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Status Review of Hawaiian Insular False Killer Whales (*Pseudorca crassidens*) under the Endangered Species Act

Erin M. Oleson, Christofer H. Boggs, Karin A. Forney,
M. Bradley Hanson, Donald R. Kobayashi, Barbara L. Taylor,
Paul R. Wade, and Gina M. Ylitalo

Pacific Islands Fisheries Science Center
National Marine Fisheries Service
National Oceanic and Atmospheric Administration
U.S. Department of Commerce

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EXECUTIVE SUMMARY

Background

On October 1, 2009, the National Marine Fisheries Service (NMFS) received a petition from the Natural Resources Defense Council requesting that Hawaiian insular false killer whales (*Pseudorca crassidens*) be listed as endangered under the Endangered Species Act (ESA) and that NMFS designate a critical habitat for these whales. NMFS reviewed the petition, decided that the petition presented substantial scientific information indicating that an ESA listing may be warranted as evidenced by the small population size and recent population decline, and committed to conducting an ESA status review of Hawaiian insular false killer whales. NMFS formed a Biological Review Team (BRT) made up of scientists with diverse backgrounds to conduct the status review. The BRT considered a variety of scientific and technical information. This document reports the results of its comprehensive ESA status review of the Hawaiian insular false killer whale.

Approach of the BRT

The BRT acknowledged that there is considerable uncertainty surrounding all aspects of the Hawaiian insular false killer whale's biology, abundance, trends in abundance and threats. Such uncertainties are expected for a naturally uncommon species that is primarily found in the open ocean where research is expensive and knowledge is consequently poor. The team decided to treat the uncertainty explicitly by defining where it exists and using a point system to weigh various plausible scenarios, taking into account all available data on false killer whales but also considering information on other similar toothed whales. The team's objectives in taking this approach were to make the process of arriving at conclusions as transparent as possible and to provide assurance that the team was basing its decisions on a common understanding of the evidence. Details are given in Appendix A. DPS Designation and Risk Assessment Scores.

Status of Hawaiian Insular False Killer Whales

Hawaiian insular false killer whales are genetically distinguishable from pelagic false killer whales based on significant differences in both mitochondrial (mt) and nuclear DNA (Chivers et al., 2007, 2010). The mtDNA and nuclear DNA data show a strong differentiation of Hawaiian insular false killer whales at other spatial scales, including when they are compared at a broad scale with whales in the central North Pacific and eastern North Pacific, and at a finer scale with pelagic whales in Hawaiian waters and with whales in Mexico, Panama, and American Samoa (all comparisons to Hawaiian insular false killer whales have p-values < 0.001). The presence of two unique, closely related haplotypes in the Hawaiian insular population is consistent with little to no immigration from other areas. The pattern of primarily closely related haplotypes shown in Hawaiian insular false killer whales is consistent with a social system that excludes immigrants or a habitat specialization that makes survival of immigrants unlikely (or

both). The magnitude of differentiation for nDNA is somewhat less than for mtDNA, which could suggest small amounts of male mediated gene flow, but there are alternate reasons for the difference that are also possible.

The range of Hawaiian insular false killer whales may be assessed from a number of data sources, including ship and aerial survey sightings, and satellite tag telemetry. The location data from satellite telemetry indicate that individual insular false killer whales move widely and quickly among the main Hawaiian Islands and use waters up to 112 km from shore (Baird et al., 2010; Forney et al., 2010). Wide movement among the main Hawaiian Islands is also shown by re-sightings of photographically identified individuals over several years (Baird et al., 2005; Baird et al., 2008a; Baird, 2009). Tagged individuals were tracked over a broad range of depths, from shallow (< 50 m) to very deep (> 4000 m), with similar water depths and types of habitat used on both the windward and leeward sides of all islands (Baird et al., 2010). Hawaiian insular false killer whales share a portion of their range with the genetically distinct pelagic population of false killer whales (Forney et al., 2010). Satellite telemetry locations of a single tagged individual from the pelagic population, as well as sightings from ship and small boat surveys suggest that the distribution of the insular and pelagic populations overlap in the area between about 42 km and 112 km from shore.

The 2009 Stock Assessment Report for Hawaiian insular false killer whales (Carretta et al., 2010) gives the current best-estimate of population size as 123 individuals (CV = 0.72), citing Baird et al. (2005). Recent reanalysis of photographic data has yielded two new estimates of population size for the 2006–2009 period. Two estimates are presented because two groups photographed near Kauai have not (perhaps yet) been observed to associate into the social network of false killer whales seen at the other islands. These animals may come from the pelagic population, may come from another undocumented population in the Northwestern Hawaiian Islands, or may represent a portion of the insular population that has not been previously documented photographically. The current best estimates of population size are 151 individuals (CV = 0.20) without the animals photographed at Kauai or 170 individuals (CV = 0.21) with them.

Genetic data were used to estimate effective population size, which reflects the size of an idealized population that would experience genetic drift in the same way as the actual (census) population. Effective population size relates to inbreeding coefficients and an effective size of 50 is used as a rule of thumb below which inbreeding depression becomes likely. The estimate for the effective population size of Hawaiian insular false killer whales is 45.8 individuals (95% CI 32.4–69.4) (Chivers et al., 2010).

Aerial survey sightings noted since 1989 suggest that the insular population of Hawaiian false killer whales has declined over the last two decades. A survey conducted on the leeward sides of the main islands in June and July 1989 reported 14 sightings of false killer whales, including three large groups (group sizes 470, 460, and 380 individuals) very close to shore off the island of Hawaii (Reeves et al., 2009). The largest group seen in 1989 is more than three times larger than the current best estimate of the population size. From 1993 to 2003 five systematic aerial surveys indicated declining false killer

whale encounter rates, with 8 groups seen in 1993, 9 in 1995, 1 in 1998, and no false killer whales seen in 2000 and 2003 (Mobley et al., 2000, Mobley, 2004). The large group sizes observed in 1989, together with the declining encounter rates from 1993 to 2003 suggest that Hawaiian insular false killer whales have declined significantly in recent decades.

It is possible that weather or other survey conditions were at least partially responsible for the decline in sighting rates from 1993 to 2003; however, there was no trend in the sighting rates for the four most commonly seen species of small cetaceans (spinner dolphin, bottlenose dolphin, spotted dolphin, and short-finned pilot whale). These four species represent nearshore and pelagic habitat preferences and span a range of body sizes from smaller to larger than false killer whales. It can be inferred from this evidence that variability in sighting conditions during the survey period did not have a major effect on sighting rates and, therefore, the rate for insular false killer whales has, in fact, declined. The specific cause of such a decline is unknown, but a number of possible causes and possible threats to the population were examined by the BRT.

Determination of Distinct Population Segments

The BRT concluded that Hawaiian insular false killer whales are a distinct population segment (DPS) of the global false killer whale taxon. The BRT found that Hawaiian insular false killer whales are discrete from other false killer whales, based on the uniqueness of their behavior related to habitat use patterns and their existence in a unique ecological setting. The false killer whale is typically considered a wide-ranging pelagic species not generally associated with coastal or island habitats; however, they have been described as common in some shelf waters and at some islands (IWC, 2007). Hawaiian insular false killer whales are behaviorally unique because they are the only population of the species whose movements are known to be restricted to the vicinity of an oceanic island group. Further, Hawaiian insular false killer whales remain close to the islands and primarily use waters that are relatively shallow and productive compared to surrounding oligotrophic waters, an ecologically unique setting for the taxon. The BRT found Hawaiian insular false killer whales to be significant to the taxon based on marked genetic differences and ecological and cultural factors. Hawaiian insular false killer whales differ from other populations of the species in their genetic characteristics: there is a strong phylogeographic pattern of mtDNA haplotypes and an indication of low levels of gene flow. Hawaiian insular false killer whales sometimes consume prey associated with islands and movement and photographic resighting data imply a unique, island-associated habitat-use strategy on the part of this population compared to other false killer whale populations. Additionally, although their abundance is quite low, Hawaiian insular false killer whales have a high density (animals per km^2) relative to other false killer whale populations, suggesting the nearshore habitat or a unique habitat-use strategy may support a higher density of animals. Finally, culture, or knowledge passed through learning from one generation to the next, is likely to play an important role in the evolutionary potential of false killer whales because they are highly social with a long period of dependence by calves as they learn to forage cooperatively. The combination

of genetic, ecological, and cultural significance factors lead the BRT to conclude that Hawaiian insular false killer whales are significant to the taxon. The remaining uncertainty is based primarily on the shortage of information on the biology and ecology of false killer whale populations in other island areas and of the adjacent population of pelagic false killer whales offshore the coast of Hawaii and to gaps in genetic sampling to the west of Hawaii. However, the BRT did not find this lack of information sufficient to alter the significance finding for Hawaiian insular false killer whales.

Risk Assessment

The BRT qualitatively assessed the severity, geographic scope, and level of certainty of 28 potential individual threats to Hawaiian insular false killer whales. Because the severity and scope of individual threats may change through time, each threat was evaluated based on its historic impact and its current or future potential for impact. The factors believed to have had the greatest potential for contributing to the decline of Hawaiian insular false killer whales historically are reduced prey biomass and prey size, ecological interactions (competition) with commercial fisheries, accumulation of natural and anthropogenic contaminants, live captures prior to 1990, operational interactions with nearshore fisheries (primarily within about 40 km) and offshore longline fisheries (leading to injuries or deaths), and reduced genetic diversity. Some of the possible causes of the historic decline have been eliminated in recent times (e.g., live captures), and others have been limited in their geographic scope (e.g., interactions with the commercial longline fishery following the implementation of the 25–75 nmi longline exclusion boundary in 1991). Other threats have emerged over time or have increased in severity or geographic scope, making them more relevant to the current and/or future decline of the population. Reduced prey biomass and prey size, ecological interaction (competition) with commercial fisheries, the accumulation of natural or anthropogenic contaminants, potential interactions in nearshore fisheries, and reduced genetic diversity remain as potentially severe threats to the population. The severity and geographic scope of some other threats may have increased, including a potential increase in disease susceptibility because of exposure to immunosuppressive environmental contaminants and short- and long-term climate change; interactions with troll, handline, shortline, and kaka line fisheries; an increase in anthropogenic noise events; and susceptibility to the adverse biological effects of small population size including inbreeding depression and Allee effects. Overall, the BRT considered the effects of small population size, hooking, entanglement, and intentional harm by fishers to be the most substantial threats to Hawaiian insular false killer whales. Fisheries interactions were postulated to be a major threat because extensive unobserved troll, handline, and other hook-and-line fisheries target large pelagic fish and continue to operate at near-record levels in the Hawaiian insular false killer whale core area, and sparse documentation of these fisheries suggests false killer whale interactions do occur. Threats were considered substantial by the BRT if they were highly plausible, even without direct evidence.

The team conducted a quantitative analysis of extinction risk using a Population Viability Analysis (PVA). This modeling exercise was conducted to evaluate the probability of

actual and near extinction (or quasi-extinction), defined as fewer than 20 animals, given measured, estimated, and inferred information on population size and trends, and varying impacts of catastrophes, environmental stochasticity and Allee effects. The PVA models did not consider causation but, rather, used the trend in the available data to represent the consequences of the threats acting in concert on the population. A variety of alternative scenarios were evaluated to allow for assessment of uncertainty in the PVA results. Forty-four of 45 models, including models allowing for a change in growth rate following the evident decline in the 1990s, indicated a greater-than-5% chance of decline to fewer than 20 individuals within 75 years. All single growth-rate models indicated declines to fewer than 20 individuals within 75 years as near certainty. All plausible models allowing a change in growth rate indicated a greater-than-20% chance of decline to fewer than 20 individuals within 75 years. A model showing positive population growth allowed growth up to 30% per year, a highly improbable rate of growth for any cetacean species. The BRT did not regard this optimistic model to be biologically plausible. The other model that showed a 1.3% chance of extinction in 75 years was run as a sensitivity test to examine only the effects of small population size.

Given the possible threats to the population and the results of the PVA analysis, the BRT agreed by consensus that Hawaiian insular false killer whales are at high risk of extinction as a result of either small scale incremental impacts over time or a single catastrophic event. The BRT defined high risk as more than a 5% chance over three generations (75 years) that the population will fall below a level where recovery may not be possible. Because false killer whales are highly social animals this level is estimated to be 20 animals, which is about the average group size.

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List of Acronyms

BRT	Biological Review Team
CEC	chemical of emerging concern
CLIOTOP	Climate Impacts on Top Predators
CML	commercial marine license
CPUE	catch-per-unit effort
CV	coefficient of variation
DIP	Demographically independent populations
DLNR	Department of Land and Natural Resources
DPER	daily prey energy requirement
DPS	discrete population segment
EEZ	Exclusive Economic Zone
EIS	Environnemental Impact Statèrent
EPO	Eastern Pacific Ocean
ESA	Endangered Species Act
ETP	Eastern Tropical Pacific
FEP	Fishery Ecosystem Plan
FMP	Fishery Management Plan
HAB	harmful algal bloom
HCB	Hexachlorobenzene
HCH	Hexachlorocyclohexanes
HDAR	Hawaii Division of Aquatic Resources
HLCC	Hawaiian Lee Countercurrent
IUCN	International Union for the Conservation of Nature
KBWF	Kona Blue Water Farms
LOF	List of Fisheries
LOQ	Limits Of Quantization
MHI	main Hawaiian Islands
MMPA	Marine Mammal Protection Act
MPA	marine protected area
MR	mark-recapture
MSA	Magnuson-Stevens Fisheries Conservation Act
MSY	maximum sustainable yield
mtDNA	mitochondrial Deoxiribose Nucleic Acid (DNA)
nDNA	nuclear DNA
NEC	North Equatorial Current
NHRC	North Hawaiian Ridge Current
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NPSG	North Pacific Subtropical Gyre
NPTZ	North Pacific Transitional Zone
NRDC	Natural Resources Defense Council
NRKW	Northern Resident killer whales
NWHI	Northwestern Hawaiian Islands
PAH	Polycyclic aromatic hydrocarbon

PBDEs	polybrominated diphenyl ethers
PCB	polychlorinated biphenyl
POP	persistent organic pollutants
PSRG	Pacific Scientific Review Group
PVA	population viability analysis
SRKW	Southern Resident killer whales
SST	sea surface temperature
SSTF	South Subtropical Front
STCC	Subtropical Countercurrent
STF	Subtropical Front
SWFSC	Southwest Fisheries Science Center
TRP	Take-Reduction Plan
TRT	Take-Reduction Team
USFWS	U.S. Fish and Wildlife Service
UTC	Units to Conserve
WPWP	Western Pacific Warm Pool

1. Introduction

1.1. Scope and intent of the status review

On October 1, 2009, the National Marine Fisheries Service (NMFS) received a petition from the Natural Resources Defense Council (NRDC) requesting, in light of the small population size, evidence of a declining population trend, and many potential threats to the population, that the Hawaiian insular false killer whales (*Pseudorca crassidens*) be listed as endangered under the U.S. Endangered Species Act (ESA). In addition, the petitioners requested that NMFS designate critical habitat for Hawaiian insular false killer whales. The petitioned population is currently defined by NMFS under the Marine Mammal Protection Act (MMPA) as the Hawaii insular stock and consists of all animals occurring within 40 km of the main Hawaiian Islands, as well as a proportion of the animals within 140 km of the islands. The range of the insular population overlaps with that of the Hawaiian pelagic population of false killer whales in the area between 40 km and 140 km from shore. The petitioners state that while the cause of the recent decline in the population is unknown, multiple factors may threaten the continued survival of Hawaiian insular false killer whales.

NMFS reviewed the petition and concluded that the petition presented substantial scientific or commercial information indicating that the ESA listing may be warranted (75 *FR* 316; January 5, 2010). As a result of the finding, NMFS committed to completing an ESA status review of Hawaiian insular false killer whales and convened a team of scientists with diverse scientific backgrounds, the Biological Review Team (BRT) [1], to conduct the review. During its status review, the BRT considered a variety of scientific information from the literature, unpublished documents, and direct communications with researchers working on false killer whales, as well as technical information submitted to NMFS. All information not previously peer-reviewed was formally reviewed by the team, and only the information found to meet the standard of best-available science was considered further. Analyses conducted by individual team members were subjected to independent peer review prior to incorporation into the Review. This document reports the results of the BRT's comprehensive status review of Hawaiian insular false killer whales.

This document is a compilation of the best available scientific and commercial information and a description of past, present, and likely future threats to the insular population of Hawaiian false killer whales. It does not represent a decision by NMFS on whether this population should be proposed for listing as threatened or endangered under the ESA. That decision will be made by NMFS after reviewing this document, other

[1] The Biological Review Team for the insular population of Hawaiian false killer whales included the following members: (1) from PIFSC, Dr. Erin Oleson (Team Leader; population biology/acoustics), Dr. Donald Kobayashi (Hawaii and tropical ecology), Dr. Chris Boggs (fisheries/fish population assessment), (2) from SWFSC Dr. Karin Forney (conservation biology/fisheries interactions), and Dr. Barbara Taylor (genetics/risk assessment), (3) from NWFSC, Dr. Brad Hanson (habitat/foraging ecology/behavioral ecology), and Gina Ylitalo (contaminants), and (4) from AFSC, Dr. Paul Wade (risk modeling/population biology).

relevant biological and threat information not included herein, efforts being made to protect the species, and all relevant laws, regulations, and policies. The decision will be posted on the NMFS Web site (refer to: http://www.nmfs.noaa.gov/pr/species/) and announced in the *Federal Register*.

1.2. Assessing plausibility in the face of non-quantified uncertainty

The BRT adopted formal methods to express plausibility for use in guiding its analysis of Hawaiian insular false killer whales as a DPS and in assessing the risks to the population. The plausibility point methods were employed although other methods were employed also, as described later. These formal methods are important in a setting where quantitative measures of uncertainty derived from the empirical data are unavailable. This point method is often referred to as the "FEMAT" method because it is a variation of a method used by scientific teams evaluating options under the Northwest Forest Plan (Forest Ecosystem Management Assessment Team (FEMAT), 1993). Each BRT member was asked to distribute 10 plausibility points among the choices for a given decision, reflecting his or her opinion of how likely that choice or option correctly reflected the population status. If a BRT member was certain of a particular option, or felt it was the only plausible scenario, he or she could assign all 10 points to that option. A BRT member with less certainty about which option best reflected reality or best reflected the population's status could split the points among two or more options. This method has been used in all status review updates for anadromous Pacific salmonids since 1999, as well as in reviews of Southern Resident killer whales (SRKW) (Krahn et al., 2002; Krahn et al. 2004), West Coast rockfishes (Stout et al., 2002), Pacific herring (Stout et al., 2001), Pacific groundfish (Gustafson et al., 2000), North American green sturgeon (Adams et al., 2002; Adams et al., 2005), and black abalone (Butler et al., 2009). Alternative methods were occasionally used, such as when the central question was not the plausibility, but rather the degree of severity, geographic scope, or level of certainty of individual population threats. Treatment of different types of uncertainty is detailed in Section 4.3.

1.3. Key questions in ESA evaluations

1.3.1. The "species" question

For the purpose of the ESA, the term "species" includes

> "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."

A distinct population segment, or DPS, must be "discrete" from other populations and "significant" to the taxon (species or subspecies) to which it belongs. A DPS is discrete if it is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological or behavioral factors. Alternatively, the DPS may be discrete if it is delimited by international governmental boundaries within which are notable differences in management of the species or its habitat. If a population segment is considered discrete, NMFS must then consider whether the discrete segment is "significant" to the taxon to which it belongs. Significance may be measured as

persistence in a unique or unusual ecological setting, evidence that loss of the DPS would result in a significant gap in the range of the taxon, evidence that the discrete population segment represents the only surviving natural occurrence of a taxon within its historic range, or marked differentiation in its genetic characteristics. A population segment may include, but is not limited to, one of these criteria to be considered significant. This list of criteria is not exhaustive and other criteria relevant to the biology or ecology of the species may be used, as appropriate.

1.3.2. The "extinction risk" question

The ESA (Section 3) 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 shall be used in evaluating the level of risk faced by a species or a DPS. Important considerations include:

1) the present or threatened destruction, modification, or curtailment of its habitat or range;
2) overuse for commercial, recreational, scientific or educational purposes;
3) disease or predation;
4) the inadequacy of existing regulatory mechanisms; or
5) other natural or man-made factors affecting its continued existence.

According to the ESA, determining 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 BRT was not asked to review the adequacy of existing regulatory mechanisms as that determination is best made by the management portion of the agency.

1.4. Summary of information presented by the petitioner

1.4.1. The DPS question: "Discreteness"

The petitioner (Fallon, 2009) presented the following arguments that Hawaiian insular false killer whales are "discrete" from other population segments.

1.4.1.1. "The insular population of Hawaiian false killer whales is behaviorally unique from other false killer whales"

The petitioner stated that Hawaiian insular false killer whales are the only long-term island associated population of false killer whales in the world. The false killer whale is generally considered a wide-ranging pelagic species, not known to strictly associate with islands. Hawaiian insular animals reside close to shore and individuals have been re-sighted over a 21-year period (Baird et al., 2008a). Although false killer whales in other regions may approach islands, there is no evidence of long-term association. Re-sightings of individual false killer whales near islands off the coast of Costa Rica have been dismissed by the study's authors (Acevedo-Gutierrez et al., 1997) and the petitioner as pelagic animals that were only foraging near the islands.

1.4.1.2. "The insular population of Hawaiian false killer whales is genetically distinct from other false killer whales"

The petitioner stated that Hawaiian insular false killer whales are genetically distinct and demographically independent of other false killer whales based on the mitochondrial DNA haplotypes identified from false killer whales sampled near Hawaii, the eastern and western Pacific, western Atlantic and Indian Oceans (Chivers et al., 2007). The haplotypes found in Hawaiian insular animals are sufficiently divergent from all other haplotypes found in all other geographic areas to be statistically significant. This evidence suggests Hawaiian insular false killer whales have been isolated for some time and experience little maternal gene flow. Male-mediated gene flow could not be assessed at the time of the petition; however, the petitioner stated that differentiation based solely on matrilineal gene flow is sufficient given traits such as alloparenting and foraging strategies are thought to be transmitted matrilineally in many cetacean species. The nucleotide diversity of Hawaiian insular false killer whales is similar to other species with matrilineal social organization, such as pilot whales (*Globicephala macrorhynchus*), sperm whales (*Physeter macrocephalus*), and killer whales (*Orcinus orca*) (Chivers et al., 2007).

1.4.1.3. "The insular population of Hawaiian false killer whales constitutes a stock under the Marine Mammal Protection Act"

Hawaii insular false killer whales are classified as a stock under the Marine Mammal Protection Act (MMPA). Although the exact geographic boundary is unknown, as of the 2008 Stock Assessment Report, the 25–75 nmi longline exclusion boundary was used as the stock boundary (Carretta et al., 2009). Because NMFS follows the phylogeographic approach in classifying stocks, including an analysis of 1) distributional data, 2) population response data, 3) phenotypic data, and 4) genotypic data, the petitioner stated that classification of Hawaiian insular false killer whales as a stock supports the finding that the population is listable under the ESA.

1.4.2. The DPS question: "Significance"

The petitioner presented the following arguments to demonstrate that Hawaiian insular false killer whales are "significant" to the taxon to which they belong.

1.4.2.1. "The insular population of Hawaiian false killer whales occupies a unique ecological setting"

The isolation and biology of the Hawaiian Islands has resulted in high rates of endemism, demonstrating the potential for evolutionary isolation in this region. Although rare in cetaceans, the petitioner argued that a number of other cetacean species in Hawaiian waters also appears to be evolutionarily isolated from the surrounding pelagic waters. Other wide-ranging pelagic species have developed morphologically and genetically distinct populations near the Hawaiian Islands, including Bryde's whales (*Balaenoptera edeni*) and pilot whales (Chivers et al., 2007). Spinner dolphins (*Stenella longirostris*), rough-toothed dolphins (*Steno bredanensis*), and bottlenose dolphins (*Tursiops truncatus*) also show evidence of demographic independence among islands in the archipelago (Norris et al., 1994; Galver, 2002; Martien et al., 2005; Andrews et al., 2006.; Baird et al., 2008b). The petitioner quoted Baird et al. (2008a) who suggests that

increases in productivity immediately around the islands may encourage the evolution of island-associated populations of cetaceans. In addition, Hawaiian insular false killer whales are the only known island-associated population of this otherwise pelagic species, indicating that the Hawaiian population occurs in a setting unique to the taxon.

1.4.2.2. "The insular population of Hawaiian false killer whales differs markedly from other populations of the species in its genetic characteristics"

Similar to genetic arguments for discreteness, the petitioner stated that the mitochondrial haplotypes of the Hawaiian insular false killer whale are uniquely identifiable from false killer whale elsewhere in the eastern North Pacific (Chivers et al., 2007).

1.4.3. Abundance and population trends

The petitioner summarized the available abundance information for the insular population. At the time of the petition, the best estimate of population size was 123 animals (CV = 0.72) based on a mark-recapture analysis of photographic identification data (Baird et al., 2005). A previous estimate using aerial line transect data from 1993-1998 was 121 individuals (CV = 0.47) within 46 km of the main Hawaiian Islands (Mobley et al., 2000). The petitioner noted that Hawaiian insular false killer whales may have the smallest population size of any odontocete in Hawaii.

The petitioner summarized several lines of evidence indicating a decline of the insular population of Hawaii false killer whales. During an aerial survey conducted in 1989, three large groups of false killer whales (group sizes 380, 460, and 470) were observed on three different days (Reeves et al., 2009). All of these group sizes are larger than the current best estimate of Hawaiian insular false killer whale abundance. Further, when the 1989 observations are compared to encounter rates of false killer whales from aerial surveys between 1993 and 2003, further evidence for a decline is apparent; however, because the 1989 surveys were conducted with methods different from the series of surveys conducted by Mobley (Mobley et al., 2000; Mobley, 2004), it is difficult to statistically compare the results. The results of five aerial surveys (conducted by Mobley, et al.) from 1993 to 2003 show a downward trend in sighting rates of false killer whales, further supporting a decline of the insular population (Mobley et al., 2000; Baird, 2009). The petitioner cited the relatively high encounter rate during the 1989 survey (17% of sightings) compared to encounter rates during boat-based surveys from 2000 to 2006 (1.5%), as well as a decline in median group size (195 during the 1989 survey to 15 during boat-based surveys) as further evidence of the decline. Finally, the petitioner cited low resighting rates of individual false killer whales photo-identified in the 1980s and an unpublished analysis of mean annual survival rate (Baird and Barlow) to suggest a low survival rate in the 1990s relative to other long-lived cetacean species.

1.4.4. Risk factors

The petitioner asserted that Hawaiian insular false killer whales qualify as endangered under the ESA based on three factors: modification of habitat, inadequacy of existing regulatory mechanisms, and other natural or man-made factors. The petitioner cited mortality and serious injury in fishing gear, overfishing and prey reductions, potential for increased accumulation of toxic chemicals, impacts of ocean acidification, and the

potential for acoustic impacts as specific habitat-related threats. Inadequacies in Hawaii State law, the MMPA, and the Magnuson-Stevens Fishery Conservation and Management Act (MSA), as well as risks inherent to small populations and the synergistic and cumulative effects of other threats, were cited as specific threats to the population's survival.

1.4.4.1. Modification of habitat

Mortality/serious injury by fishing gear

The petitioner summarized the history of false killer whale-fishery interactions as reported by the NMFS Observer Program as well as evidence of fin disfigurements and assert that both direct and indirect evidence suggests that the insular population of Hawaii false killer whales experiences mortality and serious injury as a result of interactions with fisheries. Citing a recent study on bottlenose dolphins in Sarasota, Florida, which indicates that 2% of the population died in a single year as a result of ingesting fishing gear (Powell, 2009), the petitioner argued that incidental mortality caused by the Hawaii-based longline fleet[2], as well as potential interactions with troll and recreational fisheries, and the state-regulated shortline fishery, may present a significant risk to Hawaiian insular false killer whales. Although it is difficult to assess the relative proportions of insular and pelagic false killer whales that may be interacting with longline fisheries, fin disfigurements within the insular population, at a rate of approximately 4% (Baird and Gorgone, 2005), suggest that insular animals interact with fisheries.

Interactions with the pelagic population of Hawaiian false killer whales have been documented in fishery logbooks and by the NMFS Observer Program (Nitta and Henderson, 1993; Carretta et al., 2007), and false killer whales have been observed taking catches from longline and commercial and recreational trolling lines (Shallenberger 1981). Twenty-four false killer whales were observed seriously injured or killed between 1994 and 2005 by interactions with the longline fishery (Forney and Kobayashi, 2007). The average mortality/serious injury in the most recently reported 5-year period (2003–2007) was 7.4 animals per year (CV = 0.19) in the Hawaiian Exclusive Economic Zone (EEZ) (Carretta et al., 2009) for an estimated take of 37 false killer whales during that period. The petitioner cited the 2008 Stock Assessment Report as suggesting that some of these interactions may be occurring with the insular population.

Interactions between Hawaiian insular false killer whales and nearshore commercial and recreational fisheries have also been reported (Nitta and Henderson, 1993; Rhodes et al., 2007), including the Hawaii shortline fishery, although the extent of the interactions is not well known as no observer program is currently in place for these fisheries. Recently classified as a Category II fishery in the List of Fisheries, the shortline fishery has grown

[2] The Hawaii-based longline fishery operates as a limited entry program with approximately 130 vessels operated under active permits. In the annual List of Fisheries (LOF), the deep-set (tuna target) component of the longline fishery is designated as a Category I fishery, meaning that it experiences frequent incidental mortality and serious injury of marine mammals. 50 *CFR* § 229.2. The shallow-set (swordfish target) component is designated as a Category II fishery, in that it experiences occasional incidental mortality and serious injury of marine mammals. In the 2010 LOF, the state-regulated shortline and kaka-line fisheries were listed as Category II by analogy with gear type that is known to cause serious injury and mortality of marine mammals.

to 11 vessels as of 2008, and targets bigeye tuna (*Thunnus obesus*) and lustrous pomfret (*Eumegistus illustris*). Although there is no formal reporting system, anecdotal information suggests interactions have occurred off the north side of Maui and that the insular population may be impacted by deliberate shootings because of interactions with these small-scale fisheries (NMFS, 2009b).

Overfishing and prey reductions

The petitioner cited large-scale reduction in predatory fish populations worldwide (Baum et al., 2003; Baum et al., 2005; Sibert et al., 2006) and suggest local prey reductions may be impacting Hawaii insular false killer whales. Changes in prey populations near Hawaii, including the classification of bigeye tuna as overfished (NMFS, 2009a), a decline of yellowfin tuna (*T. albacares*) biomass (Sibert et al., 2006), a significant decline in the CPUE of yellowfin tuna in the Hawaiian troll fishery from 1987 to 2006, and reduced average body weight of mahimahi (*Corypahena* spp.) in the Hawaiian longline fishery since 1987 (WPFMC, 2006, originally cited as Anonymous 2006), may negatively influence false killer whale nutritional state.

Potential for increased levels of toxic chemicals

The petitioner stated that accumulation of persistent organic pollutants in tissues have been associated with reproductive impairment and immunosuppression in several species of marine mammals. False killer whales are at particular risk from organic pollutants because they feed at a high tropic level, increasing their exposure to organic pollutants, and they are long-lived, increasing their susceptibility to bioaccumulation. Three of nine sampled Hawaiian insular false killer whales showed levels of polychlorinated biphenyls (PCBs) above the safety recommendations for other species (Ylitalo et al., 2009). The petitioner also asserted that false killer whales are at risk of swallowing plastic debris, which has led to death in similar species (Stamper et al., 2006).

Ocean Acidification

The petitioner suggested that increasing atmospheric concentrations of CO_2 may endanger Hawaiian insular false killer whales as ocean acidification may decrease the forage base of false killer whale prey, and declining mid-water oxygen concentrations may increase the size of dead zones in the Pacific (Brewer and Peltzer, 2009) compressing favorable habitat for prey species that diurnally migrate to depth. Many species of game fish forage directly on mesopelagic nekton (Bertrand et al., 2002), such that reduction in shell-building capabilities of calcareous organisms including mollusks and crustaceans (Fabry et al., 2008) and impacts on squid may result in productivity declines that may impact primary predators such as false killer whales.

Potential for acoustic impacts on false killer whale behavior

The petitioner stated that noise-producing activities in the Hawaiian Islands, such as mid-frequency naval sonar, have the potential possibility of disrupting false killer whale behavior. Beaked whales exposed to mid-frequency sonar have been injured and killed, and a number of other species have stranded, been displaced from their habitat, and exhibited disrupted dive and vocalization patterns when exposed to naval sonar (Nowacek et al., 2004; Weilgart, 2007; Brownell et al., 2009). The U.S. Navy Hawaii

Range Complex encompasses most of the known range of the insular Hawaiian population of false killer whales and employs sonar from the unit-level to multi-strike group exercises (U.S. Navy, 2008).

1.4.4.2. Inadequacy of existing regulatory mechanisms

State law

The petitioner asserted that State of Hawaii statutes and regulations intended to protect marine mammals are limited in their applicability and have not proven effective in conserving the insular population. Hawaii Revised Statutes §195D and Hawaii Administrative Rules §§13–124 prohibit the take of endangered and threatened species and any species indigenous to Hawaii. Because Hawaiian insular false killer whales are not currently listed as threatened or endangered by any federal or state law, and because the State does not recognize the insular population as a distinct biological unit, these laws do not afford any protection to the population. The petitioner also cited a NOAA General Counsel 2008 legal opinion that these State statutes are preempted by MMPA and, therefore, to the extent that they apply to marine mammals, are unenforceable by Hawaii (Luxton, 2008).

Further, Hawaii does not monitor the bycatch of marine mammals in any of its state fisheries. As described previously, a high rate of fin disfigurements (Baird and Gorgone, 2005) and other observations suggest interactions between longline fisheries and insular false killer whales. Hawaii State Law allows the Department of Land and Natural Resources (DLNR) to adopt rules consistent with the MSA to correct overharvesting in fisheries occurring within federal and state waters. However, this discretion has not prevented the apparent decline in population or body size of the false killer whale prey species (Sibert et al., 2006).

