangered species as "any species which is in danger of extinction throughout all or a significant portion of its range." The term threatened species is defined as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." The ESA states that a variety of information should be used in evaluating the level of risk faced by a species or a DPS. Important considerations include Section 4(a)(1) of the Act which establishes whether a species is endangered or threatened based on one or more of the following five factors:

A) The present or threatened destruction, modification, or curtailment of its habitat or range;
B) Overutilization for commercial, recreational, scientific, or educational purposes;
C) Disease or predation;
D) The inadequacy of existing regulatory mechanisms; or
E) Other natural or man-made factors affecting its continued existence.

According to the ESA, the determination of whether a species is threatened or endangered should be made on the basis of the best scientific information available on its current status, after taking into consideration conservation measures that are proposed or are in place. This document is a compilation of biological data and a description of past, present, and likely future threats to Cook Inlet belugas. It does not represent a decision by NMFS on whether this taxon should be proposed for listing as threatened or endangered under the ESA. That decision will be made by NMFS after reviewing this document, other relevant biological and threat data not included

herein, and all relevant laws, regulation, and policies. The results of the decision will be announced in the *Federal Register*.

2. UPDATES ON THE BIOLOGY, BEHAVIOR, ECOLOGY, AND POPULATION DYNAMICS OF COOK INLET BELUGAS

This section includes new or revised information that has become available since publication of the 2006 and 2008 Status Reviews. Additional information on each topic is provided in Section 2 of Hobbs et al. (2006, 2008).

2.1. Beluga Biology and Behavior

2.1.1. Identifying Characteristics

Information about identifying characteristics is provided in Hobbs et al. (2006, 2008).

2.1.2. Distribution of Beluga Populations

Information about distribution of beluga populations is provided in Hobbs et al. (2006, 2008).

2.1.3. Prey Preferences and Feeding Behavior

Information about prey preferences and feeding behavior is provided in Hobbs et al. (2006, 2008).

2.2. Ecology of Cook Inlet Belugas

2.2.1. Temporal Changes in Distribution

Temporal changes in the distribution of the Cook Inlet beluga population were first described in Rugh et al. (2001). Additional data were presented in Hobbs et al. (2008) where proximity to Point Woronzof (the western tip of Anchorage) was used to calculate changes in distribution over time. This section presents revised analyses and includes the June 2008 count data.

The analysis uses results from aerial surveys with documented effort that included both the upper (north of East and West Forelands; 60° 45'N) and lower portions of Cook Inlet. The time series for examining inter-annual distributional changes was restricted to surveys that occurred in June or July (when the majority of surveys took place) to minimize intra-annual changes in whale distribution, such as might occur with the presence of sea ice in winter (Rugh et al. 2004). Most reports lacked sufficient descriptions of how and where the surveys occurred, but good documentation is available for aerial surveys conducted on 18 June 1978 and 18-22 June 1979 (Alaska Department of Fish and Game, unpubl. data) and from NMFS surveys starting in 1993 (Rugh et al. 2000; 2005a,b; NMML, unpubl. data). Each of the surveys in the 1970s was a single sample of the study area, but the NMFS surveys covered 4 to 10-day periods each year and included 3-7 repetitions of coastal flights around the upper inlet plus 1-2 days dedicated to a survey of the lower inlet. Results from NMFS surveys were weighted by number of surveys of each region.

Data were parsed into three categories: 1978-1979 (when well-documented data are available), 1993-1997 (during a decline in abundance), and 1998-2008 (when hunting was regulated and, after which, recovery was anticipated). Distributional changes were calculated (using the "Directional Distribution" tool in ArcView) by determining the proximity of belugas relative to a central location computed for all belugas observed within each time period weighted by the number of animals in each group. The distribution of belugas around each central location was calculated at one standard deviation (capturing about 68% of the whales) and two standard deviations (or about 95%). We used Student's t-tests to determine if the area in Cook Inlet

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occupied by belugas changed significantly from the time period 1978-1979 to 1993-1997, 1978-1979 to 1998-2008, and from 1993-1997 to 1998-2008.

Belugas were distributed over a relatively large area in 1978 and 1979, with the central location occurring between the McArthur and Beluga Rivers (Fig. 2.2.1-1). The area of highest concentration included the region from Drift River to the mouth of the Susitna River. From 1993 to 1997, the central location shifted northeastward to the mouth of the Susitna River and the area of highest concentration contracted to north of Moose Point and began to enter Knik Arm (Fig. 2.2.1-2). From 1998 to 2008, the central location shifted east is now occurring between the Little Susitna River and Fire Island (Fig. 2.2.1-3). The area of highest concentration now extends from the mouth of the Susitna River into Knik Arm and toward Turnagain Arm. Changes in distribution over the three time periods were significant: 1978-1979 to 1993-1997 (P = 0.042, northeast shift), 1978-1979 to 1998-2008 (P = 0.022, northeast shift), and 1993-1997 to 1998-2008 (P = 0.025, longitudinal shift eastward). In all time periods, the largest numbers of belugas were in the Susitna delta (from Beluga River to Little Susitna River). However, these numbers also dropped significantly after the population decline (P = 0.0002; 1993-1997 average = 336, 1998-2008 average = 172).

The documented shift in distribution of belugas in Cook Inlet can be explained by several possible hypotheses: 1) habitat change, such as prey availability; 2) some degree of protection from killer whale (*Orcinus orca*) predation; or 3) a range retraction into preferred habitat because the population has declined. Cook Inlet supports numerous anadromous fish runs (Moore et al. 2000); however, belugas appear to be concentrating at the northernmost of these, seemingly ignoring other options, and there is no evidence of a marked decline in southern Cook Inlet runs. Therefore, habitat change does not provide an obvious explanation for the shift in distribution.

Killer whale predation on belugas in Cook Inlet is not uncommon (Shelden et al. 2003, Section 3.2), which leads to the speculation that belugas retreat to the northern reaches of Cook Inlet to avoid killer whales. However, predation events have been documented in the north; therefore, predation pressure also fails to explain the distribution shift.

A retraction to preferred habitat as the population declined is consistent with the history of this population, indeed the range retraction appears exactly as one would expect under the model of an emptying basin (MacCall 1990). The number of belugas in the Susitna delta was significantly more than in any other region of the inlet in each time period. This does not explain why the upper inlet is preferred, but it is intuitive that peripheral ranges would be abandoned when population density diminishes, particularly for a social species like belugas that frequently clusters in relatively few large groups.

Alternatively, perhaps this remnant population is limited to optimal habitat areas where prey concentration in shallow river channels maximizes feeding opportunities (Goetz et al. 2007); this is especially critical in June and July when whales must build up blubber reserves and when calving begins (Hobbs et al. 2005, 2008). The resultant concentration of these whales adjacent to an industrialized area is surprising. Numerous studies of large mammals (reviewed in Frid and Dill 2002) consider the detrimental effects of human-caused disturbance on behavior, reproductive success and parental investment, as well as indirect effects on populations and communities. If and when the Cook Inlet beluga population begins to increase, a reoccupation of peripheral habitats may be the first indication of recovery, possibly before there is a statistically significant indication of an increase in abundance or positive growth trends.
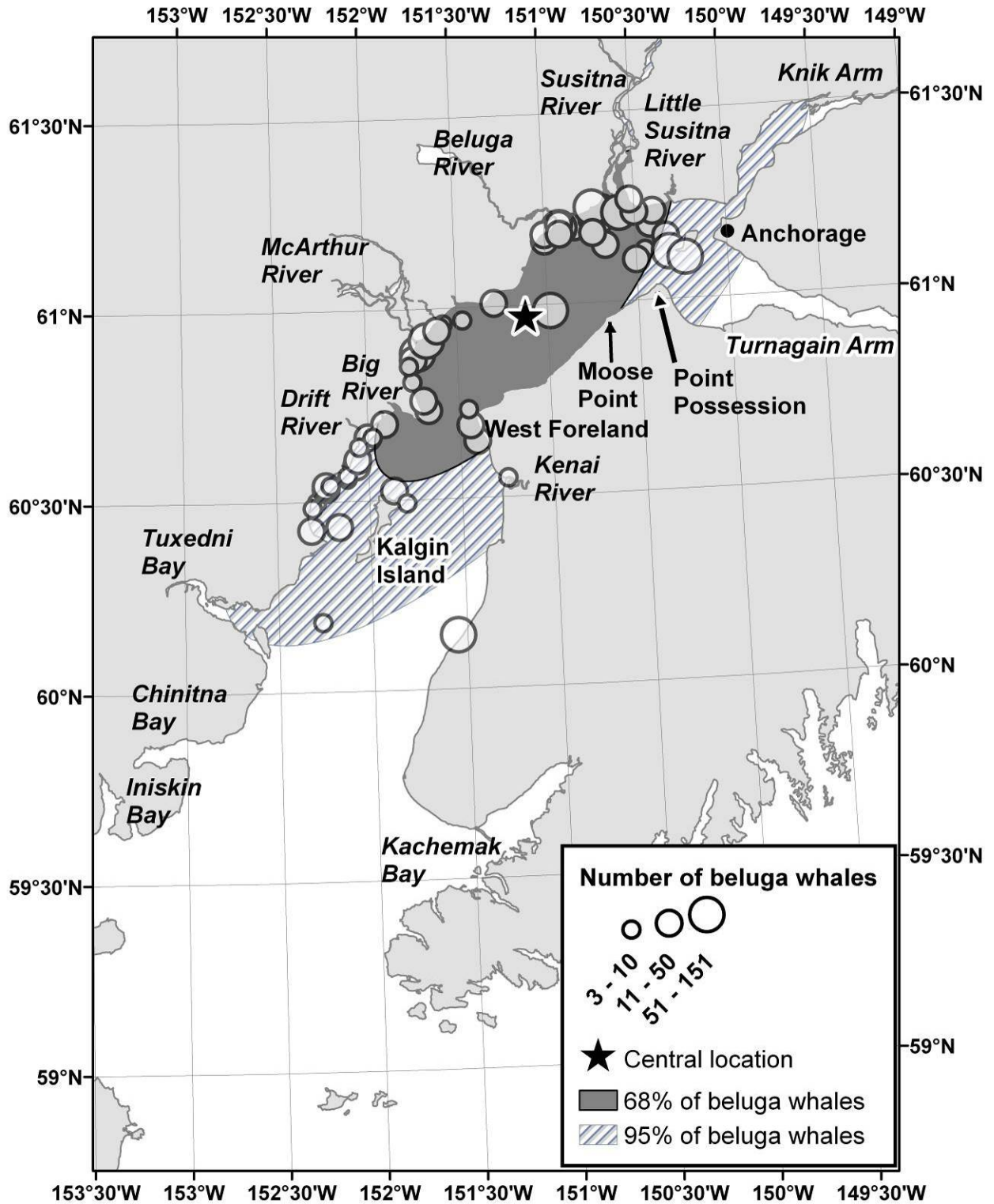
Figure 2.2.1-1. Areas occupied by belugas in Cook Inlet, Alaska, in June/July 1978-1979.
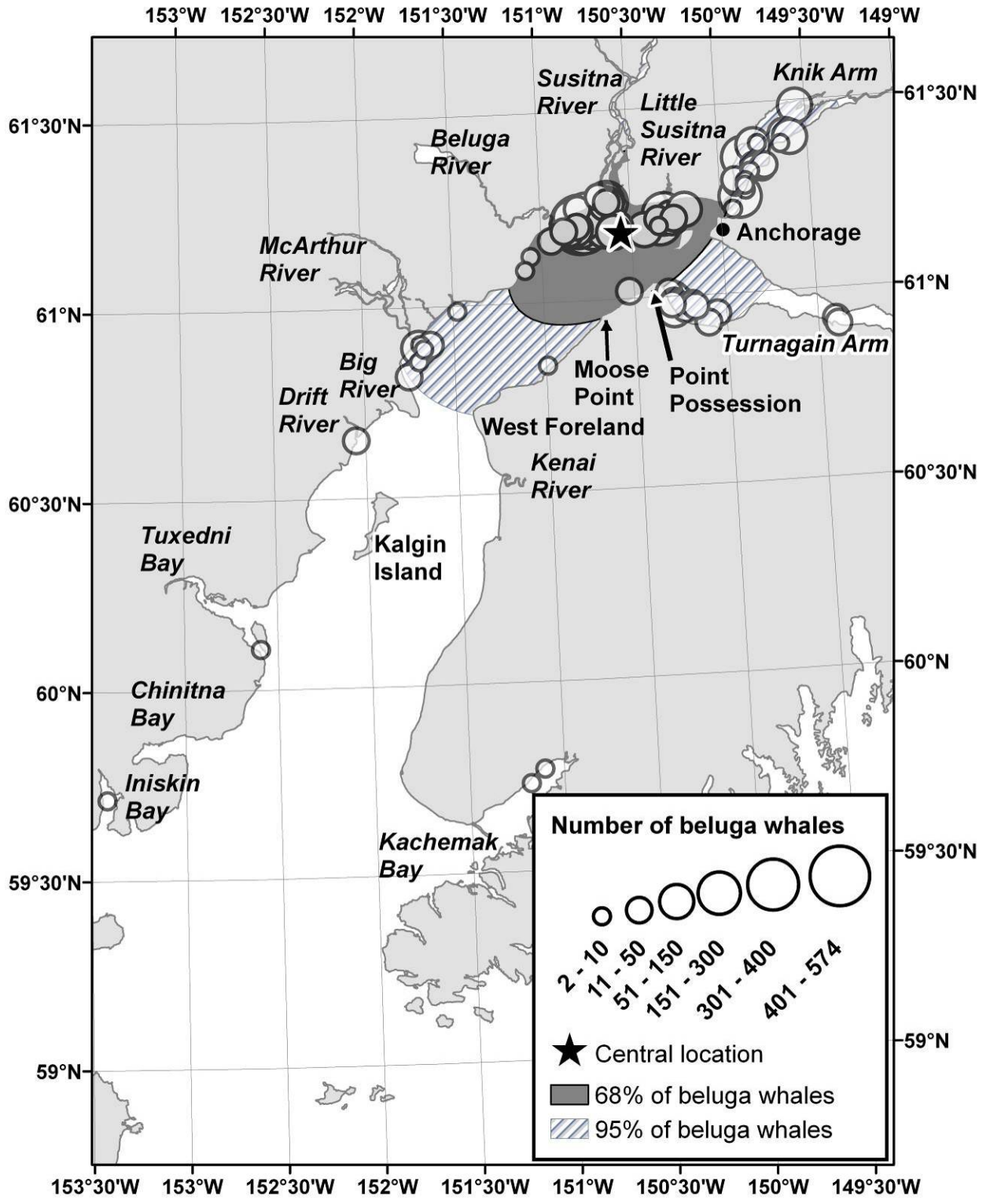
Figure 2.2.1-2. Areas occupied by belugas in Cook Inlet, Alaska, in June/July 1993-1997.
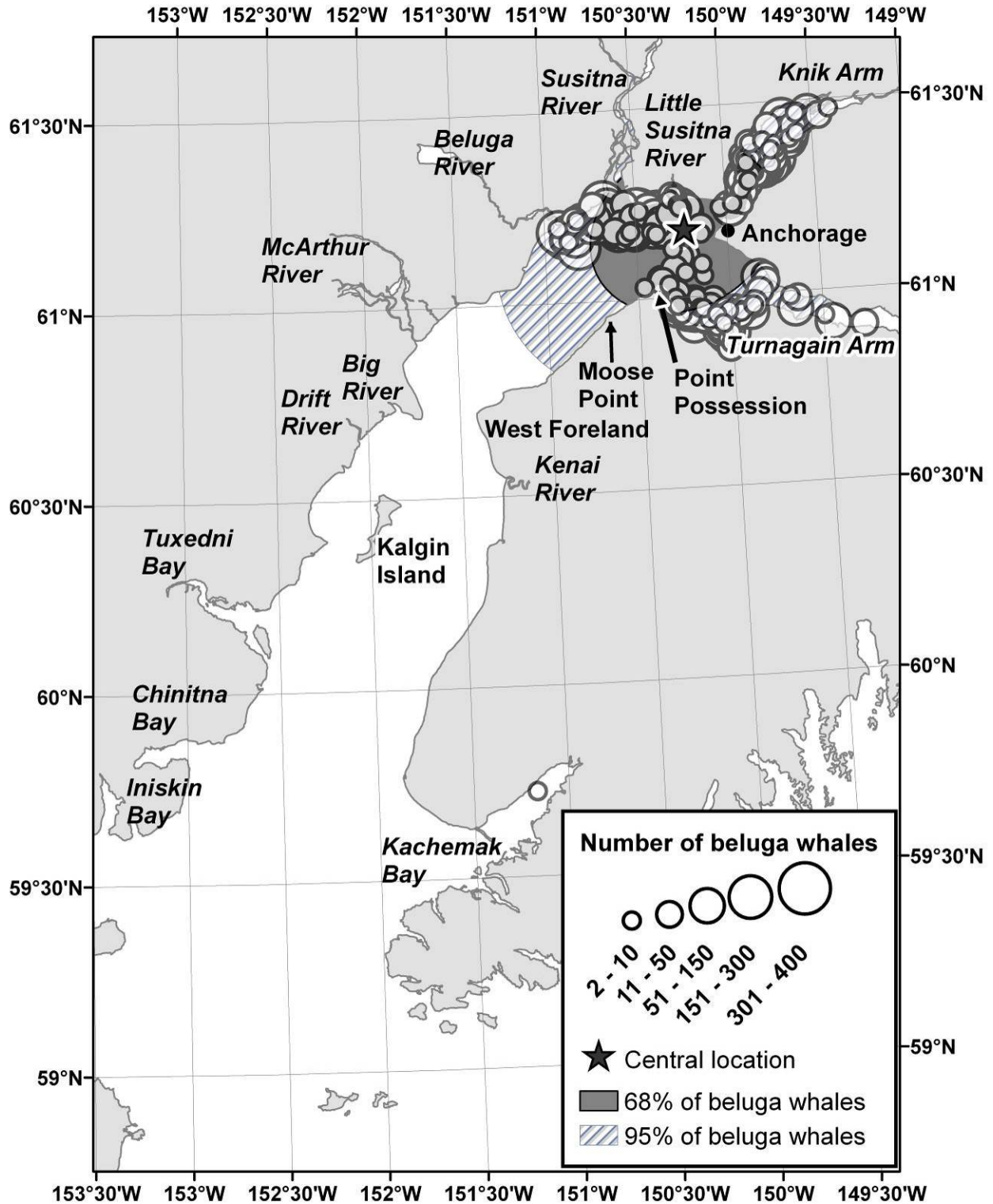
Figure 2.2.1-3. Areas occupied by belugas in Cook Inlet, Alaska, in June 1998-2008.