Federal law

MMPA—The petitioner asserted that MMPA has not proven adequate to protect Hawaiian insular false killer whales because it has inadequately protected the population from bycatch and other fisheries interactions, and from other threats including overfishing, toxic contamination, and direct shootings of animals by local fishers. NMFS does not currently recognize the insular population as a “strategic stock” under MMPA, despite the known overlap of fisheries with the insular population’s range. The petitioner argued that although the shallow-set longline fishery maintains 100% observer coverage, other commercial fisheries, including the deep-set fishery (20% observer coverage) and the shortline fishery (no observer program) are not adequately observed. The 2008 Stock Assessment Report sets a limit on potential biological removal at less than one animal per year (Carretta et al., 2009). However, incidental takes occurring within the longline fisheries are not assigned to this stock, so the stock is not strategic and, therefore, has not benefited from a take-reduction plan for any of the Hawaii fisheries. The petitioner further argued that even if afforded strategic status under MMPA, the population would not be adequately protected because NMFS did not convene a take-reduction team for the pre-2008 combined pelagic and insular stock. Finally, the strategic status for the insular stock may not improve observer coverage of the shortline and deep-set longline fisheries

because NMFS first allocates funds to observer programs covering species listed as endangered or threatened under ESA.

MSA—The petitioner stated the MSA, the leading federal statute governing marine fisheries in U.S waters, is ineffective in protecting Hawaiian insular false killer whales because the Act does not apply in Hawaii State waters, the provisions applying to longline fisheries interaction with protected species have not proven effective to reduce hooking or entanglement of false killer whales, and the Act has not been successful in preventing the depletion of fisheries. Longline vessels fish within a portion of the insular population's range for 4 months each year. High rates of dorsal fin disfigurements suggest interactions with this fishery may be occurring and, as such, the MSA is not providing the insular population with sufficient regulatory protections to prevent mortality and serious injury. Further, the MSA had failed to arrest the declines in population or body size of the primary prey species of insular false killer whales.

Foreign and international law

The petitioner noted that Hawaiian insular false killer whales occur entirely within the EEZ of the United States, and no other nations have jurisdiction nor have adopted laws to protect this population. No international conventions exist that address threats to this population and because it does not predictably occur outside of U.S waters, the Convention on Migratory Species does not apply. Further, the International Whaling Commission management provisions do not apply to this species. False killer whales are listed generically under Appendix II of the Convention of International Trade in Endangered Species of Wild Fauna and Flora and international trade in false killer whales does not constitute a threat to the insular population.

1.4.4.3. Other factors

Risks inherent to small populations

The petitioner cited Purvis et al. (2000) in describing the four most important risk factors for extinction as 1) high trophic level, 2) low population density, 3) slow life history, and 4) small geographic size. The petitioner noted that the insular population of Hawaiian false killer whales does meet all of these risk factors given the population feeds primarily on large pelagic fishes (Baird et al., 2008a), has low population density (Baird et al., 2005), is slow to mature with long calving interval of approximately 7 years (Stacey et al., 1994), and has a range limited to the main Hawaiian Islands (Baird et al., 2010).

The petitioner also discussed other threats to small populations including demographic and environmental stochasticity, risks of local catastrophes, slow adaptation rates, deleterious effects of inbreeding, and "mutational meltdown" or the expression of harmful alleles (Franklin, 1980). Island endemics may also be subject to the effects of overexploitation or introduced species, as these populations may have small population sizes and ranges and may have evolved in isolation from a diversity of competitors and predators. Finally, of particular concern for low-density animals with low reproductive rates is the potential for depensation (Allee effects) or the reduced ability of individuals to find mates, causing a decline in per capita reproduction. This may be a factor in limiting the recovery of other depleted cetacean species (e.g., Whitehead et al., 2000)

Synergistic and cumulative effects
The petitioner noted that the potential cumulative and synergistic impacts of all of the aforementioned threats must be considered. These cumulative and synergistic effects arise when the impacts of multiple stressors are greater than the sum of the stressors considered in isolation. Increased exposure to toxic contaminants, when combined with Allee effects or a reduction in the prey base may affect long-term abundance or result in a lower survival rate. Specifically, a reduction in the prey base may cause nutritional stress to the animals, resulting in the mobilization of stored fat. If this fat has high levels of contaminants, metabolizing this fat could cause acute health problems (e.g., Colbern and Smolen, 1996).

1.5. Treatment of data and arguments presented in the petition

The data and arguments presented in the petition (Fallon, 2009) were considered by the BRT along with all other public comments submitted in response to the 90-day finding (75 *FR* 316; January 5, 2010) requesting new scientific and commercial data. The petition and its arguments are occasionally referenced throughout this Status Review; however, the Status Review will not respond to each of the petitioner's arguments in turn.

1.5.1. Factors not considered by the BRT

Delineation of a population as a "stock" under MMPA does not satisfy the requirements for the designation of that population as a DPS. The definition of "stock" under the MMPA is "a group of marine mammals of the same species or smaller taxa in a common spatial arrangement that interbreed when mature." Further guidance on stocks clearly indicates that stocks should be management units. In contrast, one goal of the ESA is to conserve species that are important from an evolutionary standpoint. Thus, a stock defined under the MMPA, which fit the criteria for a DPS could be a classified as a DPS; however, MMPA stocks do not automatically equate to ESA DPSs. Thus, the fact that the Hawaiian insular false killer whales have been designated a "stock" under the MMPA has no direct bearing on whether the population should qualify as a DPS. Although the stock is not automatically a DPS, the information used to assess stock structure under MMPA will also be used, along with additional data, to assess whether Hawaiian insular false killer whale qualify as a DPS under the ESA.

2. Background Information on the Biology and Habitat of False Killer Whales

2.1. General false killer whale biology

2.1.1. Identifying characteristics

2.1.1.1. Size and shape

The false killer whale is a slender, large delphinid, with maximum reported sizes of 610 cm for males (Leatherwood and Reeves, 1983) and 506 cm for females (Perrin and Reilly, 1984). Length at birth has been reported to range from 160 cm to 190 cm, and length at sexual maturity is 334–427 cm in females and 396–457 cm in males (Stacey et al., 1994; Odell and McClune, 1999) . Estimated age at sexual maturity is about 8–11 years for females, while males may mature 8–10 years later (Kasuya, 1986). The maximum reported age has been estimated as 63 years for females and 58 years for males (Kasuya, 1986). Both sexes grow 40–50% in body length during their first year of life, but males subsequently grow faster. Growth ceases between 20 and 30 years of age, and there is evidence of geographic variation in asymptotic body length. Off the coast of Japan, asymptotic length is 46 cm (females) and 56 cm (males) longer than off the coast of South Africa (Ferreira, 2008). Large individuals may weigh up to 1400 kg. Coloration of the entire body is black or dark gray, although lighter areas may occur ventrally between the flippers or on the sides of the head. A prominent, falcate dorsal fin is located at about the midpoint of the back, and the tip can be pointed or rounded. The head lacks a distinct beak, and the melon tapers gradually from the area of the blowhole to a rounded tip. In males, the melon extends slightly further forward than in females. The pectoral fins have a unique shape among the cetaceans, with a distinct central hump creating an S-shaped leading edge.

2.1.1.2. Internal anatomy—skeleton

The skull of the false killer whale is characterized by a short, broad rostrum, with a length that is at least 1.5 times the width. Mean condylobasal length was reported to be about 59 cm based on 99 individuals from the North Atlantic, as summarized in Odell and McClune (1999). False killer whales generally have 7–10 conical teeth in each side of the upper jaw and 8–10 teeth in each side of the lower jaw. Wear patterns on teeth in older animals indicate that false killer whales can make backwards and lateral jaw movements to break up large prey (Ross, 1984). The postcranial skeleton most commonly has 48–50 vertebrae, but as few as 47 and as many as 52 have been reported (Stacey et al., 1994). There are 9–12 pairs of ribs, four of which attach directly to the sternum. The phalanges of the flippers show a wide degree of variation in structure. Facial structure, skull symmetry, and skew are similar to those reported for other delphinids, with the greatest similarities to bottlenose dolphins (Mead, 1975).

2.1.1.3. Internal anatomy—organs

Most information on internal organs has been obtained from drive fisheries off the coast of Japan and from mass strandings in the North Atlantic and in South Africa (Kasuya, 1986; Odell and McClune, 1999; Ferreira, 2008). Blood parameters from stranded individuals and from animals under human care in oceanaria or research facilities are similar to those measured for other small cetaceans (Stacey et al., 1994; Odell and

McClune, 1999). The musculature and soft tissues of the head are involved in sound production, but there are differences in the amount of connective tissue anterior to the melon in males and females (Mead, 1975). The diploid chromosome count for false killer whales ($2n = 44$) is typical for cetaceans. At sexual maturity, male testes weigh 1.0–1.7 kg, with maximum testes mass reported as 8.2 kg (Odell et al., 1980). Females ovulate spontaneously one or more times per year (Stacey et al., 1994), and calving in tropical waters may occur year-round. Ovulation rates decrease with age, and females older than 44 years are thought to be post-reproductive (Kasuya, 1986; Ferreira, 2008). The proportion of females pregnant each year has been estimated as 14–21% (Perrin and Reilly, 1984; Kasuya, 1986).

2.1.2. Taxonomy—Genus and species: Pseudorca crassidens (Owens, 1846)

The false killer whale, *Pseudorca crassidens* (Owen, 1846), is considered a single species with no proposed subspecies. Kitchener et al.(1990) found considerable morphological divergence in skull morphology among groups of animals stranded in Scotland, on the Atlantic side of South Africa, and in Australia. In the most distinct group, from Scotland, adults had generally larger skulls. A covariate analysis had a correct classification rate of 76% to specimen origin. The groups also exhibited different somatic growth rates and possibly differences in the extent of sexual dimorphism. The authors made no taxonomic judgments but suggested the differences could prove sufficient to warrant the recognition of the southern populations as distinct from the Scottish population, and possibly also recognition of separate African and Australian populations. They did not define what was meant by “population”.

Cetacean taxonomy has traditionally been hindered by the inadequacy of samples of skulls and this problem has applied particularly to the taxonomy of pelagic species, such as the false killer whale (Reeves et al., 2004). Biopsies are now often acquired from free-ranging animals, but obtaining a set of samples representative of the global distribution, which is ideal for addressing taxonomic questions, remains a distant goal for many species. A recent example of the changing state of cetacean taxonomy is the announcement of several new species and/or subspecies of killer whales based on genetic data (Morin et al., 2010). Settling the full taxonomy of killer whales may take many decades because sampling of truly pelagic killer whales is difficult. However, for certain regions, multiple lines of evidence based on behavioral, morphological and genetic factors have been accumulating to identify several new species and/or subspecies.

No global comparison to examine taxonomic questions has been conducted using genetic data for false killer whales because of the limitations of sample distribution. For a species like the false killer whale, with its distribution restricted to tropical and temperate waters and with strong social structure, the state of their taxonomy below the species level remains uncertain. A 2004 workshop that was convened to consider cetacean taxonomy rated false killer whales as having a medium level of taxonomic uncertainty (Reeves et al., 2004).

2.1.3. Global distribution

False killer whales are found in all tropical and warm-temperate oceans (Figure 2-1), generally in deep offshore waters but also in some shallower semi-enclosed seas and gulfs (e.g., Sea of Japan, Yellow Sea, Persian Gulf) and near oceanic islands (e.g., Hawaii, Johnston Atoll, Galapagos, Guadaloupe, Martinique) (Leatherwood et al., 1989). Sightings have also been reported as "common" in Brazilian shelf waters (IWC, 2007) where animals could be seen from shore from Rio de Janeiro feeding in an upwelling site that concentrates prey. There are occasional records in both the northern and southern hemispheres of animals at latitudes as high as about 50° (Stacey and Baird, 1991; Stacey et al., 1994). In the western Pacific off the coast of Japan, false killer whales appear to move north-south seasonally, presumably related to prey distribution (Kasuya, 1971), but seasonal movements have not been documented elsewhere. The distribution depicted in Figure 2-1 does not reflect differences between areas where false killer whales are commonly found and areas where false killer whales occur on a rare basis (extralimital occurrences). For example, sightings of false killer whales along the west coast of the United States and Canada are rare, and this region is not primary false killer whale habitat. A range map showing the false killer whale's primary distribution together with effort information would be a useful contribution to furthering the understanding of false killer whale habitat.

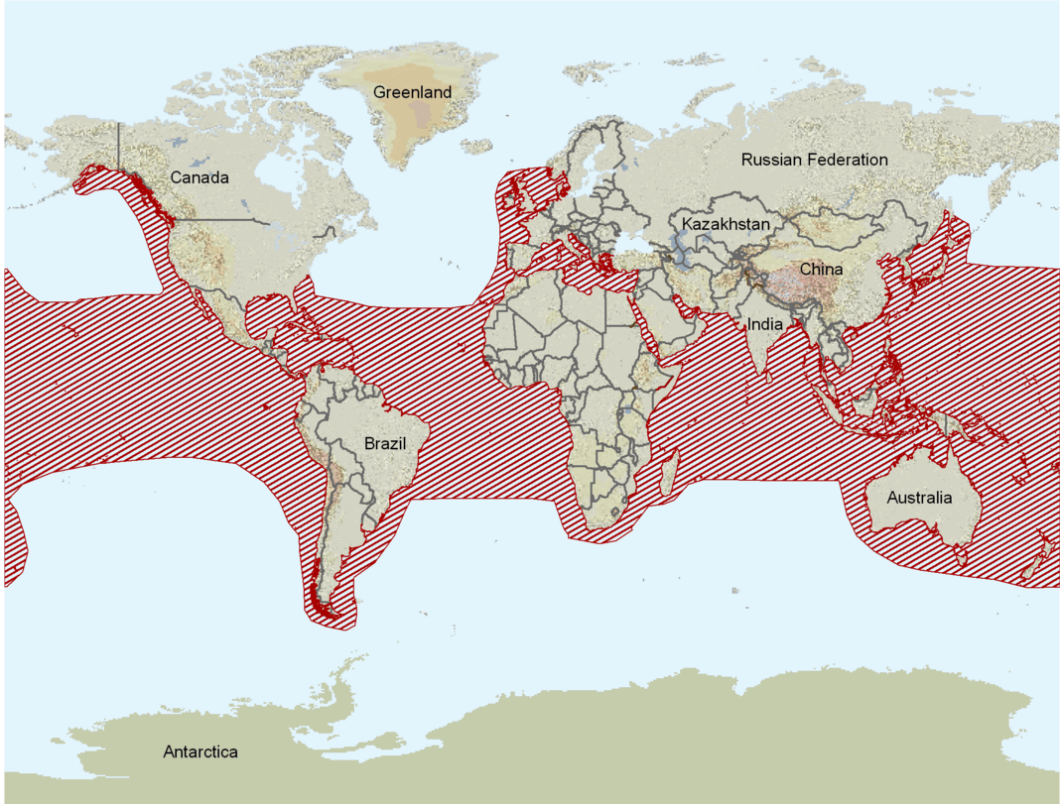


Figure 2-1. Approximate global distribution of false killer whales (Source: IUCN Red List of Threatened Species, Taylor et al., 2008), including areas of regular occurrence and extralimital records.

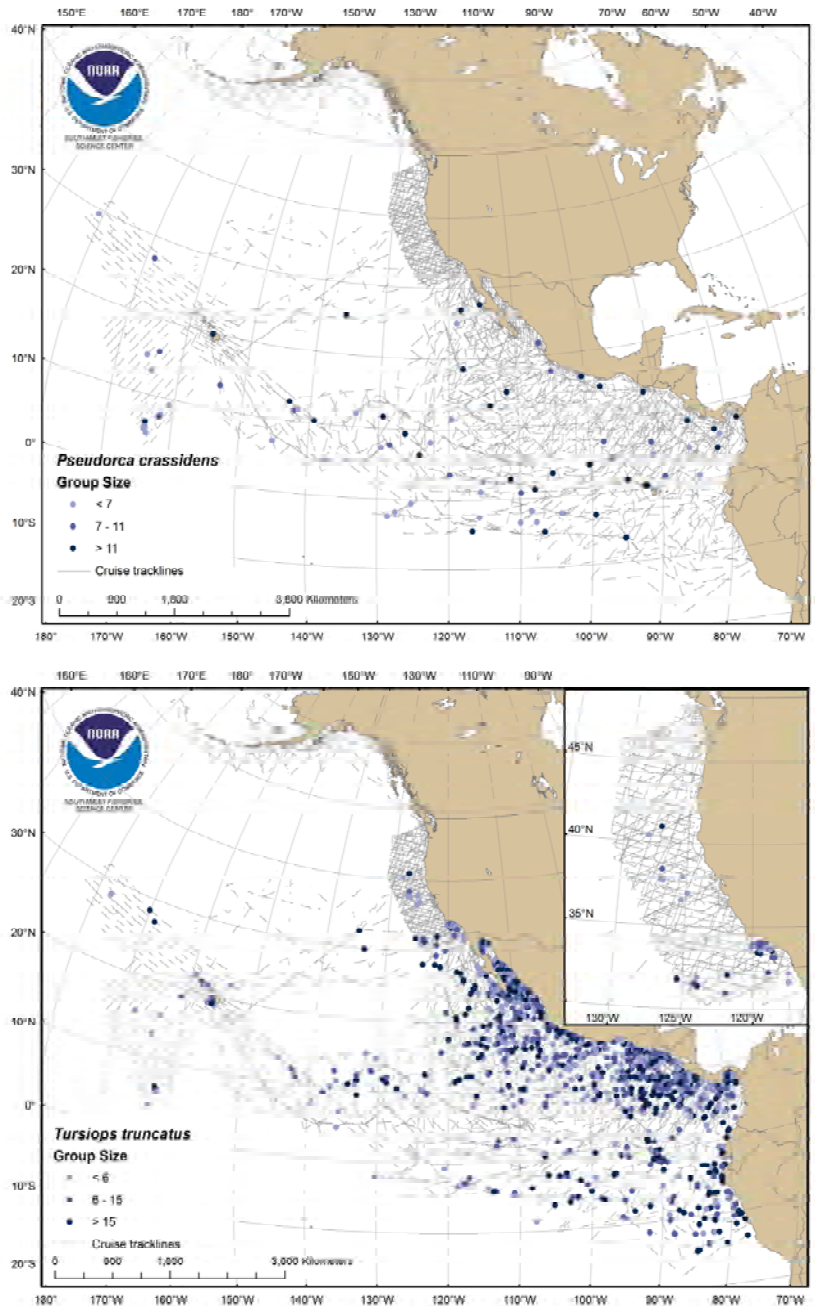


Figure 2-2. Sighting locations and cruise tracks for false killer whales (top) and bottlenose dolphins (bottom) during Cetacean and Ecosystem Assessment Research Cruises conducted by NOAA's Southwest Fisheries Science Center, 1986–2005 (Source: Hamilton et al., 2009).

Estimates of false killer whale abundance and density are available for the eastern tropical Pacific (Wade and Gerrodette, 1993), waters surrounding the Hawaiian Archipelago (Barlow, 2006), and in waters between Hawaii, Palmyra Atoll and Johnston Atoll (Carretta et al., 2007). From these studies, densities in the central and eastern Pacific range from 0.02 to 0.38 animals per 100 km^2 (Wade and Gerrodette, 1993; Mobley et al., 2000; Ferguson and Barlow, 2003; Carretta et al., 2007), with the lowest

densities reported for waters north of about 15°N off Baja California, Mexico, and within the U.S. EEZ of Hawaii, and highest densities reported in waters surrounding Palmyra Atoll. Unlike other species that can be found both along continental margins and in offshore pelagic waters (e.g., bottlenose dolphins), false killer whales densities do not appear to increase closer to coastlines (Figure 2-2).

Although false killer whales are found globally, genetic, morphometric and life history differences indicate there are distinct regional populations (Kitchener et al., 1990; Mobley et al., 2000; Chivers et al., 2007; Ferreira, 2008). Within waters of the central Pacific, three stocks of false killer whale are currently recognized for management under the U.S. MMPA: a Hawaiian pelagic stock, a Hawaiian insular stock, and a Palmyra Atoll stock (Carretta et al., 2010).

2.1.4. Population dynamics

False killer whales are long-lived social odontocetes. Much of what is known about life history comes either from examination of dead animals, originating from drive fisheries in Japan (Kasuya and Marsh, 1984; Kasuya, 1986) or strandings (Purves and Pilleri, 1978; Ferreira, 2008). Ferreira (2008) compared the large samples sizes from the Japanese drive fisheries and a stranding event on the Atlantic coast of South Africa and found the following general trends: females were about 84% of the length of males and had reached maturity at 9.2 years; males between the ages of about 8 and 18 were not found in the groups sampled, indicating dispersal of subadult males until they reach full sexual maturity; both males and females stopped growing between 25 and 30 years of age. The oldest estimated age (based on growth layers in teeth) was 63 for females and 59 for males, with females becoming reproductively senescencent at about age 44. The sex ratio was biased towards females at about 0.63. The social system has been described as matrilineal (Ferreira, 2008). However, this is not consistent with two factors: 1) males leaving their natal group when they begin to become sexually mature and 2) finding females within a single group with different haplotypes indicating that even among females, groups are composed of more than near-relatives (Chivers et al., 2010). (Ferreira, 2008) suggested the mating system may be polygynous based on the large testes size of males, but actual understanding of the mating system remains poor.

The only reported birth interval, 6.9 years between calves, is from Japan (Kasuya, 1986). However, annual pregnancy rates were reported for Japan as 11.4% and 2.19% for South Africa (Ferreira, 2008). A rough inter-birth interval can be calculated by taking the inverse of the annual pregnancy rate, which would yield an 8.8 year interval and a 4.5-year interval for Japan and South Africa respectively. Only 1 of 37 adult females was pregnant in the South African stranding suggesting that this group may be insufficient to estimate pregnancy rates.

Comparisons of the life history parameters inferred from the Japanese drive fishery samples and the South African stranding sample indicated that the whales in Japan attained a larger asymptotic body size and grew faster. Also, a suite of characteristics of the whales in Japan indicated a higher reproductive rate: the ratio of reproductive to post-reproductive females was higher and the pregnancy rate was higher than in South

Africa. Possible reasons given by Ferreira (2008) for the apparently higher reproductive rate in Japan are: 1) the Japan whales are exhibiting a density-dependent response to population reduction as a result of exploitation, 2) the colder waters near Japan are more productive, or 3) they are eating yellowtail (*Seriola quinqueradiata*) (Kasuya, 1985) as opposed to the South African whales that eat primarily cephalopods (Best, 2007).

The estimated reproductive rates in both Japan and South Africa are low compared to those of other delphinids and especially to the two species with the most similar life history (Table 2-1).

Table 2-1 Comparative life history parameters where AFR = age at first reproduction and IBI = inter-birth interval (Taylor et al., 2006, except the false killer whale IBI calculated above).

Species	AFR	IBI	Oldest Mother	Oldest Female	Generation Length
False killer whale	10.7	8.8+	44	63	25[3]
Killer whales	14.5	5	41	75	26
Short-finned pilot whale	11	6.9	43	60	25
Sperm whale	12	5	59	75	32

Reproductive senescence is quite rare in cetaceans but has been documented in false killer whales and other social odontocetes (Table 2-1). Ferreira (2008) devoted a chapter to full examination of the reproductive tracts of all the South African females and concluded that there was good evidence for reproductive senescence. The two primary reasons given for reproductive senescence are increasing survival of offspring as a result of care given by multiple females of multiple generations (grandmothering) and transmission of learning across generations allowing survival in lean periods by remembering alternative feeding areas or strategies (McAuliffe and Whitehead, 2005; Ferreira, 2008).

Wade and Reeves (2010) argue that odontocetes are more vulnerable to exploitation than mysticetes and have delayed recovery when numbers are reduced because of the combination of their life history, which results in exceptionally low maximum population growth rates, and the potential for social disruption. Particularly if older females are lost, it may take decades to rebuild the knowledge required to achieve maximum population growth rates. They give numerous examples from both cetaceans (beluga whales—*Delphinapterus leucas*, killer whales and sperm whales are particularly pertinent) and elephants, which are similarly long-lived social animals with reproductive senescence.

There is quite a bit of variance in estimates of group size of false killer whales. At least some of the variability stems from estimation methods and time spent making the group size estimate. Most group sizes estimated from boats or planes average from 20 to 30, and groups size estimates increase with encounter duration up to 2 hours (Baird et al., 2008a). The sizes of stranded groups are 4–5 times higher (Ferreira, 2008). It is possible

[3] Estimated from short-finned pilot whales. They have a very similar life history and a better estimate for interbirth interval.

that the groups seen on typical surveys are only part of a larger group spread over many miles (e.g., Baird et al., 2010) that are in acoustic contact with one another. The strandings may represent these larger aggregations.

2.1.5. Feeding ecology and food requirements

2.1.5.1. Feeding ecology

False killer whales are top predators, eating primarily fish (Peacock et al., 1936; Scheffer and Slipp, 1948; Bullis and Moore, 1956; Tsutsumi et al., 1961; Schevill, 1965; Brown et al., 1966; Shallenberger, 1981; Silas et al., 1984; Kasuya, 1985; Evans and Awbrey, 1986; Baird et al., 1989; Baird, 2009) and squid (Deraniyagala, 1945; Bullis and Moore, 1956; Ross, 1984; Baird et al., 1989; Cagnolaro et al., 2002; Hernandez-Garcia, 2002) but also occasionally taking marine mammals (Perryman and Foster, 1980; Hoyt, 1983; Palacios and Mate, 1996; Rinaldi et al., 2007). These impressions are based on relatively limited data from various parts of the species extensive range. The data include both stomach contents from stranded animals and observations of feeding by free-ranging whales, and the sources may have differing biases. In Japan it was noted that prey shifted seasonally from mackerel to squid (Tsutsumi et al., 1961).

The large, dispersed groups in which false killer whales typically occur (Baird et al., 2008a; Baird et al., 2010), and their patchily distributed prey suggest that this species forages cooperatively. Further evidence for the social nature of false killer whale foraging is the observation of prey sharing among individuals in the group (Connor and Norris, 1982; Baird et al., 2008a). False killer whale feed both during the day and at night (Evans and Awbrey, 1986; Baird et al., 2008a).

2.1.5.2. Energetic needs

Several evaluations of energy needs have been conducted for captive false killer whales. An initial estimate of 4.7% of body weight per day was made by Sergeant (1969). However, it was noted by Van Dyke and Ridgway (1977) that a 454 kg adult consumed 40 kg/day and a 353 kg/juvenile consumed 50 kg/day. Kastelein et al. (2000) reported estimates of daily consumption rates of 2.9% to 6.1% of body weight and Baird et al. (2009) reported values of 3.0 to 4.2%.

2.1.6. Diving behavior

Limited information is available on the diving behavior of false killer whales. Maximum dive depth was estimated at 500 m (Cummings and Fish, 1971). Time depth recorders have been deployed on four false killer whales (R. Baird et al., pers. comm.) totaling approximately 44 hours. The deepest dive recorded during a 22-hour deployment, was estimated to have been as deep as 700 m (estimate based on duration past the recorders 234-m limit and ascent and descent rates). However, only seven dives were to depths greater than 150 m, all of them accomplished in the daytime. Other day dives ranged from 5–20 m and lasted for a minute or less. Nighttime dives were all shallow (30–40 m maximum), but relatively lengthy (approximately 6–7 minutes).

Indirect evidence of dive depths by false killer whales can be inferred from prey. Mahimahi has been noted as a prominent prey item (Baird, 2009). Based on the catch

rates of longlines instrumented with depth sensors and capture timers (Boggs, 1992) in the daytime, mahimahi are caught closer to the surface than other longline caught fish, primarily in the upper 100 m. Other prey species, such as bigeye tuna, typically occur much deeper, down to at least 400 m (Boggs, 1992). The deepest dives by the instrumented false killer whales approach the daytime swimming depth of swordfish (*Xiphias gladius*) near 700 m (Carey and Robinson, 1981).

2.1.7. Social behavior

2.1.7.1. Pod structure

False killer whales are most commonly observed in groups of about 10 to 20 animals (Wade and Gerrodette, 1993; Baird, 2009; Baird et al., 2010), but they can also be found in widespread aggregations of small groups, totaling hundreds of individuals (Wade and Gerrodette, 1993; Carretta et al., 2007; Baird, 2009; Reeves et al., 2009). These large aggregations can be spread over tens of kilometers yet appear to have coordinated movement directions (Baird et al., 2008a). Mass strandings of large groups of false killer whales (range 50 to 835; mean = 180) have been documented in many regions, including New Zealand, Australia, South Africa, the eastern and western North Atlantic, and Argentina (Ross, 1984). Groups of 2–201 individuals (mean = 99) have also been driven ashore in Japanese drive-fisheries (Kasuya, 1986). Analysis of age, sex, and maturity status from these mass mortality events indicates that these large groups include about equal numbers of males and females of various sizes (Odell and McClune, 1999). The social organization of smaller groups has been studied most extensively in false killer whales near the main Hawaiian Islands (Baird et al., 2008a), where individuals are known to form strong long-term bonds. False killer whales are also known to associate with other cetacean species, especially bottlenose dolphins (Leatherwood et al., 1988). Interestingly, records also show false killer whales attacking other cetaceans, including sperm whales and bottlenose dolphins (Palacios and Mate, 1996; Acevedo-Gutierrez et al., 1997).

2.1.7.2. Breeding

Little is known about the breeding behavior of false killer whales in the wild, but some information is available from false killer whales held in oceanaria (Brown et al., 1966). Gestation has been estimated to last 11–16 months, and females off the coast of Japan are reported to give birth on average every 6.9 years (Kasuya, 1986; Odell and McClune, 1999). Females with calves lactate for 18 to 24 months (Perrin and Reilly, 1984). In captive settings, false killer whales have mated with other delphinids, including short-finned pilot whales and bottlenose dolphins. Bottlenose dolphins in captivity have produced viable hybrid offspring with false killer whales (Odell and McClune, 1999).

2.2. Oceanographic environment of the tropical and warm temperate Pacific and insular waters of the Hawaiian Archipelago

2.2.1. General features

2.2.1.1. Hawaiian Islands

The Hawaiian Islands are located at the northern extent of Polynesia and are made up of 137 islands, islets, and coral atolls. The exposed islands are part of an undersea mountain range known as the Hawaiian-Emperor Seamount Chain, which was formed by a hot spot

beneath the Pacific Plate. Such long island chains are formed when the plate moves over a stationary hot spot. The hot spot erupts on the ocean floor creating seamounts that can eventually reach the ocean surface as volcanic islands. The Pacific Plate moves at ~ 10 cm/year and, over geologic time, forms islands in a chain as the volcano reaches the surface of the ocean. The Hawaiian Islands themselves extend for nearly 2400 km from Kure Atoll in the northwest to the island of Hawaii in the southeast. The Hawaiian Islands are often grouped into the Northwestern Hawaiian Islands (Nihoa to Kure) and the main Hawaiian Islands (Hawaii to Niihau). The total land area of the 19 primary islands and atolls is approximately 16,600 km^2, and more than 75% of the 1.3 million humans in Hawaii live on the island of Oahu. Three of the Pacific Remote Island Areas (PRIAs: Johnston Atoll, Palmyra, and Kingman Reef), areas of land and water under U.S. jurisdiction but separate from Hawaii, are potentially of interest in this report and are briefly mentioned here.

2.2.1.2. Johnston Atoll

Johnston Atoll is located at 16° 44' N latitude and 169° 31' W longitude and is approximately 1300 km southwest of Honolulu. French Frigate Shoals in the NWHI is the nearest land mass (~ 830 km to the northwest) and, because of its proximity to the Hawaiian Islands, there is believed to be genetic and larval connectivity between Johnston Atoll and the Hawaiian Islands. Johnston Atoll is an egg-shaped coral reef and lagoon complex residing on a relatively flat, shallow platform approximately 34 km in circumference (205 km^2). Johnston Atoll comprises four small islands totaling 2.8 km^2. Johnston Island, the largest and main island, is natural in origin but has been enlarged by dredge-and-fill operations. Sand Island is composed of a naturally formed island (eastern portion) connected by a narrow, man-made causeway to a dredged coral island (western portion). The remaining two islands, North Island and East Island, are completely man-made from dredged coral. Currently, the U.S. Fish and Wildlife Service (USFWS) manages Johnston Atoll as a National Wildlife Refuge. Recreational fishing occurs within the refuge. The Coral Reef Ecosystems Fishery Management Plan/Fishery Ecosystem Plan (FMP/FEP), the current regulatory framework with jurisdiction over this region, established a low-use marine protected area (MPA) from 0 to 50 fathoms around Johnston Atoll.

2.2.1.3. Palmyra Atoll and *Kingman Reef*

Palmyra Atoll comprises approximately 52 islets surrounding 3 central lagoons. This low-lying coral atoll system is approximately 1950 km south of Honolulu and is located at 5° 53' N latitude and 162° 05' W longitude. Palmyra Atoll and Kingman Reef occur at the northern end of the Line Islands Archipelago, which is situated halfway between Hawaii and American Samoa. Palmyra Atoll is located in the Intertropical Convergence Zone (ITCZ), an area of high rainfall. Palmyra Atoll is surrounded by extensive reef flats on all sides. Palmyra Atoll has a higher diversity of corals, anemones, and fishes than other Pacific Remote Islands because it is located within the eastward flowing Equatorial Counter Current which flows from areas in the western Pacific with high levels of biodiversity (Brainard et al., 2005). Palmyra Atoll is managed cooperatively by the USFWS and the Nature Conservancy as a nature preserve with limited recreational fishing (e.g., flyfishing for bonefish). The USFWS administers the atoll as a National

Wildlife Refuge. The Coral Reef Ecosystems FMP/FEP established a low-use MPA from 0 to 50 fathoms around Palmyra Atoll.

Kingman Reef, which is located 61 km northwest of Palmyra Atoll at 6° 23' N and 162° 24' W, is a series of fringing reefs around a central lagoon. Kingman Reef does not have any emergent islets that support vegetation. The USFWS administers the reef area as a National Wildlife Refuge. The Coral Reef Ecosystems FMP/FEP established a no-take MPA from 0 to 50 fathoms around Kingman Reef.