2.2.2. Habitat Use and Requirements

Information about habitat use is provided in Hobbs et al. (2006).

2.3. Population Dynamics

2.3.1. Population Size

NMFS began comprehensive, systematic aerial surveys of the beluga population in Cook Inlet in 1993. These surveys documented a decline in abundance of nearly 50% between 1994 and 1998 (Fig. 2.3.1-1) from an estimate of 653 (CV = 0.43) whales to 347 (CV = 0.29) whales (Hobbs et al. 2000a). Estimates since 1998 have ranged from 435 (CV = 0.23) to 278 (CV = 0.18) whales (Fig. 2.3.1-1).
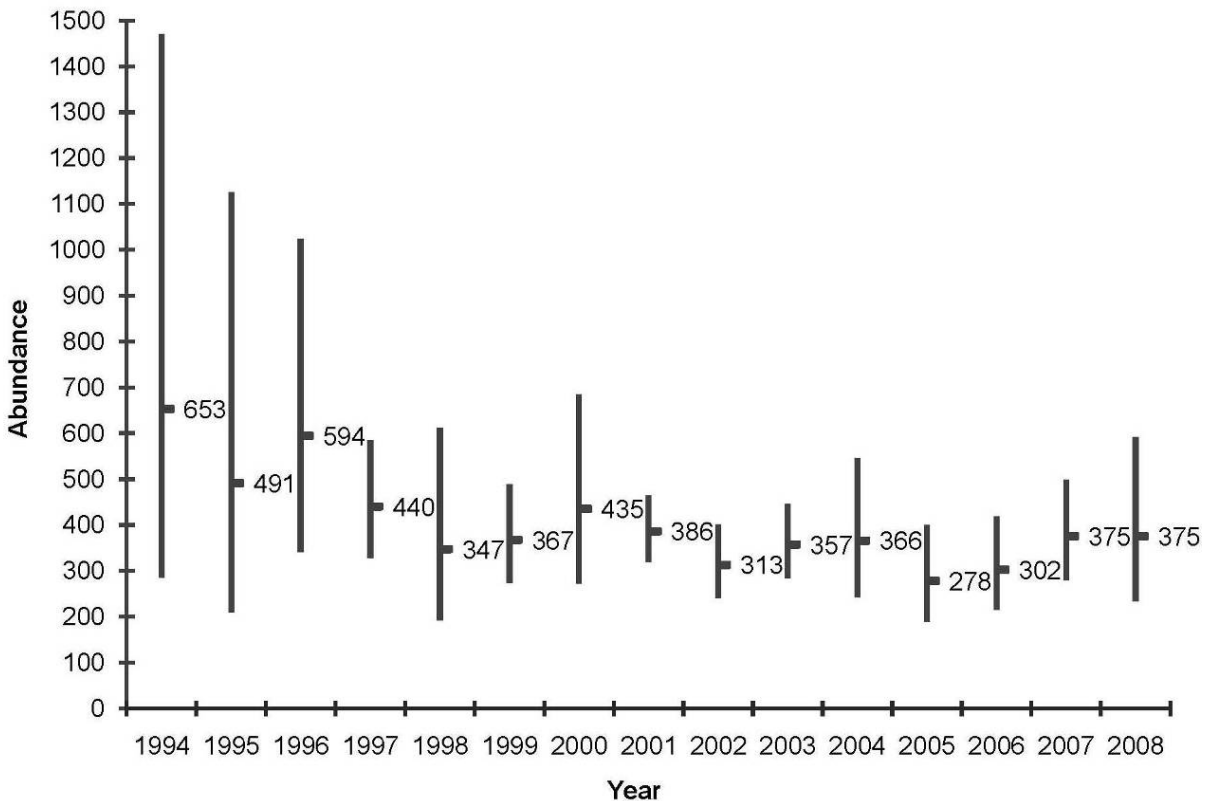
Figure 2.3.1-1. Estimated abundance of Cook Inlet belugas from NMFS annual aerial surveys, 1994-2008, showing average abundance and 95% confidence interval for each year.

2.3.2. Population Trends

Over the period from 1994 to 2008, a fitted growth model shows an average annual rate of decline of -2.91% (SE = 0.010) with an annual rate of decline of -15.1% (SE = 0.047) during the years 1994 -1998, when the harvest was unrestricted. With the very limited hunt between 1999 and 2008, NMFS anticipated that the population would begin to recover at a rate of 2% to 6% per year. When only the 1999-2008 time series of abundance estimates is considered, the rate of decline is -1.45% (SE = 0.014) per year (Fig. 2.3.2-1). While this is not significantly less than a growth rate of 0% per year, it is significantly less than a growth rate of +2% per year (P < 0.02) and, therefore, the population is not recovering at the minimum rate expected.
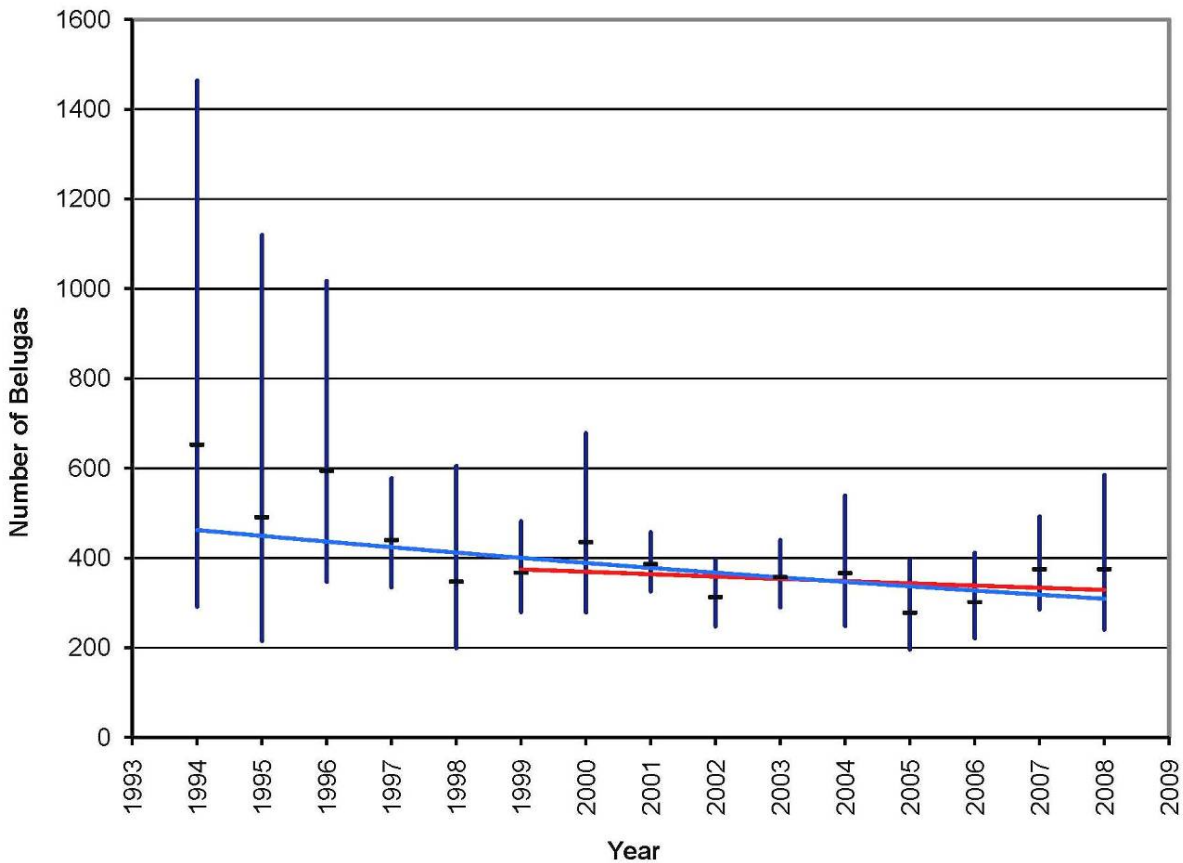
Figure 2.3.2-1. Estimated abundance and 95% confidence interval of Cook Inlet belugas from NMFS annual aerial surveys 1994-2008 with trend line for 1994-2008 and 1999-2008.

To determine which model best fits the available data, we compared the unconstrained 1999-2008 time series to a model that constrained growth to greater than 2% during this same time period using corrected (for small sample sizes) Akaike's Information Criteria (AICc) (Akaike

1974, Burnham and Anderson, 2004). AIC is a tool that ranks competing models such that the model having the lowest AIC is considered the best. Results of this analysis show the unconstrained model with the rate of decline at -1.45% is more likely (Table 2.3.2-1: Model Comparison A).

Table 2.3.2-1. Corrected Akaike's Information Criteria (AICc) results comparing fitted growth models for the 1999-2008 abundance estimates for the Cook Inlet beluga population.

Model Comparison	Growth rate	AICc	Probability
A	Unconstrained, constant	3.69	94%
	Greater than 2%/year	9.36	6%
B	Unconstrained, constant	3.69	73%
	Changes in 2005-2008	5.67	27%

Although the time series from 2005-2008 gives the visual impression of a change in trend (Fig. 2.3.2-1), it should be noted that by selecting the lowest point in a time series, it is likely that the following points will show an upward trend.   For example, the average abundance estimate in the 1999-2008 time series is 355 belugas (SD = 46). The abundance in 2005 (278 whales) is at the 3.6% percentile of this distribution. If the three following abundance estimates are drawn at random from this distribution, there is a 90% probability that all three will be greater than the 2005 abundance. Furthermore, a linear regression fit to these points has an 88% probability of showing a growth rate greater than 2%. If we assume the growth rate did indeed change in 2005 rather than remaining constant, AIC results do not strongly support this assumption (Table 2.3.2-1: Model Comparison B).   Therefore, results from these models indicate that there is a high likelihood that the population is not recovering (Model Comparison A) nor has it had a change in growth rate in the last few years (Model Comparison B).

2.3.3. Life History Parameters

Information about life history parameters can be found in Hobbs et al. (2006, 2008). In response to reviews and inquiries of Hobbs et al. (2008), sampling locations and sizes are now provided in Table 2.3.3-1.

Table 2.3.3-1. Review of female beluga life history parameters found in the published literature.

Parameters	Data		Sources
Age at sexual maturity	9-11 growth layer groups (GLGs) (mean=10, excluded one immature animal age 15 GLGs, sample sizes not provided).		1
	7-13 GLGs (mean=10 GLGs), 5-6 to 11-12 GLGs (mean=9 GLGs, n=33, calculated from data collected by Khuzin (1961) in the Kara and Barents seas, Russia).		2
	0% at 8-9 GLGs, 33% at 10-11 GLGs, 94% at 12-13 GLGs, 100% at 16-17 GLGs (n=207).		3[a]
	9.1 ± 2.8 GLGs (captive beluga studies, n=23).		4
Age at color change (gray to white)	12 GLGs (minimum age)		1
	14 GLGs (minimum from Mackenzie Delta), 17 GLGs (minimums from western Hudson Bay)		2
Age at 1st conception	54% at 8-9 GLGs (n=12 of 22)		3
	41% at 10-11 GLGs (n=9 of 22)		
	4% at 12-13 GLGs (n=1 of 22)		
Age at senescence	42-43 GLGs (arbitrarily assumed by Kleinenberg et al. (1964))		1
Pregnancy and birth rates	with small fetuses: 0.055 at 0-11 GLGs 0.414 at 12-21 GLGs 0.363 at 22-45 GLGs 0.267 at 46-57 GLGs 0.190 at 58-77 GLGs	with full-term fetuses or neonates: 0.000 at 0-11 GLGs 0.326 at 12-21 GLGs 0.333 at 22-45 GLGs 0.278 at 46-51 GLGs 0.182 at 52-57 GLGs 0.125 at 58-77 GLGs	3
Lifespan	60-61 GLGs		1
	50-53 GLGs		2[b]
	>60 GLGs (oldest female estimated at 70+ GLGs)		3
Adult annual survival	0.9064 (average based on mean annual mortality rate = 0.0936)		3
	0.91-0.92		5, 6
	0.842 and 0.905 (assuming 2GLGs/yr vs. 1 GLG/yr)		7
	0.96-0.97		8
	0.935		9
Immature annual survival	0.905 (for neonates in first half year of life, mortality rate=0.095)		2
	0.955 (based on pilot whale net recruitment)		10
Reproductive rate	0.13 (ratio of calves to adult females, modeled)		2
	0.143 (ratio of calves to adult females)		2
	0.114-0.117 (ratio of calves to whales)		2
	0.104 (a model population of 1,000 that included 94 calves)		3
	0.097 (ratio of calves to whales)		6
	0.08-0.10 (ratio of calves to whales)		10
	0.12 (ratio of calves to whales)		11
	0.056-0.10 (ratio of calves to whales)		12
	0.08-0.14 (ratio of calves to whales)		13
	0.08 (unknown)		14
Lactation period	At least 2 years		1
	21 months on average (based on length of gestation (14 months) x 33 lactating/22 pregnant whales)		2
	23 months (range:18-32 months, analysis of data collected by Seaman and Burns (1981))		6
Calving interval	3 years		1, 2[c], 3[d]
	>2 years (based on the assumption that females produce 10 calves within a 14-15 year active breeding period)		6[e]

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1. Brodie (1971) [Canada] Cumberland Sound, Baffin Island, population, n=124 animals (86% captured in nets which biased the sample toward females with newborns), Fig.3 appears to show 51 females in the sample. 2 Sergeant (1973) [Canada] Churchill and Whale Cove in western Hudson Bay, additional information from the Mackenzie Delta, Beaufort Sea and Kara/Barents seas, Russia. 3. Burns and Seaman (1986) [Northwest Alaska]; 4. Robeck et al. (2005) [captive belugas]; 5. Allen and Smith (1978) reviewed in 6. Braham (1984); 7. Ohsumi (1979); 8. Béland et al. (1992) {Canada] St. Lawrence population; 9. Lesage and Kingsley (1998) [Canada] St. Lawrence population; 10. Brodie et al. (1981) [Canada] Cumberland Sound, Baffin Island; 11. Ray et al. (1984); 12. Davis and Finley (1979) [eastern Arctic]; 13. Davis and Evans (1982) [eastern Beaufort Sea and Amundsen Gulf]; 14. Breton-Provencher (1981) [Poste-de-la-Baleine region].

[a] Sampling occurred in June, a time when most Alaskan belugas are born. It is possible non-pregnant 8-9 GLGs belugas would have conceived before their 10-11 GLGs birth date.
[b] Found differences in maximum age based on sampling technique. Life span of netted whales tended to be lower (40 GLGs at Whale Cove) than those selected and harpooned (50 GLGs at Churchill, 53 GLGs at Mackenzie Delta). Similar results were reported by Brodie (1971) for whales netted in Cumberland Sound (40 GLGs).
[c] In 7 of the 29 pregnant females examined from Whale Cove, lactation was still occurring and for some analyses a 2 year calving cycle was assumed for 25% of the adult female population (p. 1084). Sergeant (1973) concluded "overlap of pregnancy and previous lactation is infrequent so that calving occurs about once in 3 years.
[d] For some female belugas. This was a tentative conclusion based on high conception rates noted in some females between the ages of 12-13 GLGs and 44-45 GLGs.
[e] Braham (1984) based this assumption on data from Brodie (1971) and Sergeant (1973) that age at first pregnancy is 6 years (12 GLGs) and last pregnancy is about 21 years (42 GLGs) resulting in a 14-15 year breeding period, which would allow only 6 calves rather than the 10 calves predicted by the authors if a female's reproductive cycle is 3 years. However, this calculation was based on 2 GLGs = 1 year, using 42-12 = a 30-year breeding period and a 3-year reproductive cycle would produce 10 calves.

3. POTENTIAL RISK FACTORS FOR COOK INLET BELUGAS

The following section provides an update to previous Status Reviews (Hobbs et al. 2006, 2008) and discussion about potential factors which are believed to have some impact on the Cook Inlet beluga population and the mitigation measures that are currently in place. At reduced numbers and with contraction of their range, this population is far more vulnerable to losses due to stranding, predation, or disease. This population relies heavily on several fish species that are available only seasonally and are also of considerable commercial interest. Disturbances that cause belugas to temporarily or permanently abandon summer feeding areas could reduce their ability to survive through the winter months. These risk factors are also described within the MMPA Conservation Plan (NMFS 2005).

3.1. Stranding Events

The following table summarizes strandings of belugas documented in Cook Inlet from 1988 to September 30, 2008 (Table 3.1-1). Additional information on stranding events is provided in Hobbs et al. (2006, 2008).

Table 3.1-1. Yearly summary of carcasses and live stranding events of Cook Inlet belugas (Moore et al. 2000, Shelden et al. 2003, Vos and Shelden 2005, NMFS unpublished data).