2.2.2. Environmental history

The Hawaiian Islands were colonized by humans in 300–800 A.D. by Polynesians and, by all accounts, shoreline foraging and nearshore fishing were common practices as seafood was one of the dietary mainstays (Titcomb, 1972; Titcomb et al., 1978). There is very little information on the condition of marine resources during this time (Dye and Graham, 2004), but the *kapu* system (strict laws punishable by death), the *ahupua'a* system (community stewardship of resources), and the extensive array of fish ponds for raising food, among other important cultural practices, could have been effective measures for promoting sustainable use of wild and captive marine resources in premodern Hawaii. It should be noted, however, that many endemic species such as birds were driven to extinction as a result of collecting, habitat loss, and invasive species. The native Hawaiian human population was nearly 1 million (Stannard, 1989) prior to the islands being discovered by western civilization. Following the 1778 landing by Captain Cook in Hawaii, the human population decreased substantially over the next hundred years as a result of introduced diseases, and then the population slowly increased with extensive European and Asian colonization. From 1900 to present, approximately a tenfold increase has occurred in the population. The large size of the human population and the cultural loss of traditional conservation practices led to substantial declines in marine resources during the 1800s. Commercial fishing for pelagic species increased rapidly in the late 1800s and early 1900s because of Japanese immigration (Kuykendall, 1967). While there is much uncertainty regarding the condition of marine resources prior to 1900, evident declines in benthic species and coastal pelagic species occurred after this time based on catch trends (Shomura, 1987). By the time statehood occurred in 1959, several of Hawaii's fisheries were characterized as "dying" (Pooley, 1993). Presently, most fisheries have some form of management, including regulations such as area closures, closed seasons, and size limits, and most Federally regulated fisheries have quotas or harvest guidelines, limited entry, gear restrictions, etc., and a few fisheries are entirely closed such as the lobster and bottomfish fisheries in the Northwestern Hawaiian Islands. Most coastal marine resources are thought to be in less-than-pristine condition, particularly in the more populous areas of the lower islands. However, catastrophic collapses or local extirpations have not been widely documented in the marine environment in contrast to the many terrestrial extinctions (Loope et al., 1988). It should be noted that cetaceans did not appear to be a significant target or dietary component in traditional Hawaiian culture (Titcomb, 1972; Akimichi, 1992).

2.2.3. Geological and climatic history

The climatic pattern, which affects the distribution of terrestrial and aquatic communities throughout the Hawaiian Islands and most of the adjacent Pacific, is largely driven by the prevailing trade winds. Wind-born weather fronts lose some of their moisture passing over the mountainous portions of islands in this region. Thus, windward (northeastern) slopes tend to have higher rainfall than leeward (southwestern) slopes. Windward areas tend to have more wave action and embayments with estuarine development than leeward areas, while leeward areas generally tend to be calmer, dryer, and warmer. The balance between the degree of protection from wind and waves, the amount of rainfall and sedimentation, and the availability of shallow shelf influences the extent of reef development and types of ecosystems in windward and leeward areas. There are few stream-fed estuaries in Hawaii; groundwater is considered one of the more important freshwater inputs to inshore areas (Carlquist, 1980; Peterson et al., 2009).

2.2.4. Geomorphological and oceanographic features

2.2.4.1. Hawaii currents

In the North Pacific Ocean the geostrophic currents form a large basin-scale movement called the North Pacific Subtropical Gyre (NPSG) centered at about 28°N. The Hawaiian Archipelago lies in this NPSG, which rotates clockwise in response to trade wind and westerly wind forcing. Hence, the main Hawaiian Islands, which is located in the southern portion of the gyre, experiences weak mean currents flowing from east to west, while the northern portion of the Hawaiian Archipelago experiences a weak mean current flowing from west to east. Between about 18°N and 22°N, the currents are strongly influenced by the islands (Juvic and Juvic, 1998). The North Equatorial Current (NEC) forks at the Big Island; the northern branch becomes the North Hawaiian Ridge Current (NHRC) and intensifies near the islands with a typical width of 65 miles (100 km) and speed of 0.5 knots (25 cm/s). West of the islands, two elongated circulations appear. A clockwise circulation is centered at 19°N, merging to the south with the southern branch of the NEC. A counterclockwise circulation is centered at 20°30'N. Between them is the narrow Hawaiian Lee Countercurrent (HLCC), extending in longitude from 170°W to 158°W. Surface currents over the western islands and northeast of the NHRC are variable, and their average is smaller than can be estimated from existing data. Within the NPSG, the westward flowing northern edge of the NEC grazes the Hawaiian Islands near the Big Island. The NHRC can be thought of as a small part of the NEC that turns northwest to flow along the windward side of the chain instead of turning southwest to pass south of Hawaii Island. The Subtropical Counter Current (STCC) is an eastward flowing surface current found typically along 24°N from 130°E to 160°W. The eastward flowing HLCC is generally located along 20°N and extends from about 150°E to just west of the Hawaiian Islands (Kobashi and Kawamura, 2002). The formations of the STCC and HLCC have recently been attributed to the “wake effect” that results from the combination of the westward trades winds blowing over the Hawaiian Archipelago.

2.2.4.2. Temporal cycles

Climatological cycles, winds, and currents, can greatly affect the depth of the thermocline and the rate of nutrient recharge. These events and cycles may be quite transitory, with annual duration events such as the El Niño, La Niña, or longer duration events lasting

many years. During an El Niño event, the normal easterly trade winds weaken, resulting in a weakening of the westward equatorial surface current and a deepening of the thermocline in the central and eastern equatorial Pacific including the Hawaiian area. Water becomes warmer and more vertically stratified with a substantial drop in surface chlorophyll. A La Niña event exhibits the opposite conditions when the easterly trade winds strengthen, resulting in a strengthening of the westward equatorial surface current and a shoaling of the thermocline. Water becomes cooler and less vertically stratified with a substantial increase in surface chlorophyll.

Physical and biological oceanographic changes have also been observed on decadal time scales. These low frequency changes, termed regime shifts, can impact the entire ocean basin. Recent regime shifts in the North Pacific have occurred with both physical and biological impacts (Polovina et al., 1995; Polovina, 1996). Changes in the Aleutian Low Pressure System are an example of how variation in a prominent Pacific Ocean weather feature can profoundly affect the abundance and distribution of marine species. Polovina et al. (1994) found that between 1977 and 1988 the intensification of the Aleutian Low Pressure System in the North Pacific resulted in a deeper mixed layer depth, which led to higher nutrients levels in the photic zone. This, in turn, led to an increase in phytoplankton production, which resulted in higher productivity levels and higher abundance levels for some species of macrofauna in the Northwestern Hawaiian Islands. Changes in the Aleutian Low Pressure System and its resulting effects on phytoplankton productivity are thought to occur generally every 10 years. Productivity changes at all trophic levels in the Northwestern Hawaiian Islands varied by 30 to 50% as a result of this documented decadal-scale climate cycle. The phenomenon is often referred to as the Pacific Decadal Oscillation (Mantua and Hare, 1997). Thus, it is important to understand that any carrying capacity or potential productivity of an ecosystem is dynamic and may fluctuate in response to oceanographic conditions as mediated by climatological cycles and events. There has been an increasing awareness within the scientific community of the occurrence and importance of long-term (decadal or longer) oceanographic cycles (e.g., Chavez et al., 2003) and of their relationship to cycles in the population sizes of some marine species, although there have not been any studies of relationships to Hawaiian insular false killer whale population dynamics or geographic range. These naturally occurring cycles can either mitigate or accentuate the impact of anthropogenic effects and, in general, the scientific community is becoming more aware of the need to recognize the possibility of large natural swings in marine populations and to incorporate this dynamism into management models. False killer whales are long-lived animals with a substantial investment in care of offspring and, hence, they have a low reproductive rate. This life history is generally buffered against environmental shifts by various means of maintaining high adult survival rates. False killer whales may respond to long-term shifts in productivity through a foraging strategy that uses long-term social bonds and transfer of knowledge between generations.

2.2.4.3. Eddies

Generally within the lee of an archipelago, and specifically downstream from the Hawaiian Islands, there is an abundance of mesoscale eddies created from a mixture of wind, current, island, and sea floor interactions. Eddies can rotate either clockwise or

counter clockwise and have important biological impacts. Eddies create vertical fluxes, with regions of divergence (upwelling) where the thermocline shoals and deep nutrients are pumped into surface waters enhancing phytoplankton production, and also regions of convergence (downwelling) where the thermocline deepens. Eddies are generally short-to-medium-term water movements that can play important roles in regional climate (e.g., heat exchange) as well as the distribution of marine organisms (transport and retention). Large-scale eddies spun off of the major surface currents often blend cold water with warm water, the nutrient rich with the nutrient poor, and the salt laden with fresher waters (Bigg, 2003). The edges of eddies, where the mixing is greatest, are often targeted by fishermen as these are areas of high biological productivity. Eddies are very common off the leeward coast of Hawaii, a region found to be highly used by insular false killer whales. Some of these eddies are quasi-stationary, yet most slowly propagate to the west. There is some understanding of eddy dynamics with respect to lower trophic levels (e.g., Seki et al., 2001), yet linkages with productivity of higher trophic levels are not well understood.

2.2.4.4. Fronts

Fronts represent sharp boundaries or small areas of rapid change in a variety of physical parameters including temperature, salinity, chlorophyll, and sea surface height (Roden, 1980; Niiler and Reynolds, 1984; Olson et al., 1994; Seki et al., 2002). These features can be located in either the horizontal or vertical dimension. Commonly observed are convergent temperature fronts which, in many cases, involve dense, cooler nutrient-rich water sinking below adjacent warmer water thereby creating a convergence of phytoplankton (Roden, 1980; Polovina et al., 2001). Buoyant organisms, such as jellyfish as well as vertically swimming zooplankton, can maintain their vertical position in the weak downwelling, and aggregate in the front to graze on the down-welled phytoplankton (Olson et al., 1994; Bakun, 1996). The increased level of biological productivity in these zones attracts higher trophic level predators, and ultimately a complete pelagic food web is assembled. There are two prominent frontal zones located near Hawaii. These frontal zones are associated with two isotherms (17° C and 20° C), and they are climatologically located at latitudes 32°–34° N (the Subtropical Front or STF) and latitudes 28°–30° N (the South Subtropical Front or SSTF) (Seki et al., 2002). Both the STF and SSTF represent important habitats for a variety of marine species. Variations in their position play a key role in catch rates of swordfish and albacore tuna (*Thunnus alalunga*), and distribution patterns of Pacific pomfret (*Brama japonica*), flying squid (*Ommastrephes bartramii*), loggerhead turtles (*Caretta caretta*), and seabirds (Seki et al., 2002; Howell et al., 2008). These frontal zones have been found to be migratory pathways across the Pacific for loggerhead turtles (Polovina et al., 2000; Kobayashi et al., 2008). Hawaii-based longline fishing vessels targeting swordfish set their lines where the fish are believed to be moving south through the fronts following squid, the primary prey of swordfish (Seki et al., 2002). Squid are also the primary prey of albatrosses (Harrison et al., 1983). Hence, a wide variety of species including several of concern such as loggerhead turtles and albatross have considerable overlap with longline fishing vessels in these areas of high biological productivity. To the south of the Hawaiian Islands, spanning latitudes 15° N to 15° S, lies the equatorial current system consisting of alternating east and west zonal flows with adjacent fronts. Spawning in yellowfin tuna

has been correlated to sea surface temperatures (SSTs), mainly above 24–26°C and may also be related to a frontal region at the edge of the Western Pacific Warm Pool (WPWP). The WPWP is the largest oceanic body of warm water with surface temperatures consistently above 28°C (Yan et al.,1992). The edge zones of this warm area are convergence zones which bring up nutrient rich waters and create high productivity areas resulting in high densities of tuna forage (i.e., baitfish such as pelagic anchovy, *Encrasicholina punctifer*) and, thus, large numbers of tuna. False killer whales in these regions would likely have increased foraging success and such "hot spots" of biological activity need further examination with respect to the spatial and temporal patchiness of forage in oligotrophic regions.

2.2.4.5. Pelagic-benthic coastal interface

Connectivity of the different marine environments is an important factor in ecosystem function. The pelagic or open ocean ecosystem is very large compared to any other marine ecosystems; however, other marine communities are vitally important to pelagic species, in part, because of the food-poor nature of much of the pelagic environment. The interface of pelagic environments with coastline/islands and seafloor is one such ecologically important coupling with consequences such as the "Island Mass Effect" which attributes regions of higher productivity to land-based inputs such as nutrients and freshwater and wake effects (Doty and Oguri, 1956; Gilmartin and Revelente, 1974). Another example of this interface is the mesopelagic boundary area. This is the region adjacent to shorelines, between 200 and 1000 m deep, and bordered by the photic and the aphotic zones. This area provides habitat for a unique community of fishes, crustaceans, mollusks and other invertebrates which become prey for tunas and other pelagic and insular coastal species (Reid et al., 1991). Acoustic sampling studies off the coasts of Oahu and Kona were implemented by Benoit-Bird et al. (2001) to assess the spatial heterogeneity, horizontal and vertical migration patterns, relative abundance, and temporal patterns of the mesopelagic community as well as the linkages among this community, the influence of the coastlines, and oceanographic parameters. The horizontal component of the mesopelagic community migration indicates a clear link between the nearshore and oceanic ecosystems in the Hawaiian Islands which, in turn, affects the presence and abundance of the pelagic predator species. Additional studies near the Hawaiian Islands indicate that concentrations of spawning tuna near the islands may be a result of increased forage species in these areas associated with elevated primary productivity (Itano, 2000). Offshore areas of high pelagic catch rates and spawning frequencies were found around several productive seamounts which also exhibit high productivity based on interactions of submarine topography, current gyres and location in the lee of the main Hawaiian Islands (Itano, 2000). Trophic linkages such as those evident in tunas, whereby pelagic anchovy are a primary forage species which themselves feed primarily on copepods, provide a critical link between zooplankton and larger pelagic species (Ozawa and Tsukahara, 1973). Understanding these linkages is an essential component of successful ecosystem management.

2.2.4.6. Global climate change

The global mean temperature has risen 0.76° C over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC, 2007).

Ample evidence now exists supporting the wide-ranging ecological impacts of global climate change (Walther et al., 2002). There is high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. These changes include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC, 2007). The seasonal north-south movements of many large pelagics in the NPTZ appear to track the similar peak migration of primary productivity. Using remotely sensed chlorophyll concentrations from satellite observations, Polovina et al. (2008) found that over the past decade primary productivity in the subtropical and transition zone has declined an average of 1.5% per year with about a 3%-per-year decline occurring at the southern limit of the NPTZ. The expansion of the low chlorophyll waters is consistent with global warming scenarios, based on increased vertical stratification in the mid-latitudes. Expanding oligotrophic portions of the subtropical gyres in the world's oceans in time will lead to a reduction in chlorophyll density and carrying capacity in the larger subtropical gyres, thus impacting the abundance of pelagic species. An international program, the Climate Impacts on Oceanic Top Predators (CLIOTOP), is currently gathering information on climate change and its effects on pelagic ecosystems. Within this group, analytical models are being applied to investigate the future management of some marine species in the context of climate and ecosystem variability, as well as to investigate potential changes due to greenhouse warming. The relationship of climate change (e.g., global warming) on cetacean growth and mortality rates in the Hawaiian Archipelago is not well understood and requires further research in order to be taken into account by managers (IPCC, 2007).

2.2.4.7. Acidification

The oceans have absorbed about 50% of the carbon dioxide (CO_2) released from the burning of fossil fuels, resulting in chemical reactions that lower ocean pH. This has caused an increase in hydrogen ion concentration (acidity) of about 30% since the start of the industrial age through a process known as “ocean acidification”. A growing number of studies have demonstrated adverse impacts of acidification on marine organisms. These impacts include a reduction in the rate at which reef-building corals produce their skeletons, reduced ability of marine algae and free-swimming zooplankton to maintain protective shells, and reduced survival of larval marine species, including commercial fish and shellfish. Such impacts have serious implications for coral reef and pelagic ecosystem productivity. The net effect on higher trophic levels remains unclear, yet is likely to be negative. Acidification can be tenuously linked to increases in some gelatinous zooplankton (Richardson et al., 2009). Shifts in the pelagic species composition could have many direct and indirect effects on the distribution and abundance of large pelagic predators.

2.2.4.8. Benthic environments

The benthic (or demersal) environment is regarded as extending from the high-tide mark to the deepest depths of the ocean floor. Benthic habitats support a wide range of marine organisms forming complex communities and biogenic substrata. This section presents a simple description of the two benthic zones in the Hawaiian Archipelago that may have direct or indirect impacts on Hawaiian insular false killer whales: deep reef slope and

banks/seamounts. Habitat for the Hawaiian insular false killer whales is likely related to the abundance of deep reef slope habitat fringing all of the emergent islands since their range appears to be limited to waters extending out to only approximately 112 km from shore, with a core range that is much smaller. Similarly, there are a multitude of banks and seamounts that are likely used if ocean interfaces with either coastline or shallow submarine features are important characteristics of the preferred environment.

Deep reef slope

As most Pacific Islands are oceanic islands versus continental islands, (i.e., they were never connected to continents), they generally lack an extensive shelf area of relatively shallow water extending beyond the shoreline. For example, the average global continental shelf extends 75 km, with a depth of around 60 m (Postma and Zijlstra, 1988). While lacking a shelf, many oceanic islands have a deep reef slope, which is often angled between 45° and 90° toward the ocean floor. The deep reef slope is home to a wide variety of macrofauna that are important fisheries target species, such as snappers and groupers found in areas of high vertical relief. Such fauna could be potential insular false killer whale forage. Biological zonation occurs on the reef slope, and is related to the limit of light penetration beyond 100 m. For example, reef-building corals can be observed at depths less than 100 m, but at greater depths gorgonian and black corals are more readily observed (Colin et al., 1986). The deep reef slope is a large expanse of habitat that is not well studied but likely has a large role in the delineation of nearshore habitat used by Hawaiian insular false killer whales based on sighting and satellite-tag telemetry locations.

Banks and seamounts

Banks are generally volcanic structures of various sizes that occur both on the continental shelf and in oceanic waters. Coralline structures tend to be associated with shallower parts of the banks as reef-building corals are generally restricted to a maximum depth of 30 m. Deeper parts of banks may be composed of rock, coral rubble, sand or shell deposits. Banks thus support a variety of habitats that, in turn, support a variety of marine life (Levinton, 1995). Seamounts are undersea mountains, mostly of volcanic origin, which rise steeply from the sea bottom but do not reach the surface of the sea (Rogers, 1994). Seamounts have complex effects on ocean circulation. One effect, known as the Taylor column, relates to eddies trapped over seamounts to form quasi-closed circulations. It is hypothesized that this helps retain pelagic larvae around seamounts and maintain the local populations. Although evidence for retention of larvae over seamounts is sparse (Boehlert and Mundy, 1993), endemism has been reported for a number of marine species at seamounts (Rogers, 1994). Numerous banks and seamounts can be found in the Hawaiian Archipelago, with more in the Northwestern Hawaiian Islands rather than in the main Hawaiian Islands. In the main Hawaiian Islands, the largest bank is Penguin Bank which is located southeast of Oahu. Other notable features in the area include Middle Bank to the northwest of Kauai, and Cross Seamount to the southwest of the Big Island. These habitats may be frequented by Hawaiian insular false killer whales if they display similar characteristics pertinent to island-association behavior; at present, these characteristics are unknown but may be related to such things as patterns in bottom topography, currents, or productivity.

2.2.5. Marine species in the central North Pacific and Hawaiian Archipelago

2.2.5.1. Pelagic ecosystem

Most surface water in Hawaii is deficient in nutrients because the lens of warm surface water rarely mixes with the cooler, nutrient-rich deep water. However, some areas are enriched by upwelling, eddies, and fronts as mentioned above. Floating microscopic phytoplankton or algae forms the base of food production in the ocean. Productivity is somewhat higher in the summer than in the winter. The annual primary productivity of the open ocean near Hawaii may be only one-third of that of temperate coastal zone areas and one-tenth of that in open ocean upwelling. The small plankton, from protozoans to larval fish, graze on the algae in the ocean water. These primary consumers are in turn eaten by secondary consumers, either carnivorous zooplankton or fish. Hawaiian areas, like most tropical and subtropical waters, are usually considered to have five trophic levels. Trophic efficiency is generally regarded to be ~ 10% (Roger and Grandperrin, 1976; Pauly and Christense, 1995). Hence, of every 10,000 units of energy produced by Hawaiian phytoplankton, only one unit winds up in a yellowfin tuna after four energy transfers through five trophic levels. In contrast, productive areas off the coast of California with higher ecological efficiencies and shorter food chains produce 400 times more energy to top-level predators, such as large fishes, seabirds, cetaceans, and humans (Ryther, 1969). An important phenomenon in the Hawaiian pelagic ecosystem is the daily vertical migration in the water column of certain planktonic creatures. Unlike phytoplankton, zooplankton is not restricted to the sunlit upper 300 ft of the ocean but can be found at all depths. Many of these creatures undertake extensive migrations, usually moving towards the surface during the night and descending hundreds of feet during the day. Migrations enable the organisms to feed in the productive euphotic zone yet hide in deep water during the day. Larger predators, including omastrephic squids follow vertical migrations, although some squids do remain near the surface all day in Hawaii (Parry, 2003). Other mid-trophic level species, such as flyingfishes (*Exocetidae*) and scads (*Decapturus* spp) may also remain near the surface all day.

2.2.5.2. Fish species

The primary marine species relevant to the ecology of the Hawaiian insular false killer whales are their prey species and competitors. These are primarily the large pelagic fishes like tunas, billfishes, and pelagic sharks of the epipelagic zone of tropical and subtropical oceans that frequent the Hawaiian Archipelago (Table 2-2), but also may include some insular species (Table 2-3) such as carangids and coastal sharks and, possibly, also bottomfishes like snappers or groupers.

Large pelagic fish live in tropical and temperate waters throughout the world's oceans, and they are capable of long movements that reflect complex relationships to oceanic environmental conditions. The larvae and juveniles of most species are more abundant in tropical waters, whereas the adults range into temperate latitudes. Distribution varies with seasonal changes in ocean temperature and local abundance is seasonal, but all of these fishes are found year-round in Hawaii. The more tropical tuna species such as yellowfin, skipjack (*Katsuwonus pelamis*), and bigeye appear to roam extensively within a broad expanse of the Pacific centered on the equator. Billfish species that are capable of

transoceanic movement, and some seasonal regularity suggestive of migration has been noted. The pelagic ecosystem responds to ambient climatic and oceanographic conditions on a variety of spatial and temporal scales such that fish population sizes fluctuate somewhat independent of the impacts of removals by fisheries or predation. This variation is more pronounced for short-lived pelagic species. Most of the tunas can reproduce by their third year of life and average less than 5 years of age. Skipjack tuna and mahimahi reproduce in less time. Billfish grow very fast but also live longer and reproduce later than tuna.

Table 2-2. Large pelagic fishes that commonly occur in the Hawaiian Archipelago.

Official Common Name	Other (Local) Common Name	Scientific Name
Tunas		
Albacore	Ahi tombo	*Thunnus alalunga*
Bigeye tuna	Ahi mebachi	*T. obesus*
Yellowfin tuna	Ahi	*T. albacares*
Skipjack tuna	Aku	*Katsuwonus pelamis*
Kawakawa	Kawakawa	*Euthynnus affinis*
Frigate and bullet tuna	Keokeo	*Auxis* spp.
Billfishes		
Blue marlin	Kajiki	*Makaira mazara*
Striped marlin	Nairagi, Kajiki	*Kajikia audax*
Shortbill spearfish	Hebe	*Tetrapterus angustirostris*
Swordfish	Shutome	*Xiphias gladius*
Other teleosts (bony fishes)		
Dolphinfishes	Mahimahi	*Coryphaena* spp.
Wahoo	Ono	*Acanthocybium solandri*
Moonfish	Opah	*Lampris* spp.
Oilfish family	Walu	*Gempylidae*
Pomfret family	Monchong	*Bramidae*
Sharks	Mano	
Thresher sharks		*Alopias* spp.
Silky shark		*Charcharinus falciformis*
Oceanic whitetip shark		*Carcharhinus longimanus*
White shark		*Carcharodon carcharias*
Blue shark		*Prionace glauca*
Mako shark		*Isurus* spp.

Pelagic fish movements are not restricted to the horizontal dimension. In the oligotrophic tropical ocean, light and temperature diminish with depth, and photosynthesis is limited to the epipelagic zone, within about 200 m of the surface. Most pelagic fish migrate vertically, descending in the daytime and moving toward the surface at night to feed on prey species that exhibit similar diurnal vertical migrations. Certain species, such as swordfish, have wide depth ranges (i.e., down to 800 m) but are most vulnerable to fishing when they are concentrated near the surface at night. Others such as bigeye tuna (depths to > 500 m) are caught mostly when hooks are deployed in deep water during the day, and this is how the largest sector of the Hawaii longline fishery operates. Shallow-set longline fishing at night, as well night handline fishing (Yuen, 1979) catch bigeye, as does offshore handline fishing near weather buoys and seamounts (Impact Assessment, 2007) where bigeye appear to remain closer to the surface in the day. Based on

instrumented longline catch rates (Boggs, 1992) yellowfin tuna, marlins, spearfish (*Tetrapturus angustirostris*), and mahimahi have generally shallower depth distributions during the day, and these may be more easily preyed upon by false killer whales, given the low frequency of deep dives by false killer whales (Section 2.1.6). These species are also the main targets of the surface-fishing troll and handline fisheries.

Many of the pelagic stocks in the Pacific are thought to be exploited at or near the maximum sustainable yield (MSY) level. Achieving MSY is the goal of fishery management, and in recent years concerns over the harvest of yellowfin and bigeye tuna (Harley et al., 2009; Langley et al., 2009) have prompted international action to reduce fishing mortality on these species. Similar concerns are motivating recommendations to limit fishing effort on striped marlin (*Kajikia audax*) and albacore (ISC, 2007). The status of skipjack tuna and swordfish stocks appear to be healthy. The status of populations of many other species is not known. Some studies indicate that as the biomass of tuna and billfish has been reduced by fishing, the abundance of intermediate trophic level species such as mahimahi and pelagic pomfrets has increased (Polovina et al., 2009).

Table 2-3. Examples of some insular fishes in Hawaii that could be important to Hawaiian insular false killer whales.

Common Name	Other (Local) Common Name	Scientific Name
Reef Fishes		
Scrawled filefish		*Aluterus scriptus*
Threadfin jack	Kagami ulua	*Alectis ciliaris*
Rainbow runner	Kamanu	*Elegatis bipinnulatus*
Great barracuda	Kaku	*Sphyraena barracuda*
Bottomfish		
Amberjack	Kahala	*Seriola dumerili*
Grey Snapper	Uku	*Aprion virescens*
Pink Snapper	Opakapaka	*Pristipomoides filamentosus*
Sea Bass	Hapu‘upu‘u	*Epinephelus quernus*
Red Snapper	Ehu	*Etelis carbunculus*
Longtail red snapper	Onaga	*E. coruscans*
Coastal Sharks		
Galapagos shark	Mano	*Carcharhinus galapagensis*
Sandbar shark	Mano	*Carcharhinus plumbeus*
Tiger shark	Niuhi	*Carcharhinus cuvier*

If the insular false killer whales are adapted to use island-associated prey types, then pelagic species that are most associated with the islands could be important. For example ono (wahoo, *Acanthocybium solandri*) are commonly caught by troll fishermen near shore on their way from or back to port, during trips taken farther offshore to target tuna, marlin, or mahimahi. Kawakawa (*Euthynnus affinis*) and Keokeo (*Auxis* spp.) are also common nearshore, and *Auxis* are among the most common tuna larvae found near Hawaii (Boehlert and Mundy, 1994) even though these fish are a very minor component of Hawaii fisheries. Lustrous pomfret are commonly caught near seamounts and submarine ridges, and in the Hawaiian Islands where they are caught in the bottomfish fishery.

Some insular species (Table 2-3) occur far enough offshore (> 40 km) to be caught in the longline fishery, such as rainbow runner (*Elegatis bipinnulatus*), amberjack (*Seriola dumerili*), and barracuda (*Sphyraena barracuda*). Rainbow runner and barracuda are among the incidental species depredated by false killer whales (Section 4.1.1.1, Table 4-2, bony fish category). The few coral reef species that have been observed in the diet of insular false killer whales (Baird, 2009) include scrawled filefish (*Aluterus scriptus*) and threadfin jack (*Alectis ciliaris*). These species are sometimes found in open water, although their pelagic distribution is coastal. Scrawled filefish are found around floating objects in deep water (Gooding and Magnuson, 1967) along with rainbow runner, great barracuda, amberjack, other jacks (*Caranx* spp.) and large pelagic species (mahimahi, tuna, etc.). Threadfin jack are found in coastal waters to a depth of about 100 m, while the young are usually pelagic and drifting (Smith-Vaniz in Carpenter and Neim, 1999).

Hundreds of other coral reef fish species exist in Hawaii, but there is no way to know which of them might be important to Hawaiian insular false killer whales. Most harvested reef species appear to have been greatly reduced in abundance over the last century (Shomura, 1987).

Some of the deep-slope fishes (Table 2-3) that figure prominently in Hawaii's bottomfish fishery may or may not be important to false killer whales, which have not been observed to interact with Hawaii's bottomfish fishery (Section 4.1.1.4). The bottomfish are predominantly distributed several hundred meters deep, although a few species have shallower distributions, such as amberjack and grey snapper (*Aprion virescens*). Amberjack are also sometimes caught by pelagic fisheries. Fishing for seven predominant bottomfish species around the eight main Hawaiian Islands is limited by an annual catch limit instituted to prevent localized stock depletion. However, the index of abundance for these species in the MHI has showed a continued decline from 1988 to 2004 (Moffitt et al., 2006).

Large pelagic and coastal sharks might be important competitors or predators for Hawaiian insular false killer whales. The only shark species in Hawaii that has been assessed is the blue shark (Kleiber et al., 2009) and, according to the assessment, the population is close to MSY. The biomass of that population is much reduced from its pristine condition, as one could assume would be likely for all large sharks, that are slow to grow and reproduce compared to pelagic fishes, but which have experienced similar mortality from fisheries. Depredation of sharks in the longline fishery by false killer whales is rare and, in some cases, it appears that false killer whales may avoid taking sharks (NMFS, unpublished data).

2.2.5.3. Cetaceans, sea turtles, and sea birds

Twenty-four species of cetaceans are known to occur within the Hawaiian Islands and the tropical Pacific. Both odontocetes and mysticetes are common. Similar to Hawaiian insular false killer whales, a number of other cetacean species in Hawaii are known or are thought to have separate island-associated and pelagic populations, including the common bottlenose dolphin, spinner dolphin, pantropical spotted dolphin, pygmy killer whale (*Feresa attenuata*), melon-headed whale (*Peponocephala electra*), short-finned

pilot whale, rough-toothed dolphin, Blainville's beaked whales (*Mesoplodon densirostris*) and Cuvier's beaked whale (*Ziphius cavirostris*). The diversity of cetacean fauna in Hawaii fits within a variety of ecological niches, with no other species known to be a direct competitor to false killer whales. Pilot whales, pygmy killer whales, bottlenose dolphins, melon-headed whales, and rough-toothed dolphins do have some overlap in prey species with false killer whales. Pilot whales are also known to depredate catch in the Hawaii-based longline fishery NMFS, PIRO Observer Program, unpublished observations).

Five species of sea turtles are found in the vicinity of the Hawaiian Archipelago: olive ridley (*Lepidochelys olivacea*), leatherback (*Dermochelys coriacea*), hawksbill (*Eretmochelys imbricata*), loggerhead, and green sea turtles (*Chelonia mydas*). All are listed under the ESA as threatened or endangered, and all are highly migratory, or have a highly migratory phase in their life history (WPFMC, 2009). None are known to have any direct role in the ecology of the false killer whale, although as bycatch in the Hawaii longline fishery, turtles have had an important influence on management controls for this fishery, which itself is an important aspect of the whale's environment (Section 4.1.1.1). Regulatory definition of deep-set versus shallow-set fishery sectors to control sea turtle bycatch in 2001 eliminated a mixed-method type of longline fishing that previously had set some shallow gear in closer proximity to the Hawaiian Islands than the shallow-set fishery targeting swordfish (He et al., 1997). And in 2004, circle hooks and fish bait were required in the shallow-set swordfish fishery to reduce sea turtle bycatch.

The Hawaiian Islands also are home to a variety of seabirds, including albatrosses (*Phoebastria* spp.), boobies (*Sula* spp.), shearwaters (*Puffinus* spp.), petrels (*Pseudobulweria* spp. and *Pterodroma* spp.), tropicbirds (*Phaethon* spp.), frigatebirds (*Fregata* spp.), noddies (*Anous* spp.), and terns (*Sterna* spp. and *Gygis alba*). Seabirds primarily prey on flying fish (Exocetidae), mackerel scads (*Decapterus* spp), and squids (Teuthida) which might also be forage for false killer whales, although the whales seem to prefer larger species and prefer fish over squid. Seabirds feed on surface prey, and some feed in association with tuna schools that drive prey towards the surface, making the prey more available to seabirds, and perhaps also to shallow-diving cetaceans.

Albatrosses follow ships, and frequent interactions with the Hawaii longline fleet resulted in many thousands of black-footed albatross (*Phoebastria nigripes*) and Laysan albatross (*Phoebastria immutabilis*) deaths per year in the 1990s. Mandatory deterrents have reduced current albatross mortality in the longline fishery to a few hundred per year. The commonly employed seabird bycatch mitigation methods include night setting, side setting, the use of weighted branch lines and thawed bait to sink bait more rapidly, and use of blue dyed bait (Gilman et al., 2008).

2.3. Genetics of false killer whales

2.3.1. Introduction to Units to Conserve and uses of genetic data for inference

This report considers genetic data as a potential line of evidence for whether Hawaiian insular false killer whales are discrete and whether the genetic data show a marked difference consistent with significance. Interpreting the genetic data for "discreteness"

and “significance” requires some background in how inferences are drawn from genetic data and where DPSs fit in the continuum of genetic differentiation that ranges from family units to species.

Genetic data allow inference on several different levels of population structure and at several different timescales. Within the genetic makeup of every organism, certain portions code for protein products and are, therefore, acted on by selection. Selection can be strong or weak and, therefore, selected genes are not conducive to making inferences on how long different groups of organisms have been separated. For that reason, most genetic studies that focus on differentiation, and the length of time that units have been separate, use neutral genetic markers. These markers are not under direct selection. The markers examined for false killer whales are neutral genetic markers (though see discussion below on mitochondrial DNA). The markers used for false killer whales and for the case studies detailed below all use the standard markers for population genetics: mitochondrial DNA sequence data and nuclear markers called microsatellites (details below).