Year	Carcasses (beach-cast or floating) per year	No. with evidence of killer whale predation[a]	No. of belugas per live stranding event (mortalities associated with live-stranding)	Date of live stranding event	Location
1988	0		27 (0)	Oct. 23	Turnagain Arm
1989	3				
1990	2				
1991	1	1	70-80 (0)	Aug. 31	Turnagain Arm
1992	5	2	2 (2)	Oct. 6	Kenai River
1993	3	1[b]	10+ (0)	July 6	Turnagain Arm
1994	10		186 (0)	June 14	Susitna River
1995	3				
1996	12		63 (0)	June 12	Susitna River
			60 (4)	Aug. 28	Turnagain Arm
			20-30 (1)	Sept. 2	Turnagain Arm
			1 (0)	Sept. 8	Knik Arm
			10-20 (0)	Oct. 2	Turnagain Arm
1997	3				
1998	10		30 (0)	May 14	Turnagain Arm
			5 (0)	Sept. 17	Turnagain Arm
1999	12	5[c]	58 (5)	Aug. 29	Turnagain Arm
			12-13 (0)	Sept. 9	Turnagain Arm
2000	13	2	8 (0)	Aug. 27	Turnagain Arm
			15-20 (0)	Sept. 24	Turnagain Arm
			2 (0)	Oct. 24	Turnagain Arm
2001	10				
2002	13				
2003	20	1	2 (0)	April 18	Turnagain Arm
			46 (5)	Aug. 28	Turnagain Arm
			26 (0)	Sept. 6	Turnagain Arm
			32 (0)	Sept. 14	Turnagain Arm
			9 (0)	Oct. 6	Turnagain Arm
2004	13				
2005	6		7 (1)	Aug. 24	Knik Arm
2006	8		12 (0)	Sept. 12	Knik Arm
2007	15				
2008	11	1	28 (0)	Aug. 7	Knik Arm
			20-40 (0)	Sept. 28	Turnagain Arm

[a] Killer whales chasing and/or consuming belugas were also reported in 1985 (Turnagain Arm), Sept. 1990 (Chickaloon Bay), Sept. 1999 (Chinitna Bay), Oct. 2000 (Kenai River), 2000 (Kachemak Bay), May 2001 (Turnagain Arm), Sept. 2002 (Turnagain Arm) (see Shelden et al. 2003 for details), 14 June 2007 (Anchor Point), and 10 Sept. 2008 (Turnagain Arm) (NMFS unpubl. data). In these instances, any subsequent reports of strandings could not be conclusively linked to the predation event.
[b] Presumed dead: stranded killer whale regurgitated a large chunk of beluga blubber before dying (Shelden et al. 2003).
[c] Deaths were the result of stranding after being chased by killer whales (Shelden et al. 2003).

3.2. Predation

Since publication of Shelden et al. (2003), the following accounts of predation on belugas by killer whales have been reported:

1) The carcass of a male beluga (474 cm (15'7")) examined August 25, 2003) found in Knik Arm had obvious trauma in the form of killer whale tooth marks and internal hemorrhaging (Vos and Shelden 2005);

2) On June 14, 2007, a large pod of killer whales was observed chasing and feeding on a beluga just outside of Kachemak Bay near Anchor Point (video and photographs were obtained). The killer whales swam next to a fishing boat and several times swam under the boat and alongside carrying the beluga in their mouths. Mel Erickson, the Captain of the fishing vessel, reported "When the killer whales killed the beluga there was blood and guts all over the surface" (NMFS unpubl. data);

3) On September 10, 2008, an adult beluga was killed by two killer whales (what appeared to be a mother/calf pair) in Turnagain Arm, midway between Beluga Point and Hope. The adult killer whale was observed to grab the beluga in her mouth and drag it around. Blood sprayed into the air from the injured beluga.   The killer whales remained with the floating beluga, which appeared to be dead. On 19 September, a dead beluga was reported on the mudflats near Indian Creek. NMFS personnel performed a full necropsy. The whale was on its right side. Fluke tips had been chewed off and the peduncle was covered with killer whale teeth marks. Blubber had been removed from most of the body from the chin to the genital slit and the left pectoral fin was completely gone. The base of the skull appeared to be crushed and killer whale teeth marks were also evident near the throat.

Additional information on predation is provided in Hobbs et al. (2006, 2008).

3.3. Parasitism and Disease

Information on parasitism and disease is provided in Hobbs et al. (2006, 2008).

3.4. Ice Entrapment

Information on ice entrapment is provided in Hobbs et al. (2006).

3.5. Small Population Effects

Information on small population effects is provided in Hobbs et al. (2006). Also see revised extinction risk models in Section 5.

3.6. Fishery Interactions

Information on fishery interactions is provided in Hobbs et al. (2006).

3.7. Anthropogenic Sound

Information on anthropogenic sound is provided in Hobbs et al. (2006).

3.8. Pollution

Information on pollution effects is provided in Hobbs et al. (2006, 2008).

3.9. Ship Strikes

Information on ship strikes is provided in Hobbs et al. (2006).

3.10. Subsistence Hunting

Since publication of the 2006 Status Review (Hobbs et al. 2006), NMFS entered into only one co-management agreement to hunt one beluga in 2006, but the hunt was not successful. No co-management agreement and, therefore, no hunt occurred in 2007 and 2008.

3.11. Research

Information on research is provided in Hobbs et al. (2006).

3.12. Summary of Potential Risk Factors

The potential risk factors and their possible effects on Cook Inlet belugas at the individual and population level are summarized in Table 3.12-1 in Hobbs et al. (2008).

In order to begin to determine factors that may affect the recovery of Cook Inlet belugas, a population comparison study was initiated in May 2008 focusing on Bristol Bay belugas. The Bristol Bay population is found during the summer months about 1,500 km away by sea and separated from Cook Inlet by the Alaska Peninsula that extends 3 degrees of latitude south of the southern limit of the Bristol Bay beluga population. This population, in comparison to Cook Inlet, is increasing at about 4.5% annually and numbers about 2,000 animals (L. Lowry, University of Alaska Fairbanks, pers. comm.). Points of comparison will include: seasonal movements and dive behavior, habitat use, relative dependence on summer salmon feeding for total annual calories, annual caloric requirements, and types and prevalence of disease and parasites. In May 2008, 10 belugas were captured and had satellite transmitters attached and blood, skin, gastric, fecal and blubber samples collected. Each whale was scanned with an ultrasound to determine blubber thickness and density. In September 2008, eight more whales were tagged and underwent similar testing. Results from these samples will be used to determine fitness and health status as a baseline for similar tests and samples from belugas in Cook Inlet. Additional years of research will be added as funding allows.

4. DETERMINATION OF DPS

4.1. ESA Discreteness and Significance

Joint NOAA/USFWS policy defines a population to be a DPS if it is both discrete and significant relative to the taxon to which it belongs (61 FR 4722, February 7, 1996). Under the policy, a population may be considered discrete if it satisfies one of the following conditions:

- It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors. Quantitative measures of genetic or morphological discontinuity may provide evidence of this separation.
- It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of Section 4(a)(1)(D) of the ESA.

Data relevant to the distinctiveness question include the physical, ecological, behavioral, and genetic data that are presented in Section 2 and summarized below. If a population segment is considered discrete, NMFS must then consider whether the discrete segment is "significant" to the taxon to which it belongs. A discrete population segment needs to satisfy only one of the following criteria to be considered significant:

- persistence of the discrete segment in an ecological setting unusual or unique for the taxon,
- evidence that loss of the discrete segment would result in a significant gap in the range of the taxon,
- evidence that the discrete segment represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historical range, or,
- evidence that the discrete segment differs markedly from other populations of the species in its genetic characteristics.

The policy also allows for consideration of other factors if they are appropriate to the biology or ecology of the species. Data relevant to the significance question include the morphological, ecological, behavioral, and genetic data presented in Section 2 and summarized below.

4.2. Evaluation of ESA Discreteness

The evaluation has not changed since Hobbs et al. (2006). It is not clear from available data whether the group of belugas found in Yakutat Bay is isolated from Cook Inlet.

4.3. Evaluation of ESA Significance

The evaluation has not changed since Hobbs et al. (2006).

5. ASSESSMENT OF EXTINCTION RISK

5.1. Population Viability Analysis

A detailed population viability analysis (PVA) model was developed for the Cook Inlet beluga population in Hobbs et al. (2006) to assess the extinction risks faced by this small population. The age- and sex-structured model included immature and mature phases of both sexes and focused on behavior of a declining population at sizes less than 500 belugas. Small population effects were taken into account by examining survival and fecundity under a range of scenarios that considered demographic stochasticity, harvest, density-dependent and density-independent effects, Allee effects (Allee et al. 1949), constant mortality effects (e.g., predation), and unusual mortality events (e.g., catastrophes). In response to the CIE review, several models were added in the 2008 status review (Hobbs et al. 2008) to further test the sensitivity of the PVA: 1) environmental variability was added to the range of scenarios, 2) a likelihood model using only 1999 to 2007 abundance estimates, and 3) a likelihood model assuming over half of the small, gray animals under 10 years of age are missed during the aerial surveys. In this document the sensitivity analysis has been expanded further to include: 1) several additional models using the

1999-2008 likelihood, 2) the relationship between the delayed response of the population and the fraction of females and the total harvest, and 3) the relationship between the magnitude of the unusual mortality events and the probability of extinction and other forms of unusual mortality. It is not the intent of this exercise to attempt to identify a particular mechanism that affects the population trajectory but instead to try a variety of plausible mechanisms that fit the existing data to explore the range of likely outcomes.

5.1.1. Methods

To foresee the growth or decline of this beluga population in the future, a PVA model was developed using life history and population parameters estimated for this and other beluga populations (see Table 2.3.3-1 in Section 2.3.3). In addition to the selection of parameters, mechanisms affecting small populations (as described above) and time lags inherent in long-lived populations (which can result in a delayed response to changes in mortality probabilities) were also considered (c.f. Litzky 2001). To account for the time lag from birth to sexual maturity and the preference of hunters for adult animals, an age-structured model was used with adult age classes lumped together. Females and males were also modeled separately to incorporate sex-structure into the model and allow for unequal harvest of males and females.

Demographic stochasticity, the random variations in the number of individuals that happen to die or reproduce in a given year (Begon et al. 1996:927), was included in the model projection from one year to the next. To this extent, survival from year to year and births each year were modeled using a binomial draw. To model the harvest and the stochastic effects of injuring or killing a whale but not retrieving it (struck and lost), we used a binomial draw from the remaining population with the reported landings, the number of adults less the number landed and the struck and loss probability determining the probability of mortality. The resulting stochastic age- and sex-structured model was used to model the current population and project the possible outcomes.

The model was fit to the available abundance estimates for the years 1994 to 2008 (Table 5.1.1-1) using Bayesian statistical methods. The abundance of the Cook Inlet beluga population and

subsistence harvest removals from this population were estimated for each year between 1994 and 2008 (Table 5.1.1-1, Fig. 2.3.1-1). Limited information is available to determine the behavior of this population during periods prior to 1994, including the original or pristine size of this population and its sustainable harvest level. A value of 1,300 belugas was used in the model as pristine population size or carrying capacity (K) (see Calkins 1989) sensitivity trials with other values are included. Finally, although K is included as a parameter, the purpose of this exercise was to model the behavior of the population at sizes below 500 animals. By relaxing the constraint of no population growth or decline at K, a greater variety of possible trajectories were available to the model within the range of interest.

Table 5.1.1-1. Time series used in the Bayesian analysis. Median aerial counts are the median of all observer counts from complete surveys of upper Cook Inlet. Estimated abundance was calculated from observer and video data. Harvest landings and struck and lost data were from Mahoney and Shelden (2000) and NMFS Alaska Region Office, unpublished data. Where conflicting sources occur, all are listed with the numbers used in the model in bold. Note that killed but lost are included with the struck and lost.

Year	Median aerial count	Estimated abundance	Abundance CV	Harvest landings (struck and lost)
1994	281	653	0.430	**19(2)**
1995	324	491	0.440	60(14), **52(22)**, 42(26)
1996	307	594	0.280	**49(49-98)**
1997	264	440	0.140	35(30-40), **35(35)**
1998	193	347	0.290	**21(21)**
1999	217	367	0.140	**0(0)**
2000	184	435	0.230	**0(0)**
2001	211	386	0.087	**1(0)**
2002	192	313	0.120	**1(0)**
2003	174	357	0.110	**0(0)**
2004	187	366	0.200	**0(0)**
2005	187	278	0.180	**2(0)**
2006	150	302	0.160	**0(0)**
2007	224	375	0.140	**0(0)**
2008	126	375	0.230	**0(0)**

It is important to note that the abundance data used in the likelihood function for fitting the model resulted from aerial surveys conducted each June from 1994 to 2008 that used essentially the same methods through the entire time series (Hobbs et al. 2000a, b). During a 2-week period in early June, three to seven surveys of the upper inlet and one survey of the lower inlet were

conducted. During each survey the entire coastline to approximately 1 km off shore and all river mouths are surveyed. Transects across the inlet are flown as well (Rugh et al. 2000; 2005a, b). When a group of whales is encountered it is circled in a racetrack pattern 4 to 16 times to allow multiple counts by researchers and the collection of video data.

Two video cameras are used: one to collect a view of the entire group for counting and a second to collect a zoomed in view of a portion of the group to estimate the fraction of missed animals. The video data are the primary source of group size estimates. Useable video sequences are reviewed frame by frame and all individuals are counted. The zoomed video is also reviewed frame by frame and individuals in each zoomed frame are accounted for in each frame of the counting video. Those not found in the counting video are included in the fraction missed. The video counts are also corrected to account for animals that were under water during the video sequence using dive data from radio tags (Lerczak et al. 2000). For groups with no usable video, a correction for the researcher counts is developed by comparing researcher counts of groups to video group size estimates (Hobbs et al. 2000a). Group size estimates for each survey of the upper inlet are summed to get several independent estimates of abundance. Another correction includes data from paired, independent observers to estimate the fraction of whales missed when an observer does not see a group.

The above methods account for belugas missed within the area surveyed. Whale groups can also be missed because they have moved out of the survey area or they were in a portion of the survey area that could not be surveyed on that day. It is not possible to correct for a large group that was missed in this way with the current survey design, instead survey days are reviewed for completeness and either dropped or retained for the abundance estimate. For survey days with unusually low estimates (less than approximately 60% of the highest estimate) the flight paths are reviewed to determine if a group seen on other survey days could have been missed either because the area was unavailable due to weather or air traffic or the group could have moved to an adjacent area that was not surveyed. These survey days were not included in the abundance estimate. While this is not an ideal approach, it reduces the possibility of biasing the abundance estimate downward by not including survey days where a large group was likely to have been missed.

With the distinctive geography of Cook Inlet and the affinity of the belugas for specific locations, a randomized survey design is not feasible. However, as recommended by one reviewer, a randomized survey of the areas adjacent to the primary habitat areas could be used to estimate a correction. It is possible that a positive bias could occur because a survey was dropped that just happened to have an unusually low estimate, though it is also possible that a negative bias could occur because a group was missed on all survey days that year. While each of these biases may affect a particular year, when the time series as a whole is considered, they have limited impact.

The remaining survey days are averaged to complete the abundance estimate. While the survey methodology has remained the same, the video cameras have not, and over the period from 1995 to 2008 several upgrades have occurred. There is concern that the fraction of small, gray animals missed may have declined through the time series (this issue was raised during the CIE review), therefore, this is now tested in the model analysis.

Life history parameters of particular interest for modeling purposes were: survival probability, birth interval comprised of gestation period and lactation period, and age at first birth. With the exception of survival probability, life history sample sizes from Cook Inlet were not sufficient to estimate the other model parameters. These data were instead obtained from the available literature on several other beluga populations (see Table 2.3.3-1 in Section 2.3.3). Upper and lower bounds for the model parameters are described below.

Survival data for Cook Inlet belugas consist of annual summaries of beach-cast and floating carcasses reported to the NMFS Alaska Regional Office and consequently represents a minimum estimate of mortality for this population. From 1999 to 2005, years in which a limited harvest occurred (Table 5.1.1-1), an average of 12 mortalities were reported each year (Vos and Shelden 2005) during a time when the population size averaged around 350 animals. This provided an estimated annual survival probability of 0.97/year which was used as the upper bound for the model. From the literature, survival probabilities have been estimated at or above 0.90/year (Table 2.3.3-1). One estimate of 0.842/year was based on an assumption of two growth layer

groups per year (GLGs/yr) which underestimates the ages of whales by half, and an estimate of survival rate assuming one GLG/yr is 0.905. For modeling purposes, 0.87/year was used as the lower bound for the annual survival probability to ensure an uninformed lower bound for the prior distribution of this parameter.

The annual probability of giving birth is derived from the data on pregnancy rates and birth interval. The one study reported full-term fetuses or neonates found with approximately one-third of females between 12 and 45 GLGs of age. The birth interval, roughly the inverse of the annual probability of giving birth for the average mature female in most beluga populations was thought to be 3 to 4 years (Table 2.3.3-1) resulting in an annual probability of giving birth (between 0.25 and 0.33) for each adult female. Birth intervals as short as 2 years have been reported, but are considered atypical. A 3-year interval is consistent with a 14-month gestation period and 22-month lactation period. The ratio of calves to adult females was as low as 0.13 (Table 2.3.3-1). Ratios of calves to whales were as low as 0.056 which, if 30% to 40% of whales are adult females, corresponds to a minimum ratio of 0.14. To keep the model simple, an average value was used for all adult females including senescent females. A reduced birth probability in the Cook Inlet population resulting from external effects such as pollution or poor fish runs suggests a lower value than 0.13 is possible so the interval 0.10 to 0.33 was used in the model. Note that this is higher than the minimum value (0.05) used in Hobbs et al. (2006, 2008).

Female belugas in several wild populations reach sexual maturity at an average age of 10 years with a range of 8 to 13 years for populations in northwestern Alaska (Table 2.3.3-1). The gestation period lasts about 14 months, so average age at first birth would be 10 years and range from 9 to 16 years if each female becomes pregnant during the first year of maturity. However, no females with full term fetuses or neonates were younger than 12 years in the same northwestern Alaska sample (Table 2.3.3-1), suggesting that newly mature females may require a year or two before beginning a successful pregnancy. We used 9 years in this model, noting that the probability of giving birth distributes the age at first birth over the birth interval so that if the probability of giving birth is 0.20, then the age at first birth would be spread over the ages 9 to 13. In Hobbs et al. (2008) this parameter was set to 10 years, we have changed it to 9 years to more closely match results for wild populations in Alaska. In Hobbs et al. (2006), age at first

birth had been set to 5 years based on life history studies of several beluga populations (Table 2 in Hobbs et al. (2006)). The ages in these studies were determined by counting GLGs in tooth sections. Previously, the accepted practice interpreted two GLGs as indicating 1 year of age so 8-9 GLGs were interpreted as 4 years old (see Section 2.1.1. in Hobbs et al. (2008)). Recent research has shown that each GLG should be interpreted as one year resulting in a doubling of the age at first birth when the life history data are reinterpreted in light of this new information. The age of first birth is of importance for the model and is referred to hereafter as the age of maturity or age at first birth. The lactation period typically lasts longer than one year so calf survival was modeled as dependent on the survival of the mother, including the mother's risk of mortality in the harvest, during the first year after birth. Survival probabilities and age at maturity also have been estimated for males. However, these estimates were not sufficiently different from those for females to require additional parameters in the model.

At about the time a beluga reaches maturity, its skin changes from gray to white (Burns and Seaman 1986). Hunters have stated that they focus their hunting effort on white adult animals so vulnerability to harvest was set in the model to coincide with the age at first birth. While not all animals are mature before they turn completely white this was considered a reasonable approximation to simplify the model.

The population was projected as:

$$f_{0,t} = B\left(f_{mat,t}, \frac{b_t}{2}\right) \qquad\qquad m_{0,t} = B\left(f_{mat,t}, \frac{b_t}{2}\right)$$

$$f_{1,t+1} = B\left(f_{0,t}, s_t^2 \frac{f_{mat,t} - H_{f,t}}{f_{mat,t}}\right) \qquad m_{1,t+1} = B\left(m_{0,t}, s_t^2 \frac{f_{mat,t} - H_{f,t}}{f_{mat,t}}\right) \qquad (1)$$

$$f_{a+1,t+1} = B(f_{a,t}, s_t) \qquad\qquad m_{a+1,t+1} = B(m_{a,t}, s_t)$$

$$\text{for } a = 1 \text{ to } (a_{mat}\text{-}2) \qquad\qquad \text{for } a = 1 \text{ to } (a_{mat}\text{-}2)$$

$$f_{mat,t+1} = B(f_{mat,t} - H_{f,t} + f_{a_{mat}-1,t}, s_t) \qquad m_{mat,t+1} = B(m_{mat,t} - H_{m,t} + m_{a_{mat}-1,t}, s_t),$$

where,

$f_{a,t}, m_{a,t}$ is the number of females and males, respectively, of age a at the beginning of year t;

$f_{mat,t}, m_{mat,t}$ is the number of mature females and mature males, respectively, at the beginning of year t;

$B(x, p)$ is a binomial random variable with x trials and p probability of success;

s_t is the probability of an individual in year t surviving to year $t+1$;

b_t is the probability of a mature female giving birth to a live offspring in year t;

a_{mat} is the age of maturity or the age at which a female could first give birth; and

$H_{f,t}$, $H_{m,t}$ is harvest mortality (both landings and struck and lost) of females and males, respectively, in year t.

Harvest mortality was modeled as the sum of the landed whales plus estimates for those struck and lost. During the years 1995-1998 (Table 5.1.1-1), landings were fairly well documented and struck and lost was estimated at between one-half and two whales lost for each whale landed. For the model, this uncertainty in the level of struck and lost for the years 1979 to 1998 was accounted for by drawing from a binomial draw from the adult population after the harvest was removed with the probability as the struck and loss rate multiplied by the ratio of the landed harvest to the number of adults remaining after subtracting the harvest. Note that this is a change

from the model in Hobbs et al. (2006, 2008) where a negative binomial draw was used, which occasionally resulted in a harvest mortality larger than the existing population which was both not possible and caused the computer program to stop running. The struck and loss rate for each realization of the model was drawn from a uniform distribution between one-half and two (U[1/2, 2]). For the years 1999 and later, where harvests have been regulated, the number of struck and lost was set to zero.

While harvest is not well documented and likely underestimated before 1994, averaging the harvest reported in Mahoney and Shelden (2000) for the years 1987-1990, 1992, and 1993 (no data are available for 1991) equals 10 belugas/year. The largest harvest in these years occurred in 1993, with an estimate of 30 belugas based on a retrospective analysis that attempted to estimate the full number of landed whales using a variety of sources (Hill and DeMaster 1998, Mahoney and Shelden 2000). Using these two values as upper and lower bounds, we have assumed that constant landings occurred from 1979 through 1993 with landings for these years drawn from a uniform distribution between 10 and 30 belugas (U[10, 30]). For the years 1994 to 2008, actual landings (Table 5.1.1-1) were used in the model. For the purposes of the model, no harvest occurred after 2008. Data on the sex of whales killed in the hunt are sparse. From 1992 to 1998, 19 male and 15 female belugas were documented during the harvests (Mahoney and Shelden 2000) corresponding with approximately 55% probability that an animal landed in the harvest was male. Variability in this probability was accounted for by drawing a value for each model realization from a triangular distribution between 0.40 and 0.70 with the peak at 0.55 (TR[0.40,0.55,0.70]). This approximated the beta distribution of relative probabilities for this parameter, without the tails.

The harvest mortality model is

$$H_t = CIBL_t + B\ (f_{mat,t} + m_{mat,t} - CIBL_t,\ \ SLR\ (CIBL_t\ )/(\ f_{mat,t} + m_{mat,t} - CIBL_t))$$
$$H_{m,t} = B(H_t, \mathbf{Pr}(Harvest\ Male)) \tag{2}$$
$$H_{f,t} = H_t - H_{m,t}\ ,$$

where,

H_t is total harvest mortality (both landings and struck and lost) in year t;

$CIBL_t$ is the recorded harvest landings for 1994 to 2008 and a constant harvest landing per year for 1979 through 1993 in year t;

SLR is the struck and lost rate or in other words the ratio of animals killed in the hunt but not recovered, to the number landed, drawn from U[1/2,2] for years 1979 to 1998, for 1999 and after this is always 0, so all animals struck are landed and, therefore, struck and lost is zero; and

Pr(*Harvest Male*) is the probability that an animal taken in the harvest is a male, drawn from TR[0.40,0.55,0.70].

To allow for density dependence in the annual growth multiplier (ϕ) (discussed in greater detail below), both survival (s) and fecundity (b) in Equation 1 were made density-dependent with the following equation:

$$s_t = \left[s_0 - (s_0 - s_k)\left(\frac{N_t}{K}\right)^z \right](s_e) \tag{3}$$

$$b_t = \left[b_0 - (b_0 - b_k)\left(\frac{N_t}{K}\right)^z \right](b_e) \; ,$$

where,

s_0, s_k, b_0, b_k are the values for s and b when the size of the population is close to 0 and at K, respectively;

s_e, b_e are multipliers for s and b that reduce survival or fecundity independent of density;

$N_t = \sum\limits_{all\ ages} f_{a,t} + m_{a,t}$ is the size of the population at time t;

K = the carrying capacity (1,300); and

z = a shape parameter (2.39).

The annual survival probability and annual fecundity probability consisted of three components: a compensatory density-dependent survival or fecundity (Equation 3: within the square brackets); a density-independent component (s_e, b_e); and a modifier such as Allee effects, unusual mortality events, variable environment, and constant mortality effects which will be discussed later. The density-dependent component used the discrete logistic formulation to decrease the probability of survival and probability of giving birth as the population increases. Parameters were chosen so that the annual growth multiplier (ϕ) of 1.02 to 1.06 (i.e., annual per capita increase between 2% and 6%; cf. Wade and Angliss 1997) fell between these values when the population was small and declined to 1.00 (zero growth) when the population reached carrying capacity. The density-independent components (s_e, b_e) can be set to 1 to model a healthy population with annual growth between 2% and 6% or they can be set to values less than 1 to model processes that decrease survival or fecundity for each individual such as contaminants or ship strikes.

Choosing efficient and still uninformative prior distributions for the parameters of the annual fecundity probability and annual survival probability requires a bit of tuning to the model in question and the parameter constraints. If we treat fecundity (b) and survival (s) as constant parameters and consider the deterministic projection of the expected values of the abundance with harvest at zero, we have a recursion model in expected births by year. The characteristic equation for this recursion model can be written as:

$$1 = \frac{b s^{a_{mat}+1} \phi^{-a_{mat}}}{2(1-s\phi^{-1})} \qquad ,$$
(4)

which then yields an equation for b if s and ϕ are known:

$$b = \frac{2(1-s\phi^{-1})}{s^{a_{mat}+1} \phi^{-a_{mat}}} \qquad ,$$
(5)

where,

ϕ is the annual growth multiplier for an expected stable age distribution.

The density-dependent components (Equation 3: square brackets) represent the basic model for a healthy cetacean population with an annual growth multiplier of 1.02 to 1.06. To create a uniform prior distribution for the annual growth multiplier, ϕ_0 was drawn at random from U[1.02, 1.06]. At K, ϕ_K is, of course, 1.00, indicating the population is no longer growing upon reaching carrying capacity. The upper bounds for s_0 and b_0 were 0.97 and 0.33, respectively, and both s_K and b_K were nonnegative. Values for s_0 were then drawn from U[0.87, 0.97] and s_K from U[$s_0 - 0.9(\phi_0 - 1)$, s_0] which allow anywhere from none to all of the density dependence to effect survival while avoiding a significant range of useless parameter space. Equation 5 was then solved for b_0 and b_K. If b_0 and b_K fell in the intervals [0.10, 0.33] and [0.0, b_0], respectively, then the parameter set was retained; otherwise it was discarded and new values for s_0 and s_K were drawn. This approach allowed the density dependence to entirely affect survival or fecundity or any ratio of the two while maintaining a uniform prior for ϕ_0.

The annual growth multiplier for a healthy cetacean population described in the previous paragraph requires that s_e and b_e are set to 1. To allow a full range of annual growth multipliers, to model populations that may be in decline, we included cases where s_e and b_e were less than 1 which model impacts that are independent of density (e.g., reduced survival or fecundity resulting from a pervasive pollutant). To create a uniform prior distribution for the annual growth multiplier, ϕ_0 was drawn at random from U[0.94, 1.06], where the annual per capita change ranged from -6% to +6%. In these cases, ϕ_K is not necessarily 1.00 and instead was chosen from U[$\phi_0 - 0.06$, minimum($\phi_0 - 0.02$, 1.00)] so that the annual per capita change would be 0% or < 0% when the population was at K. Density-independent components, s_e and b_e, were multiplied through the density-dependent portion of Equation 3 (square brackets) to form composite parameters: $s_0 s_e$, $s_K s_e$, $b_0 b_e$ and $b_K b_e$. As in the Healthy Population model (where s_0 and b_0 were bounded by 0.97 and 0.33, respectively, and both s_K and b_K were non-

negative), the upper bounds for the composite parameters $s_0 s_e$ and $b_0 b_e$ were also set to 0.97 and 0.33, respectively, and both $s_K s_e$ and $b_K b_e$ were non-negative. Values for $s_0 s_e$ were then drawn from U[0.87, 0.97] and $s_K s_e$ from U[$s_0 s_e$ -0.90(ϕ_0 - ϕ_K), $s_0 s_e$] which allowed all of the density dependence to affect survival. Equation 5 was then solved for $b_0 b_e$ and $b_K b_e$. If $b_0 b_e$ and $b_K b_e$ fell in the intervals [0.10, 0.33] and [0.0, $b_0 b_e$], respectively, then $s_0 s_e$ and $s_K s_e$ were retained, otherwise they were discarded and a new set was drawn. Note that parameters must be drawn as composites in this model, which we will refer to as the Baseline model, to maintain a uniform and uninformative prior distribution. If the parameters are drawn individually and then multiplied together the resulting prior distribution would be peaked and highly informative.

Modifiers to survival and fecundity were intended to model specific processes. These processes included a constant mortality effect and a stochastic or unusual mortality event to modify survival, and an Allee effect to modify fecundity. Environmental variability was included in both survival and fecundity as a correlated normal random deviate. These were included in the model by rewriting Equation 3 as:

$$
s_t = \left[s_0 - (s_0 - s_k)\left[\left(\frac{N_t}{K}\right)^z + \varepsilon_t \right] \right] s_e \left(\frac{N_t}{N_t + C}\right)\{1 - M_e B(1, P_{Me})\}, \tag{6}
$$

$$
b_t = \left[b_0 - (b_0 - b_k)\left[\left(\frac{N_t}{K}\right)^z + \varepsilon_t \right] \right] b_e \left(\frac{f_{mat,t}}{f_{mat,t} + A}\right)\left(\frac{m_{mat,t}}{m_{mat,t} + A}\right),
$$

$$
\varepsilon_t = \rho \varepsilon_{t-1} + \sigma \sqrt{1 - \rho^2} z_t
$$

where,
C is the parameter of the constant mortality effect and represents expected annual mortalities;
M_e is the individual probability of mortality during an unusual mortality event;
P_{Me} is the probability of an unusual mortality event occurring in a given year;
A is the Allee effect parameter,

ε_t is a stationary, correlated, random environmental deviation with mean = 0, variance = σ^2 and correlation = ρ (Morris and Doak 2002:139).

z_t is a normal random deviate with mean = 0 and variance = 1.

Note that these processes were formulated so that if any of these parameters were zero then the corresponding effect does not modify survival or fecundity. The constant mortality effect was intended to model mortality resulting from annual killer whale predation in which the killer whales were thought to take a number of belugas proportional to their own needs regardless of the size of the beluga population. It could also model illegal harvest if that harvest remained constant from year to year regardless of the population size. The values for C represent the average mortalities per year due to killer whale predation or some other constant mortality. Shelden et al. (2003) estimate an average of one observed predation mortality per year. This was considered a minimum since unobserved predation events may also be occurring. Values considered for the parameter (C) were 0, 1, 2, and 5, with no mortalities per year occurring when the constant mortality effect was absent and five mortalities per year included as a reasonable maximum case.

The unusual mortality event (P_{Me} within the curly brackets of Equation 6) models random events such as mass stranding mortality, oil or toxic waste spills, tsunamis, volcanic activity, failure of salmon runs, etc. In this formulation, it included a mortality fraction and a binomial draw which determined whether or not an event occurred that year. The expected mortality from this source was the product of the mortality fraction and the probability of occurrence so that for the values used here ($M_e = 0.20$ and $P_{Me} = 0.05$), the expected or average annual unusual mortality event when it was included was 0.01 (i.e., an increase of average annual mortality of 1% of the population).

The Allee effect (Allee et al. 1949) is thought to occur in small populations where small numbers of adult females and adult males results in reduced mating opportunities or reduced variety of mate selection with consequent declining fecundity. Although other mechanisms affecting both fecundity and survival have been included under the definition of the Allee effect (Courchamp et al. 1999, Stephens and Sutherland 1999), for the purposes of this modeling exercise, its effect

was applied to fecundity only (*A* in Equation 6). Note that in the formulation above the birth probability is zero when either sex is not present. The Allee parameter was set to 0.5 or 0.0 depending on presence or absence of the effect, respectively. There is little information on which to base a choice of this parameter instead it was tuned to affect the population when there were fewer than 50 whales in total as a proxy for a variety of small population effects.

To date, no environmental time series and mechanism has been identified as impacting survival or fecundity of the Cook Inlet beluga population, so environmental variation is included as random variation in both the probability of birth and survival in proportion to the effect of density- dependence. A new environmental time series is drawn for each run of the model and results from this analysis will indicate the response of the model to autocorrelated variation in fecundity and survival rather than a specific environmental time series. The environmental variation is in the form of a stationary, correlated, normal random deviation with mean = 0, variance = σ^2 and correlation = ρ (Morris and Doak 2002:139). When this feature is included in the model, the value of σ is set at 0.2 of the growth rate range of the density dependence, between 2% and 6%. Consequently 95% of the variation will fall within ± 0.8% to ± 2.4% per capita annual growth for density-dependent ranges from 2% to 6%. However, because these are applied to survival and birth probabilities, they remain subject to the biological constraints of survival in the interval zero to one and birth rate in the interval zero to 0.333. A measure of the effect of the correlation is the distribution of runs of positive or negative variation (i.e., good conditions leading to a positive bias in growth or poor conditions leading to a negative bias in growth). The correlation is set at 0.8 which gives a median run of 9 years, with 25% of runs at 15 years or longer. While this choice is arbitrary, it typically provides one change or no change of environment during the time series of abundance estimates in most cases such that each run of the model will experience good conditions or poor conditions on average during this period. Consequently, the resulting average environment would be either worse or better than the environment during the years that the model is fit to data resulting in an altered expectation in the long term.

The remaining parameter in the age-structured model was the age at first birth (a_{mat}) which was set to start at 9 years. Note that because birth is a discrete event, either a female gives birth to a

calf or not, the value of *b* determined the distribution of ages of first giving birth for the model population. For instance, if *b* was 0.25 then 25% of females first gave birth at age 9, 25% at age 10, etc., resulting in an age at first birth distributed from ages 9 to 12. By the same reasoning if *b* was 0.14 then age at first birth was distributed from ages 9 to 15, or if *b* was 0.10 then age at first birth was distributed from ages 9 to 18.