Distinct Population Segments (DPSs) are at one level in a range of levels of population structure from social units, like families, to species. Units to conserve (UTCs), including DPSs, subspecies and species, are points on a continuum of differentiation that characterizes the process of speciation. In attempting to delimit any particular UTC, we seek to determine the present position along that continuum of the units we study. Chivers et al. (2007) focused on delineating Demographically Independent Units, which are relevant to the ecological time-scale pertinent to the MMPA. The goal of the MMPA is to maintain population stocks as functioning elements of their ecosystem. The timescale that relates to this goal is on the order of decades. Analytical methods focus on showing that the frequency of genetic markers differ sufficiently that gene flow on the decadal time scale is relatively trivial and indicates effective dispersal of less than a percent per year. The DPS discreteness criterion states “Quantitative measures of genetic or morphological discontinuity may also provide evidence of this separation.” It is not clear what a genetic “discontinuity” is, but presumably a genetic character that allowed almost certain identification of an individual as a member of that DPS would match a definition of “discontinuity”.

The DPS significance criterion relating to genetics requires evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics. Presumably, the strength of evidence for “significance” should be greater than for “discreteness” and we interpret “markedly” in this context to mean that the degree of genetic differentiation is consistent with a population that could have genetic adaptations to the local habitat. The magnitude of genetic differentiation is expected to be large (consistent with less than one migrant per generation), and phylogeographic patterns (haplotypes or genotypes that are found nearly exclusively in one geographic region indicating evolution has occurred within that geographic region) are expected to be present. Regions that have phylogeographic structure will always also be demographically independent, but the converse is not necessarily the case. There are no genetic criteria to distinguish any of the “boundaries” between UTCs. Generally, as time

passes with units not interchanging genes, the gene pools drift further apart and new genetic types evolve that are only found within one unit.

In each cell there is nuclear DNA (nDNA) and mitochondrial DNA (mtDNA). These different types of DNA have different modes of inheritance. mtDNA is inherited only from the mother and is likened to inheritance of a family name. It is useful for determining Demographically Independent Populations (or DIPs) where internal population dynamics are far more important than external dynamics (immigration) to maintain local abundance. However, mtDNA often provides signals at an evolutionary timescales. Closely related mtDNA haplotypes found in only one location can indicate separation for a long enough time to evolve local haplotypes. This degree of separation can be used as a line of evidence for DPS (significance criterion), subspecies or species designations.

Genetic patterns in mtDNA can plausibly result from several different histories. Two important concepts for interpreting patterns of mtDNA that might arise when either island or coastal habitats are colonized from pelagic populations are called the "founder effect" and "lineage sorting". When a small population is formed from a larger population, the small population will initially have the higher genetic diversity of the large population. Through a process called genetic drift, genetic diversity will be lost and eventually reach an equilibrium lower diversity within the small population. The reduction in genetic diversity can happen very quickly, in evolutionary timescales, if the population is very small. This loss of haplotypes can sometimes result in a population with only a few haplotypes that differ from one another by many base-pair differences. These quite different haplotypes did not evolve within the small population but resulted from the loss of intermediary haplotypes.

Lineage sorting is another process that makes it difficult to interpret the time that has elapsed between neighboring small populations along a coastline. Consider, for example, a case where population A lost all but one haplotype and its neighbor population B lost all but one haplotype and that those haplotypes differed by 5 basepairs. Evaluating the time since these populations have been separated based on the time it would take to evolve haplotypes that differ by 5 basepairs would greatly overestimate separation time. Thus, history can influence genetic patterns and the best interpretations will assess the plausibility of lineage sorting given the data and give the reader appropriate caveats.

Nuclear DNA (nDNA) is inherited from both parents. Although nDNA codes for most genetic traits subject to selection, we are again only looking at neutral markers. Different neutral markers have different mutation rates. The markers used in the studies examined here for Hawaiian insular false killer whales are called microsatellites. These markers are repeats of nucleotide groups and are quantified by length. For example, CACACA is a repeat of the CA group with a 3-repeat length. The mutation rate for these markers is high so it is useful for identifying individuals. In the studies examined here, these markers are used to identify individuals and eliminate cases where individuals were accidentally sampled multiple times. First-order relatives (mother-offspring or father-offspring) can also be detected because they will share one allele at each microsatellite

marker. Microsatellites are also used to estimate the magnitude of genetic differentiation and level of gene flow between different units.

Interpreting genetic data to assess the magnitude of separation between strata is often conducted with statistics that compare within strata variation to between strata variation. These statistics, which for simplicity here are called F_{ST} statistics, have a convenient theoretical underpinning because they relate demography to a measure of genetic differentiation with simple formulas: $F_{ST} = 1/(2N(m+\mu) + 1)$ for mtDNA and $F_{ST} = 1/(4N(m+\mu) + 1)$ for nDNA, where N is the effective population size, m is the number of migrants per generation and μ is the mutation rate. If we use one-migrant per generation as a rule of thumb for the level of gene flow that is so low that selection can outweigh the effects from neighboring population, then we can use F_{ST} to get a rough idea of the level of differentiation consistent with the DPS significance criterion. The mutation rate is relatively low for mtDNA control region sequence, so the expected value for $F_{ST} = 0.33$ for one-migrant per generation. The mutation rates for microsatellites are high and range between about 0.01 and 0.001, which means that the expected values for F_{ST} assuming one-migrant per generation would be between 0.031 and 0.075. These expected values are quite small and make interpretation of the nDNA weak for the purposes of estimating gene flow. The interpretation of genetic data for false killer whales will return to caveats necessary for understanding what the nDNA can tell us about the magnitude of gene flow.

The studies considered here examine a particular sequence within the mtDNA genome (called the control region) that is commonly treated as a neutral marker. The mitochondria is inherited as a unit and has functional genes under selection. Some authors suggest that there is extensive evidence for selective sweeps in mtDNA data (Bazin et al., 2006). Readers should be aware that one plausible explanation for mtDNA haplotypes being found in one location and nowhere else is that these haplotypes resulted from a selective sweep. Such evidence, should selection be operant, would strengthen the case for being markedly genetically different and hence meet the DPS significance criterion. Because the region examined in these studies does not code for a protein product, however, one cannot conclude that selection drove the genetic pattern and we take the more conservative interpretation that the pattern results from genetic drift and mutation in a neutral marker.

Genetic studies most often use analytical methods that assume that the population is in equilibrium (i.e., a population that has remained constant for many generations and has experienced a constant level of gene flow from neighboring populations). All biological populations violate this assumption. In some cases, where population history is well documented, the effects of violating this assumption can be tested by simulating the population history (Archer et al., 2010). For others, as is the case for all cetacean populations in the vicinity of the Hawaiian Archipelago, the consequences of violating this assumption should be discussed. The potential consequences of Hawaiian insular false killer whales being out of equilibrium will be discussed below.

To put the results concerning false killer whales in context (given in detail in Section 2.3.2), a number of case studies for other species are shown below. The case studies focus on island or coastal units compared to pelagic units. For mtDNA, results are presented in the form of minimum spanning networks that allow the reader to easily view the degree to which haplotypes differ from one another, together, and how the differences relate to the geographic context. These networks were generated using the program Harlequin, version 3.11 (Excoffier et al., 2005) to examine the concordance between the distribution of mtDNA control region haplotypes and geographic region. Concordance would constitute evidence for evolutionary processes driving the patterns revealed. Optimal minimum spanning networks incorporate information about haplotype frequency to obtain the most parsimonious network for haplotype evolution. For example, haplotypes that are 'rare' or occur at low frequencies would be most likely to have been derived from haplotypes that are 'common' or occur in high frequencies rather than from another 'rare' haplotype for a given series of mutation events (Excoffier et al., 1992; Excoffier & Smouse, 1994).

2.3.1.1. Case studies

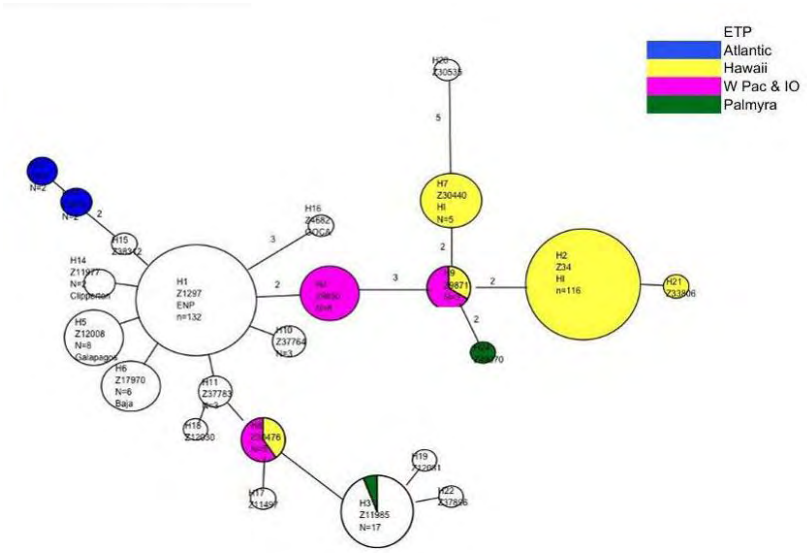


Figure 2-3. Minimum spanning network for the short-finned pilot whale. Each circle represents a haplotype, and the size of the circle represents the relative haplotype frequency with most haplotypes being represented by a single individual. Connecting branches are also scaled to represent the minimum number of base pair changes between haplotypes. All haplotypes have, at a minimum, from 1 to 5 base pair changes. Each region shows distinct haplotypes, and each group is labeled with a descriptive name for the region. A haplotype identified from animals sampled away from the main Hawaiian Islands is identical or closely related to haplotypes identified from the Indo-Western Pacific (indicated by the pink circle) and one individual from Hawaii with a haplotype related to the ETP region.

Short-finned pilot whales—Short-finned pilot whales are similar to false killer whales based on their general life history, strong social structure, and tendency to forage as groups. Studies using photographic identification and satellite tagging near Hawaii have

demonstrated fidelity of groups to Hawaiian coastal waters and movement at least among the main Hawaiian Islands (Baird et al. pers., comm.). The whales in Hawaii are genetically differentiated from those in the Eastern Tropical Pacific (ETP) and western Pacific/Indian Ocean (Figure 2-3) (Chivers, 2004). Some haplotypes may have evolved in Hawaiian insular waters, but sampling of adjacent pelagic waters is poor making it possible that the Hawaiian haplotypes could be found elsewhere. Several Hawaii individuals have a haplotype that is distantly related to the common Hawaii haplotypes, and it is also found in other areas. No nDNA were examined in this study.

Spinner dolphins—In spinner dolphins, genetic differences were found in nDNA (10 microsatellites) and in the frequencies of mtDNA haplotypes between different regions within the Hawaiian Archipelago (Fig. 2-4—Fig. 2 from Andrews et al., 2010). Although American Samoa is the only distant region sampled, the pattern within the archipelago is much more complex than for pilot whales and is consistent with immigration from groups with relatively distantly related haplotypes (i.e. haplotypes that differ by many basepairs with none of the intermediary haplotypes present). Strong differences ($p < 0.001$) were found between American Samoa ($n = 16$) and Hawaii for both mtDNA and nDNA. The magnitude of differentiation appears larger as measured by F_{ST} for mtDNA (> 0.2) than for nDNA (< 0.09). Several possible explanations for such differences will be discussed in detail for false killer whales in Section 2.3.3.

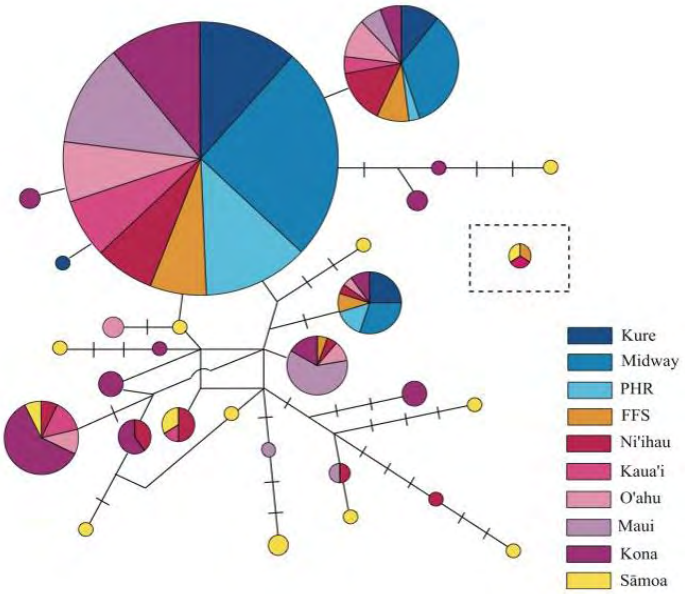


Figure 2-4. Minimum spanning network for the spinner dolphins.

Pantropical spotted dolphins—Only preliminary data are available on pantropical spotted dolphins (*Stenella attenuata*) (Courbis et al., 2008). Structure within the Hawaiian Archipelago is evident, but no direct comparisons are made with distant regions. The number of haplotypes is dramatically fewer than for spinner dolphins with one common haplotype, a few at medium frequency and many found only once (indicating low sampling relative to diversity). The two haplotypes next to most the

common differ by 8 and 4 basepair differences, which is consistent with multiple immigration events.

Bottlenose dolphins (Hawaii and California coastal vs. offshore)—Within the Hawaiian Archipelago both photographic identification (Baird et al., 2009) and genetic data suggest no contemporary movement of bottlenose dolphins among the main Hawaiian Islands (Martien et al., 2006). The minimum spanning network (Figure 2-5) reveals strong evidence of multiple immigration events originating from distantly related groups.

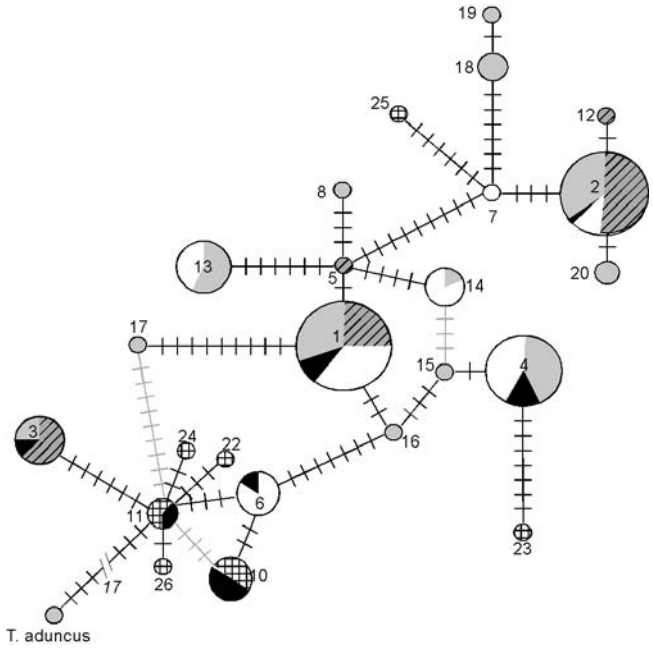


Figure 2-5. Minimum spanning network showing relationships among the mtDNA haplotypes for Hawaiian bottlenose dolphins. Each circle represents a haplotype. The sizes of the circles are proportional to the frequencies of the haplotypes. Each circle is color coded to indicate the fraction of individuals with that haplotype that come from each island group: checked = Palmyra, black = Hawaii, gray with black lines = 4-Islands region, white = Oahu, and solid gray = Kauai/Niihau. The dashes on the lines separating the haplotypes indicate the number of mutational steps between haplotypes. The T. aduncus individual had a haplotype most similar to this species but was within a group of T. truncatus and did not appear different than those individuals in nDNA. Photographs of this individual did not reveal obvious external differences.

Bottlenose dolphins from coastal waters off of California and Baja California have been described with different skull morphology than pelagic bottlenose dolphins (Walker, 1981). Genetic studies have confirmed the differentiation (Lowther, 2006). The minimum spanning network (Figure 2-6) reveals a cohesive group of haplotypes within the coastal group that is consistent with locally evolved haplotypes. Similar differences were found in nDNA (using microsatellites). The much more abundant offshore group has much higher diversity with many relatively distantly related haplotypes. There are

other cases of differences between pelagic and coastal populations too numerous to give details here (e.g., Querouil et al., 2007; Escorza-Treviño et al., 2005).

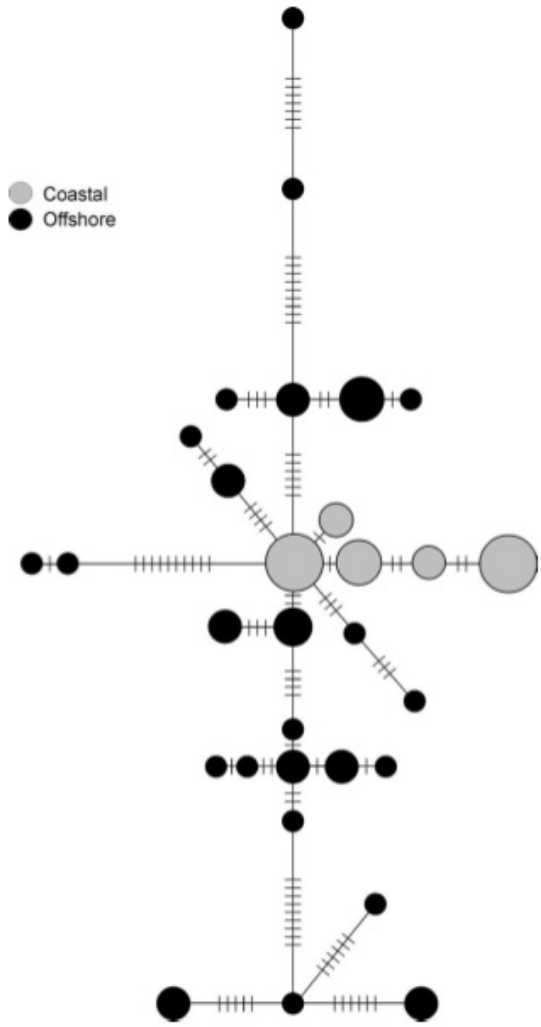


Figure 2-6. Minimum spanning network for coastal (gray) and offshore (black) bottlenose dolphins where hash marks indicate basepair differences.

Cook Inlet beluga whales—Cook Inlet belugas were recently listed as endangered, having met the DPS criteria. Genetic data were used as part of the evidence for distinctness. The Φ_{ST} values comparing Cook Inlet to other stocks ranged from 0.577 for the closest neighboring stock in Bristol Bay to 0.200 for the Eastern Chukchi stock. The Cook Inlet stock has low haplotypic diversity, which is also consistent with its being genetically distinct from other stocks (O'Corry-Crowe et al., 1997). Figure 2-7 shows the minimum spanning network for different stocks of beluga whales in Alaska. Cook Inlet belugas have no unique haplotypes (i.e., all haplotypes are found in other beluga stocks). The haplotype differences between Cook Inlet belugas and other Alaskan stocks are consistent with its being a stock that is demographically independent from other stocks

but has not been separated for a sufficient length of time to evolve unique haplotypes. Although these genetic results are sufficient to meet discreteness criteria, the phylogeographic patterns of Cook Inlet belugas are not considered markedly different and, hence, do not justify meeting the significance criterion. The justification for the significance criterion was that they occupied a habitat unique to the taxon. O'Corry-Crowe et al. (1997) does not report nDNA results.

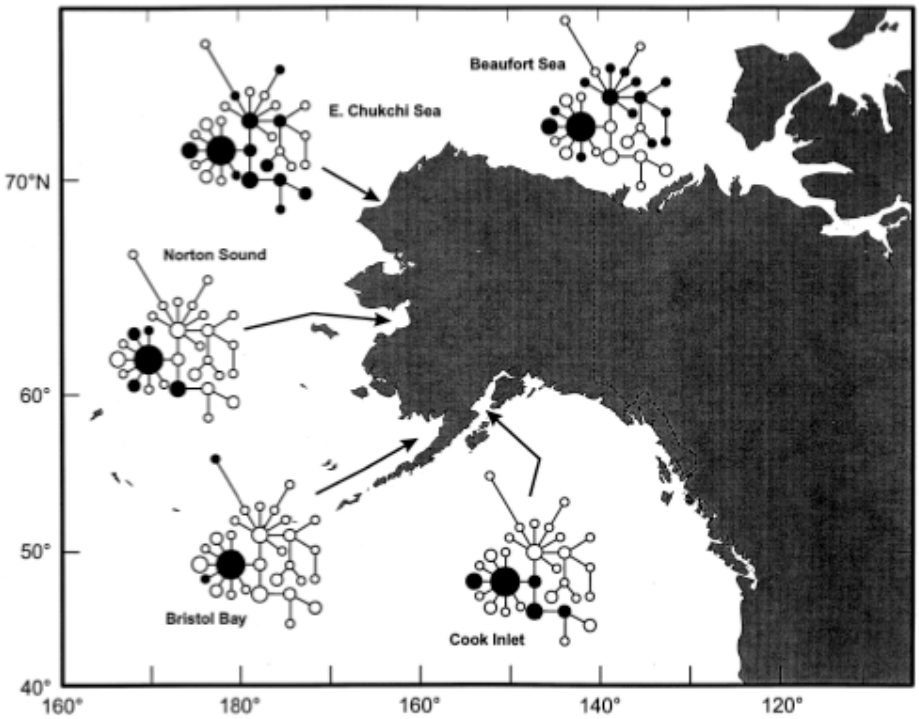


Figure 2-7. Minimum spanning network for beluga in Alaska. Networks are shown for the 5 stocks recognized under the Marine Mammal Protection Act. Solid circles indicate when that haplotype was present in the stock.

Killer whales—Killer whales are similar to false killer whales in life history (late age of maturity, long inter-birth interval and reproductive senescence) and sociality (tight social networks and group foraging). Several species and/or subspecies of killer whales have been proposed (Morin et al., 2010) that divide along foraging specialization types. Like false killer whales, killer whales globally have low genetic diversity (within all potential killer whale species). SRKWs are listed as endangered under the ESA and were found to be genetically distinct from their nearest neighbors the Northern Residents (Krahn et al., 2004; Reeves et al., 2004). SRKWs differ by one fixed basepair from Northern Residents in the mtDNA haplotype based on the d-loop. However, the same haplotype that the Southern Residents have is found in another group of Resident killer whales farther to the west. Thus, the genetic signal was not considered "significant" in determining whether the Southern Residents had "marked" genetic differences (Krahn et al., 2004).

The Resident killer whales have a matriarchal social structure that shows both male and female offspring stay with their maternal group. As a result, social groups have only one haplotype present. This differs from false killer whales where even within the Hawaiian insular false killer whales both haplotypes are found within a single foraging group.

2.3.1.2. General remarks from case study examples

These examples reveal that population structure, at least to the Demographically Independent Population (DIP) level, is common among cetacean species with the ability to move long distances. Numerous DIPs for different species occur around the Hawaiian Islands, but minimum spanning trees suggest different histories of acquiring haplotypes with immigration from different areas being likely for most species. Higher diversity is expected for more abundant populations and this pattern generally holds, with the more abundant species of dolphins maintaining higher diversity than the less abundant species like false killer whales, pilot whales and the island populations of bottlenose dolphins. All of the cetacean species studied thus far exhibit some degree of structure within the Hawaiian Archipelago, and the patterns shown in minimum spanning networks are consistent with multiple immigration events. Short-finned pilot whales have low diversity similar to pilot whales elsewhere and to killer whales generally and, as will be shown later, to false killer whales. Most individuals are part of a cohesive group of haplotypes with the potential of local evolution of haplotypes. The other case where such local evolution seems likely is for California coastal bottlenose dolphins.

2.3.2. Genetics of false killer whales

Chivers et al. (2007) delineated false killer whales around Hawaii into two separate populations- Hawaii insular and Hawaii pelagic. This analysis focused on delineating demographically independent populations of false killer whales, which are units relevant to the ecological time-scale pertinent to the MMPA. That work has recently been extended with new samples, addition of nuclear markers and an analysis with a broader interpretation of the data (Chivers et al., 2010). The new analysis examined mitochondrial DNA (mtDNA) using sequences of 947 base pairs from the d-loop and nuclear DNA (nDNA) using 8 microsatellites. Additional samples since the 2007 analysis also moderately mprove the same distribution (Figure 2-8).

Three stratifications of the data examined genetic differentiation at different spatial scales (Chivers et al., 2010). The first *a priori* stratification, called the broad-scale stratification, recognized three strata: Hawaii Insular, Central North Pacific (CNP), and Eastern North Pacific (ENP) (Fig. 2-10a). Samples were assigned to the Hawaii insular population if they were collected from a group containing any individuals that were part of the Hawaii insular social network as determined by analyses of the photo-identification catalog and association pattern data (Baird et al., 2008a). In the second *a priori* stratification, called the fine-scale stratification, five strata were recognized: Hawaii Insular, Hawaii Pelagic, Mexico, Panama, and American Samoa (Fig. 2-10b). The third *a priori* stratification, referred to as the interisland stratification, recognized each of the main Hawaiian Islands as strata as in Chivers et al. (2007).

After extensive quality checking some haplotypes and duplicate individuals were removed from the earlier findings. All but one Hawaiian insular false killer whale had 1 of 2 closely related haplotypes that have not been found elsewhere (Fig. 2-11).

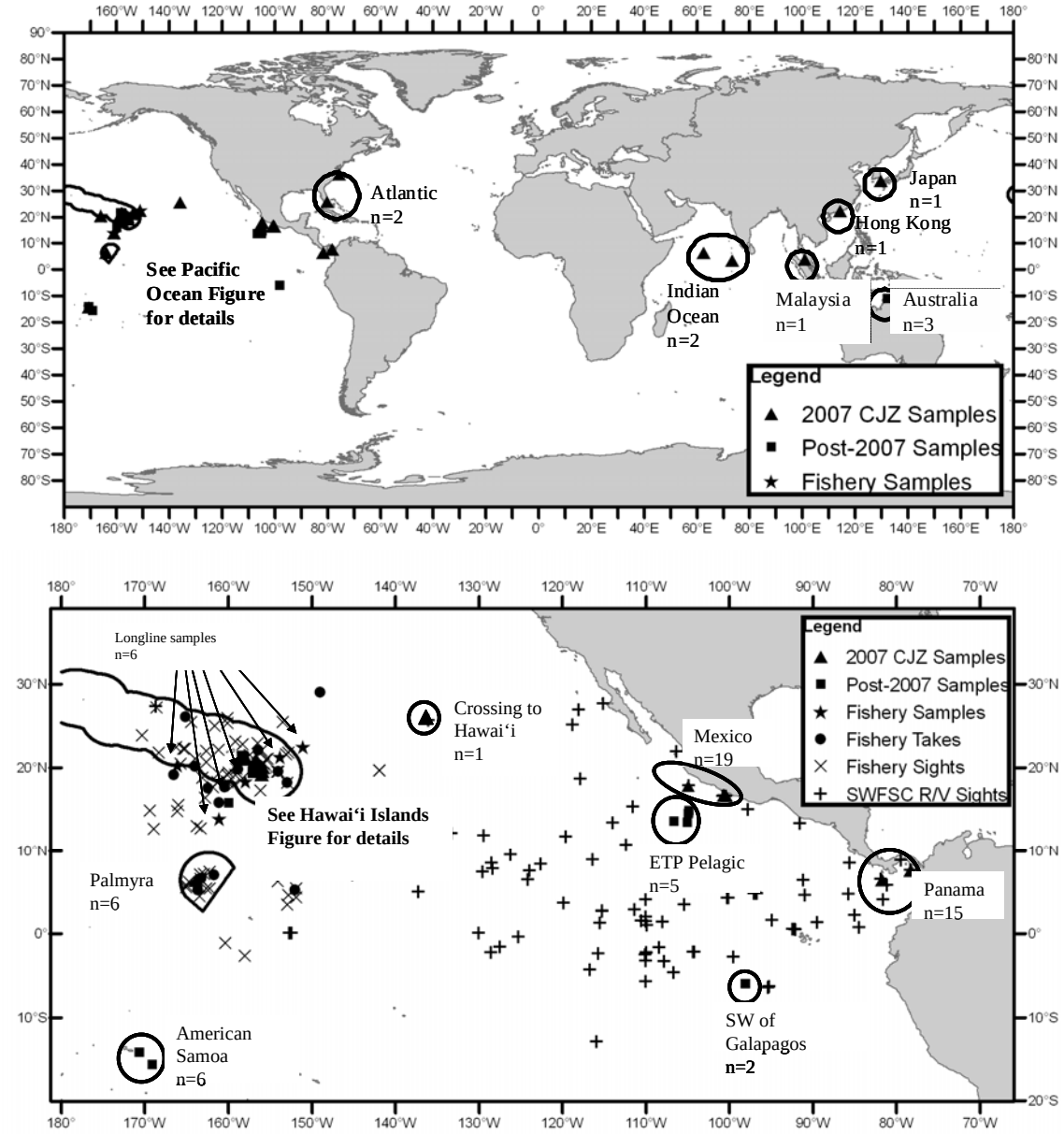


Figure 2-8. (a, upper panel) Collection locations of all samples used in Chivers et al. (2010). (b, lower panel) Samples collected in the eastern Pacific Ocean are shown here with research and fishing vessel sightings of false killer whales. The sightings data were collected on aerial and shipboard surveys conducted between 1974 and 2005 (Baird et al., 2005; Barlow, 2006; Mobley et al., 2000; Gerrodette and Forcada, 2005) and by observers working aboard longline fishing vessels between 1994 and 2004 (NOAA, NMFS, PIRO). See legend for guide to symbols; all collection locations for samples used in Chivers et al. (2007; denoted as "2007 CJZ") are indicated by a solid triangle with the samples added to this analyses indicated by a solid square, and the fishery collected samples indicated by a solid star.

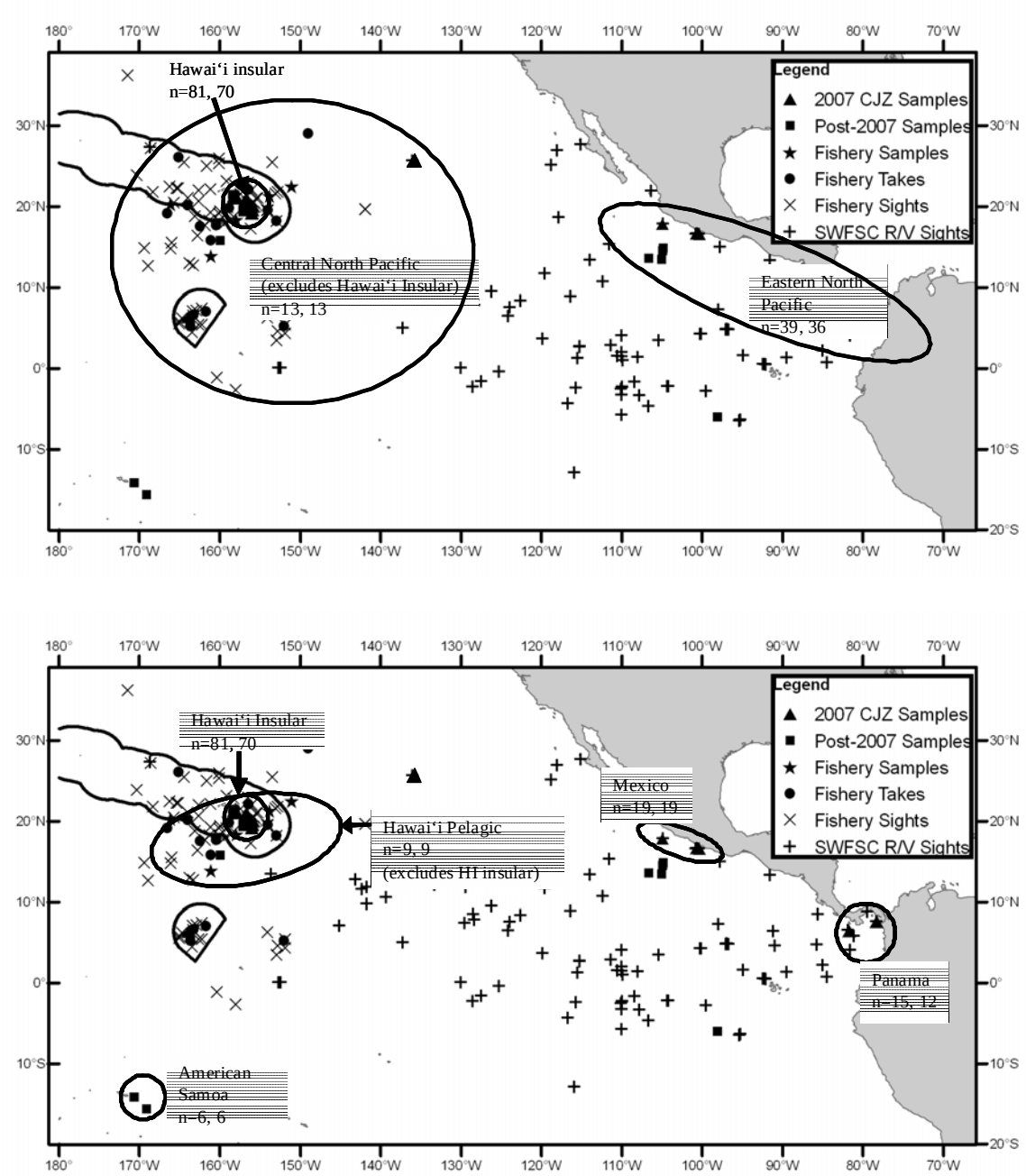


Figure 2-9. The mitochondrial and nuclear DNA data sets were analyzed as shown for the broad-scale (a) and fine-scale (b) stratifications.

The presence of two distinct, closely related haplotypes in Hawaiian insular false killer whales is consistent with Hawaiian insular false killer whales not accepting immigrants regularly from other areas. This pattern differs from those of Hawaiian stocks of bottlenose, spinner and spotted dolphins that all have minimum spanning networks suggesting multiple immigration events (see Section 2.3.1.1). The pattern of primarily closely related haplotypes shown in Hawaiian insular false killer whales is consistent with a strong social system that excludes immigrants or strong habitat specialization that

makes survival of immigrants unlikely (or both). One single individual was found in among Hawaiian insular false killer whales with haplotype 5. Although there is no photograph of the individual male with haplotype 5 to connect it directly to Hawaiian insular false killer whales, it was sampled within a group with such direct connections and assignment tests could not exclude that it belongs to the insular group. Given the low power of the current assignment test (with few microsatellite markers), the possibility of immigration (permanent membership in Hawaiian insular false killer whales but with an origin outside that group) cannot be ruled out. Likewise, the possibility that this individual was a visitor from the pelagic population cannot be excluded, nor is it possible to rule out that this individual is a lifelong member of the Hawaiian insular killer whales with a rare haplotype. The rare haplotype is sufficiently distantly related that it seems most plausible that this resulted from a separate immigration event (i.e., that immigrants are accepted on rare occasions).

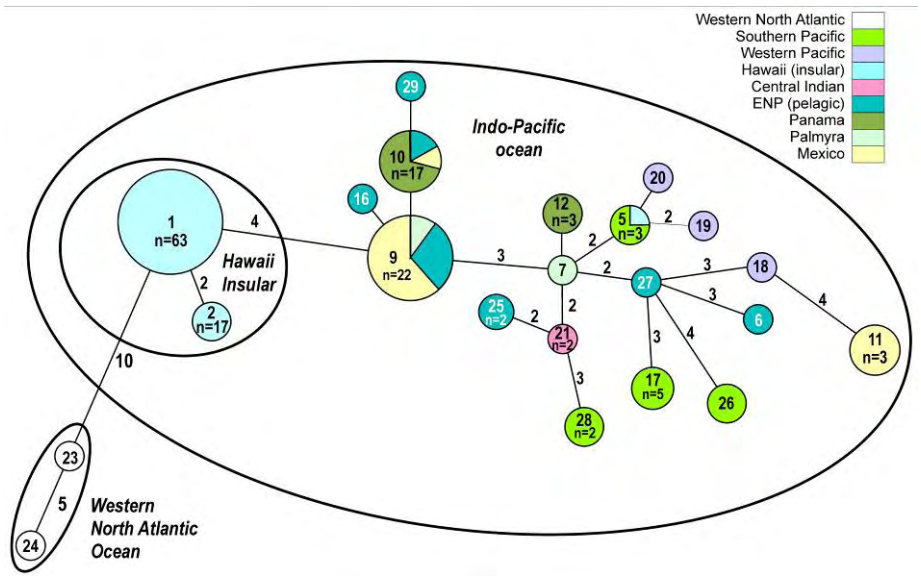


Figure 2-10. Minimum spanning tree for the 22 haplotypes identified for false killer whales. Each haplotype is identified by a number, which corresponds to the haplotype numbers (Table 2, Chivers et al., 2010). The observed frequency (n) for haplotypes found in more than one sample is given below (e.g., 63 individuals had haplotype 1). Each connecting branch is labeled with the minimum number of base pair changes if > 1.

The mtDNA data also show strong differentiation (with all *p*-values for Fisher exact and Φ_{ST} less than 0.0001) between Hawaiian insular false killer whales (n = 81) and both broad-scale strata (central North Pacific (n = 13) and eastern North Pacific (n = 39)) and fine-scale strata (Hawaiian pelagic (n = 9), Mexico (n = 19), Panama (n = 15) and American Samoa (n = 6)). Genetic divergence (Φ_{ST}) between the Hawaiian insular false killer whales and other strata examined ranged from 0.686 to 0.856. These magnitudes of differentiation were all consistent with less than one migrant per generation, which would roughly be a genetic divergence (Φ_{ST}) of 0.33. No significant differences were found among Hawaii, Oahu, and Maui once all potential duplicates were removed.

Nuclear results were also highly significant with all Fisher exact *p*-values ≤ 0.001 (Hawaiian Insular (*n* = 69), central North Pacific (*n* = 13) and eastern North Pacific (*n* = 36), Hawaiian pelagic (*n* = 9), Mexico (*n* = 19), Panama (*n* = 12) and American Samoa (*n* = 6)). The estimates of divergence between the Hawaiian insular strata and other strata ranged from 0.0189 to 0.056 for F_{ST} and from 0.004 to 0.043 for Jost's D. F_{ST} is expected to be less for microsatellites than when estimated from mtDNA because the mutation rate is higher and the formula differs because of the mode of inheritance. The expected value for one-migrant per generation would be between 0.031 and 0.075. Observed levels are a bit less, but there are numerous caveats that make interpretation to estimate gene flow of dubious value. There are many assumptions made in the model relating genetic differentiation to gene flow that are known to be violated, such as the population being in genetic equilibrium (see Whitlock and McCauley (1999) for detailed discussion of the pitfalls of indirect measures of gene flow). Further, there is no way to determine whether the level of differentiation is a result of ongoing migration or of recent common ancestry (Holsinger and Weir, 2009). The smaller magnitude of differentiation for nDNA than for mtDNA could indicate the potential for some male-mediated gene flow. Tests for differences between currently living males and females in level of differentiation were not significant for either mtDNA or nDNA. However, this test has no ability to detect differences in male versus female gene flow in the past. Chivers et al. (2010) give a number of hypotheses for the apparently different magnitude of signals between mtDNA and nDNA:

- There is a low level of male-mediated gene flow that was not apparent because of insufficient sampling of nearby groups of false killer whales and/or the test for male-mediated gene flow can only detect first-generation male migrants;
- The magnitude of nDNA differentiation is underestimated because of the high mutation rate of microsatellites;
- The magnitude of differentiation is not inconsistent with cases where selection has been shown to be strong enough for local adaptation.

The aforementioned uncertainties will best be resolved with more extensive sampling of false killer whales in nearby pelagic waters. Although the sample distribution is improved since the 2007 anlaysis, it remains poor in pelagic areas. The only full-scale cetacean survey of Hawaiian pelagic waters resulted in only 2 sightings of false killer whales in 4 months of effort, and the weather was too poor to obtain any high-quality identification photographs or biopsies (J. Barlow, pers. comm.). Fisheries observers are trained to obtain identification photographs and biopsy samples; however conditions during disentanglement usually result in photographs difficult to identify due to darkness, and prevent successful biopsy. Thus, obtaining an adequate sample of nearby pelagic waters may take decades. Estimating the level of male-mediated gene flow indirectly using these genetic data is of limited utility given the sample distribution and the questions generally about the quality of such indirect estimates by using simple models where most assumptions are violated.

The strongest data are the mtDNA data. The Hawaiian insular false killer whales have approximately half of the population sampled, and all but one individual has one of two

closely related haplotypes that have not been found elsewhere. This pattern alone argues for a strong possibility of a high degree of separation. The nuclear data show strong differentiation with very low *p*-values between Hawaiian insular false killer whales and all outside strata, including the Hawaii pelagic strata. Reliable inferences cannot be made about the magnitude of gene flow using the microsatellite data.

Chivers et al. (2010) used the analytical method described in Cornuet and Luikart (1996) and implemented in the program BOTTLENECK (Piry et al., 1999) to test for evidence of a recent decline in abundance within the Hawaiian insular population. The analysis takes advantage of the fact that when the effective size of a population is reduced, the allelic diversity of the population is reduced more rapidly than its heterozygosity, resulting in an apparent excess of heterozygosity given the number of alleles detected. BOTTLENECK detected evidence of a recent decline in N_e in Hawaiian insular false killer whales ($P = 0.015$), with all 8 loci exhibiting heterozygosity excess. Seven out of 8 loci also exhibited heterozygosity excess in the Mexico stratum. However, this result was not statistically significant ($P = 0.103$).

The microsatellite data were also used to estimate the effective population size of Hawaiian insular false killer whales as 45.8 (95% CI = 32.4–69.4). Because this population may have recently declined and the animals are long-lived, many of those individuals still alive likely were born prior to the decline. Thus, if anything, the estimate of effective population size is likely too high. Domestic animals have been shown to start displaying deleterious genetic effects (lethal or semi-lethal traits) when effective population size reaches about 50 individuals (Franklin, 1980). While negative genetic effects cannot be predicted for a group of individuals that are probably naturally uncommon with a strong social structure that limits genetic diversity, the current low effective population size is a concern.

2.4. Ecology of Hawaiian insular false killer whales

2.4.1. Current range and distribution

The range and boundaries of Hawaiian insular false killer whales may be assessed using ship and aerial survey sightings and location data from satellite-linked telemetry tags (Fig. 2-11). Satellite telemetry location data from 7 groups of individuals tagged off the islands of Hawaii and Oahu indicate that the whales move widely and quickly among the main Hawaiian Islands and use waters up to 112 km offshore (Baird et al., 2010; Forney et al., 2010). Regular movement throughout the main Hawaiian Islands is also documented by resightings of photographically-identified individuals over several years (Baird et al., 2005; Baird, 2009; Baird et al., 2010). Movements between islands can occur over the course of a few days, and although individuals were tagged on the leeward sides of the islands, they used both windward and leeward waters, moving from the windward to leeward side and back within a day (Baird, 2009; Baird et al., 2010). Ship survey sightings with photographs of individuals also confirm that Hawaiian insular false killer whales occur on both the windward and leeward sides of the main Hawaiian Islands (Forney et al., 2010). Some individual false killer whales tagged off the island of Hawaii have remained around that island for extended periods (days to weeks), but individuals from all tagged groups eventually ranged widely throughout the main Hawaiian Islands,

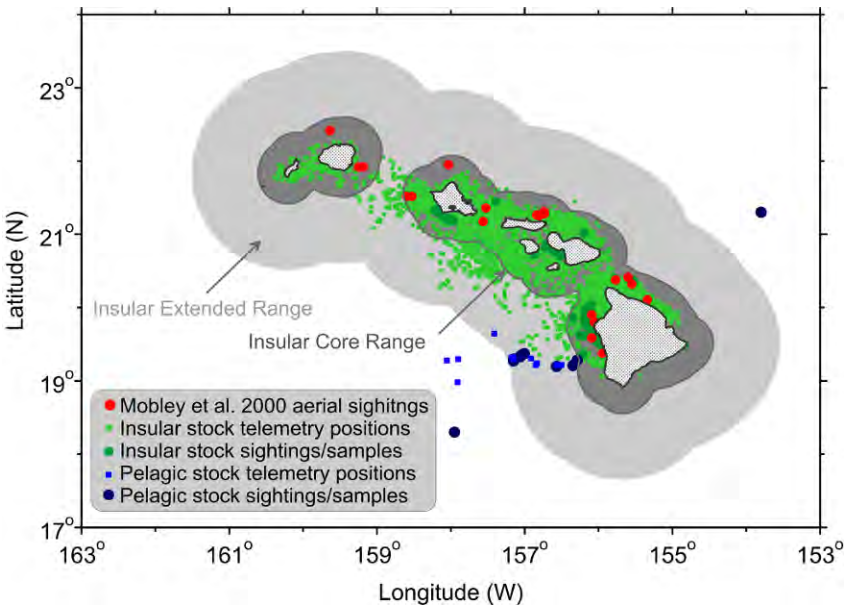


Figure 2-11. Sighting and satellite telemetry locations of tagged false killer whales around the main Hawaiian Islands. Data are available from 20 individuals have been tagged from 7 groups encountered off the islands of Hawaii and Oahu. Known pelagic false killer whale occurrence close to the islands is also shown. The gray shading delineates the current stock boundaries recognized in the NMFS 2010 Stock Assessment Report.

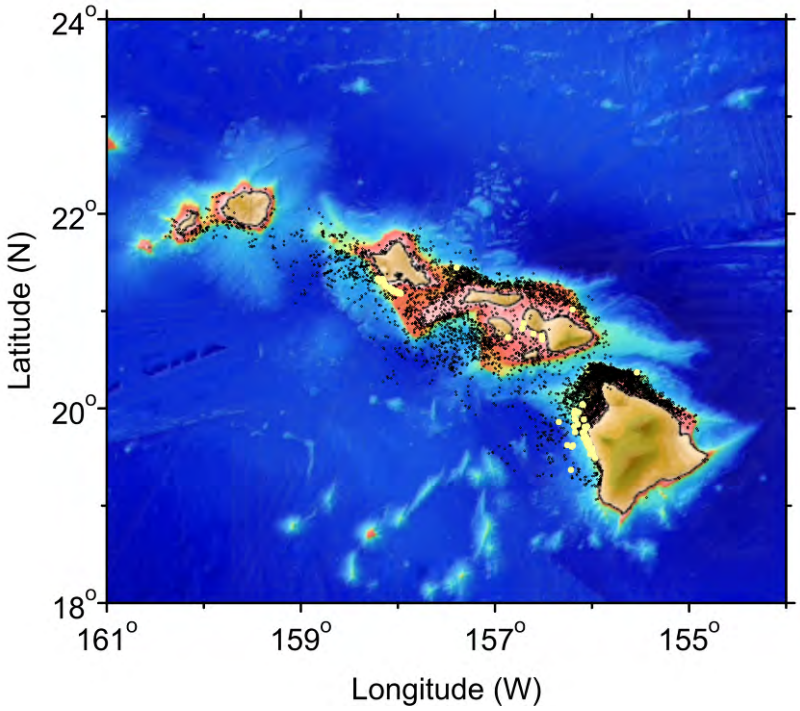


Figure 2-12. Hawaiian insular false killer whale satellite tag telemetry (black dots) and sighting locations (yellow dots) overlaid on 50-m resolution multibeam (nearshore) and 2-minute satellite altimetry (offshore). Satellite altimetry from Smith and Sandwell (1997).

including movements to the west of Kauai and Niihau (Baird, 2009; Forney et al., 2010). Based on locations obtained from 20 satellite-tagged insular false killer whales, the minimum convex polygon range was estimated to encompass 77,600 km^2 (M.B. Hanson, unpublished data).

The greatest offshore movements occurred on the leeward sides of the islands, although on average, similar water depths and habitat were utilized on both the windward and leeward sides of all islands (Baird et al., 2010). High rates of movement were documented, with one individual moving from Hawaii to Maui to Lanai to Oahu to Molokai, covering a minimum distance of 449 km over a 96-hr period (Baird et al., 2010). Tagged, as well as sighted individuals utilize habitat overlaying a broad range of water depths, varying from shallow (< 50 m) to very deep (> 4000 m) (Baird et al., 2010) (Figure 2-12). Tagged false killer whales have often demonstrated short- to medium-term preferences for individual island areas before ranging widely among islands and adopting another short-term residency pattern. It is likely that movement patterns of the whales vary over time depending on the density and movement patterns of their prey species (Baird, 2009).

Hawaiian insular false killer whales share a portion of their range with the genetically distinct (Chivers et al., 2007) pelagic population of false killer whales (Forney et al., 2010). Satellite telemetry locations from a single tagged individual from the pelagic population, as well as shipboard and small boat survey sightings, suggest that the ranges of the two populations overlap in the area between 42 km and 112 km from shore (Baird et al., 2010; Forney et al., 2010). The 2010 Stock Assessment report for false killer whales recognizes an overlap zone between insular and pelagic false killer whales between 40 km and 140 km from the main Hawaiian Islands based on sighting, telemetry, and genetic data (based on justification in Forney et al., 2010). Aerial survey sightings are within 40 km of shore are assumed to insular false killer whales based on what is known of the movements of the insular and pelagic populations. Because photographic or genetic identification of individuals is often required to determine the population identity of false killer whales in Hawaiian waters, we cannot be absolutely certain that sighting from the 1989 or 1993–2003 aerial surveys came from the insular population. Similarly, sightings of false killer whales by observers aboard fishing vessels cannot be attributed to the insular population when no identification photographs are obtained.

2.4.2. Population dynamics

There is no information available to assess whether the life history of Hawaiian insular false killer whales differs markedly from other false killer whale populations (see Section 2.1.4). However, there is also no evidence to show they are similar. As discussed earlier, false killer whales in Japan were larger and had a higher reproductive output than those in South Africa, and these differences were attributed to one or all of the following: colder more productive waters, response to exploitation and different food in the two regions (Ferreira, 2008). It remains uncertain whether Hawaiian insular false killer whales are more like those from Japan or South Africa.

2.4.2.1. Social structure

Molecular genetics results (Section 2.3) support the separation of Hawaiian insular false killer whales from the more broadly distributed Hawaiian pelagic false killer whales (Chivers et al., 2007, 2010). Matches from photo-identification of individual insular false killer whales also suggest functional isolation from the overlapping pelagic population of falsekiller whales (Baird et al., 2008a). Based on 553 identifications available as of July 2009, with the exception of 4 small groups of false killer whale observed (2 near Kauai and 2 off the Big Island), all false killer whales observed within 40 km of the main Hawaiian Islands link to each other through a single large social network. A large group of 19 identified individuals seen 42 km from shore and identifications from a number of other sightings of smaller groups do not link into the social network, providing even stronger evidence that the populations do not associate (Baird, 2009).

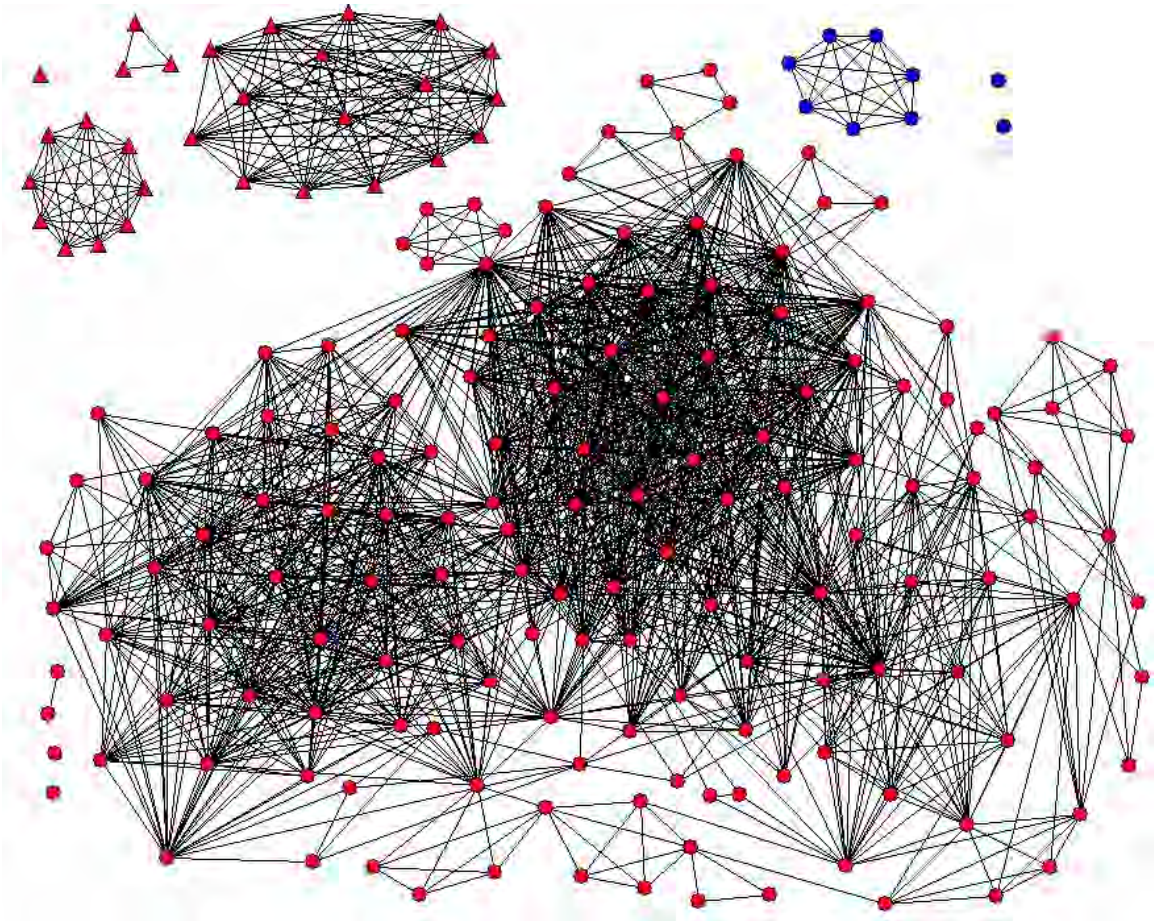


Figure 2-13. Social network diagram of false killer whale s photo-identified around the main Hawaiian Islands and offshore (40–130 km) from 1986 through May 2009 (Baird, 2009). Individuals encountered close to shore around Oahu, Maui, Lanai, and Hawaii are shown in red circles. All except four individuals photo-identified around these islands are linked in a single social network. Individual encountered close to shore around Kauai are shown in blue circles. Individuals encountered > 40 km from shore are shown in red triangles.

2.4.2.2. Historical population size

Historical population size is unknown. BRT members used density estimates from other areas together with the range inferred from telemetry data to suggest plausible ranges for historical abundance in the Population Viability Analysis (Appendix B). Using the estimated density of false killer whales around Palmyra Atoll, where the highest density of this species has been reported (Barlow and Rankin, 2007), and extrapolating that density to the 202,000 km^2 area within 140 km of the main Hawaiian Islands (proposed as a stock boundary for Hawaiian insular false killer whales in the Draft 2010 Stock Assessment Report), a plausible historical abundance is around 769 animals. To allow for uncertainty in this estimate, higher potential abundances are also estimated as the mean plus one standard deviation (1392 animals), and the upper 95% lognormal confidence limit (2461 animals). There are several important caveats. Even though Palmyra has a density that is high relative to other areas, it is unlikely that this represented a pristine population during the 2005 survey on which the estimate is based. Given the depredation tendencies of false killer whales, known longlining in the Palmyra area, and the fact that false killer whales are known to become seriously injured or die as a result of interactions with longlines, the possibility that current densities are lower than historical densities cannot be discounted. Although Palmyra is situated in more productive waters than the Hawaiian Islands, we do not understand enough about the feeding ecology, behavior, and social system(s) of false killer whales to know how or whether productivity might be related to animal density for false killer whales. Finally, we do not know whether the range of Hawaiian insular false killer whales extends into the Northwestern Hawaiian Islands, which would significantly increase the area and, hence, the estimates of historical abundance.

2.4.2.3. Current Abundance

Abundance of false killer whales in Hawaiian waters has been estimated using shipboard line-transect surveys of the entire Hawaiian EEZ, as well as aerial line-transect surveys and photographic capture-recapture analysis. The aerial surveys and photo-identification efforts covered only portions of the range of false killer whales in Hawaii. Hawaiian insular false killer whales were recognized as being separate from pelagic false killer whales only after the genetic results of Chivers et al (2007) has become available. Thus, many previously published estimates refer to both the insular and pelagic populations combined. Separate estimates of abundance are now available for both populations, with the best estimate of the size of the insular population being the result of capture-recapture analysis of photo-identification data. The 2009 Stock Assessment Report for insular Hawaiian false killer whales (Carretta et al., 2010) gives this as 123 individuals (CV = 0.72) (Baird et al., 2005). Recent reanalysis of the photographic data using more recent sighting histories and open population models has yielded new estimates of population size (Baird et al. presentation to 2009 PSRG), for two time periods, 2000–2004, and 2006–2009. These abundance estimates are considered overestimates because they do not account for known missed matches of individuals within the photographic catalog (Baird, pers. comm.). Two estimates of population size are presented for the 2006–2009 period because two groups of whales photographed near Kauai have not been seen to associate into the social network of false killer whales seen at the other islands (Baird, 2009). Those animals may come from the pelagic population, may come from another

undocumented population in the Northwestern Hawaiian Islands, or may represent a portion of the insular population that has not been previously encountered and photographically documented. Table 2-4 summarizes the current estimates of abundance for the Hawaiian insular population. The Hawaiian pelagic population numbers 484 individuals (CV=0.93) within the U.S. EEZ surrounding Hawaii (Barlow & Rankin 2007), though the population range is known to extend into high-seas waters, though the specific boundaries of that population are unknown.

Table 2-4. Current estimate of population size presented in the 2009 Stock Assessment Report and new estimates of abundance using more recent resighting histories and open population models.

Time period of estimate	Population size	Source of estimate
2000–2004	123 (CV = 0.72)	Baird et al., 2005
2000–2004	162 (CV = 0.23)	Baird et al., unpublished (PSRG 2009)
2006–2009 (w/out Kauai)	151 (CV = 0.20)	
2006–2009 (w/Kauai)	170 (CV = 0.21)	

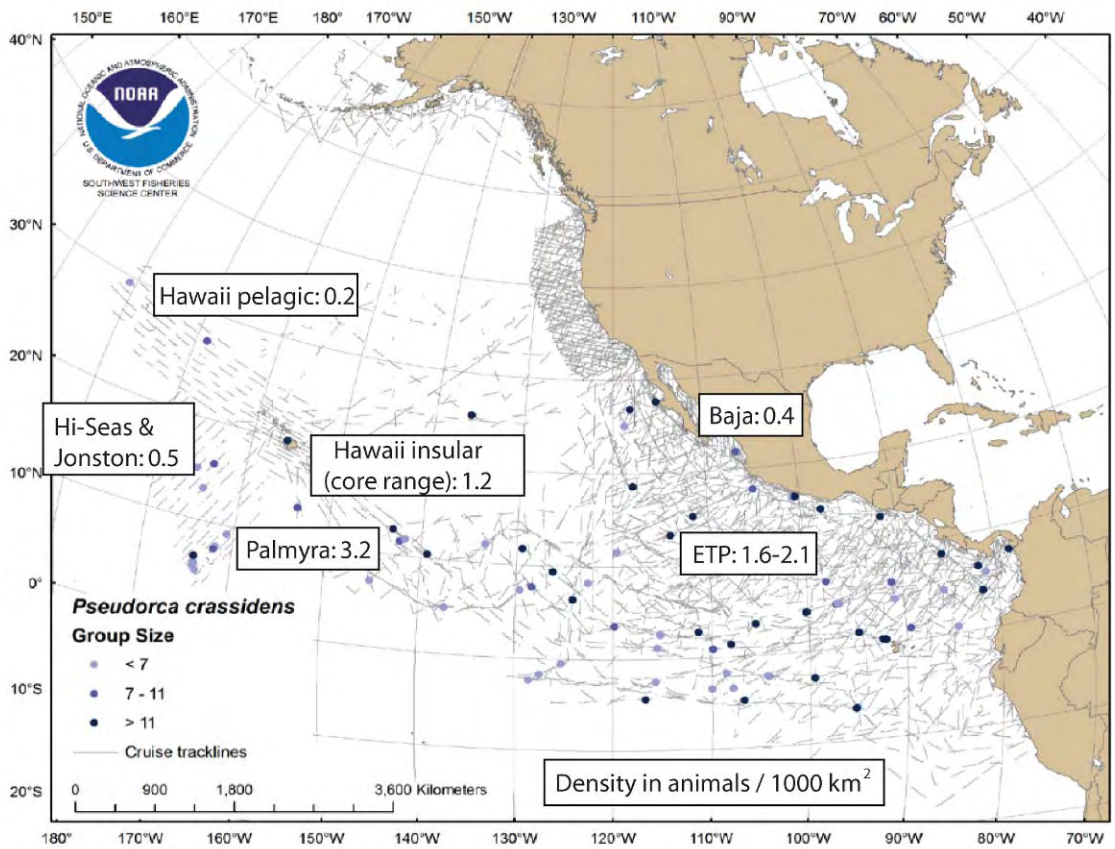


Figure 2-14. Visual line-transect survey sightings, color-coded by group size, overlaid on survey effort conducted during Southwest Fisheries Science Center cetacean assessment cruises in the Pacific (Hamilton et al., 2009). The estimated density of false killer whales in different survey regions is shown in the boxes on the figure. The "core range" of the insular population is the region within which only insular false killer whales are known to commonly occur (within 40km of the main Hawaiian Islands).

Although considered out-of-date by NMFS standards (NMFS, 2005), a previous estimate of insular false killer whale abundance based on aerial surveys from 1993 to 1997 was similar to the 2000–2004 estimate. Mobley et al. (2000) estimated 121 false killer whales (CV = 0.47) obtained from aerial survey sightings pooled over 3 survey years (1993, 1995, 1997). That estimate was likely negatively biased because the survey aircraft did not allow detection of cetaceans directly below the plane, and no adjustment was made for availability bias (animals below the surface and unavailable for sighting as the aircraft passed overhead).

In February 2009 a line-transect survey was conducted within 75 nmi of the main Hawaiian Islands where there were 6 sighting of false killer whales. Four on-effort sightings yielded a preliminary abundance estimate of 635 animals (CV = 0.77); however, not all groups could be attributed to the insular population. Some sightings occurred in the overlap zone, such that they may be pelagic false killer whales. Visual and acoustic observations of false killer whale behavior during the survey and the shape of the detection function (based on all sightings of false killer whale on previous NMFS Southwest Fishereis Science Center surveys) indicate this is an overestimation of abundance, particularly due to likely attraction of false killer whales to the survey vessel. Vessel attraction in other species has resulted in overestimation by as much as 400% (Turnock and Quinn II, 1991). Without a correction factor, which is currently not available, the Pacific Scientific Review Group noted (PSRG, 2010), and the BRT agreed, that this estimate does not represent best-available science because of a substantial upward bias and the lack of a correction factor for this species.

Although the absolute abundance of Hawaiian insular false killer whales is small, the core-area (within 40 km) population density (1.2 animals/1000 km^2) is among the highest reported for this species. False killer whale sightings, survey effort, and density estimates from the NMFS cetacean cruises in the Pacific (Figure 2-14) illustrate the general preference for more productive equatorial waters, with waning occurrence in the subtropics. The high density of the Hawaiian insular population suggests a unique habitat capable of supporting a larger population density than nearby oligotrophic waters.

2.4.2.4. Trends in Abundance

Aerial survey sightings since 1989 suggest that the Hawaiian insular false killer whale population has declined over the last 2 decades. A survey was conducted in June and July 1989 on the leeward sides of Hawaii, Lanai, and Oahu to determine the minimum population size of false killer whales in Hawaiian waters. False killer whales were observed on 14 occasions with 3 large groups (group sizes 470, 460, and 380) reported close to shore off the island of Hawaii on 3 different days (Reeves et al., 2009). The largest group seen in 1989 is more than 3 times larger than the current best estimate of the size of the insular population. Although the animals seen during the 1989 surveys are assumed to come from the insular population, based on their sighting location near the Big Island, it is possible that they represent a short-term influx of pelagic animals to waters closer to the islands. The average group size during the 1989 survey (195 animals) is larger than the typical average group size for the insular population (25 animals for encounters longer than 2 hours) during more recent surveys (Baird et al.,

2005), and is larger than that observed for the pelagic population (12 animals) (Barlow and Rankin, 2007). However, because of the location of the sighting and lack of evidence of pelagic animals occurring that close to the islands, it is likely that this group did consist of insular animals.

Five systematic aerial surveys were conducted between 1993 and 2003 covering both windward and leeward sides of all of the main Hawaiian Islands, including channels between the islands, out to a maximum distance of about 46 km from shore (Mobley et al., 2000; Mobley, 2004). A regression of sighting rates (8 groups in 1993, 9 in 1995, 1 in 1998, and none seen in 2000 and 2003 (Mobley et al., 2000; Mobley, 2004)) from these surveys suggests a significant decline in the population size ($p = 0.028$, $r^2 = 0.8429$ (Baird, 2009). The large groups sizes observed in 1989, together with the declining encounter rates from 1993 to 2003 suggest that Hawaiian insular false killer whales have declined significantly in recent decades.

It is possible that weather or other survey conditions are at least partially responsible for the decline in sighting rates from 1993 to 2003; however, there was no trend in the sighting rates for the four most commonly seen species of small cetaceans (spinner dolphin, bottlenose dolphin, spotted dolphin, and short-finned pilot whale). These four species represent nearshore and pelagic habitat preferences and span a range or body sizes from smaller to larger than false killer whales. It can be inferred from this evidence that variability in sighting conditions during the survey period did not have a major effect on sighting rates and therefore the rate for insular false killer whales has, in fact, declined.

A number of additional lines of evidence, summarized in Baird (2009), support a recent decline in Hawaiian insular false killer whale population size. Individual researchers in Hawaii have noted a marked decline in encounter rates since the 1980s and the relative encounter rate of false killer whales during the 1989 aerial survey was much higher than current encounter rates.

3. DPS Determination

3.1. 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.

If a population segment is considered discrete, NMFS must then consider whether the discrete segment is significant to the taxon to which it belongs. In carrying out this examination, the Service will consider available scientific evidence of the discrete population segment's importance to the taxon to which it belongs. This consideration may include, but is not limited to, the following:

- 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.

Because precise circumstances are likely to vary considerably from case to case, it is not possible to describe prospectively all the classes of information that might bear on the biological and ecological importance of a discrete population segment. Thus, in addition to the four criteria listed above, 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 distinctiveness and the significance questions include the ecological, behavioral, and genetic data summarized in Sections 2.2 and 2.3 above.

The BRT considered a number of factors related to Hawaiian insular false killer whale behavior, ecology, and genetics that relate to both the discreteness and significance criteria as defined above. However, some discreteness and significance factors as defined by the DPS guidance were not considered in detail as the team determined there was no evidence to suggest that those factors were applicable to Hawaiian insular false killer whales. In particular, there is no evidence to suggest that Hawaiian insular false killer whale are discrete from other false killer whale populations based on marked physical or

physiological separation, nor is the population delimited by international governmental boundaries, as it occurs entirely within U.S. waters. The team dismissed significance criteria based on loss of the discrete segment resulting in a significant gap in the range of the taxon because Hawaiian insular false killer whales already overlap in range with the Hawaii pelagic stock and the remaining nearshore area that would be vacated if the insular population were lost is geographically very small compared to the overall species range. The team also dismissed significance criteria based on the discrete segment representing the only surviving natural occurrence within its historical range, as false killer whales are known to occur throughout tropical and subtropical waters of all oceans.

Each remaining discreteness and significance criteria relevant to Hawaiian insular false killer whales was discussed separately and the arguments for and against each factor were examined in detail. Because the policy allows for consideration of other factors that are appropriate to the biology and ecology of the species, the BRT also considered whether loss of the discrete segment would result in the loss of cultural diversity for the taxon. Culture (knowledge passed through learning from one generation to the next) was evaluated as a significance factor within the Status Review of Southern Resident killer whales (Krahn et al., 2004), as culture is likely to play an important role in the viability of killer whale populations. Because false killers are thought to be similar to killer whales by having long-interbirth intervals suggesting a long period for young to learn effective foraging techniques and by having reproductive senescence in females suggesting older females may be important because of their long-term knowledge of prey distribution, this BRT evaluated the role of cultural significance during the determination of the DPS.

As noted in the Introduction, the BRT used the plausibility point method to evaluate each discreteness and significance factor potentially relevant to Hawaiian insular false killer whales, with each team member asked to allocate 10 points to the arguments for and against each factor. Point allocations from all team members were combined to produce percentage plausibility, where a score greater than 50% indicates that the arguments in favor of that factor carry more weight than the arguments against, and where higher percentages indicate greater confidence in those arguments than do lower values. Individual arguments for and against each factor, as well as all of the individual point-allocations are listed in Appendix A.