To set up the initial age structure and a nearly uniform prior for the population abundance in 1994 (N_{1994}), N_{1994} was drawn from a uniform distribution ranging from 450 to 950 belugas (U[450,950]), a constant harvest level H_{79-93} was chosen from U[10,30] and when environmental variation is included, an environmental time series is drawn. The population starts in 1979 (N_{1979}) and is projected forward to 1994 to set up the age structure. The required value for N_{1979} is found using a bisection method between the extremes of 450 and 2,950 which covers the 95% confidence interval around the abundance estimate in 1979 of 1,300 belugas if it is given the same CV as the 1994 abundance estimate. The bisection is conducted by choosing extreme values for the 1979 abundance with the low end of the abundance, $NL_{1979} = 400$ and the high end of the abundance, $NH_{1979} = 3,000$ and a trial value, NT_{1979}, half way between 400 and 3,000. A stable age distribution was set up using s_{NT1979} (the survival rate calculated based on NT_{1979}) and setting ϕ to 1.00. Age and sex classes were filled as a multinomial distribution of NT_{1979} by density at age for each sex. The population was then projected from 1979 to 1994. The population size in 1994, NT_{1994}, is then compared to N_{1994} to determine if the value for NT_{1979} is a useable starting point. If NT_{1994}, is within $N_{1994} \pm 25$ it was considered sufficiently close and the simulation was continued from that point—the stochastic nature of the model prevented an exact match. If $NT_{1994} > N_{1994} + 25$, then the value of NL_{1979} is set to NT_{1994} - 50, if $NT_{1994} < N_{1994} - 25$, then the value of NH_{1979} is set to NT_{1994} + 50, a new NT_{1979} is calculated and the process is repeated until a useable value of NT_{1994} is found or 12 trial values of NT_{1979} are discarded and a new value for N_{1994} is drawn. Projecting the model through 15 years prior to 1994 allowed the juvenile ages to be filled with values derived from the population model and the adult segment to be subject to the pre-1994 harvest level while maintaining nearly uniform and independent prior distributions for N_{1994} and ϕ_0. Each population was then projected from 1994 to 2008 and likelihood was calculated as:

$$L_j = \prod_{t=1994}^{2008} T\left(\frac{N_{t,j} - \overline{N}_t}{\overline{N}_t CV(\overline{N}_t)}, DF = 10 \right) \qquad , \qquad (7a)$$

where,

L_j is the relative likelihood of the *jth* population projection;

$T(X, DF = 10)$ is the density of Student's-t distribution at *X* with 10 degrees of freedom;

$N_{t,j}$ is the population size of the *jth* projection in year *t*; and

$\overline{N}_t, CV(\overline{N}_t)$ are the estimated abundance (point estimate) and associated coefficient of variation in year *t*.

In the case where small or gray animals are less likely to be counted either by observers or in the video analysis,

$$L_j = \prod_{t=1994}^{2008} T\left(\frac{S_{t,j} - \overline{N}_t}{\overline{N}_t CV(\overline{N}_t)}, DF = 10 \right) \qquad , \qquad (7b)$$

where,

$S_{t,j} = \sum_{all\ ages} w_a \left(f_{a,t,j} + m_{a,t,j} \right)$ is the observed size of the population excluding the missed animals

at time *t*;

$w_a = \begin{cases} a/10 & for\ a = 0\ to\ 9, \\ 1 & for\ a \geq 10. \end{cases}$ are weights such that animals in older age classes are more likely to

be seen and counted, with all adults seen.

The Student's-t distribution was chosen for the likelihood model as the best fit compared to the gamma distribution, log-normal distribution and normal distribution to bootstrap results from annual abundance estimates for this population (R. Hobbs, NMML-AFSC, unpublished data). Projections to 2008 with likelihoods less than $10^{-10} \times$ the maximum possible likelihood (i.e., the likelihood if the model N_t was equal to the abundance point estimate in all years) were discarded

as having no contribution to the posterior distribution. A Sampling-Importance-Resampling (SIR) algorithm was followed (Rubin 1988) in which the acceptable parameter sets were weighted by their relative likelihoods from projections to 2008, and a resample drawn with replacement to give a posterior distribution of outcomes. Projections to 2308 (300 years into the future) for this posterior parameter set were done to estimate the probability of decline and extinction during that period. Model comparisons between the various models were done using the Bayes factor (Kass and Raftery 1995, Wade 2002), calculated as twice the natural logarithm of the ratio of the average likelihoods of the two resamples:

$$BayesFactor(x, y) = 2\ln\left[\frac{\sum_{j=1}^{SIRtot} L_{j,x} \Big/ SIRtot}{\sum_{j=1}^{SIRtot} L_{j,y} \Big/ SIRtot}\right] \quad , \tag{8}$$

where,

$BayesFactor(x,y)$ is the Bayes factor comparing model x and model y;

$\ln[]$ is the natural logarithm of the value in $[]$;

$L_{j,x}$ and $L_{j,y}$ are the likelihoods of the jth projection of model x and model y, respectively; and

$SIRtot$ is the number of projections in the SIR subsample.

Where the Bayes factor had absolute value greater than 2 the model with the higher average likelihood was considered to be the more likely of the two, otherwise the models were of equivalent likelihood.

All models were compared to the Baseline (model a) (U[0.94, 1.06]). The probability of the Healthy Population (model b) (U[1.02, 1.06]) was compared using the Bayes factor. The three options for modifying the Baseline model, the constant mortality effect (C), the unusual mortality event (P_{Me}) and the Allee effect (A) were each considered. The time series of abundance and harvest data covered a sufficient range of population sizes (270-660 belugas) to compare between the Baseline and the Healthy Population models but not among the remaining

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options. Six models with the modifiers for survival and fecundity were considered, three of the Baseline with the C parameter at 1, 2 or 5 mortalities per year (models d-f), one of the Baseline with the Allee parameter at 0.50 (model c), one of the Baseline with an unusual mortality event ($P_{Me} = 0.05$) (model g), and one of the Baseline with the C parameter at 1 and an unusual mortality event ($P_{Me} = 0.05$) (model h). Two additional models were included to test the sensitivity of the parameters: a Baseline (model i) and Healthy Population (model j) that included an unusual mortality event and a C of five mortalities. Several additional cases were included as variations of the model itself which are provided as tests of the underlying assumptions of the model: The Baseline model and the Baseline with the C parameter at 1 and an unusual mortality event ($P_{Me} = 0.05$) with missed small and gray animals (models k, l); missed small and gray animals but with the fraction missed declining to zero by 2004 (models m, n); environmental variation (models o, p); and with survival of immature animals set to 90% of the adult survival (models q, r); a set that compared the Baseline (model s) to the Healthy Population (model t) with only the years 1999-2008 included in the likelihood. A series of parameter sensitivity trials are also included: the Baseline model with K set to 650 (model u), 1,000 (model v) and 2,000 (model w); variations of the unusual mortality event models with the probability reduced to 3% ($P_{Me} = 0.03$) with $C = 1$ (model x) or $C = 2$ (model y), and ($P_{Me} = 0.05$) with $C = 1$ but with total mortality capped at 100 animals (model z); a run with the harvest prior to 1994, H_{79-93}, drawn from U[20,50], the range of the harvests between 1994 and 1998 (model aa); a run with the fraction of females in the harvest set to 60% in the Baseline model (model bb); and variation in the age at first birth at age 8 (model cc) and 10 (model dd) in the Baseline model and at age 5 (model ee) to duplicate the model in Hobbs et al. (2006).

For each model, 100,000 trials were projected to 2008 and the likelihood was calculated. Each population projection was fully defined by the 11 parameters: s_0, s_K, b_0, b_K ,(or $s_0 s_e$, $s_K s_e$, $b_0 b_e$, $b_K b_e$), N_{1989}, H_{79-93}, C, M_e, P_{Me}, A, and a_{mat}, though, the stochastic nature of the projection meant two projections with identical parameters would have different outcomes. A sample of 10,000 of these trials, weighted by the likelihoods, was drawn with replacement for the SIR algorithm resample for further analysis. For all populations the population size in 1994, 2008, 2108, 2208, and 2308 was retained. A population with 1 or 0 individuals or only one sex was considered extinct.

5.1.2. PVA Results

The first eight models allowed a range of possible behaviors for the theoretical populations as they became small while behaving similarly within the range of actual abundance estimates (278-653). Examples of the deterministic annual growth multipliers associated with each model are given in Figure 5.1.2-1. For these examples, Equation 4 was solved iteratively for ϕ using values for *s* and *b* calculated at population sizes varying from 1 to 500 (Equation 6). In all of the solid line examples, density-dependent survival parameters were chosen so that at a population size of 350, the annual growth multiplier was 1.01. These "tuned" survival parameters were then used throughout the range. For the dashed line example the survival parameters were tuned such that $\phi = 1.00$ at a population size of 350. The strong density-dependence example set $\phi_0 - \phi_K = 0.06$ while the weak density-dependence example set $\phi_0 - \phi_K = 0.02$, all other examples used strong density-dependence. Where growth increased as population declined crossing the value 1.00 (the dashed example), a stable equilibrium point was formed and, without stochastic variation, the population settled at this size (350 belugas). Where growth decreased as population declined crossing the value 1.00, an unstable equilibrium point resulted forming a population size threshold below which, without stochastic variation, the population continued to decline, and above which, without stochastic variation, the population increased. Consequently, if stochastic variation in the form of demographic stochasticity or unusual mortality events pushed the population below the population size threshold, the population would likely continue to decline to extinction. Note that in these examples, the annual growth multiplier fell below 1.00 for the Allee effect at a population size of about 15 belugas. For varying levels of *C*, the population size thresholds occurred around 60 belugas for 1 mortality per year, 120 for 2 mortalities, and 200 for 5 mortalities. This demonstrates the possibility of thresholds at different population sizes depending on the parameters used in each model.
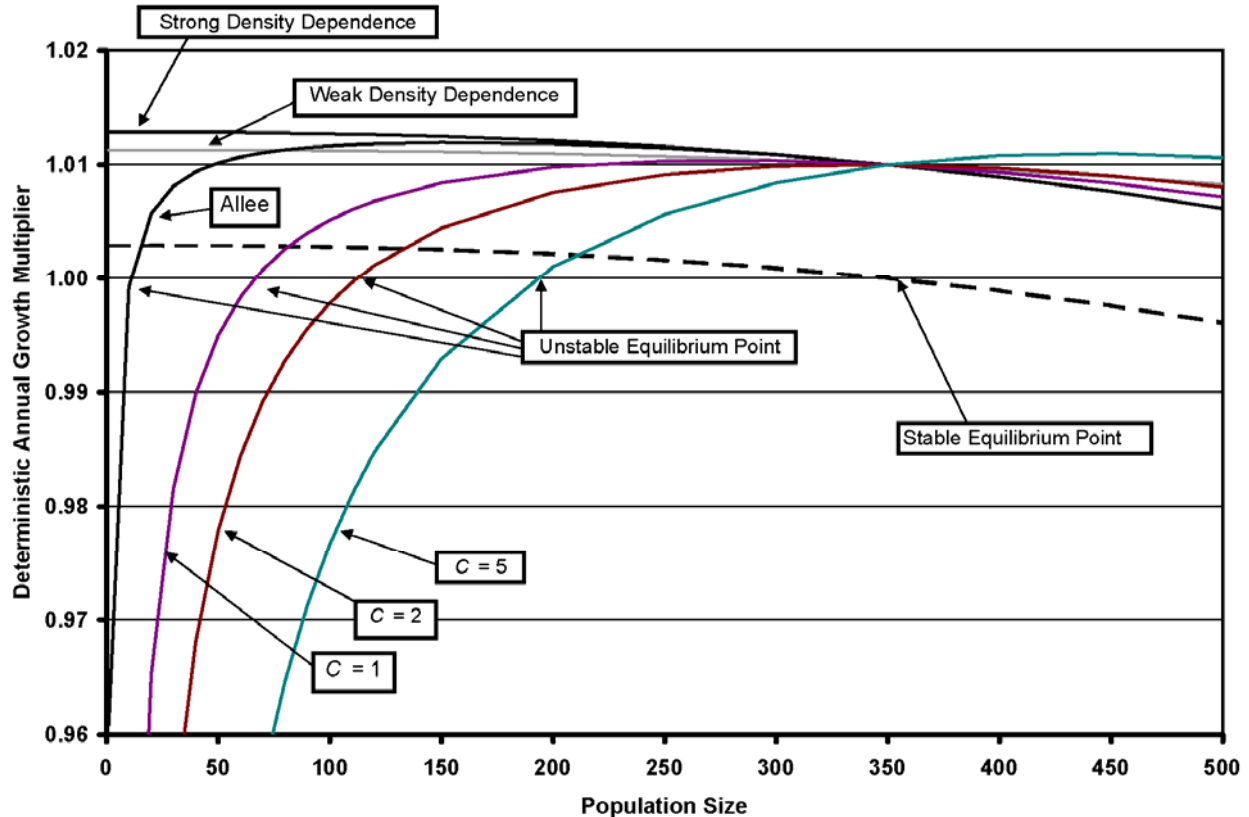
Figure 5.1.2-1. Solutions for the annual growth multiplier (ϕ) to the characteristic equation (Equation 4 in text) by population size for examples of the models. Solid line examples include density-dependent survival parameters for a population of 350 belugas chosen to set $\phi(350) = 1.01$. The dashed line example was tuned to $\phi(350) = 1.00$. Strong density dependence was set at $\phi_0 - \phi_K = 0.06$; weak density-dependence at $\phi_0 - \phi_K = 0.02$; all other examples used strong density dependence. C is the annual constant mortality effect parameter.

These growth multipliers and resulting behavior of the populations are reflected in the abundance time series for these example populations (Fig. 5.1.2-2a-h). Note that although the projections match the abundance time series closely during the period from 1994 to 2008, after 2008 there was considerable variation in behavior. Although there was no harvest in these models after 2008, the examples with $C > 0$ could be considered examples of the effect of a constant harvest level. Considering the Baseline model (Fig. 5.1.2-2a), which allowed a growth multiplier between 0.94 and 1.06, we had three typical behaviors: a slow decline to extinction, an increase to K, and an approach to stable equilibria between extinction and K. When only a limited range of values for the growth multiplier (between 1.02 and 1.06) was considered, as in the Healthy

Population (Fig. 5.1.2-2b) only one behavior resulted, an increase until the population leveled off near K.

Inclusion of an effect that created a threshold, such as predation (C) or Allee (Fig. 5.1.2-2c-f), added an additional behavior of a rapid decline to extinction, with the steepness determined by the height of the threshold effect. This threshold was particularly pronounced when the predation parameter was two or greater (Fig. 5.1.2-2e-f) causing a visible break point below which there was little probability of avoiding extinction. Including an unusual mortality event (Fig. 5.1.2-2g-h) had the effect of raising the population size threshold because populations above but near the threshold were still at risk of falling below after an unusual mortality event, and once below the threshold the population most likely would continue to decline. Because unusual mortality events periodically reduced the population, this prevented these populations from settling near an equilibrium. Extreme values of the C parameter alone (Fig. 5.1.2-2f) and mixed with unusual mortality events provided for sensitivity analysis. In these examples a population size threshold occurred within the range of recent abundance estimates (278-653), and in the Healthy Population model an unusual mortality event combined with $C = 5$ resulted in a significant number of extinctions, populations that would have recovered without these effects.