3.2. BRT determination of ESA discreteness

The BRT expressed strong support for a finding that Hawaiian insular false killer whales are discrete from other false killer whales. The team found that Hawaiian insular false killer whales are markedly separated from other false killer whales based on behavioral and ecological factors. In particular, the team noted that throughout its distribution, the false killer whale is considered a wide-ranging pelagic species not typically associated with coastal or island habitats (with exceptions noted above). Hawaiian insular false killer whales are behaviorally discrete because they are the only population of the species known to have movements restricted to the vicinity of an oceanic island group. Multiple lines of evidence support this conclusion. In particular, Hawaiian insular false killer whales are linked through a tight social network, with most identified individuals linked to all others through at least 2 distinct associations and with none of the identified

individuals linking to animals outside of the nearshore areas of the Hawaiian Islands. Further, phylogeographic analysis indicates that the Hawaiian insular population is isolated with 80 of 81 sampled individuals (roughly half of the population) described by haplotypes found exclusively within the Hawaiian insular population (Chivers et al., 2007, 2010). Finally, telemetry studies show all 20 satellite-tagged Hawaiian insular false killer whales have remained within the main Hawaiian Islands (Baird et al., 2009, Forney et al., 2010). Although it is not unusual for false killer whales to be observed close to land, long-term history of exclusive use of a mainland or island system has not been documented. Although information relating to false killer whale residency patterns at other oceanic islands is sparse, as are the number and distribution of genetic samples of central Pacific false killer whales other than the insular population, the BRT concluded that the weight of the evidence did support recognition of Hawaiian insular false killer whales as behaviorally discrete from other false killer whales in the taxon (81% of plausibility points).

Hawaiian insular false killer whales inhabit a unique ecological setting because they are found only in island-associated waters that are relatively shallow and productive compared to surrounding oligotrophic waters. Movement data indicate that Hawaiian insular false killer whales have adapted to a different ecological habitat than their pelagic conspecifics. Eleven insular false killer whales instrumented with satellite tags, whose locations have been evaluated in detail, show close association with the Hawaiian Islands in waters with a median depth of 575 m, substantially shallower than global oceanic habitat and the median depth of 3844 m for a satellite-tagged Hawaiian pelagic false killer whale (Baird et al., 2009). The shallower near-island habitat supports enhanced productivity and the tight association of Hawaiian insular false killer whales with this environment suggests they have adapted to a different habitat than their oceanic conspecifics, even when foraging on potentially similar prey species. Contaminant data also suggest that Hawaiian insular false killer whales forage on island-associated prey as they maintain elevated levels of fire retardants, a contaminant associated with urban environments, in their tissues (Ylitalo et al., 2009). The allocation of plausibility points recognizes that some uncertainty does remain, primarily due to the lack of information on false killer whales in other island archipelagos with similar areas of suitable habitat, such as the Cook Islands. The team allocated 70% of plausibility points in favor of ecological discreteness.

3.3. BRT determination of ESA significance

The BRT discussed at length the significance of the Hawaiian insular false killer whale with respect to the false killer whale global taxon. In particular, the BRT focused on the following three factors:

- the ecological setting of Hawaiian insular false killer whales, including diet and habitat, and how it differs from that of other false killer whales;
- genetic information for Hawaiian insular false killer whales and differences from other false killer whales in the tropical Pacific; and
- cultural diversity within false killer whales populations and what role Hawaiian insular false killer whales play in maintaining this diversity.

The BRT concluded that Hawaiian insular false killer whales are significant to the taxon in which they belong, based on the following: existence in a unique ecological setting, marked genetic differences, and maintenance of cultural diversity. Although there was greater uncertainty among the BRT when considering the significance factors, the team concluded by a 3:2 margin that Hawaiian insular false killer whales are significant based on each individual factor. Further, the BRT found there was stronger support for finding insular Hawaiian false killer whales to be significant to the taxon based on all factors taken together and the potential synergistic effects of those factors. The arguments favoring significance are outlined below.

3.3.1. Ecological setting

The BRT evaluated ecological significance in two different ways. First, it was considered whether a population with range restricted to the Hawaiian Islands, such as Hawaiian insular false killer whales, would automatically be significant because the Hawaiian Islands represent a unique ecological setting. Defining significance based on this rationale would imply that all populations of animals whose range is restricted to the Hawaiian Islands are ecologically significant. Although there was relatively little support for this definition of ecological significance within the BRT as a whole, one team member allocated points heavily in favor of it (Appendix A). The member noted that the ecology of the Hawaiian Islands is unusual and unique in many ways, and this is reflected in a high degree of endemism of various species. Although it is difficult to describe what makes Hawaii unique in explicit terms, the unusual ecological setting is derived from being a large but very isolated archipelago, consisting of steep volcanic peaks set in deep water, in a tropical setting, with substantial shelf and slope habitat around the islands but with deep water between the islands. Most relevant to false killer whales is the fact that the Hawaiian Archipelago contains an extensive amount of mid-depth habitat that false killer whales use, and it is difficult to identify another archipelago that contains such a large extent of that type of habitat in a similar ecological setting. The fact that the archipelago is large but very isolated from other island groups in the region allows for the development of unique evolutionary potential in populations that are uniquely tied to the Hawaiian Archipelago. Other members acknowledged the uniqueness of the Hawaiian habitat in general but felt that the argument needed to be made relative to the taxon. The example of bottlenose dolphins, which have discrete populations on each of the main Hawaiian Islands and are likely to have many thousands of such populations globally, influenced team members' positions. If the habitat alone was deemed unique such that all discrete populations found there would by definition also be significant, then numerous bottlenose dolphin populations (and those of other species) would automatically qualify for DPS status. Relatively few likelihood points (39%) were allocated in favor of this option.

The second set of ecological significance arguments was more specific, that Hawaiian insular false killer whales are significant because they persist in an ecological setting unique for the taxon. In particular, the team noted that Hawaiian insular false killer whales sometimes consume prey associated with island habitat, suggesting specialized knowledge of locations and seasonal conditions that aggregate prey or make them more vulnerable to predation. In an insular habitat such foraging grounds may occur more

regularly or in more predictable locations than on the high seas. The shelf and slope waters surrounding the main Hawaiian Islands, which is occupied by the insular false killer whales, differs from the surrounding oligotrophic waters by being relatively shallow and productive. In particular, the mesopelagic boundary community supports a local food chain that may support insular false killer whales; an island-mass effect may result in regions of higher productivity that may be targeted by false killer whales and their prey, and Hawaiian insular false killer whales have been observed to consume threadfin jack and filefish, two species that are primarily found in coastal waters. Additional lines of evidence that support significance based on existence in a unique ecological setting include the relatively high population density of Hawaiian insular false killer whales relative to other false killer whale populations, and satellite-linked telemetry based movement and photographic resighting data suggesting that Hawaiian insular false killer whales employ a unique, island-associated foraging strategy compared to other false killer whales. Although absolute abundance is low, the core-area density of the insular population is second only to waters surrounding Palmyra Atoll, and about 6 times higher than the pelagic population within the U.S. EEZ around Hawaii. The home range of the insular population appears to be extremely small compared to other medium body-sized, piscivorous odontocetes, such as resident-type killer whales, which maintain larger home ranges in more productive habitats. Finally, contaminant levels are consistent with consuming nearshore prey contaminated by local urban run-off. The BRT did recognize that there is insufficient data on other false killer whale populations with which to evaluate whether the foraging strategy of this population is entirely unique or if they have truly specialized on local resources. However, the team concluded that the potential existence of other, as yet unidentified, island-associated populations would not negate the significance of the Hawaiian population, because the mid-depth habitat targeted by the Hawaiian insular population is not common at other island archipelagos, and other local populations would likely be adapted to the specific conditions of that local habitat, which would differ from the habitat found around Hawaii. For example, the Galapagos Islands lie in productive equatorial waters, where elevated productivity extends over a very broad region and is not strictly associated with the islands. Although false killer whales are socially complex adaptable animals, such that pelagic animals may be able to alter their foraging strategies to colonize insular Hawaiian waters, it is unlikely that their habitat use would change to rely entirely on the Hawaiian Islands. Sixty-five percent of the plausibility points were placed in favor of the significance of the discrete population based on the existence in an ecological setting unique or unusual to the taxon.

3.3.2. Genetic differentiation

The BRT found that Hawaiian insular false killer whales differ markedly from other populations of the species in their genetic characteristics, supported by two lines of evidence: strong phylogeographic patterns of mtDNA haplotypes with all but one individual having one of two closely related mitochondrial haplotypes found nowhere else, and indications of low levels of gene flow based on microsatellite (nDNA) analysis. The genetic patterns are suggestive of a single colonization event with local evolution of a new mitochondrial haplotype. Microsatellite data in general are weaker on the question of genetic differentiation on long time scales because of the high mutation rate and mode of mutation that allows alleles of the same length to have different evolutionary histories.

The nuclear data do suggest strong differentiation of the Hawaiian insular population (all *p*-values < 0.001 for fine and broad-scale strata). Geneticists use one effective-migrant per generation as a rule of thumb for the level of gene flow below which adaptation to local habitat is likely. Comparisons using mtDNA of the Hawaiian insular animals to those in all other geographic strata indicate less than one migrant per generation. The molecular genetics results supporting the BRT conclusions are described in detail in Section 2.3.

The BRT acknowledged that uncertainty remains in the genetic comparisons of the Hawaiian insular population to other Pacific false killer whales. Although the Hawaiian insular population is very well sampled with approximately half of the individuals represented, pelagic false killer whale genetics are poorly sampled with large sampling gaps to both the west and east of Hawaii. It is possible that mtDNA haplotypes found in Hawaii could be found elsewhere in these inadequately sampled areas. Low levels of male-mediated gene flow also cannot be ruled out based on the genetic results. However, the weight of the evidence available was in favor of marked differentiation of the discrete Hawaiian insular false killer whale population from other populations of the species in its genetic characteristics (64% of plausibility points): thus making it significant to the taxon.

3.3.3. Cultural diversity

Finally, the team found that culture (knowledge passed through learning from one generation to the next) is likely to play an important role in the evolutionary potential of false killer whales because of the role transmitted knowledge may play in adapting to local habitats. The plausibility of cultural transmission is increased by the long period of dependence of calves as they learn to cooperatively forage and reproductive senescence in females, which is consistent with those females enhancing survival rates in the group through knowledge about local conditions that can only be acquired over decades. Documenting the relation of cultural transmission to the evolutionary potential of false killer whales would take data on birth and death rates of individual whales spanning decades and hence, there is no direct evidence. Learning is a common feature of other social odontocetes, including killer whales and belugas. The movements of insular false killer whales demonstrate a clear preference for particular areas of the inshore waters. Transfer of this knowledge from one generation to the next is consistent with the false killer whale life history traits of a long interbirth interval and reproductive senescence in females. These factors suggest that knowledge of local conditions may be important to successfully persist in this unique Hawaiian habitat. Learning to persist in this unique habitat may take many generations and, therefore, it is unlikely that the habitat would be quickly recolonized by the pelagic population if the insular population were removed. The BRT found Hawaiian insular false killer whales represent a significant cultural unit independent of other false killer whale populations (64% of plausibility points).

3.3.4. Synergy of ecological setting, genetic differentiation, and cultural diversity

The BRT decided to further evaluate the significance criteria based on all of the factors taken together. The particular combination of qualities makes this population truly unique; the population has adapted to this particular environment in a way that likely has

not and cannot occur with this species anywhere else in the world. The team concluded that the overall plausibility that Hawaiian insular false killer whales met the significance criterion was greater because three factors each met the significance criterion (71% of the plausibility points).

3.4. Conclusions for DPS determination

The BRT concluded that Hawaiian insular false killer whales are a DPS of the global false killer whale taxon. Uncertainty that remained, as measured by the minority of points assigned to arguments against individual discreteness and significance factors, was primarily a result of the lack of information on the biology and ecology of false killer whales at other island areas, and the gaps in genetic sample distribution to the west and east of Hawaii. However, lack of information on other potential island-associated populations does not negate the information that is presently available and the application of the best-available science standard, which does support designation of this population as a DPS.

Table 3.1. Summary of BRT evaluation of discreteness and significance based on point allocations to arguments for and against each factor considered in detail by the team (see Appendix A for a complete list of point allocations by team member). Ten points were assigned to arguments for and against each criterion by each team member, such that the total point allocation for each criterion is 80 points.

Criteria	**Points 'For'**	**Points 'Against'**	**Percentage (%) 'For'**	**Percentage (%) "Against'**
Discreteness				
Behavioral discreteness	65	15	81	19
Ecological discreteness	56	24	70	30
Significance				
Ecological significance	52	28	65	35
Genetic significance	51	29	64	36
Other factors (i.e. Cultural significance).	51	29	64	36
Synergy of ecological, genetic, and cultural factors	57	23	71	29

4. Assessment of Extinction Risk

4.1. Potential risk factors for Hawaiian insular false killer whales

Several factors may put the insular population of Hawaiian false killer whales at risk of extinction. The following sections describe each of these risk factors and, when possible, provide an assessment of the degree of risk.

4.1.1. Incidental interactions with commercial and recreational fisheries and aquaculture facilities

Only limited information is available on the nature and extent of interactions between insular false killer whales and commercial fisheries, recreational fisheries, or aquaculture facilities in Hawaiian waters. Potential threats associated with these activities include 1) hooking or entanglement of false killer whales in gear, 2) gear ingestion, 3) direct shooting or injury of false killer whales by fishermen, and 4) competition with fisheries for prey, such as tuna and billfish.

False killer whales have been documented taking catch or bait during commercial and recreational fishing operations around the Hawaiian Islands since at least the 1940s (Shallenberger, 1981; Nitta and Henderson, 1993), but little information is available to document the affects of these interactions on false killer whales. Animals may become hooked or entangled, and in some cases, fishermen have reported shooting at false killer whales and other dolphins or using explosives or chemicals to avoid losing catch or bait (Schlais, 1985; Nitta and Henderson, 1993; TEC, 2009). Based on photographs of Hawaiian insular false killer whales, Baird and Gorgone (2005) documented a high rate of dorsal fin disfigurements that were consistent with injuries from unidentified fishing line (3 out of 80 individuals or 3.75%, compared to 0–0.85% for other studied cetacean populations). Interactions with false killer whales have been reported for troll fisheries (Shallenberger, 1981; Zimmerman, 1983; Nitta and Henderson, 1993), deep-set and shallow-set longline fisheries (Nitta and Henderson, 1993; Forney and Kobayashi, 2007; McCracken and Forney, 2010), and possibly shortline or kaka-line fisheries (anecdotal reports of 'blackfish' interactions that may have been false killer whales, cited in Baird 2010). Some recreational fisheries in Hawaii target the same species as commercial fisheries (e.g., tuna, billfish) and use the same or similar gear, and might also be expected to experience interactions with false killer whales. The fisheries and the nature of reported interactions are described in more detail below.

4.1.1.1. Longline fisheries

Unless otherwise noted, fishery descriptions are summarized from Boggs and Ito (1993) and from the Western Pacific Regional Fisheries Management Councils most recently published Pelagics Annual Report (WPFMC, 2010). The Hawaii longline fishery primarily targets bigeye tuna using deep-set gear in a very broad radius around the Hawaiian Archipelago. The fishery also targets broadbill swordfish with shallow set gear, fishing mostly in the North Pacific Transition Zone north of Hawaii (Bigelow et al., 1999) where there is little chance of interacting with insular false killer whales, but with some effort south of the main Hawaiian Islands and inside of the potential range of the insular population north of the main Hawaiian Islands. Until 1980, distant-water

longliners from Japan caught between 1300 and 5000 t of tuna and billfish annually within the U.S. EEZ around Hawaii (Yong and Wetherall, 1980). Since 1980 no foreign longline fishing has been legally conducted in this zone, but the U.S. Hawaii-based fishery now harvests similar quantities of tuna and billfish in the EEZ.

Longline fishing gear consists of a mainline strung horizontally across 1–100 km of ocean, supported at regular intervals by vertical float lines connected to surface floats. Descending from the main line are branch lines, each ending in a single, baited hook. The main line droops in a curve from one float line to the next, allowing some control over fishing depth, which affects the efficiency with which species are captured. The Hawaii longline fishery began in 1917 off Waianae, Oahu, and by the 1930s longliners landed most of the 1000 t (ca. 2 million lb) of bigeye tuna, yellowfin tuna, and albacore, landed in the Territory of Hawaii. In the early years most of the catch was within catch reporting zones extending about 40 km off the coast of Waianae, Oahu, and off the coasts of Kona, Hilo, and Hamakua, Hawaii, These fishing areas remained important until a longline exclusion area extending a minimum of 46 km was established in the early 1990s. The fishery peaked in the 1950s and then declined through the mid 1970s. Local sale of fresh fish, mostly for raw consumption, provided a limited market that was easy to saturate, driving down the price.

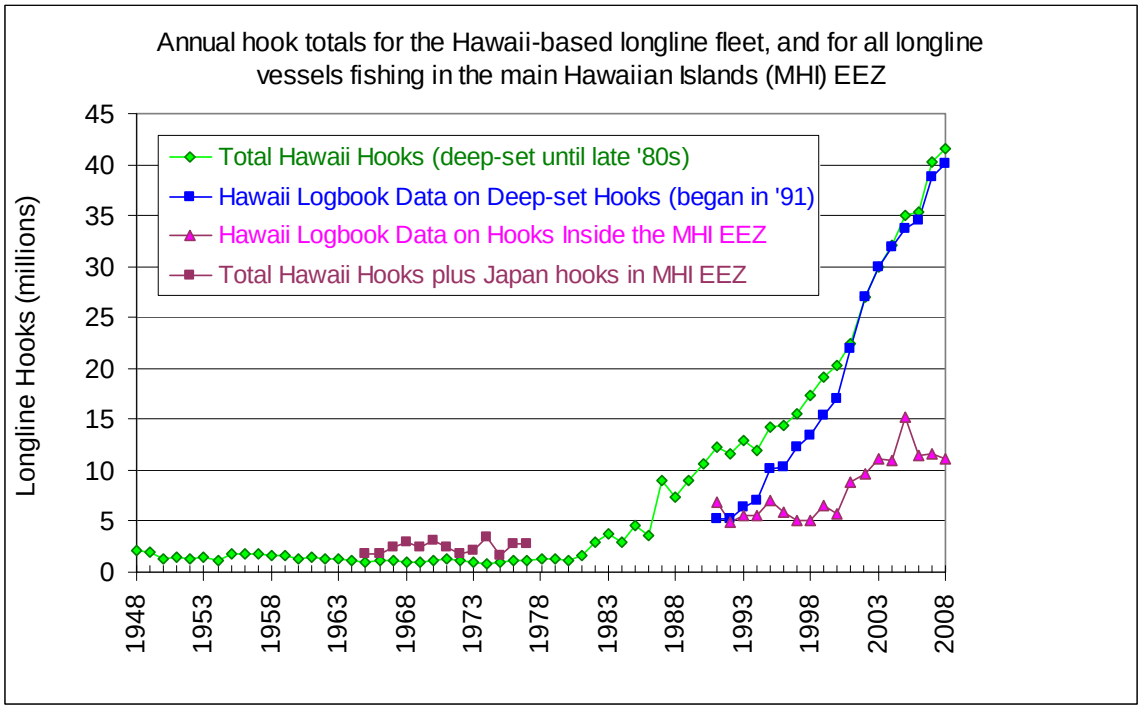


Figure 4-1. Total number of hooks set in the Hawaii longline fishery, 1949–2008.

Although the number of vessels declined after the 1950s, the amount of fishing gear deployed by each vessel increased. Using interpolation between occasional published descriptions (e.g., Shomura, 1959) of effort details that were not recorded by the Hawaii Division of Aquatic Resources (HDAR) fishermen's catch reporting system, Boggs and

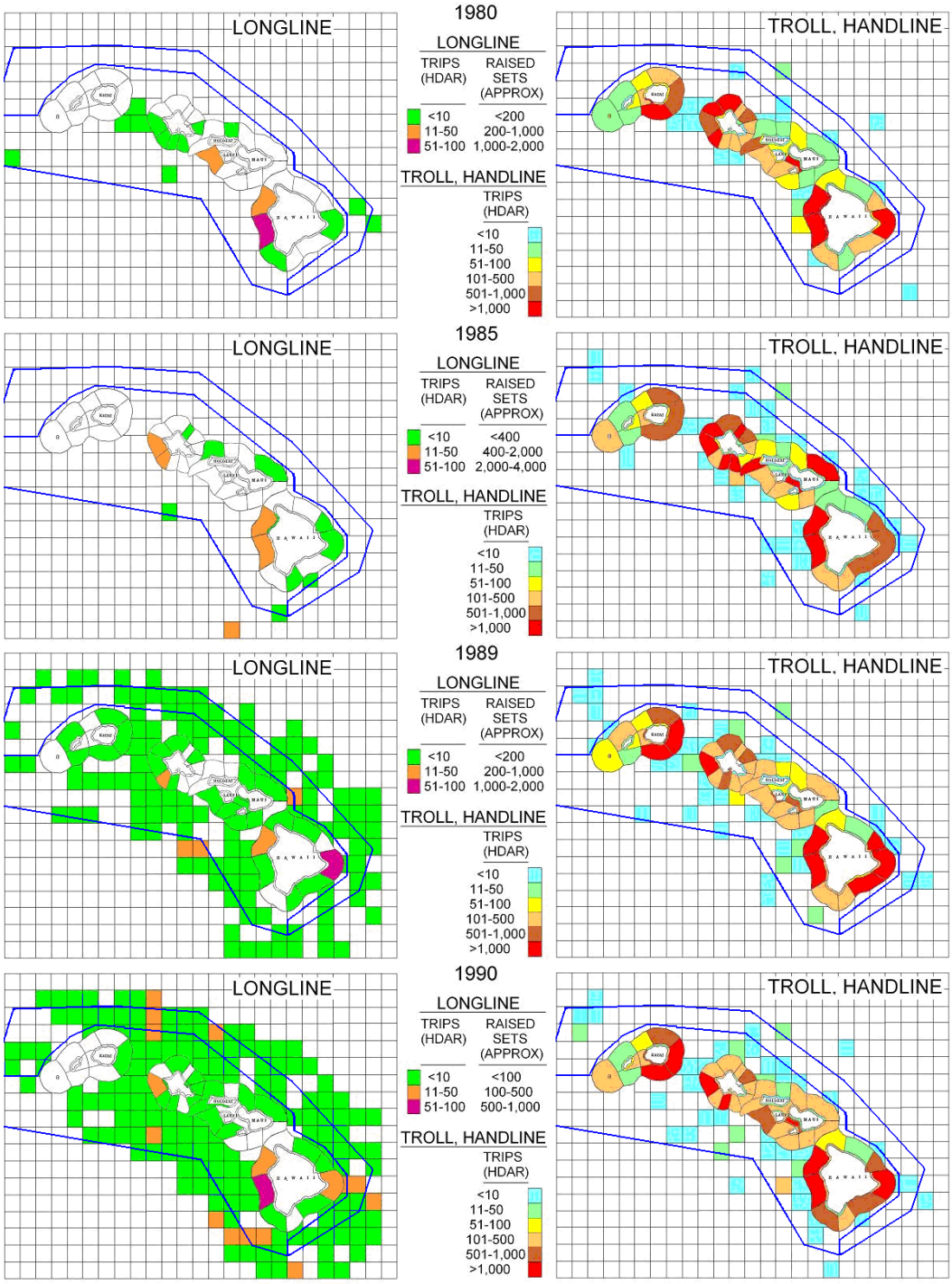


Figure 4-2. HDAR (unpublished data) on the distribution of longline, troll, and handline (palu ahi, ika shibi and deep-sea handline gear categories) during the rapid expansion of longline fishing nearshore from 1980 to 1990. Longline trips represent many days (with a set made on most days). Troll and handline trips usually represent one day of fishing. HDAR data on longline trips were raised and expressed as numbers of sets following methods outlined in Boggs and Ito (1993). Raising factors differ by year. The very narrow coastal areas wrapping around the islands indicate effort within 5 km of shore. The next concentric ring of areas around the islands extends to 42 km, and is the zone defined as "nearshore" in this document. The longline exclusion zones are shown as dark blue outlines and explained further in Figure 4-3.

Ito (1993) estimated the number of hooks that were deployed per trip from the late 1940s to the advent of the Federal logbook system in 1990. The total effort in annual hooks suggested by the interpolations is shown in Figure 4-1. In terms of total hooks deployed by the U.S. domestic fishery, the fishery declined slightly in the 1960s and 1970s, and then began to grow again in the 1980s. Considering days fished in the main Hawaiian Islands EEZ by Japanese longliners (Yong and Wetherall, 1980) converted to hooks assuming 2000 hooks per day fished and adding this to estimated domestic effort, total hooks in the MHI EEZ in the period of 1965 and 1977 was around 1.6 to 2.9 million hooks per year (Figure 4-1). This is a maximum estimate since the domestic hooks were assumed to all be in the main Hawaiian Islands EEZ. As the domestic fishery declined in the 1960s and 1970s foreign fishing in the main Hawaiian Islands EEZ increased, and then ceased in 1980.

The revitalization of domestic longlining in the 1980s was based on the development of new markets for fresh tuna on the U.S. mainland and in Japan, and also by the introduction of new shallow-set swordfish fishing methods in the late 1980s. When targeting bigeye tuna 12–25 hooks are deployed between floats with lots of sag to reach as deep as 400 m. In contrast, when targeting swordfish, only a few hooks are deployed between floats and the line is kept relatively taut so that it stays in the upper 30–90 m of water. The swordfish technique also soaks the gear primarily at night, employing fluorescent "light sticks" as an attractant, and using fewer hooks per set than deep-set tuna fishing. Data distinguishing deep-set from shallow-set fishing was not available until logbooks were required in 1991, so the increase in hooks set through the 1980s is described by total hooks (Figure 4-1). Shallow-set fishing declined after 1994 and was halted because of turtle bycatch in 2001–2004. Domestic fishing inside the main Hawaiian Islands EEZ was not distinguishable prior to logbooks. Hooks deployed inside the main Hawaiian Islands EEZ in the 1990s were double that estimated for the 1970s, and doubled again in the 2000s (Figure 4-1).

Participation in the Hawaii longline fishery approximately doubled from 37 vessels in 1987 to 75 in 1989 and doubled again to 156 (vessels with permits) by the end of 1991. As the Hawaii-based longline fishery expanded during the late 1970s through the early 1990s, longline fishing effort increased in waters near the Hawaiian Islands and within the range of insular false killer whales. The expansion in these nearshore waters within the 40-km core habitat of the Hawaiian insular false killer whales was pronounced during an influx of new fishery participants in the late 1980s (Ito 1991) and this led to conflicts in the predominant fishing areas of troll and handline fishermen (Figure 4-2). Longline fishing effort doubled from 1985–1986 to 1987–1988, reaching a level 5 times higher than the historical average (Figure 4-1), Effort in the main Hawaiian Islands EEZ had doubled over its 1970s levels by 1991, and nearshore (within about 40 km) longline effort increased as the fishery also greatly expanded its geographic range (Figure 4-2). Although the longline fishery subsequently expanded to an area from the equator to 45 N latitude, and from the eastern Pacific Ocean almost to the International Dateline, the deep-set tuna-targeting sector maintained the highest density of fishing effort within a 600 nm radius around the main Hawaiian Islands (He and Boggs, 1997). The growing

conflict between commercial longliners and near-shore troll and handliners was finally resolved in 1991 with exclusion zone limiting nearshore longlining.

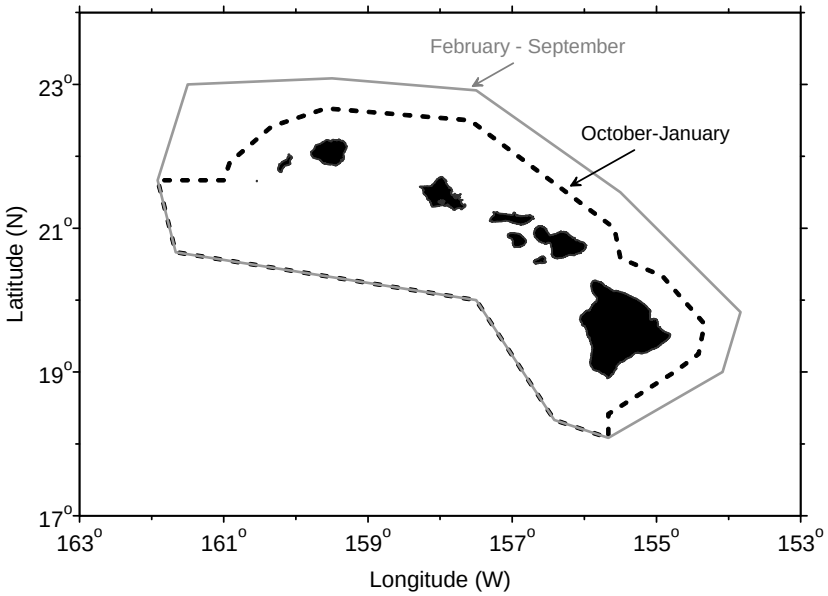


Figure 4-3. Main Hawaiian Islands longline exclusion zones established in 1991–1992. The solid line represents the February–September, 50–75 nmi longline exclusion zone. From October to January, that boundary contracts toward the islands (dashed line) yielding an exclusion boundary 25 to 75 nmi from the main Hawaiian Islands.

A new international system of catch limits that was first implemented in the Hawaii fishery in 2009 may prevent further increases in longline fishing effort for tuna. Although the fraction of total longline tuna catches that are from the EEZ around the main Hawaiian Islands has declined from about half to about a quarter over the last two decades, the absolute quantity caught in the EEZ fishery continued to increase through 2005, declining moderately thereafter (WPFMC, 2010).

No quantitative information on interactions with false killer whales is available during the expansion period of the longline fishery, but false killer whales were known to take large tuna (50–100 kg) from lines, leaving only the fish heads (Yuen, 1977). Logbooks were required beginning in December 1990, and in 1991 a longline exclusion zone was established to reduce conflicts between longline and other nearshore pelagic fisheries (56 *FR* 28116, June 1991; 56 *FR* 47701, September 1991). The exclusion zone was modified for the final rule in 1992 to allow some seasonal fishing closer to shore during October–January (Fig. 2-14; 57 *FR* 7661, March 1992).

Some longline effort continued within the longline exclusion zone (Table 4-1) because 4 small vessels were granted exemptions under certain circumstances (50 *CFR* Part 665, Subpart F, § 665.807). However, only two of these vessels used the exemptions, and longline effort within the October–January longline exclusion zone dropped from the logbook-recorded peak of 770 sets during 1991 (Table 4-1). It is not clear why sets continued to be recorded in logbooks inside the exclusion zone after the mid 1990s, when permits for exempted fishing expired. It has not been determined whether these sets

represent illegal fishing, errors in reported location (i.e., some reported locations are on land), or shortline sets (see Section 4.1.1.5) recorded in longline logbooks and mistakenly summarized as longline data.

Table 4-1. Annual effort (number of sets) reported in longline logbooks within three nearshore areas: a) the October–January longline exclusion zone; b) a 40-km radius around the main Hawaiian Islands that has been identified as a core area for insular false killer whales based on telemetry studies and sightings; and c) an expanded 140-km radius, which has been proposed as a new offshore range limit for insular false killer whales based on recent satellite telemetry results that show individuals moving at least 112 km offshore (Draft 2010, SAR; Carretta et al., 2010).

Year	Oct.–Jan. Longline Exclusion Zone	40-km core range of insular false killer whales	140-km extended range of insular false killer whales
Dec. 1990	267	56	429
1991	770	202	1723
1992	180	78	822
1993	80	46	713
1994	75	45	638
1995	60	26	727
1996	105	86	611
1997	108	65	363
1998	29	5	381
1999	31	21	536
2000	23	11	285
2001	17	6	677
2002	35	22	1198
2003	10	2	935
2004	4	1	1018
2005	4	0	1100
2006	6	1	821
2007	7	4	955
2008	1	1	668
2009	4	2	618
Total	**1816**	**680**	**15218**

There was possibly even more effort in the longline exclusion zone in the 1980s than indicated by the initial logbook data for 1990–1991, based on estimates of sets derived from HDAR data (Figure 4-2). By the time logbooks were established in late 1990, the State of Hawaii was already negotiating a “gentlemen’s agreement” whereby longliners agreed to avoid fishing within 20 nmi of shore (Boggs and Ito, 1993). Raised estimates of numbers of longline sets (Figure 4-2) are based on under-reported HDAR data on trips (Boggs and Ito, 1993). In 1980 and 1985 there were less than a dozen HDAR statistical areas adjacent to the coast with reported longline trips whereas in 1989–1990 there were about two dozen such areas. Underreporting may not have been evenly distributed by area, so the raised set estimates for these areas are somewhat speculative. The three or four coastal areas with the highest density of longline effort in 1980–1985 were off the coasts of leeward Hawaii (Kona and Kawaihae) leeward Lanai, and leeward Oahu

(Waianae and Maile). Estimated sets in these high density areas ranged from > 100 to > 1000 sets per area. In 1989–1990, a similar number of high-density areas within 23 nm was located off the coasts of Hilo, Kona, Kawaihae, Waianae, or Maili. One area off the coast of Hilo in 1989 had > 1000 sets, and one off the coast of Kona had > 500 sets.

Beginning in 1994, onboard observers in Hawaii-based longline fisheries have systematically recorded information on interactions with protected species, including sea turtles, seabirds, and marine mammals. Observer coverage initially was about 4% for all longline effort combined, but increased beginning in 1999. Since 2004, observer coverage has been 100% for shallow-set (swordfish targeting) trips and 20% for deep-set (tuna targeting) trips. Both fisheries operate on the high seas and within the U.S. EEZ around the Hawaiian Islands, Palmyra Atoll, and Johnston Atoll. False killer whales have been the most frequently hooked or entangled cetacean, primarily during tuna-targeting longline sets (Forney and Kobayashi, 2007; McCracken and Forney, 2010). Average mortality and serious injury, based on 31 observed interactions between 1994 and 2008, has been about 13 (CV = 0.37) false killer whales per year (calculated from estimates in Forney and Kobayashi, 2007; McCracken and Forney, 2010). Eleven false killer whales were observed injured or killed during 2009 throughout the range of the fishery[4].