a. Baseline Model ϕ_0 **drawn from U(0.94-1.06)**	
b. Healthy Population Model ϕ_0 **drawn from U(1.02-1.06)**	
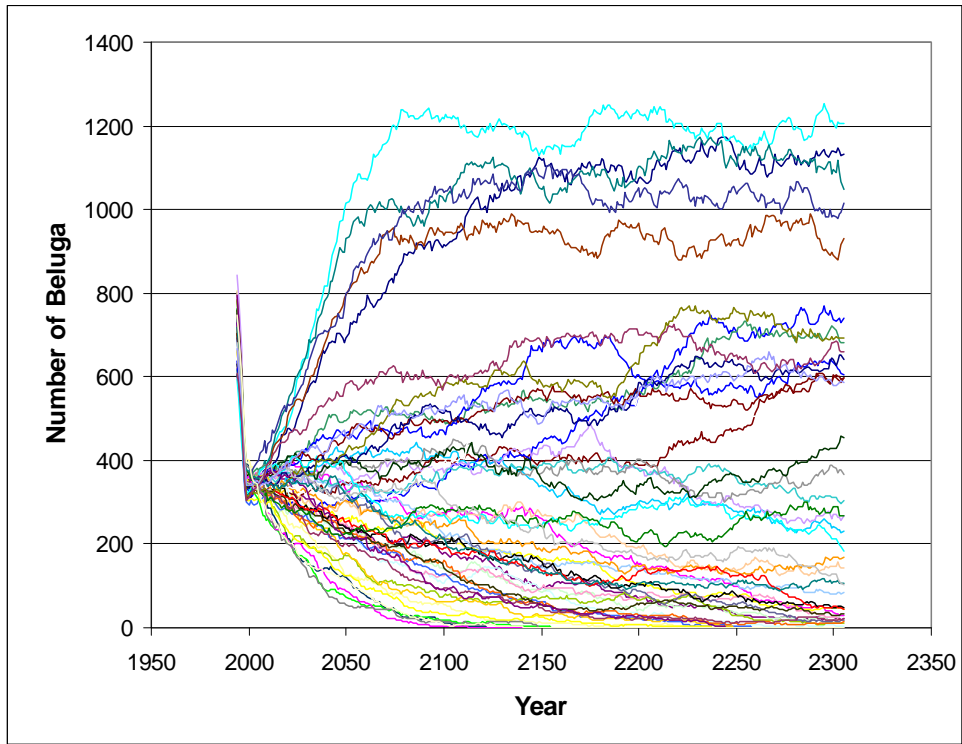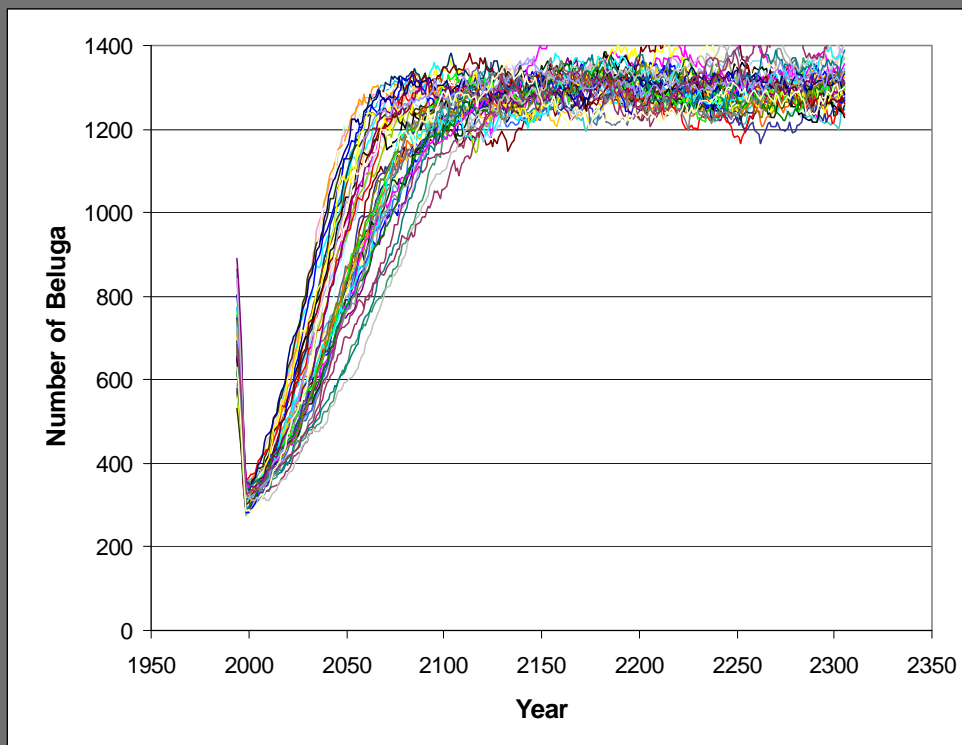
c. Baseline Model + Allee effect

d. Baseline Model + C = 1

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e. Baseline Model $+ C = 2$	 Number of Beluga vs. Year
f. Baseline Model $+ C = 5$	 Number of Beluga vs. Year
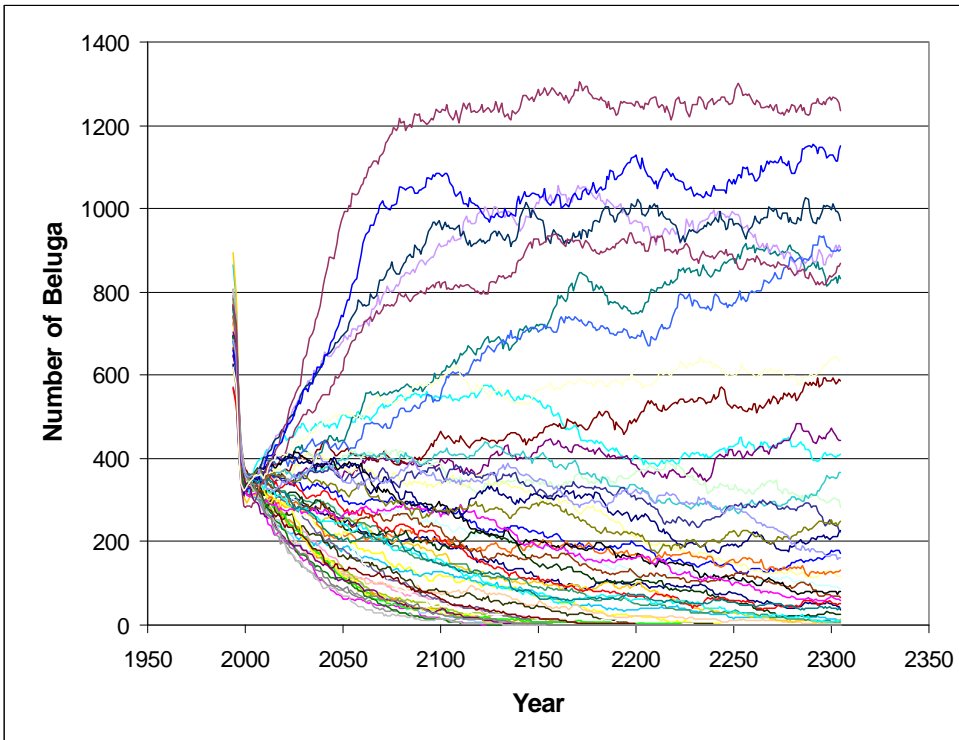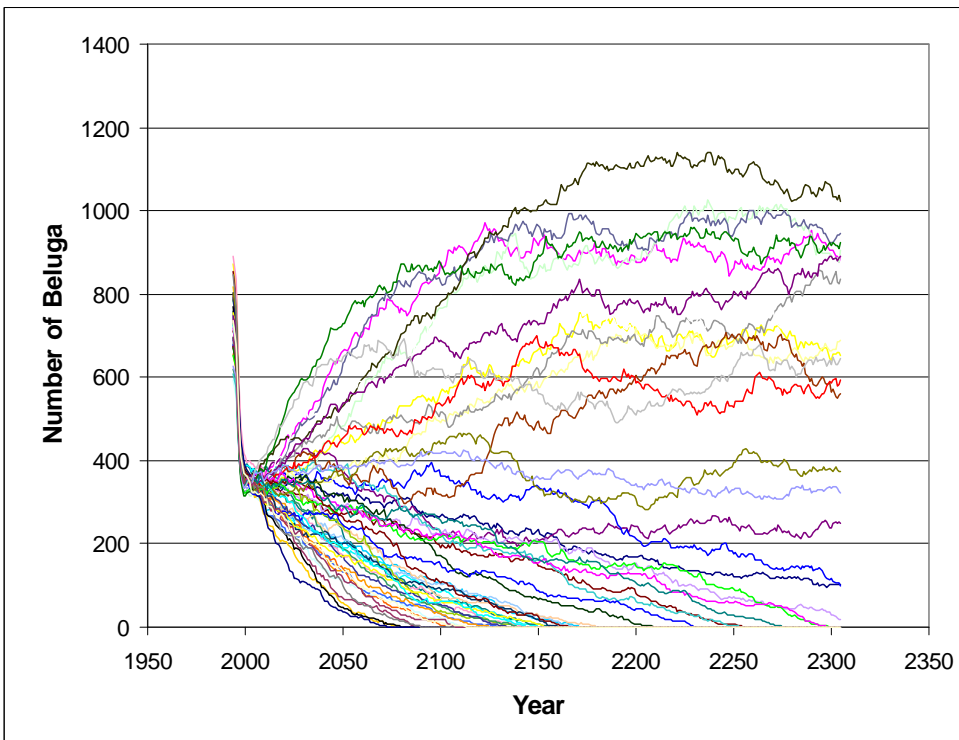
g. Baseline Model
$+ P_{Me} = 0.05$
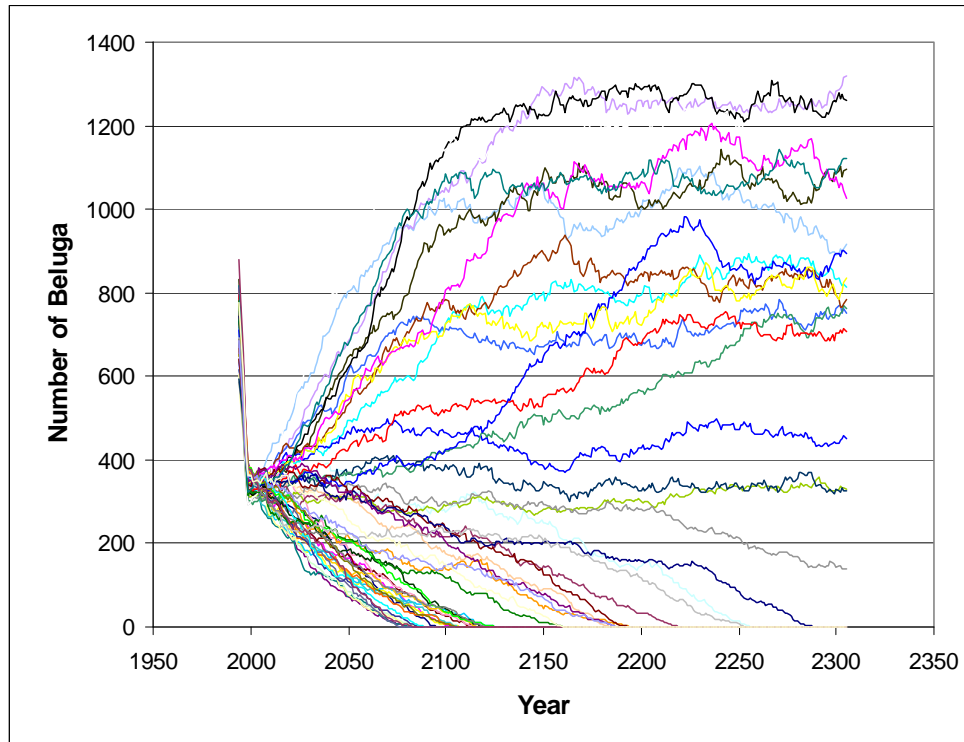
h. Baseline Model
$+ P_{Me} = 0.05$
$+ C = 1$
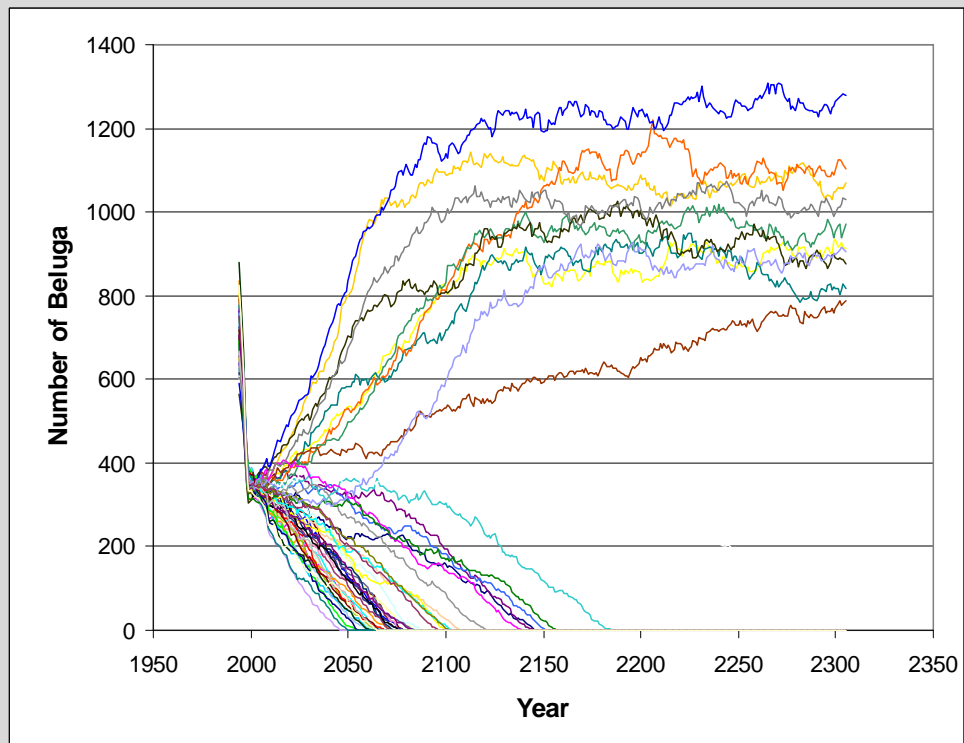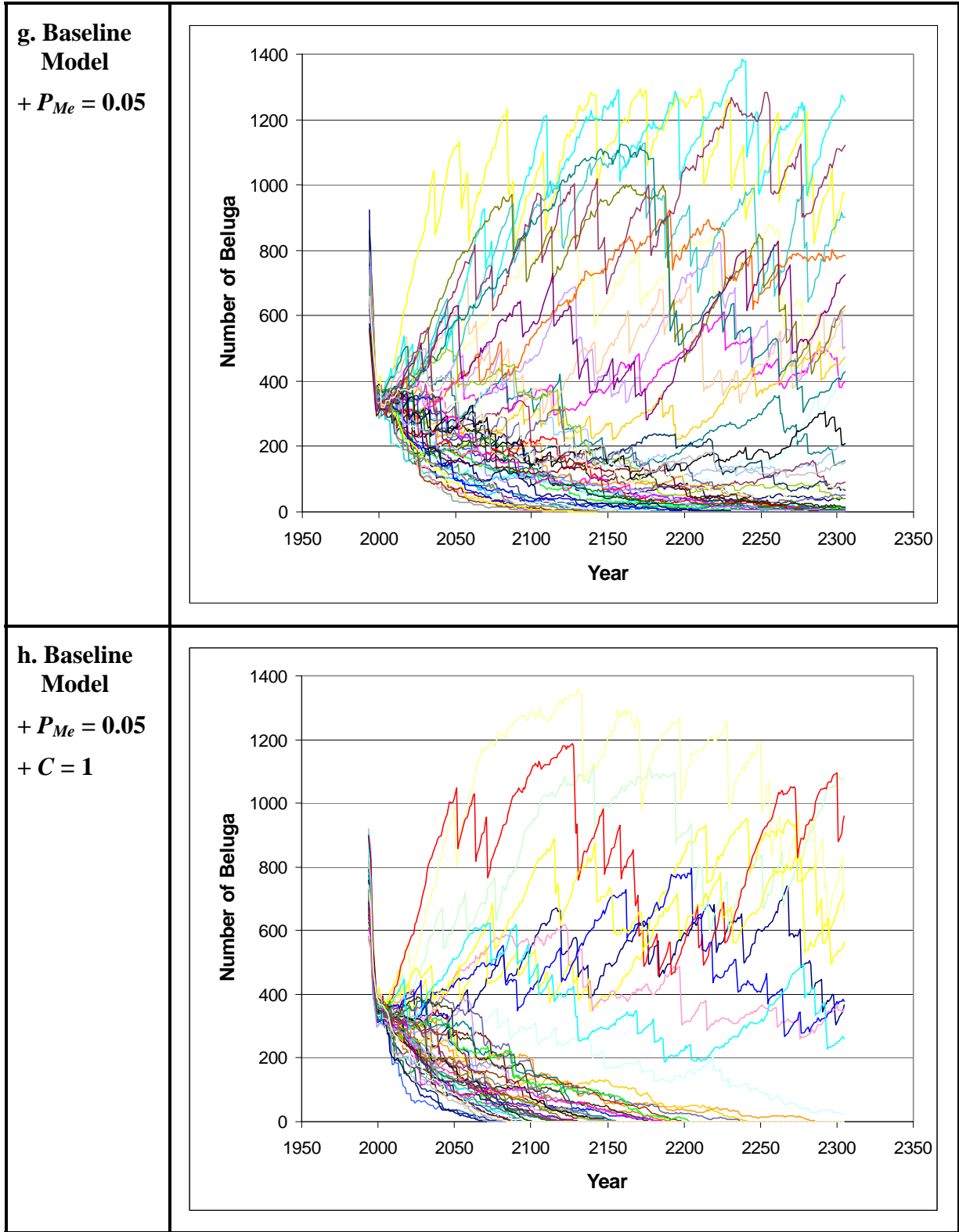
Figure 5.1.2-2. Projections of 50 example cases from the posterior sample of 10,000 trials for eight models (a-h). The Healthy Population (b) and Baseline (f) used parameters and model variations outside the range supported by the available Cook Inlet beluga data and are meant for sensitivity analysis only.

The SIR algorithm provided a posterior distribution for ϕ_0 (Fig. 5.1.2-3) shown here for the Baseline model in the cumulative (black line and left axis) and the density (bars and right axis) forms. The value of ϕ_0 is the annual growth multiplier for a small population (approaching zero) and can be interpreted as similar to $R_{max}+1$ in the potential biological removal (PBR) population model (Wade and Angliss 1997). However, it should be noted that unlike the PBR model these are idealized growth multipliers and the average per capita growth will be less than these values due to demographic stochasticity. The median value for ϕ_0 is 0.995 and 90% of the probability falls between 0.974 and 1.021. Also note that 5% of the probability is above $\phi_0 = 1.02$ (i.e., the minimum 2% growth that was anticipated for a healthy, recovering population).
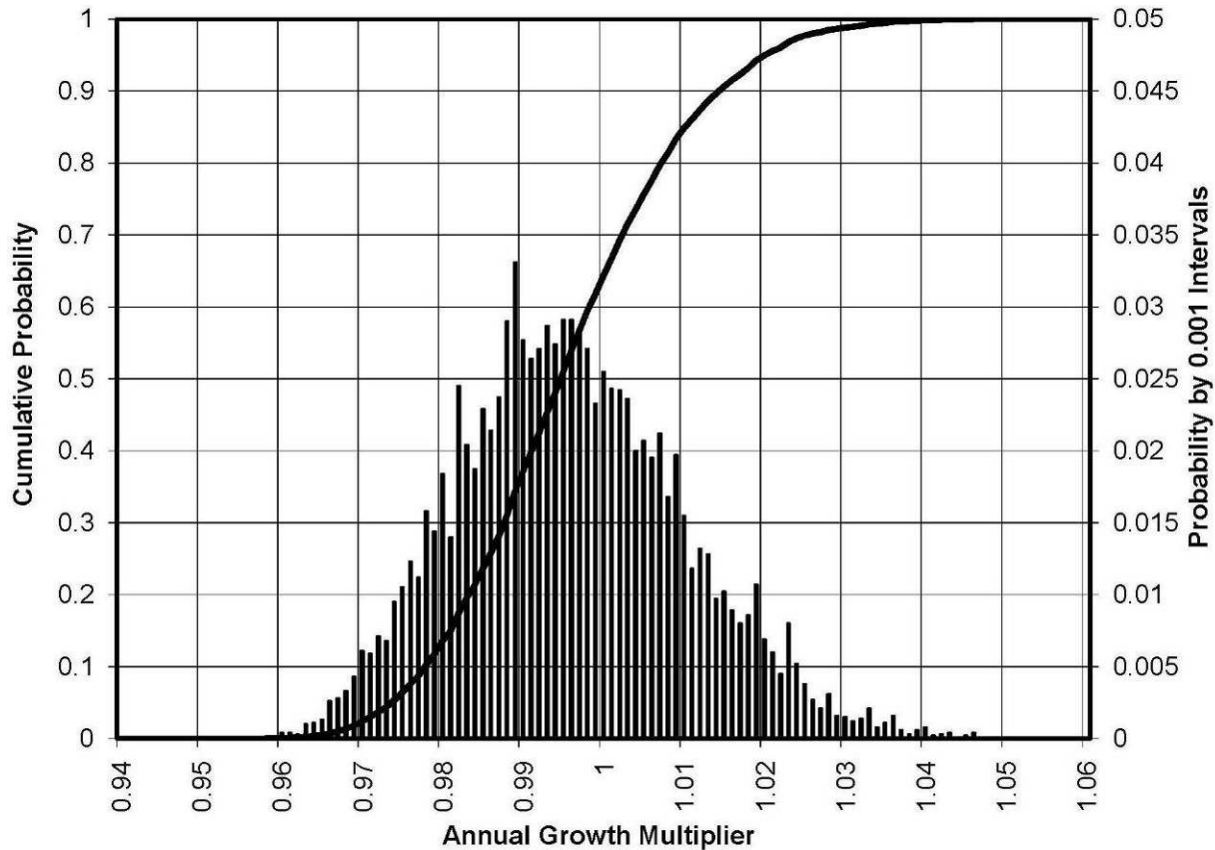
Figure 5.1.2-3. The posterior distribution of ϕ_0 from the Baseline model which had a prior distribution for ϕ_0 of U(0.94, 1.06) and no Allee effect or constant mortality effect; the solid line is the cumulative distribution (left axis). Note that there is a 5% probability that $\phi_0 > 1.02$ (healthy population range) and that there is 62% probability that $\phi_0 < 1.00$; the vertical bars are the probabilities of values of ϕ_0 in 0.001 increments of the distribution (right axis).

As suggested by Figure 5.1.2-2, there was little variation in the fit of the different models to the time series data (1994 to 2008). In a closer examination of the Baseline model (a) results, the abundance in each year from the SIR resample of population trajectories provide posterior distributions for the abundance in each year that account for the population dynamics as well as the annual abundance estimates from surveys. Comparison of the median values and the 2.5 percentiles and 97.5 percentiles of these posterior distributions to the annual abundance estimates from surveys, indicates that the model has a smoothing effect on the time series and gives an indication of the measurement errors that occur in each year (Fig. 5.1.2-4) (Wade 2002). The median values are an estimate of abundance in each year with the 2.5 percentiles and 97.5 percentiles forming a 95% credibility interval which is narrower than the one standard error range in most years and much narrower than the 95% confidence intervals for the individual abundance estimates from the aerial surveys (Fig. 2.3.1-1) (Punt et al. 2004, Brandon and Wade 2006). The Healthy Population model (b) had a similar narrow 95% credibility interval (Fig. 5.1.2-5), but showed a greater lack of fit in several years between 1999 and 2008. Also note that while the model accounted for depletion of the adult population and the delay of 9 years from birth to first reproduction, there is an almost immediate response of the Healthy Population to the change in harvest level.
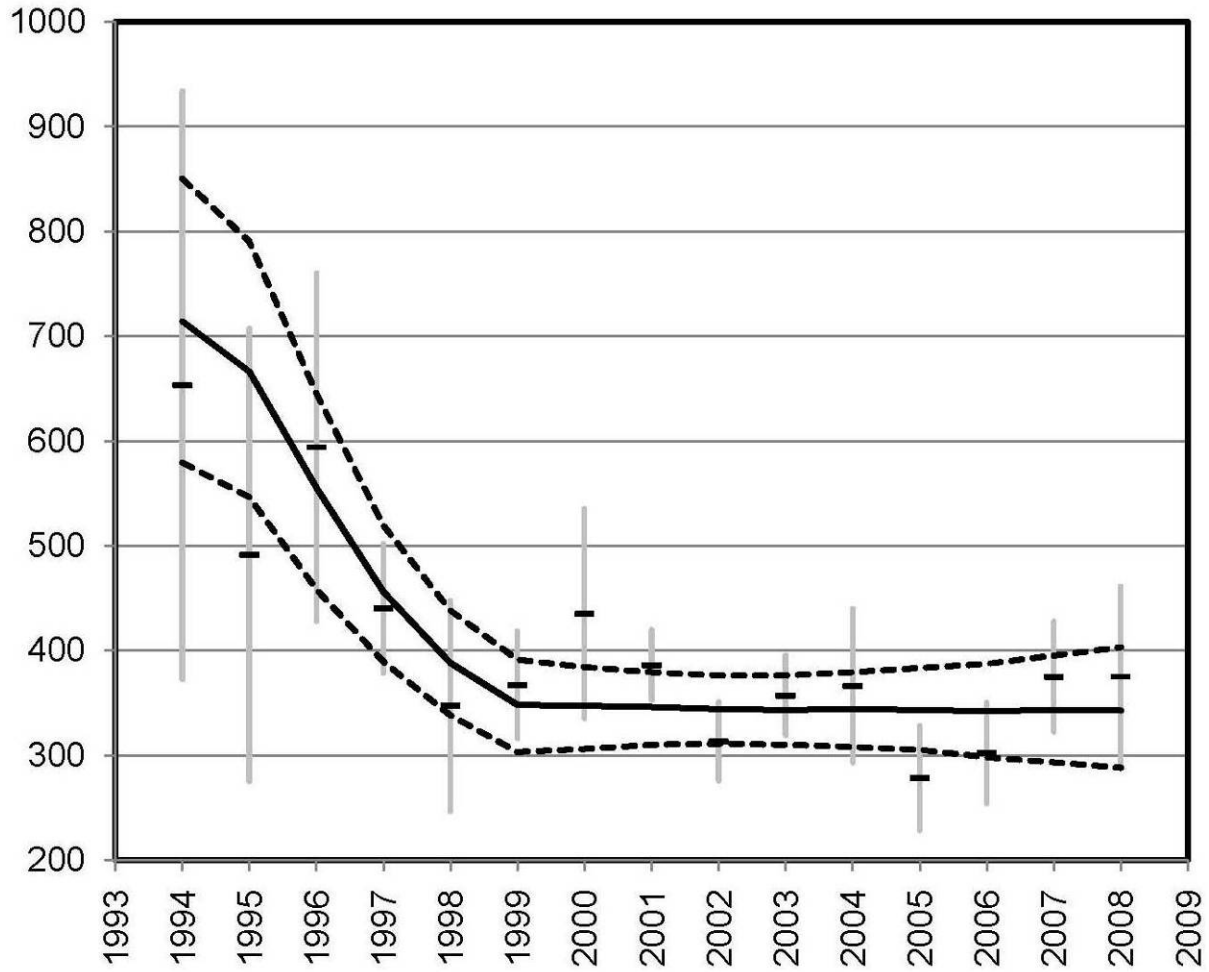
Figure 5.1.2-4. Posterior distributions of annual abundance for the years 1994-2008 from the Baseline model (a) The vertical gray bar is ± 1 standard error for each of the annual abundance estimates (black cross bar). The posterior distributions of the abundance from the population model are represented by the solid line connecting the median values and the dashed lines connecting the 2.5 percentile and 97.5 percentile values. The values between the dashed lines represent a 95% credibility interval.
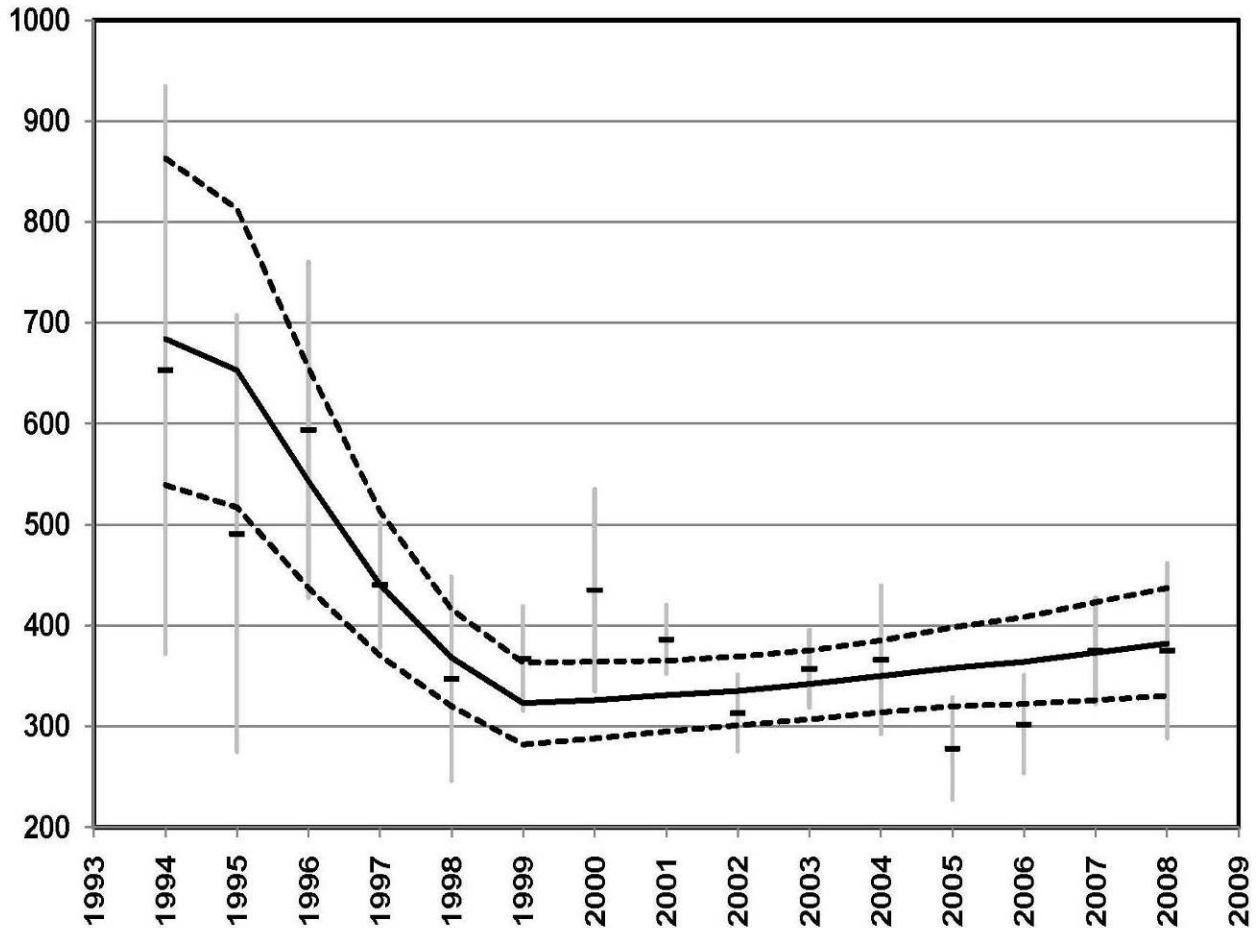
Figure 5.1.2-5. Posterior distributions of annual abundance for the years 1994-2008 from the Healthy Population model (b) The vertical gray bar is ± 1 standard error for each of the annual abundance estimates (black cross bar). The posterior distributions of the abundance from the population model are represented by the solid line connecting the median values and the dashed lines connecting the 2.5 percentile and 97.5 percentile values. The values between the dashed lines represent a 95% credibility interval.

Posterior distributions for abundance in 2008 from variations of the Baseline model were nearly identical with medians ranging between 339 and 343 (Table 5.1.2-1). The Healthy Population model indicated a somewhat higher median of 382 (Table 5.1.2-1, row b) but when combined with $P_{Me} = 0.05$ and $C = 5$ (Table 5.1.2-1, row j) the median value nearly fell in the range of the Baseline variations. The models with missed small, gray animals had higher medians for the 2008 population size because 15 to 20% of the populations were missed in the counts, these under-estimates were compared to the abundance estimate data to test the model fit. All of the variations of the Baseline model considered had probabilities similar to the Baseline model itself

with none being significantly better as indicated by the Bayes factor. The Healthy Population model fit to either the full time series or only years 1999-2008 had a much lower probability than any of the variations of the Baseline model considered and should not be considered viable models. However, the Healthy Population model with $P_{Me} = 0.05$ and $C = 5$ was significantly more likely than the Healthy Population model itself and had a probability similar to that of the variations of the Baseline model.

During the projection from 2008 to 2308, considerable variation occurred within each model run and between models (Fig. 5.1.2-2; Tables 5.1.2-2 through 5.1.2-5). However, by the year 2308, the majority of cases in each model had either gone extinct or recovered to a population size greater than 500 (Table 5.1.2-2). The Healthy Population models (b, t) were the only ones which resulted in all of the cases recovering to a population size above 500 within 300 years (Table 5.1.2-2). For the six unshaded models, the probability of extinction by 2308 was between 29% and 70% (Table 5.1.2-2).

Table 5.1.2-1. Statistics for the posterior distributions of the population size in 2008 (N2008) and the Bayes factors for each model compared to the Baseline model. The dark shading (b, t) is the Healthy Population and the light shading (f, i - s) used parameters and model variations outside the range supported by the available Cook Inlet beluga predation mortality data and are meant for sensitivity analysis only. Note that the absolute value of the Bayes factor should be greater than 2.0 before a significant difference in probability is indicated. U = uniform distribution, C = constant mortality effect parameter, P_{Me} = unusual mortality event (with a 5% annual probability of 20% mortality).

Model ID	Variation from Baseline Model	N2008 Median	N2008 5th percentile	N2008 95th percentile	Probablility Relative to the Baseline	2 × Ln Bayes Comparison to the Baseline
a	ϕ_0 U(0.94, 1.06)	343	295	393	1.00	0.00
b	ϕ_0 U(1.02, 1.06)	382	336	428	0.35	-2.13
c	Allee	341	295	393	0.99	-0.02
d	$C = 1$	341	294	391	0.99	-0.02
e	$C = 2$	339	292	391	1.00	0.00
f	$C = 5$	336	288	386	1.03	0.07
g	$P_{Me} = 0.05$	341	287	393	1.22	0.40
h	$P_{Me} = 0.05, C = 1$	340	285	393	1.14	0.26
i	$P_{Me} = 0.05, C = 5$	333	278	385	1.14	0.26
j	Healthy Population, $P_{Me} = 0.05, C = 5$	353	297	402	1.29	0.51
k	Missed small, gray	339	296	387	1.18	0.33

Table 5.1.2-1. -- Continued.

Model ID	Variation from Baseline Model	N2008 Median	N2008 5th percentile	N2008 95th percentile	Probablility Relative to the Baseline	2 × Ln Bayes Comparison to the Baseline
l	Missed small, gray, $P_{Me} = 0.05$, $C = 1$	339	285	388	1.51	0.82
m	Missed small, gray, Decreasing 1994-2003	338	296	387	0.96	-0.09
n	Missed small, gray, $P_{Me} = 0.05$, $C = 1$ Decreasing	336	284	387	1.46	0.75
o	Variable environment	343	294	393	0.99	-0.02
p	Var. environment, $P_{Me} = 0.05$, $C = 1$	340	285	391	1.21	0.38
q	Immature survival 90% of adult surv.	337	295	381	1.05	0.09
r	Immature survival 90% of adult surv., $P_{Me} = 0.05$, $C = 1$	331	279	376	1.05	0.09
s	Baseline w/ 1999-2008 Likelihood	339	293	389	1.00	0.00
t	Healthy Population w/ 1999-2008 Likelihood	379	335	426	0.39	-1.90
u	Baseline w/ $K = 650$	343	298	391	0.96	-0.09
v	Baseline w/ $K = 1000$	343	297	393	1.00	0.01
w	Baseline w/ $K = 2000$	339	291	392	0.95	-0.11
x	$P_{Me} = 0.03$, $C = 1$	339	289	392	1.09	0.17
y	$P_{Me} = 0.03$, $C = 2$	338	286	390	1.12	0.22
z	$P_{Me} = 0.05$, $C = 1$ with deaths capped at 100 for N > 500	338	284	390	1.26	0.47
aa	Baseline w/ pre1994 harvest = drawn from U(20,50)	349	305	398	1.05	0.10
bb	Baseline w/ fraction of females in harvest = 0.60	334	289	381	0.81	-0.42
cc	Baseline w/ Age at first birth = 8 years	342	294	393	0.96	-0.09
dd	Baseline w/ Age at first birth = 10 years	343	297	394	0.99	-0.01
ee	Baseline w/ Age at first birth = 5 years	339	294	394	0.99	-0.02

Table 5.1.2-2. Outcomes of projections to year 2308 (300 years) for each of the models. The dark shading is the Healthy Population (b, t) and the light shading (f, i-s, u-bb) used parameters outside the range supported by the available Cook Inlet beluga predation mortality data and are meant for sensitivity analysis only.