Most of the observed interactions with false killer whales in the Hawaii-based longline fisheries occurred more than 140 km from the Hawaiian Islands, beyond the known range of insular false killer whales; however, a few interactions occurred closer to the Hawaiian Islands and may have involved insular animals (Figure 4-4). Tissue samples for genetic analysis were obtained for 6 of the 31 interactions observed during 1994–2008 (19%; representing only 3% of the total of 194 estimated takes). All of the samples were obtained at distances > 140 km from the Hawaiian Islands, and they were determined to be from pelagic, not insular, false killer whales. No samples are available for any of the false killer whales taken closer to the Hawaiian Islands. Prior to 2010, the takes were all assumed to belong to be from the pelagic stock (Carretta et al., 2009), but following a review of insular false killer whale movements and other factors, the 2003–2008 takes have been prorated to insular vs. pelagic animals based on geographic location (McCracken and Forney, 2010). During this 5-year period, one false killer whale was determined to be seriously injured within the 140-km extended range and an average of 0.60 insular false killer whales were estimated to have been killed or seriously injured annually (McCracken and Forney, 2010).

In summary, although the longline fishery has the most comprehensive data of any Hawaiian fishery on interactions with false killer whales, including bycatch estimates since 1994, there is uncertainty regarding the population identity of the false killer whales taken. Based on the locations of observed takes and the analysis of genetic samples, the majority of mortalities and injuries have probably been of animals from the pelagic population rather than the insular population, but some of the observed takes were within

[4] On January 19, 2010, NMFS announced the establishment of a Take Reduction Team (TRT) and convened the team to address the incidental mortality and serious injury of the Hawaii pelagic and insular stocks of false killer whales in the Hawaii-based, deep-set and shallow-set longline fisheries. The TRT is currently developing a Take Reduction Plan (TRP) as required in the MMPA.

the extended range of insular false killer whales. Given current observer coverage levels, only approximately 20% of all takes are observed and have known locations. Current analyses estimate that, on average, less than one insular false killer whale might be taken per year by longline fisheries (McCracken and Forney, 2010). This estimate assumes that the probability of taking Hawaiian insular vs. pelagic false killer whales is proportional to the estimated density of each population in the area where the takes occurred (NMFS, 2005). There are presently no data available to evaluate this assumption or whether there are other potential differences that might cause the two populations to behave differently with respect to longline gear. Historically, more frequent takes may have occurred when there was much greater overlap between insular false killer whales and longline fisheries.

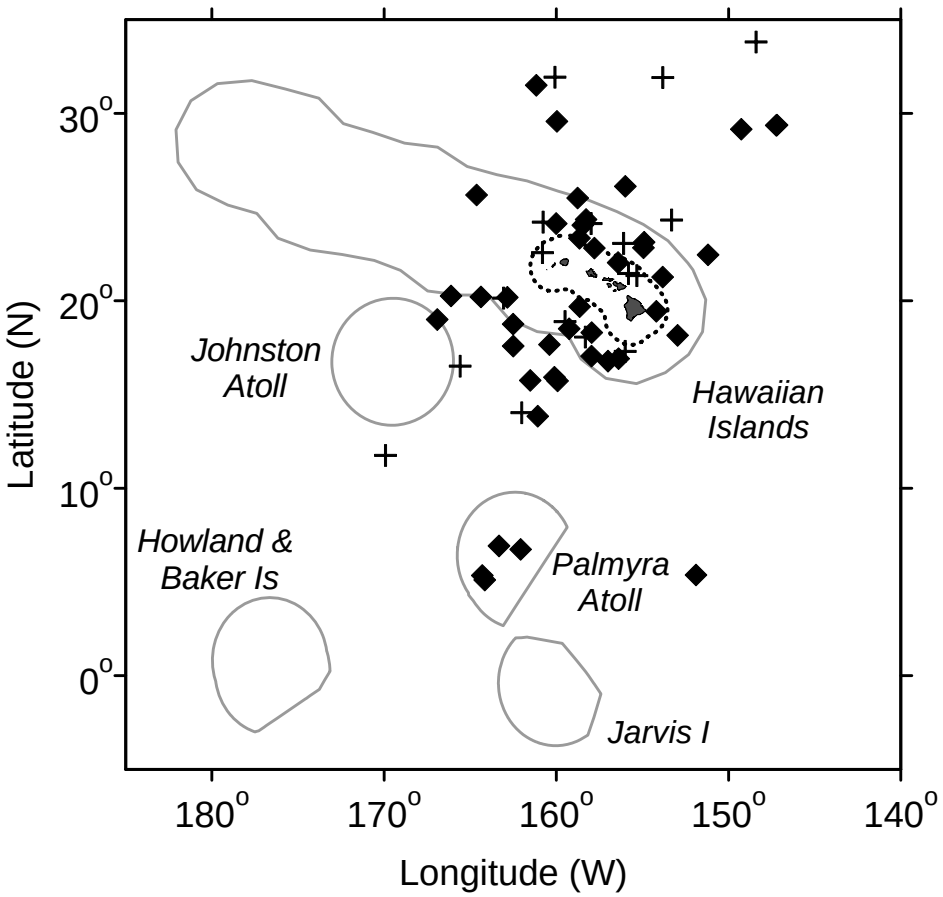


Figure 4-4. Locations of the 42 false killer whale takes observed in the Hawaii longline fisheries during 1994–2009 (◆), and the 17 possible false killer whales reported during 1994–2008 (+) (unidentified cetaceans for 2009 have not yet been reviewed). Four of the false killer whales and three of the possible false killer whales were observed taken within the 140-km extended range of the insular false killer whale population (dotted line). Gray lines are U.S. EEZ boundaries.

4.1.1.2. Nearshore troll fisheries

The evolution and operation of the Hawaii troll fishery are poorly documented (Boggs and Ito, 1993). Trolling with lures for pelagic species was a traditional Polynesian fishing method. The fishery targets epipelagic species like marlin, mahimahi, wahoo (*Acanthocybium solandri*), and the shallower-swimming tunas such as yellowfin and skipjack that will strike a lure or bait being pulled through surface waters. Commercial

catch reports to HDAR do not distinguish between different types of troll fishing (i.e., part-time, charter), and only fishermen who sell their catch are required to file reports. The available time series on troll effort consists of the annual estimates of fishing days that can be inferred from dated catches by license numbers reported to HDAR (Figure 4-5). The fishery totaled less than 3000 annual fishing days until the mid 1970s, and then increased steadily through the mid 1990s with effort declining somewhat since that time. A recreational sector of this fishery that is not represented in time-series data but that has been studied occasionally and is now being monitored by the Federal Marine Recreational Fisheries Service (WPFMC, 2010).

Although there are only few published reports of interactions between false killer whales and troll fisheries (Shallenberger, 1981; Zimmerman, 1983; Nitta and Henderson, 1993), anecdotal evidence indicates that false killer whales have been associated with troll fisheries for decades, sometimes taking catch or bait from lines. It is unknown whether animals get hooked or entangled in troll gear (as they do in longline gear). Fishermen have reported shooting at animals or taking other measures to protect their bait, catch or gear (Shallenberger, 1981), although it has been illegal to intentionally kill or injure cetaceans since the MMPA was passed in 1972. No interactions with false killer whales have been reported to NMFS under the Marine Mammal Authorization Program (required for fisheries listed on the LOF) and currently, there is no independent observer reporting system to document potential marine mammal interactions in the troll fishery.

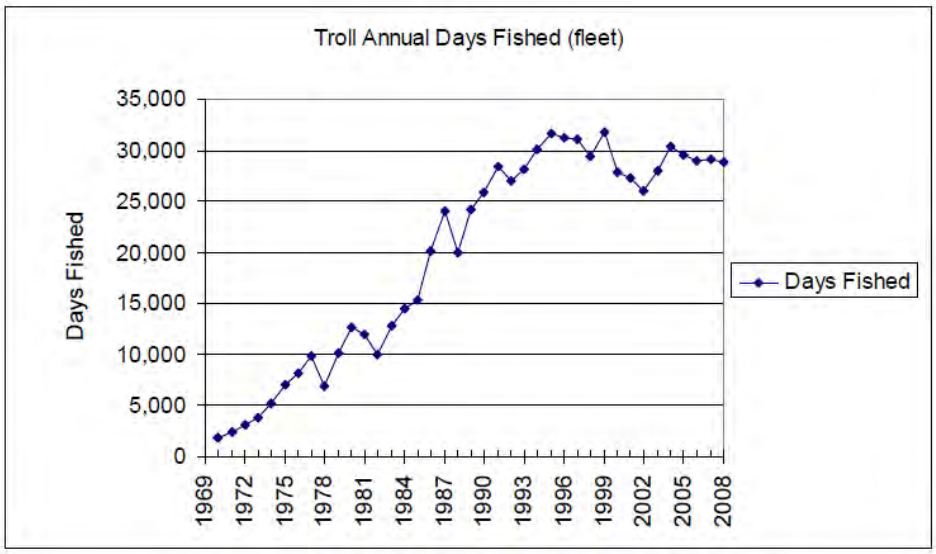


Figure 4-5. Total number of days fished by the commercial troll fishery, 1970–2008.

4.1.1.3. Handline fisheries

Handline fishing is a revitalization of an ancient Hawaiian method called “palu-ahi” for the use of “palu” (chum) to attract and hook ahi (yellowfin tuna). A baited hook on the end of the handline is laid against a stone with chum put on top, then this is covered in cloth and the line is wound around with the end tied with a slip knot that will be released. Then the line is lowered to the preferred depth (commonly 20–50 m) and jerked to untie the knot so that the baited hook and chum are released. Night-handline fishing is called

"ika shibi" from the Japanese names for squid (ika) and tuna (shibi). The ika shibi fishery is an outgrowth of a squid fishery that probably began in the 1920s and did not target tuna until the 1950s. Ika shibi fishermen attract squid to the fishing vessel with a light and catch the squid on jigs or with a gaff. The squid are then used as bait (Yuen, 1979; Ikehara, 1981; Boggs and Ito, 1993). These and several other variants of handlining (deep-sea handlining, drifting handlining, etc.) were not always distinguished as separate fishing methods by HDAR, and all methods were combined to estimate annual days fished (Figure 4-6). One handline method used on the Kona coast of Hawaii involves locating dolphins associated with yellowfin tuna schools, and dropping baited lines in front of the advancing cetaceans (C. Boggs, pers. observ., 1995). Trends in the palu ahi and ika shibi handline sectors derived from fishery data have been illustrated separately elsewhere (Impact Assessment, 2007).

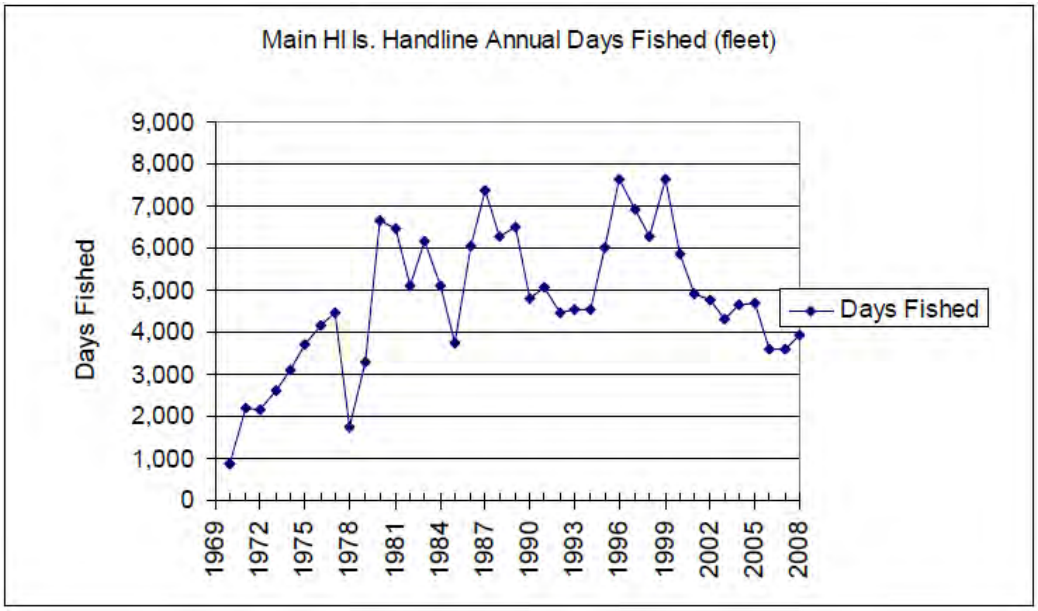


Figure 4-6. Total number of days fished by the commercial handline fishery in the main Hawaiian Islands, 1970–2008.

The fisheries described here take place primarily near the main Hawaii Islands from about 3 to 20 miles from shore, and do not include the offshore handline fishery that developed at weather buoys and the Cross Seamount during the 1990s (Impact Assessment, 2007), since that fishery operates beyond the range of the insular false killer whales. The fishery occurs primarily off the Big Island of Hawaii. Annual handline days fished in the main Hawaiian Islands increased from the 1970s through the 1990s with dramatic oscillations at 3- to 9-year intervals, and has been decreasing since 1999. All handline catches were sold on the island of Hawaii until 1971 when the rising price for tuna and reduced shipping costs made air shipment to Honolulu economically feasible, contributing to the expansion of the troll and handline fisheries. The magnitude of the recreational-subsistence sector of the handline fishery is mostly unquantified (Impact Assessment, 2007).

No interactions with false killer whales have been reported to NMFS under the Marine Mammal Authorization Program (required for fisheries listed on the LOF) and currently there is no independent observer program for Hawaii's handline fisheries. Anecdotal reports indicate that interactions between handline fisheries and cetaceans have been common since at least the 1970s. Bottlenose dolphins or rough-toothed dolphins have generally been implicated rather than false killer whales. No information is available to determine whether handline fishermen shoot at cetaceans or take other harmful measures to try to prevent the loss of bait or catch, as has been reported for the other fisheries (Shallenberger, 1981; Zimmerman, 1983; Nitta and Henderson, 1993).

4.1.1.4. Bottomfish fisheries

The Hawaiian bottomfish fishery targets a variety of snappers and groupers using hook-and-line methods that are based on Native Hawaiian techniques (Haight et al., 1993; Simonds, 2003). The fishery operates within the main Hawaiian Islands (Figure 4-7) and the Northwestern Hawaiian Islands, generally at depths between 50 and 200 fm (91–366 m). There are commercial and recreational components to the bottomfish fishery, but the distinction between the two is blurred as some recreational fishermen may sell part of their catch. Around the main Hawaiian Islands, the number of vessels targeting bottomfish has varied between 250 and 500 each year (WPFMC, 2006).

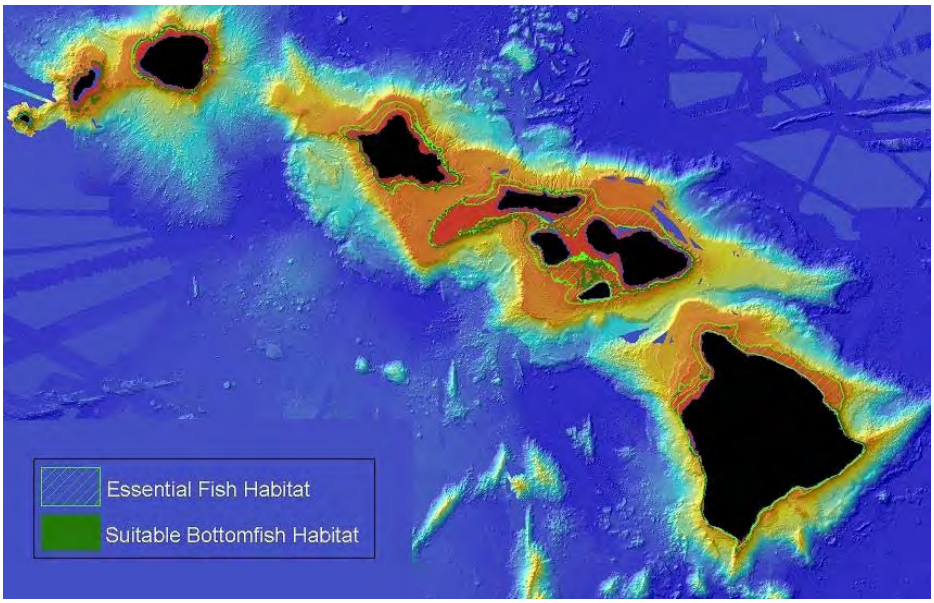


Figure 4-7. Bottomfish habitat around the main Hawaiian Islands. M. Parke, NOAA-PIFSC.

As with the handline fishery, there is no reporting system to document potential marine mammal interactions in the bottom fishery, but there is anecdotal evidence that bottlenose dolphins and rough-toothed dolphins have interacted with bottom fisheries for many decades, and fishermen have tried a variety of deterrents, seal bombs, or even illegal poisoning or shooting to deter dolphins (Schlais, 1985; Nitta and Henderson, 1993; Kobayashi and Kawamoto, 1995). There is no evidence that false killer whales have been involved in these interactions.

4.1.1.5. Shortline and kaka-line fisheries

The small-scale shortline and kaka-line fisheries have been reported in the State of Hawaii's Commercial Marine License system since 1999. The early reports were all classified as kaka lines (shortline was not an option in the logbooks), but since 2003 there have been reports of shortlines as well. The fisheries use gear similar in some ways to longline gear, but with a mainline length of less than 1 nmi and they use a greater variety of floats and hooks. Whereas longlines frequently set from 800 to several thousand hooks in a day's operation (one set) kaka-line and shortline sets involve a few hundred hooks or fewer per day's operation, spaced much closer together and suspended by fewer floats. Kaka-line and shortline operators are allowed to fish in nearshore waters. Kaka lines are set on or near the bottom or in shallow midwater to target a variety of species, including bottomfish, opelu (*Decapturus pinnulatus*), mahimahi, wahoo, yellowfin tuna, and other nearshore or reef-associated species (WPFMC, 2010). The shortline fishery was developed to target bigeye tuna or lustrous pomfret when they concentrate over the summit of Cross Seamount (290 km (180 mi) south of Hawaii), but it operates in other areas as well. The gear is set before dawn and is retrieved about 2 hours later (74 *FR* 58879 November 2009).

Any person with a State of Hawaii commercial marine license (CML) may participate in kaka-line or shortline fisheries, but only 5 to 11 vessels reported participation annually during 2003–2008. Catches by the kaka line and shortline fisheries, excluding the shortline operations at Cross Seamount which are outside the known range of insular false killer whales, declined from 45,000 pounds to 5000 pounds per year from 2003 to 2007 and rose again to about 73,000 pounds in 2008 (WPFMC, 2010). At their peak level in 2008, these catches amounted to less than 2% of the catches by troll and handline fisheries operating inside the range of the insular population of false killer whales (Section 4.1.1.2). The shortline fishery was added to the LOF in 2010 and the kaka-line fishery will be added to the 2011 LOF. No interactions between the shortline fishery and false killer whales have been reported to NMFS under the Marine Mammal Authorization Program (required for fisheries listed on the LOF) and currently, there is independent observer program for monitoring bycatch in either the shortline or kaka-line fishery. There are anecdotal reports of interactions with cetaceans off the north side of Maui, but the species and extent of interactions are unknown (74 *FR* 58879 November 2009). Based on the similarity of these fisheries to longline fisheries (with respect to gear type and target species), it is likely that false killer whales are involved; however, the nature and extent of any such interactions are unknown. Although there is no evidence to suggest a disproportionate threat from the kaka line and shortline fishery compared with other, much larger fisheries operating within the known range of insular false killer whales, the 2008 increase in catch suggests that the shortline fishery could expand rapidly. And perhaps a stationary fishery with multiple catch online at once is more attractive and rewarding to a false killer whale group than a few handlines or moving troll lines.

4.1.1.6. Recreational fisheries

Hawaii's recreational fisheries use many of the same gear types as commercial fisheries, and might be expected to interact with false killer whales in a similar manner. However, no specific information is available on interactions with false killer whales.

4.1.1.7. Aquaculture

There is a long cultural history of developing and maintaining fish ponds along the coastlines of the Hawaiian Islands (Costa-Pierce, 1987), but these fish ponds are located onshore and are not likely to affect false killer whales. .

In 2001, Cates International, Inc. received approval to rear moi (Pacific threadfin, *Polydactylus sexfilis*) and initiated a lease for a 28-acre aquaculture facility about 2 miles off the shore of Ewa Beach, Oahu, in January 2001. The moi is indigenous to Hawaii and was once reserved solely for royalty, with fishponds built along Hawaii's coastline for moi culture. The original aquaculture site included two submerged cages, and later expanded to four cages, anchored in an average water depth of 140 ft using a mooring system that maintains the cages about 30–40 ft below the surface. An expansion of the operation has been proposed (Aquaculture Planning & Advocacy, 2009) and would involve a seaward extension of the facility to encompass 61.59 acres and a total of six new 6000 m^3 cages. Cage frames are covered with a tight 35-mm mesh netting of a "Spectra" fiber, which is a strong, UV-resistant synthetic material. Divers enter through zippered openings in the mesh.

The second aquaculture facility was opened 2004 by Kona Blue Water Farms (KBWF), and is located in 50–75 m water depth on a 90-acre site located a few kilometers off Kona, Hawaii, including eight submersible Sea Station net pens (see Kona Blue Water Farms 2009, Supplemental Draft EIS). KBWF has applied for a permit to replace these pens with newer pens, including two production net pens and up to three pens to be used for nursery rearing and for research. A variety of designs have been proposed, but all would be covered by low-stretch, taut-mesh material or rigid plastic or metal. The original permit allowed pens of 15 m diameter, and the new pens would be up to 30 m in diameter. The overall production capacity of the farm would remain about the same (24,000 m^3).

A new open ocean aquaculture project has been proposed that would operate from a 1-km^2 ocean site located 4.8 km off the North Kohala coast of the island of Hawaii, in a water depth of 402 m (Final Environmental Impact Assessment, Tetra Tech, Inc., 2009). The facility would contain up to 12 large 'oceanspheres', in which 6000 t per year of yellowfin tuna and bigeye tuna would be raised by 2013. The spheres are designed to be self-contained, without sharp surfaces or dangling lines to entangle wildlife or interfere with navigation or fishing, and will be composed of an anodized aluminum frame covered in taut, anti-fouling netting made from soft, torque-free braided Kevlar. They will not be tethered, but rather will actively maintain station using an automated, dynamic positioning system that uses a Global Positioning System (GPS) and a telemetry-controlled water jet propulsion.

The potential impacts of these aquaculture facilities are unknown, but open-water pens tend to attract wild marine fishes, including sharks, and may also attract cetaceans looking for prey. As a result of this association, cetaceans may be at risk of injury from contact with or entanglement in netting or other materials, intentional harassment (e.g. through the use of acoustic harassment devices) by facility operators (Goldburg et al.,

2001), or predation by sharks associated with the facility. The taut net used for these facilities has not been documented to entangle marine mammals in other U.S. aquaculture operations (Final EIS, Tetra Tech, Inc. 2009). The toxic antifouling chemicals used on the facility have potential to enter the surrounding waters and impact wildlife, but risks to false killer whales are unknown.

4.1.2. Prey availability

4.1.2.1. Prey preferences—observational data

In Hawaii, false killer whales appear to forage primarily on large pelagic fish (Brown et al., 1966; Shallenberger, 1981; Baird et al., 2008a; Baird, 2009). Insular false killer whales have been observed feeding on yellowfin tuna, albacore tuna, skipjack tuna, scrawled file fish, broadbill swordfish, mahimahi, wahoo, lustrous pomfret, and threadfin jack (Baird, 2009). However, this information is primarily derived from observational data, with only one stranded animal, and its stomach contents, being available from this area, was notable because it included squid beaks (Shallenberger, 1981; Tomich 1986). Although false killer whales in Hawaii have not been observed eating squid, this lack of observations is surprising because they are known to bring prey to the surface and share among the groups (Baird et al., 2008a; Brown et al., 1966). Consequently, deep prey, such as squid, which have been commonly reported as false killer whale prey in other areas (Bullis and Moore, 1956) and are likely prey of pilot and beaked whales in Hawaii, might be under-represented in the evaluation of diet.

Table 4-2. Fishes depredated by cetaceans, as observed in the Hawaii deep-set tuna longline fishery. Observations are lumped here into taxonomic categories to account for the lack of identifiable remains of the most frequently depredated taxa. Percentages of the catch that are depredated are shown both for depredated sets only, and for all observed sets.

	Data from depredated sets			Data from all observed sets		
Species group	Number depredated	Number caught	Percent (%) depredated	Number depredated	Number caught	Percent (%) depredated
Billfishes	761	3,821	19.92	1,088	155,345	0.70
Wahoo (ono)	196	1,030	19.03	196	19,467	1.01
Tunas	2,611	14,791	17.65	2,654	277,272	0.96
Moonfish	84	681	12.33	84	18,684	0.45
Mahimahi (dolphinfishes)	408	4,521	9.02	422	105,445	0.40
Pomfrets	122	3,516	3.47	124	70,263	0.18
Other bony (inclusive*)	89	4,477	1.99	94	67,943	0.14
Gempylid fishes (walu)	34	2,357	1.44	68	46,611	0.15
Lancetfishes (*Alepisaurus* spp)	160	11,434	1.40	161	232,688	0.07
Sharks	1	6,669	0.01	1	167,732	0.00
Other cartilaginous fishes	0	588	000	0	7,300	0.00

*Including species not observed to be depredated. Those bony fishes not observed to be depredated are infrequently caught, whereas sharks are very seldom depredated though they are very common in the catch.

Hawaiian nearshore troll and longline fishermen have reported the loss of hooked blue marlin, spearfish, and bigeye tuna to false killer whales (Zimmerman, 1983). False killer whales have also been implicated in the depredation of catch from pelagic longline gear targeting tuna and swordfish in the central North Pacific. Since 2003, onboard observers

have systematically recorded damage to catch that is indicative of depredation by cetaceans. A wide variety of fish species have been documented with evidence of cetacean damage, including virtually all of the commercially valuable target and non-target species in the fishery. Based on the percentages of caught species depredated in longline sets (Table 4-2), false killer whales interacting with the longline fisheries appear to target wahoo, billfish, tuna, moonfish (*Lampris guttatus*), and mahimahi.

In the eastern tropical Pacific Ocean, false killer whales have been observed chasing or attacking other marine mammals, including bottlenose dolphins off the coast of Costa Rica (Acevedo-Gutierrez et al., 1997), common dolphins (*Delphinus* sp.) and dolphins of the genus *Stenella* associated with purse seining (Perryman and Foster, 1980), and sperm whales in the Galapagos Islands (Palacios and Mate, 1996). In Hawaiian waters, Hoyt (1983) reported a false killer whale attack on a humpback whale calf (*Megaptera novaeangliae).* Subsequent investigation of this event suggests that insufficient evidence is available to confirm whether an attack on humpback whales took place (D. McSweeney, pers. comm.).

4.1.2.2. Estimated prey needs

The energy needs of Hawaiian insular false killer whales were estimated based on a model developed by Noren (2010) for killer whales. Model inputs require the mass for each sex and age class in the population, and the sex and age structure of the population. Lengths of known-age animals (determined from growth layers from tooth sections) were estimated using a logistic model developed by Ferreria (2008) from male and female false killer whales taken in the drive fishery in Japan. The mass of these known-length animals, and thus their age, was estimated using the relationship developed by Kastelien et al. (2000). The average mass was calculated for 5-year age classes to match the percent age composition available from two data sets, the South Africa stranding data and Japan drive data (Ferreria, 2008). The total number of whales in each 5-year stage was estimated and the total minimum and maximum daily prey energy requirement (DPER) was calculated for a population of 170 whales. The minimum and maximum DPER for a population with an age structure based on Japan data were approximately 16.8 million and 20.2 million kcal/day, respectively, and for a population with an age structure based on South Africa data they were 18.9 million and 22.7 million kcal/day, respectively. Based on a diet composed entirely of one of the most commonly observed prey (mahimahi), the energy needs of the insular population were estimated as the number of fish potentially consumed on an annual basis. By dividing the total annual caloric needs of the whale population by the caloric value of an average-sized (8 kg) mahimahi (940 kcal/kg, based on muscle tissue, Sidwell, 1981), depending on the whale population age structure used, approximately 2.9 to 3.9 million pounds of fish would be consumed annually. For comparison, this quantity of fish is similar to the current annual retained catch in the commercial troll fishery (Figure 4-9) and 3 to 4 times greater than the annual catch in the main Hawaiian Islands handline fishery (Figure 4-10).

4.1.2.3. Trends in prey abundance

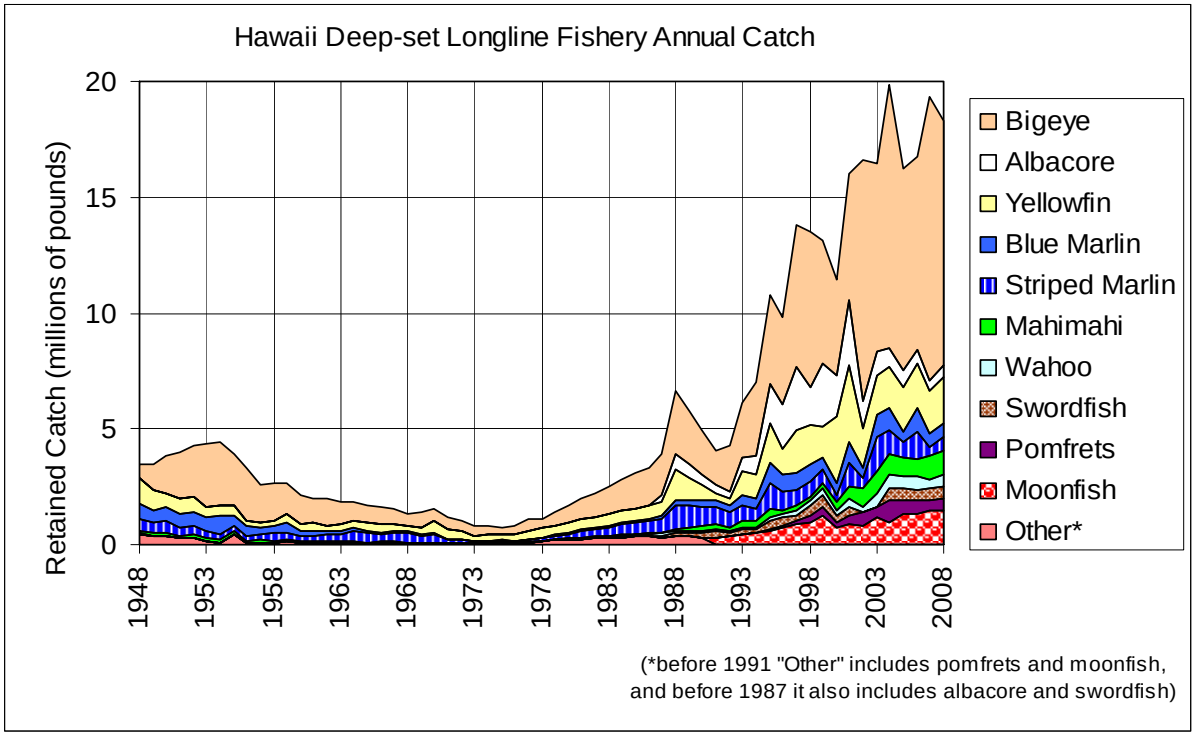
False killer whale prey includes many of the same species targeted by Hawaii's commercial fisheries, especially the fisheries for tuna, billfish, wahoo and mahimahi.

Trends in catch and catch-per-unit effort (CPUE) are assumed to reflect trends in abundance of these fish species.

The catch and CPUE data trends imply two potential threats to insular false killer whales. First, the catches by local fisheries inside the known range of insular false killer whales represent removals that compete locally in "real time" (i.e., ecological interaction) with foraging by the whales as fisheries, and the whales disrupt the local prey aggregations and remove the immediately available local fish. Second, the CPUE of fisheries that overlaps the range of insular false killer whales represents an index of the local as well as more widespread availability of prey and reflect much broader impacts of Pacific-wide fishing mortality and other factors on the forage base. Catches by local fisheries inside the habitat might represent competition for the locally available prey, and CPUE trends represent reductions in the potential forage base that integrate the impacts of fishing mortality over a longer time frame and over broad expanses of the Pacific.

A primary assumption in most ecosystem approaches to understanding multispecies population dynamics (Kitchell et al., 1999) is that prey biomass fluctuations have a strong influence on predator populations. If they do, then fishery removals of potential prey in the immediate vicinity of false killer whales (competition), as well as long-term declines in prey biomass over the range of the fish stocks (declining CPUE and biomass) both represent potential threats to Hawaiian insular false killer whales. The following discussions of fishery catch and CPUE make this assumption.

The competitive threat from catches by the longline, handline, and troll fisheries expanded from the 1970s through the 1990s, with the longline fishery reduced and stabilized inside most of the insular false killer whales' range in about 1990. The increase in possible local competition for Hawaiian insular false killer whales prey species by the longline fishery was most intense in the 1980s. In the last decade, the troll fishery has somewhat stabilized at much higher levels than in the 1970s (Figure 4-9) and remains a very active potential competitor in the insular false killer whales habitat. Handline catches seem to be most variable, with wide oscillations in the 1970s and 1980s and with a recent decline to about a third of the 1999 catches (Figure 4-10). So, the potential competition with local foraging indicated by the catch of the local handline fishery would seem to be diminishing compared to the late 1990s, and competition from the other local fisheries' catches in the false killer whale habitat seems to have stabilized in the same period. No information is available on the magnitude or trends in recreational catches, but based on the number of licensed recreational vessels in Hawaii and the estimated proportion used for recreational fishing (P. Dalzell, pers. comm.) the contribution of recreational fishing catches to potential competition for fish may be substantial.