Model ID	Percent probability that the population will be:					Probability of extinction by 2308 (%)
	> 500	< 500 & > 350	< 350 & > 200	< 200 & > 100	< 100	
a	27	5	7	7	25	29
b	100	0	0	0	0	0
c	25	6	7	6	19	36
d	26	4	4	3	5	58
e	25	3	3	1	2	66
f	22	1	0	0	0	77
g	20	4	5	5	20	45
h	17	3	3	2	3	70
i	11	1	1	0	0	87
j	49	2	2	1	1	46
k	31	5	5	4	6	49
l	22	4	4	3	4	63
m	15	4	5	5	24	47
n	12	2	3	2	3	79
o	29	6	7	7	22	29
p	18	3	3	2	4	69
q	12	6	8	10	35	28
r	0	0	1	1	2	96
s	24	5	6	6	24	35
t	100	0	0	0	0	0
u	11	21	19	11	22	15
v	24	10	9	8	25	23
w	25	3	5	6	22	39
x	20	4	3	3	4	66
y	20	2	2	1	2	73
z	19	2	2	2	3	71
aa	33	6	8	8	24	20
bb	66	5	5	4	11	8
cc	27	5	7	7	25	30
dd	27	6	7	7	24	28
ee	27	6	6	6	23	32

The probability of extinction in 100 years ranged from 1% to 27% for the unshaded models (Table 5.1.2-3). The probability of extinctions before 2058 (within 50 years) reached 1% only in model h in the unshaded models with the combination of stochastic mortality events and $C = 1$ (Table 5.1.2-4).

Table 5.1.2-3. Outcomes of projections to year 2108 (100 years) for each of the models. The dark shading is the Healthy Population (b, t) and the light shading (f, i-s, u-bb) used parameters outside the range supported by the available Cook Inlet beluga predation mortality data and are meant for sensitivity analysis only.

Model ID	Percent probability that the population will be:					Probability of extinction by 2108 (%)
	> 500	< 500 & > 350	< 350 & > 200	< 200 & > 100	< 100	
a	24	11	16	19	30	1
b	100	0	0	0	0	0
c	22	10	17	19	31	1
d	22	9	13	15	29	12
e	22	8	11	11	21	27
f	19	4	6	5	8	58
g	18	7	12	16	44	3
h	17	6	10	12	30	26
i	10	3	4	3	5	74
j	47	9	10	8	8	18
k	30	11	16	16	23	4
l	23	8	11	14	28	15
m	12	7	12	19	49	1
n	11	5	8	11	30	35
o	26	11	16	18	29	0
p	18	6	10	12	29	25
q	10	11	22	26	30	0
r	0	2	6	11	42	38
s	21	9	14	19	36	1
t	100	0	0	0	0	0
u	11	23	28	21	18	0
v	22	13	19	20	25	0
w	22	8	14	17	38	1
x	18	7	11	13	30	20
y	18	6	8	10	19	39
z	18	5	9	12	29	27
aa	30	12	18	19	20	0
bb	57	10	13	10	10	0
cc	24	10	16	19	30	0
dd	24	11	17	19	29	0
ee	24	10	15	18	32	1

Table 5.1.2-4. Outcomes of projections to year 2058 (50 years) for each of the models. The dark shading is the Healthy Population (b, t) and the light shading (f, i-s, u-bb) used parameters outside the range supported by the available Cook Inlet beluga predation mortality data and are meant for sensitivity analysis only.

Model ID	Percent probability that the population will be:					Probability of extinction by 2058 (%)
	> 500	< 500 & > 350	< 350 & > 200	< 200 & > 100	< 100	
a	19	17	30	25	8	0
b	100	0	0	0	0	0
c	18	17	30	26	9	0
d	17	16	27	26	14	0
e	17	14	24	25	20	0
f	14	10	18	18	27	12
g	16	12	24	28	21	0
h	14	11	21	26	28	1
i	9	7	12	15	29	30
j	38	21	22	12	6	1
k	29	20	29	18	4	0
l	24	14	24	23	16	0
m	8	11	27	35	19	0
n	9	8	19	26	38	1
o	21	18	28	24	9	0
p	15	11	21	25	28	0
q	7	18	40	29	6	0
r	1	4	18	35	41	0
s	16	15	28	29	12	0
t	100	0	0	0	0	0
u	9	27	41	20	4	0
v	18	19	33	23	6	0
w	17	14	26	28	14	0
x	15	12	24	26	23	0
y	15	11	20	24	28	2
z	15	10	20	26	29	1
aa	24	20	33	20	3	0
bb	43	17	24	13	3	0
cc	19	17	30	26	9	0
dd	19	18	30	25	8	0
ee	19	16	29	26	10	0

The Allee effect had a limited impact on the probability of extinction in 100 years, increasing the probability by 1% over the Baseline, however, by 300 years the risk increased 9%. Where C was 1 or 2 animals per year (models d, e, h) there was a 12% to 27% probability of extinction in 100 years and 58% to 66% in 300 years. The effect of including $C = 1$ with $P_{Me} = 0.05$ (model h) is roughly equivalent to $C = 2$ (model e) (Table 5.1.2-3). As indicated in Figure 5.1.2-1, the $C = 1$ threshold was around 60 animals and the $C = 2$ threshold was around 120 animals. Three

unusual mortality events in a short time span would nearly reduce the population by half making up the difference between the two thresholds. Increasing C to 5 mortalities per year (models f, i, j), raised the population size threshold to 200 belugas, increased the overall risk of decline and extinction in each model to the extent that a significant probability existed for extinction in 50 years (Table 5.1.2-3). Again this population size threshold was increased by including unusual mortality events (c.f. models i and j).

In general, unusual mortality events added 10% to 15% to the probabilities of extinction in 300 years in each variation of the Baseline model. In the Healthy Population model, unusual mortality events with $C = 5$ resulted in a probability of extinction in 300 years of 46% indicating that a small increase in predation by killer whales would put a recovering population at risk. Variation in the model assumptions did not result in significant variation in the results for the similar unshaded models and, of particular note, models k and l which correspond to models a and h, but include the assumption of missed small gray animals, had similar but slightly reduced probabilities of extinction and nearly identical probabilities of decline (Table 5.1.2-5).

Table 5.1.2-5. Extinction risk for each of the models by 2058, 2108, 2208, and 2308. The dark shading is the Healthy Population (b,t) and the light shading (f,i-s,u-bb) used parameters outside the range supported by the available Cook Inlet beluga predation mortality data and are meant for sensitivity analysis only. Probability of declining is the probability that $N_{2308} < N_{2008}$

| Model ID | Percent probability of extinction by: | | | | Probability of declining (%) |
	2058 (50 years)	2108 (100 years)	2208 (200 years)	2308 (300 years)	
a	0	1	14	29	68
b	0	0	0	0	0
c	0	1	20	36	69
d	0	12	47	58	70
e	0	27	58	66	72
f	12	58	74	77	78
g	0	3	28	45	76
h	1	26	61	70	80
i	30	74	85	87	89
j	1	18	39	46	49
k	0	4	35	49	64
l	0	15	51	63	74
m	0	1	27	47	81
n	1	35	70	79	86
o	0	0	14	29	65
p	0	25	60	69	79
q	0	0	11	28	81
r	0	38	87	96	99
s	0	1	19	35	71
t	0	0	0	0	0
u	0	0	6	15	68
v	0	0	11	23	66
w	0	1	22	39	72
x	0	20	56	66	76
y	2	39	67	73	78
z	1	27	63	71	79
aa	0	0	7	20	60
bb	0	0	3	8	27
cc	0	0	15	30	68
dd	0	0	13	28	67
ee	0	1	17	32	67

5.1.3. Discussion and Conclusions

Although the model structure and parameters had obvious effects on the distributions of predicted outcomes, even the best case scenario (model a) with no threshold effects resulted in population declines in 68% of the cases and extinction in 29% of the cases within 300 years. With this most optimistic scenario, with no harvest after 2005, only 27% of the cases resulted in

a population above 500 animals in 2308. The distributions of possible outcomes were sensitive to a variety of poorly known small population effects; however, the data that we do have supports the choice of the Baseline (model h) as the best approximation of the current population with the estimated mortality due to killer whale predation averaging 1 per year ($C = 1$) and allowing for uncertainty with unusual mortality events occurring on average every 20 years. This model had a 1% probability of extinction in 50 years, 26% probability of extinction in 100 years, and a 70% probability of extinction in 300 years. Although there were no data to support higher predation rates or more frequent unusual mortality events, the examples given (models f, i, j) indicate the fragile nature of this population and even when the events are less frequent, averaging three per century rather than five, an increase in the predation parameter to $C = 2$ resulted in 2% risk of extinction in 50 years (models x, y). Should the constant mortality level increase either by increased killer whale predation or other means, or if this mortality level has been underestimated, the population would have a very high probability of decline and a significant probability of extinction in 50 years (e.g., Table 5.1.2-5, row f, i).

A synergistic effect occurred when the Allee effect or constant mortality effect acted as traps for populations hit by a series of unusual mortality events, which hastened the extinction of declining populations and placed even populations with an otherwise healthy annual increase at risk. Several of the assumptions of this analysis have been questioned during reviews. To test these assumptions, we used models k-t, which indicate that the results are robust to relaxation of the assumptions. Of particular interest are the results for models k and l where over half of the small gray animals under 10 years of age are assumed to be missed during the aerial surveys. Models k and l were designed to test the results presented in models a and h, respectively, and ended up having nearly identical results to these models. In all but models o and p, the environment is assumed to be constant. Models o and p showed results similar to the constant environment models, however, without an environmental time series and mechanism forcing the population it was unlikely that environmental variability alone would do more than add to the existing variability.

Lower values for carrying capacity ($K = 650$ and 1,000) (models u, v) had the same or lower probabilities of extinction at each time frame while a higher value ($K = 2,000$, model w) had a

higher extinction risk. This results from the shape of the production curve at 350 whales (Fig. 5.1.3-1). With $K = 650$, 350 is near MNPL where the growth multiplier is reduced below the maximum growth rate by 0.007. The model is fit to the population size range between 300 and 400 so all of the models will have similar values in this range. For smaller values of K, the maximum growth rate will be higher for the same fitted value than the Baseline with $K = 1,300$. This creates a greater potential for an equilibrium point between population sizes 100 and 500 which explains why more than half of the populations remained in this range after 300 years (Table 5.1.5-2, model u). The opposite occurs when $K = 2,000$, the growth curve is nearly flat below 500 whales so an equilibrium point is unlikely to be below 350 whales.
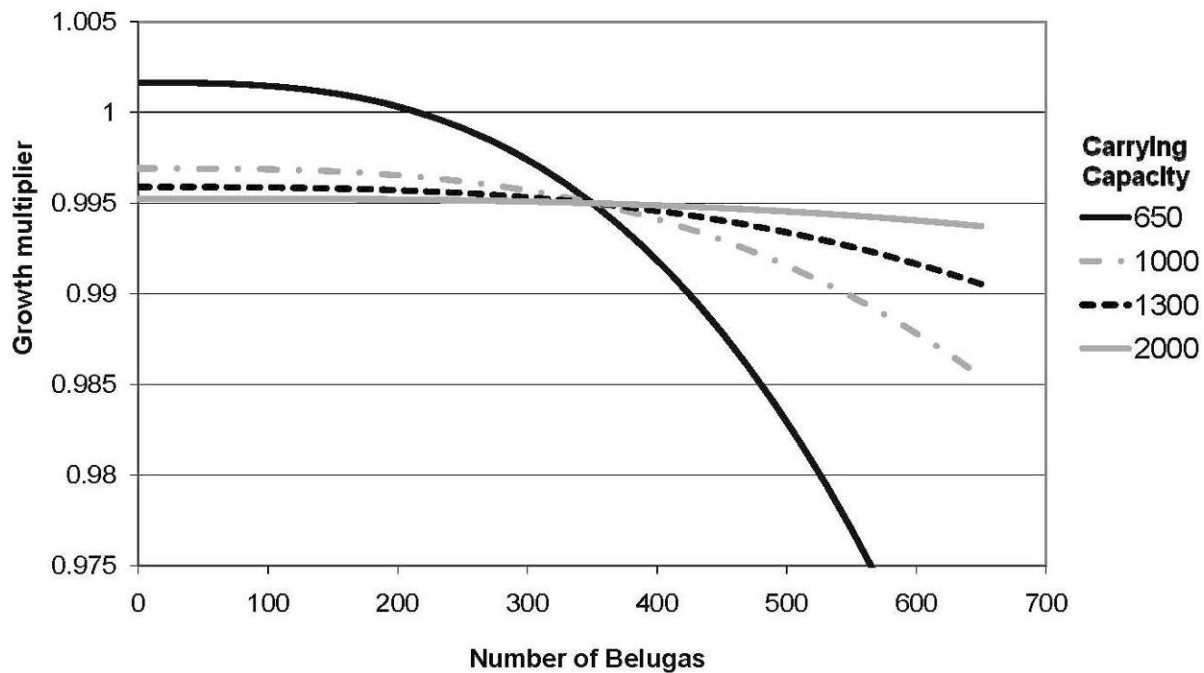
Figure 5.1.3-1. Per Capita Annual Growth Multiplier by population size for carrying capacity (K) = 650, 1,000, 1,300, 2,000 with parameters for survival and birth rate set so that the growth multiplier for a population of 350 = 0.995 the median value for the base model (See Fig. 5.1.2-3).

Models x and y are versions of model h (the NMFS preferred model) with the probability of an unusual mortality event reduced to 0.03. Note that a doubling of the C parameter (model y) results in a greater probability of extinction than model h. Model z is also a variation of model h with the total mortality from an event limited to 100 animals. In this respect, as the population

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increases a smaller fraction of the total population is lost during an event. While this increases the likelihood that populations will recover, the extinction risk remains the same as model h.

Variations to the harvest parameter are considered in model aa, a pre-1994 harvest drawn from U(20, 50), and model bb, a harvest comprised of 60% females, rather than the 45% females determined from harvest samples. Both models result in greater forcing via depletion of the adult female population during the period 1979-1993 or 1994-1998, and a stronger rebound after 1999, which leads to a lower risk of extinction and higher probability of recovery than the Baseline model.

Variation by one year in the age at first birth (models cc and dd), including the range from the available beluga life history data, and halving the age at first birth as was assumed in Hobbs et al. (2006) based on 2 GLG/yr ageing (model ee), had little effect on recovery or extinction risk.

While several of the sensitivity trials showed some improvement in extinction risk, only the assumption of a growth rate greater than 2%, the least likely model, removed the risk of decline and extinction. Taken as a whole, these modeling results indicate clearly that it is likely that the Cook Inlet beluga population will continue to decline or go extinct over the next 300 years unless factors determining its growth and survival are altered in its favor.

5.2. Application of IUCN Criteria

Information on application of IUCN criteria is provided in Hobbs et al. (2006).

6. CONCLUSIONS OF THE STATUS REVIEW

The small, isolated population of belugas in Cook Inlet has not shown appreciable signs of recovery since 1999 when hunting restrictions began. Prior to this, a significant decline in abundance was documented from 1994 to 1998, but there are little empirical data for the period between 1979 and 1994 to identify a mechanism for the apparent decline of this population from 1,300 to 650. Anecdotal reports suggest a Native subsistence hunt (enumerated through hunter interviews) was significant during the 1970s and 1980s and may have been at levels similar to the hunts reported in the mid-1990s. Also, commercial and sport hunts occurred during the 1960s and 1970s, so the highest available abundance estimate of 1,300, based on the 1979 ADF&G survey, may already represent a partially depleted population. With the very limited hunt between 1999 and 2008, NMFS anticipated that the population would begin to recover at a rate of 2% to 6% per year. However, a Bayesian analysis including the 2008 estimates of abundance indicates that there is a probability of only 5% that the annual increases of 2% or greater will occur and a probability of 62% or more that the population will decline further.

A population viability analysis was conducted to assess the extinction risks faced by this small population under a range of scenarios that considered density dependence, constant mortality, Allee effects, and catastrophes. The best case scenario, with no threshold effects, resulted in population declines in 68% of the cases and extinction within 300 years in 29%. Even with this most optimistic scenario, and with no harvest after 2008, only 24% of the cases resulted in a population above 500 animals in 2108 (within 100 years). There is a significant probability that the Cook Inlet beluga population will continue to decline or go extinct over the next 300 years unless factors determining its growth and survival are altered in its favor. The contraction of the range of this population northward into the upper inlet makes it far more vulnerable to catastrophic events with the potential to kill a significant fraction of the population. The probability of potential catastrophic events -- such as oil or toxic substance spills, failure of key fish runs, ice entrapments, or disease or parasitic introductions -- added between 15% and 40% to the probabilities of extinction in 300 years in the models. As the models demonstrate, killer whale predation which is documented on a near annual basis, could also significantly impact recovery. Since belugas spend much of their time in shallow waters, stranding is a constant risk.

Prolonged stranding events more than a few hours, although not common, may under unusual circumstances such as unusual tidal cycles, storm surge, flooding, tsunami, or earthquake uplift result in significant mortalities.

Belugas in Cook Inlet make up a small, genetically distinct population that appears to have strong site fidelity to the inlet year-round. Should this population go extinct, it is highly unlikely that Cook Inlet would be repopulated with belugas in the foreseeable future. The closest large population is in Bristol Bay, 1,500 km away by sea and separated by the Alaska Peninsula that extends 3 degrees of latitude south of the southern limit of the Bristol Bay beluga population. It is highly probable that the loss of the Cook Inlet beluga population would result in a permanent loss of range for the beluga species.

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