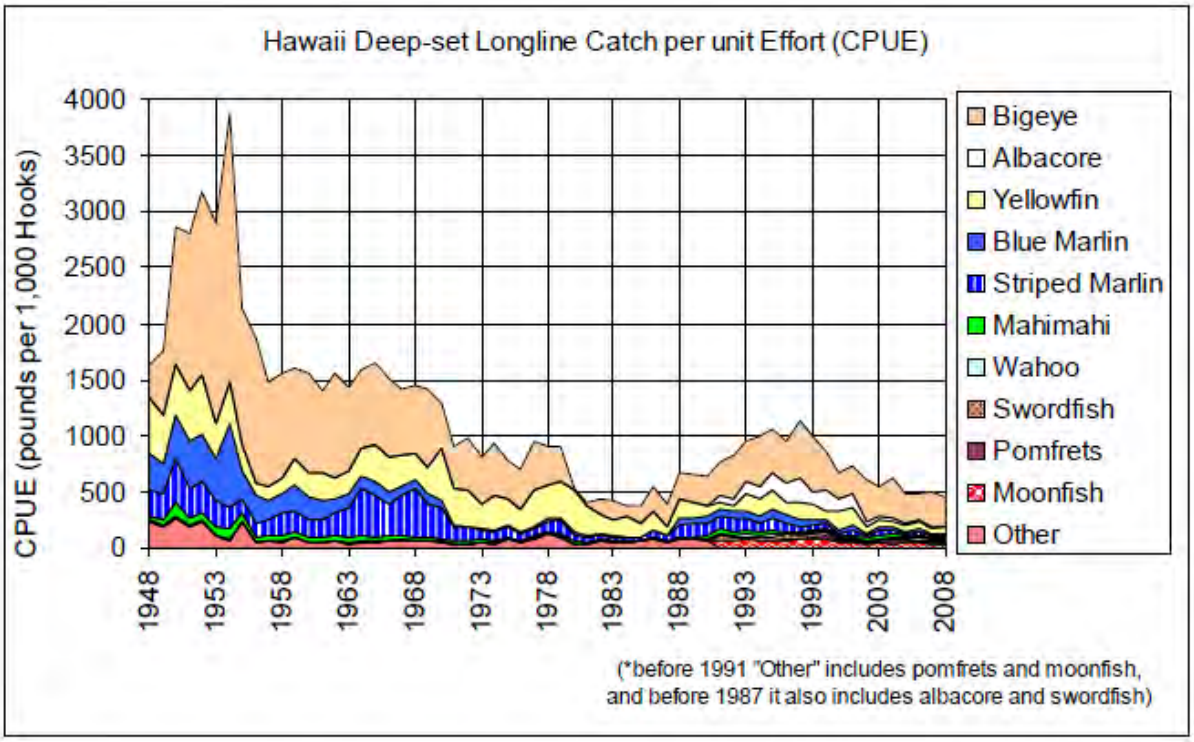


Figure 4-8. Longline fishery catch and catch-per-unit effort trends, 1948–2008.

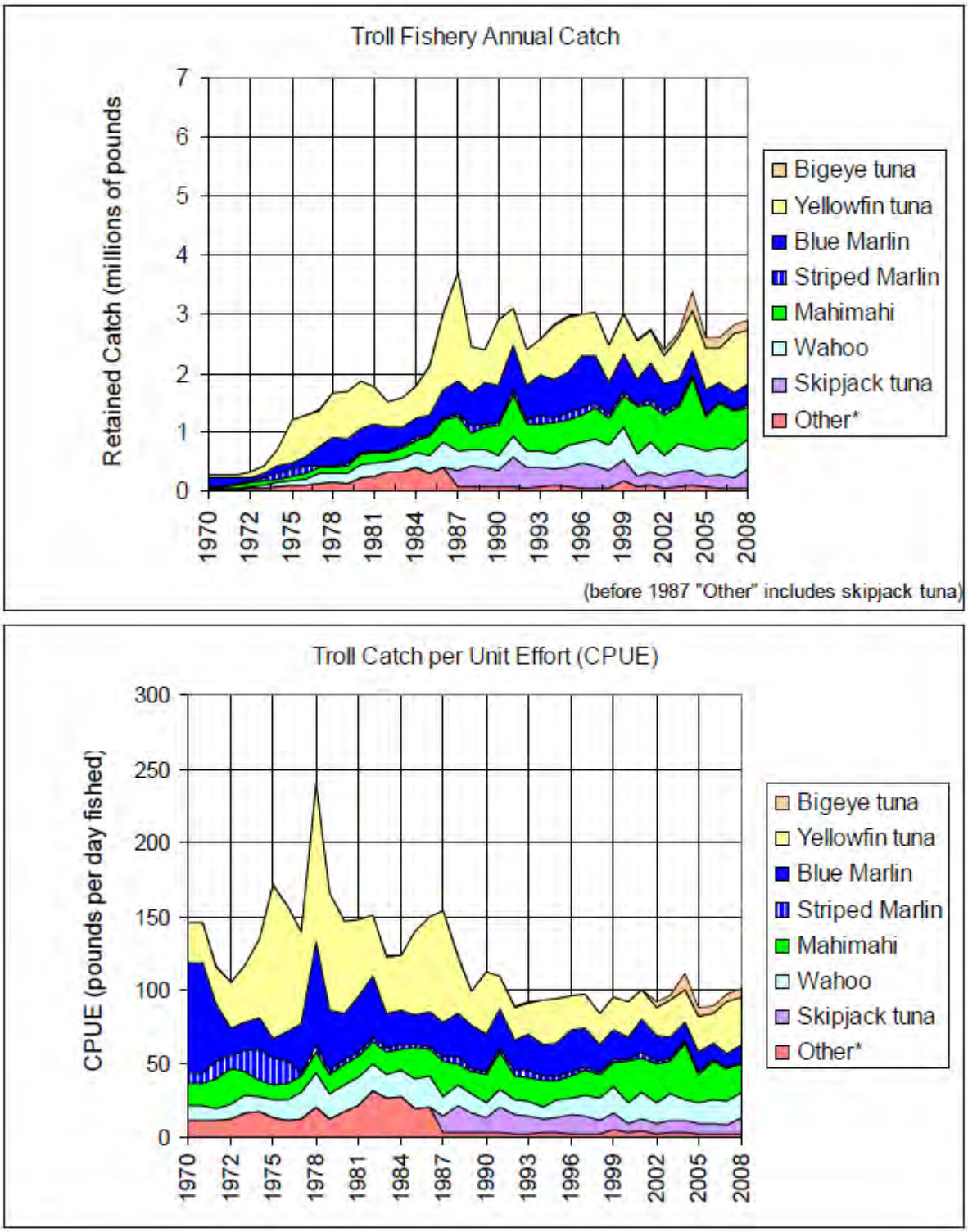


Figure 4-9. Troll fishery catch and catch-per-unit effort trends, 1970–2008.

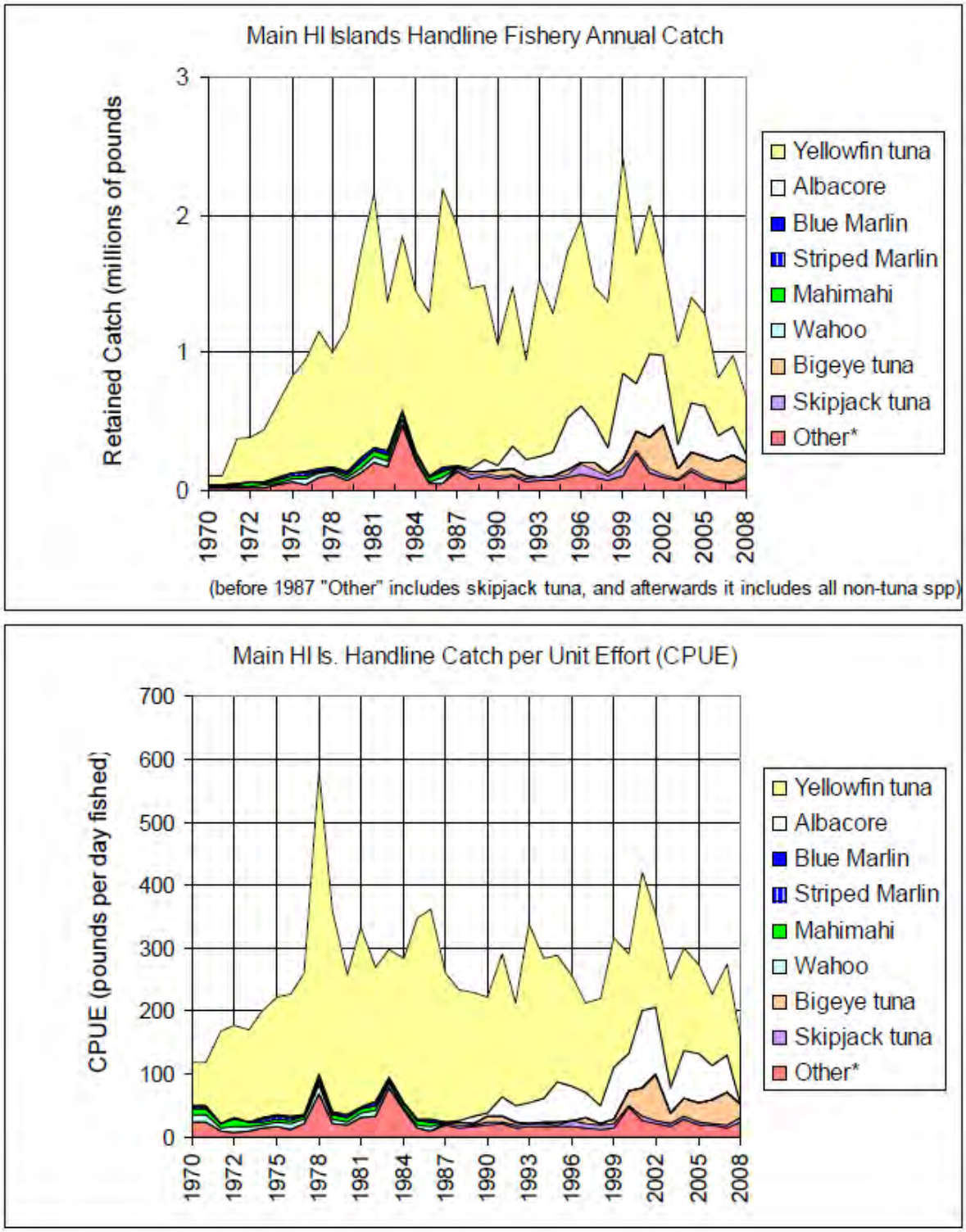


Figure 4-10. Handline fishery catch and catch-per-unit effort trends, 1970–2008.

The fisheries catch, CPUE, and average weight time series presented in this document are composites combining all species in stacked form, so that the top line represents total catch, or the CPUE for all species combined, or the sum of the average weights obtained by adding the weight of one average fish of each species. Since we do not quantitative information on the relative occurrence of prey species within the Hawaiian insular false killer whale diet, these plots allow a holistic review of all potential prey in a fishery. To review a single species in the CPUE or in the average weight time series, examine how the width of the band for each species varies over time, not its height.

In terms of prey availability and abundance reflected by CPUE, the longest record of declining biomass as a result of fishing mortality by all types of fishing throughout the region is provided by longline CPUE time series (Figure 4-8). Other fisheries CPUE time series are too recent to show the whole pattern. The most recent stock assessments of both yellowfin and bigeye tuna (Harley et al., 2009; Langley et al., 2009) clearly outline a similar pattern of substantially declining biomass in the 1960s and 1970s similar to that indicated in Boggs and Ito (1993) and in Figure 4-8. The Hawaii longline CPUE data suggest some overall increases in abundance from the mid 1980s into the 1990s, followed by a decline since 1997. Whether or not the near-MSY status of tuna biomass reflects overfishing, or an overfished condition, the thresholds defined for overfishing and overfished are mostly of human economic concern with respect to the aggregate enterprise of maximizing overall yield. These metrics do not address local availability to natural predator populations or local fisheries, which data presented here show cannot find anywhere near the abundance of fish that they could in prior decades. The current downward trends in abundance (CPUE) are less steep than in the earlier decades (Figure 4-8). The reduction of some of the larger species may result in reduced competition for food and reduced predation on smaller species, although increased harvesting by humans may absorb much of the increased prey biomass because of the removal of top predators. Several studies suggest increases of intermediate level predators with top predator reduction (Kitchell et al., 1999; Polovina et al., 2009).

Longline CPUE for some species appears to have stabilized a bit in the last 5 years (Figure 4-8). However, for bigeye and yellowfin tuna the stock assessments (Harley et al., 2009; Langley et al., 2009) indicate future biomass reductions, and indicate that international actions to reduce stock wide fishing mortality have so far been inadequate to prevent this.

The CPUE for the troll (Figure 4-9) and handline (Figure 4-10) fisheries that operate entirely within the insular false killer whales' habitat have a shorter history, but the troll CPUE shows declining total CPUE from the late 1970s to the mid 1980s, with some stability since. The handline CPUE shows less trend previously but suggests a recent decline. The handline fishery CPUE is dominated by tuna catches with little catch of other species. The troll fishery CPUE shows recent declines in marlin CPUE that are offset by increases in mahimahi and ono CPUE as have also been noted by Polovina et al., 2009).

4.1.2.4. Prey size

Few records assess the sizes of fish eaten by false killer whales in the wild, but anecdotal observations by fishermen indicate they are capable of consuming large prey when depredating fishing gear. Zimmerman (1983) reported that false killer whales removed a marlin estimated at more than 500 lbs (227 kg) from a troll line in Hawaii, and anecdotal reports indicate false killer whales routinely taking tuna weighing 50–100 lbs (22–45 kg). It is not known whether false killer whales are able to remove larger prey from lines than they would be able to capture and consume naturally, but field observations indicate they are capable of capturing large mahimahi (estimated at 18–20 lbs or 8–9 kg by Brown et al. (1966)), and that they readily attack larger species such as yellowfin tuna and broadbill swordfish (Baird et al., 2008a).

Yellowfin and bigeye tuna average weights seem to have declined between each of the periods of data available for the Hawaii longline fishery, with the greatest decline in tuna size happening half a century ago (Figure 4-11). Data on average fish weight in Figure 4-11 are taken from Shomura (1959) for 1949–1954, from Yong and Wetherall (1980) for 1965–1977, and from the Pelagics Annual Report (WPFMC 2010) for 1987–2008. A decline in average longline tuna and marlin weight persisted from the 1960s through the 1990s, with some increase in the last 5 years or so largely as a result of increased billfish sizes. Longline-caught swordfish average weights have increased since the mid 1980s (Figure 4-11). The troll and handline fish sizes also show some increase for about the last decade, again as a result of increased billfish sizes. All of these size data are reflected in the CPUE data as well, as these were presented in weight per unit of effort. Concern over fish size is based on the hypothesis of increased foraging success with greater energetic reward per successful predation event. Larger average prey may result in greater net reward per predation event, and since it isn't known what species are most important, all species were examined in a stacked component time series to provide an index of overall pelagic fish sizes and energy yield per predation event (Figure 4-11).

4.1.2.5. Prey distribution

The large pelagic prey species have very widespread distributions throughout the tropical and temperate oceans (Section 2.2.5). All occur throughout the Hawaiian insular false killer whale habitat, and there is little available documentation of prevailing patterns of abundance within this habitat. To some extent, both the fisheries (Figure 4-2) and the whales (Figure 2-11) show some preference for the leeward areas off Oahu and the island of Hawaii, but these are the areas with well developed small boat access, and with relatively sheltered water for safer boating and cetacean surveying. These areas are famous for their fishing, as is the area offshore of Hilo on the Big Island. Waters off the coast of Hilo are not sheltered, but Hilo has a very good harbor and an avid fishing population. Regardless of these areas of apparent concentration, all of the waters within the insular false killer whale habitat yield substantial harvests of the large pelagic prey species.

Variation in ocean conditions contributes to variability in prey abundance, and the pelagic-benthic coastal interface (Section 2.2.4.5) near islands appears to increase the relative abundance of large pelagic species such as tunas (Murphy and Shomura, 1972).

This “Island Mass Effect” may have contributed to the evolution of a specialized insular false killer whale population. In terms of threats, the abundance of insular fish species contributing to the forage base for nearshore large pelagics, possibly including Hawaiian insular false killer whales has been dramatically reduced by human activities in the last century (Shomura, 1987).

4.1.2.6. Seasonal availability

Long-term averages of seasonality of the harvests of large pelagic species, in aggregate, indicate availability year round (Figure 4-12), with the greater spring-summer harvests influenced partially by greater fishing activity in these months. Yellowfin tuna, skipjack

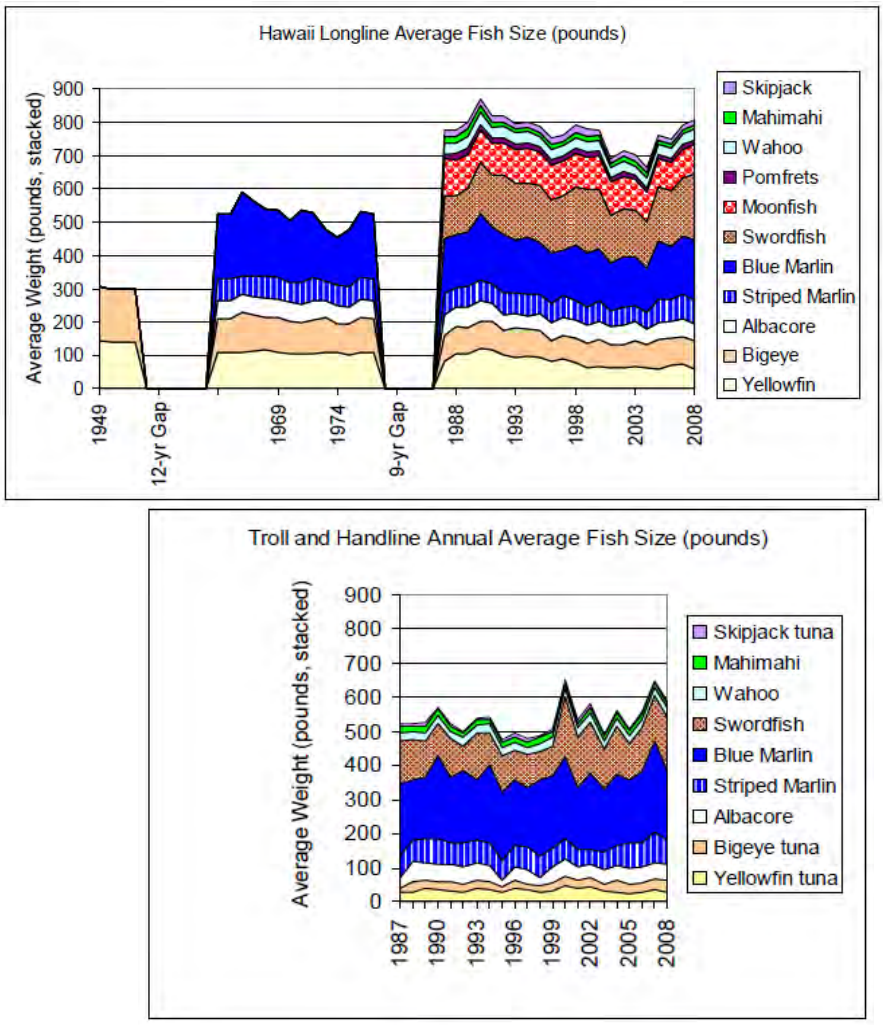


Figure 4-11. Stacked trends in average fish size from the longline (top), and combined troll and handline fisheries (bottom), over a variety of time periods for which summary data are available. The average weight of each species in a year is the width of the stacked band for that species (e.g., longline blue marlin generally average several hundred pounds each year, longline striped marlin average less than 100 lbs, longline albacore average about 50 lbs, etc.). The total of the average weights for one of each species caught are represented by the stacked total (e.g., 800 lbs for one of each longline-caught species in 2008. Since the diet composition is unknown, this figure provides some overall indication of average prey size with equal weight to all species.

tuna, blue marlin (*Makaira nigricans*), wahoo, and swordfish are more available near the islands in summer. These are also shallower-swimming species that may be more available to predation by false killer whale (Section 2.2.5.2). Bigeye tuna and striped marlin are more available near the islands in winter, and mahimahi have two seasonal peaks, one in spring and one in fall. All of these patterns, like those for prey distribution, are confounded by the effects of environment on fishermen's behavior. Longline catch per unit effort by month (Skillman and Kamer, 1992) generally reflects the same seasonal patterns as shown by long-term average harvests (http://www.pifsc.noaa.gov/wpacfin/pdf_file/h_vol24.pdf).

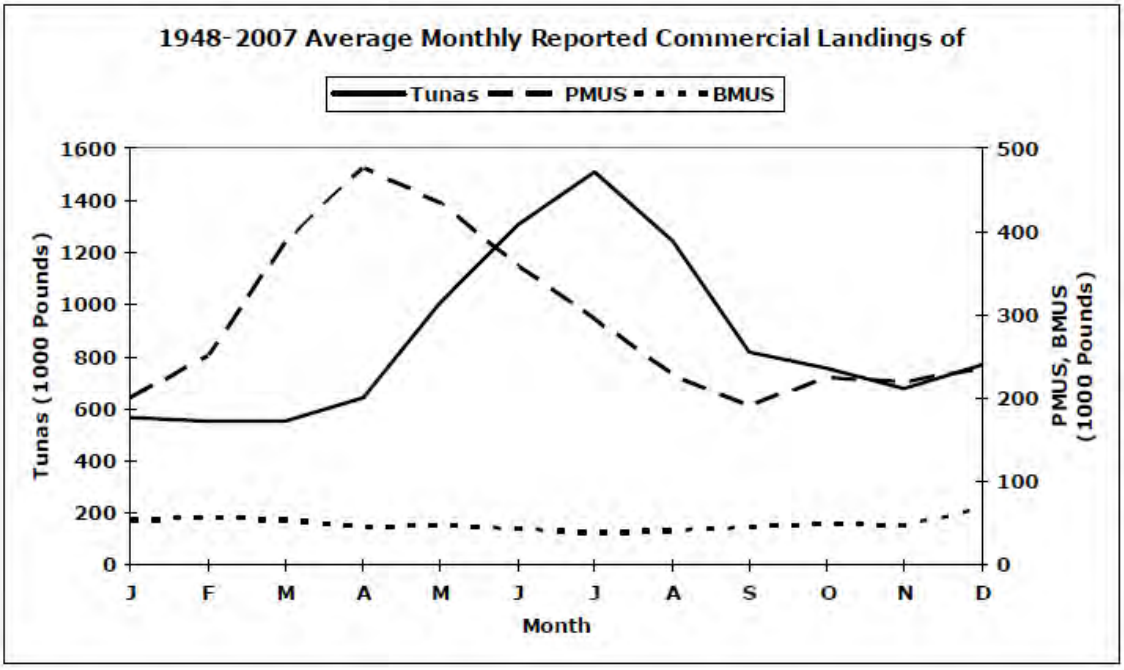


Figure 4-12. Harvest of potential Hawaiian insular false killer whale prey by month taken from http://www.pifsc.noaa.gov/wpacfin/pdf_file/h_vol24.pdf. The term "PMUS" refers to the pelagic management unit species other than tunas (i.e., billfishes, mahimahi, ono, pomfrets, etc.). BMUS refers to bottomfish management unit species. The aggregate peak long-term average harvest of pelagic species (tuna and PMUS) appears to be in May–June but substantial harvests continue year-round.

4.1.2.7. Decadal variability

The individual species seem to oscillate at a wide range of periods based on CPUE, but not in phase with each other, perhaps because the oscillations are related to life history, with strong cohorts depending on different recruitment factors for each species, and density-dependent effects. For the total CPUE of all the species in any of the fisheries there seem not to be marked oscillations at any consistent periods but no rigorous analysis has been conducted (Figure 4-8–4-10).

4.1.3. Vessel traffic

4.1.3.1. Ship strikes

Although more commonly observed in large whales, ship strikes also have potential to kill or injure smaller cetaceans including false killer whales. Propeller strikes from large vessels are likely to be fatal, while propeller injuries from small boats may cause disfigurement of the dorsal fin or other parts of the body without killing the whale outright (Wells et al., 2008). Slow-moving animals, or animals that associate with vessels, i.e., to ride the bow or depredate catch on fishing gear being retrieved, are likely the most susceptible to ship strikes. At least one juvenile killer whale that frequently approached vessels was killed by an injury caused by a ship off British Columbia, Canada (Laist et al., 2001; Gaydos and Raverty, 2007). False killer whales in waters surrounding Hawaii (belonging to both insular and pelagic stocks) are known to ride the bow or stern wake of vessels and may come into proximity of propellers (Baird, pers. comm.). No ship-strike related injuries or deaths of false killer whales have been documented in Hawaiian waters, but Baird (2009) reported a fresh head wound on one individual from the insular population photographed off Oahu in September 2009 that may have been caused by a propeller strike.

4.1.3.2. Whale watching

Whale watching operations are common in Hawaii during the winter and spring humpback whale breeding season. Most whale watching operators depart and return from the west shore of Maui; however, some operators use ports along the Kona coast of the Big Island and the west coast of Oahu. Although none of these operations target their activities on false killer whales, most will stop to watch them when seen. Similarly, dolphin watching tours are common on the Big Island, Maui, and Oahu and work year-round. There have been reports of dolphin watching tours allowing customers to enter the water with spinner dolphins, pilot whales and other cetaceans. Like humpback activities, spinner dolphin tours do not target false killer whales but some are known to take customers to offshore waters in search of species other than spinner dolphins. There have been no reported interactions or infractions of whale or dolphin watching activities with false killer whales and the level of risk to the insular population is unknown.

4.1.4. Anthropogenic noise

Odontocete cetaceans, including false killer whales, have a highly evolved acoustic sensory system. False killer whales rely heavily on their acoustic sensory capabilities for navigation, foraging, and communicating with conspecifics. Potential and measured impacts of anthropogenic noise on cetaceans have been reviewed by a number of authors (Richardson et al., 1995; Hildebrand, 2005; Weilgart, 2007). No specific studies or observations of the impacts of noise on wild false killer whales are available. However, intense anthropogenic sounds have the potential to interfere with the acoustic sensory system of false killer whales by causing permanent or temporary hearing loss, thereby masking the reception of navigation, forging or communication signals, or through disruption of reproductive, foraging, or social behavior. Experiments on a captive false killer whale have revealed that it is possible to disrupt echolocation efficiency in this species with the level of disruption related to the specific frequency content of the noise source as well as the magnitude and duration of the exposure (Mooney et al., 2009).

In recent years there has been increasing concern that active sonar and seismic operations are harmful to beaked whales (Cox et al., 2006)and other cetaceans, including melon-headed whales (Southall et al., 2006) and pygmy killer whales (Wang and Yang, 2006). The use of active sonar from military vessels has been implicated in mass strandings of beaked whales, and recent mass-stranding reports suggest some delphinids are affected as well. A 2004 mass stranding of melon-headed whales in Hanalei Bay, Kauai occurred during a multinational sonar training event around Hawaii (Southall et al., 2006). Although data limitations preclude a conclusive finding regarding the role of Navy sonar in triggering this event, sonar transmissions were considered a plausible, if not likely cause of the mass stranding based on the spatiotemporal link between the sonar exercises and the stranding, the direction of movement of the transmitting vessels near Hanalei Bay, and propagation modeling suggesting the sonar transmissions would have been audible at the mouth of Hanalei Bay (Southall et al., 2006). False killer whales have been herded using loud sounds in drive fisheries off Japan (Kishiro and Kasuya, 1993; Brownell et al., 2008), suggesting that high-intensity noise can affect the behavior of false killer whales in Hawaiian waters. The U.S. Navy Hawaii Range Complex surrounds the main Hawaiian Islands and is regularly used for training exercises that broadcast high-intensity, mid-frequency sonars sounds (U.S. Navy, 2008). The specific location and timing of mid-frequency sonar training activities near Hawaii are not known, therefore, it is not possible to assess the frequency of occurrence of sonar use within the range of the insular population of Hawaiian false killer whales.

4.1.5. Small population effects

Compared to most other marine mammal populations in U.S. waters, Hawaiian insular false killer whales have a very low abundance (Angliss and Lodge, 2002; Carretta et al., 2002). The current estimate of 123 (Baird et al., 2005) is a bit higher than that of SRKWs (85, Carretta et al., 2009) but lower than that of Cook Inlet belugas (375, Hobbs and Shelden, 2008), both populations listed as endangered.

The processes that cause small populations to have a greater risk of extinction include genetic and behavioural problems, as well as chance processes like demographic and environmental stochasticity (Shaffer, 1981; Gilpin and Soule, 1986; Goodman, 1987; Simberloff, 1988; Lande, 1993). The decrease in per capita population growth as population size declines is often referred to as the “Allee effect” or “depensation” (e.g., Allee, 1931; Allee, et al. 1949; Dennis, 1989; Fowler and Baker, 1991; Courchamp et al., 1999; Stephens and Sutherland, 1999; Stephens et al., 1999; Petersen and Levitan, 2001; Dennis, 2002; Berec et al., 2007). In essence, as the number of individuals decreases there are costs from a lack of predator saturation, impaired anti-predator vigilance or defense, a breakdown of cooperative feeding, an increased possibility of inbreeding depression or other genetic issues, decreased birth rates as a result of not finding mates, or a combination of these effects. The Allee effect increases risk to small populations directly by contributing to the risk of extinction, and indirectly by decreasing the rate of recovery of exploited populations and, therefore, maintaining populations at a smaller size where extinction risk is higher for a variety of reasons (e.g., environmental or demographic stochasticity) (Dennis, 1989; Stephens and Sutherland, 1999).

In addition, social odontocetes (such as false killer whales) may be particularly vulnerable to exploitation over and beyond the numerical loss of individuals to the population (Wade and Reeves, 2010). Some of these effects may act in a similar fashion to Allee effects or have a more pronounced effect at low population sizes. Survival and reproductive success may depend on such things as social cohesion and social organization, mutual aid in defence against predators and possible alloparental care such as 'babysitting' and communal nursing, sufficient opportunities for transfer of 'knowledge' (learned behaviour) from one generation to the next, and leadership by older individuals that know where and when to find scarce prey resources and how to avoid high-risk circumstances (e.g., ice entrapment, stranding, predation).

False killer whales share several life history traits with killer whales and belugas that make them prone to problems associated with small population size: a low intrinsic growth rate (a consequence of late maturity and a low birth rate), strong social structure demonstrated through close associations of individuals over long time periods, the potential for high adult survival enabled by the intergenerational transmission of certain types of awareness or specialized behavior, and a low effective population size compared to abundance. This last feature leads to low genetic diversity, which increases the probability that inbreeding depression (a decrease in birth or survival rates resulting from deleterious or lethal genes) will occur at a higher level of total abundance than is the case for many other species. Franklin (1980) found that inbreeding depression increases substantially when the number of reproductive animals becomes fewer than 50. Given that the adult population of Hawaiian insular false killer whale adults is probably around 65 (123 times 43% mature from killer whales, Taylor et al., 2008), it could be approaching the level at which the effects of inbreeding depression become a factor in determining whether the population is able to maintain itself or increase. The estimate of an effective population size of about 46 breeding adults (Chivers et al., 2010) makes the potential for loss of genetic diversity in the near future of concern.

A small population could also experience a breakdown of social functionality important to fecundity and survival. Although some aspects of the behavior and "culture" of Hawaiian insular false killer whales have been investigated or discussed, the mechanisms by which they might influence population growth rates are not well understood. The situation of this population could be analogous to those of other populations of large mammals in which females live well beyond their reproductive life spans (e.g., elephants, higher primates, and some other toothed cetaceans such as pilot whales) (McComb et al., 2001; Lahdenpera et al., 2004). The loss of only a few key individuals—such as the older, post-reproductive females —could result in a significant loss of inclusive fitness conveyed by "grandmothering" behavior (i.e., assistance in care of the young of other females in the pod). In addition, cultural knowledge (e.g., how to cope with environmental changes occurring on decadal scales) could be lost, leading to reduced survival or fecundity of some or all age classes. Wade and Reeves (2010) document the special vulnerability of social odontocetes giving examples of killer whales, belugas, sperm whales, and dolphins in the eastern tropical Pacific.

4.1.6. Disease and predation

Since the mid 1960s, the number of published reports that examined protozoa, parasites, bacteria, viruses, and harmful algal toxins in marine mammals has increased greatly (Gulland and Hall, 2007). However, limited published data are available on disease occurrence in false killer whales. Liver disease was examined in 18 species of cetaceans (e.g., common dolphin, Atlantic spotted dolphin (*Stenella frontalis*), and false killer whale) that stranded in the Canary Islands from 1992 through 2000 (Jaber et al., 2004). The most common lesions identified in the study, nonspecific chronic reactive hepatitis and hyaline intracyctoplasmic inclusions in hepatocytes, were found in the livers of two stranded false killer whales. Toxoplasmosis, a protozoan-caused infection, was investigated in 59 wild and captive cetaceans from Japan, including six false killer whales (Murata et al., 2004). All samples from false killer whales were seronegative using latex agglutination and indirect hemagglutination tests. A serological survey was conducted on odontocetes and pinnipeds that stranded on the coasts of northwest Europe or the Mediterranean Sea to determine exposure to a recently isolated dolphin rhabdovirus-like virus (Osterhaus et al., 1993). Several species tested positive for this virus, including false killer whales. Porpoise and dolphin morbillivirus surveys have been conducted on various odontocetes because of the large number of deaths associated with these viral infections (McCullough et al., 1991; Domingo et al., 1992; Visser et al., 1993; Lipscomb et al., 1994). Van Bressem et al. (2001) analyzed blood of 288 cetaceans from the Mediterranean Sea, as well as the Atlantic, Pacific, and Indian Oceans collected from 1995 to 1999 for the presence of dolphin morbillivirus. Although 35 animals were seropositive, none of the blood samples collected from captive dolphins (including a false killer whale) in Taiwan, Indonesia, and Japan showed seropositive titers for this virus. In another study, blood samples collected from stranded and free-ranging cetaceans from the western Atlantic Ocean between 1985 and 1992 were tested for porpoise morbillivirus (Duignan et al., 1995). Positive neutralizing titers against porpoise morbillivirus were found in blood of 47% (9/19) of the western Atlantic false killer whales examined. The authors noted that, although the impact of morbillivirus infection on the natural mortality of western Atlantic cetaceans was unknown, those species that live in close-knit social groups (e.g. false killer whales) would have greater potential for pathogen transmission. Gaydos et al. (2004) conducted a literature search of infectious diseases in odontocetes and determined that marine *Brucella* spp., cetacean poxvirus and morbillivirus, and herpesviruses would pose the greatest threat to the ESA-listed Southern Resident killer whales. Based on similarities in life history strategies (e.g., long life span, time to maturation, cohesive social group), it is likely that exposure to these pathogens also poses a risk to Hawaiian insular false killer whales. The likelihood of exposure of Hawaiian insular false killer whales to marine *Brucella* spp. and other pathogens is not known.

In March 2010, the first record of a killer whale predation event on a false killer whale was documented off the coast of New Zealand (Visser et al., 2010). After the attack, killer whales were observed feeding on the carcass of a false killer whale calf. No other predation events by killer whales on false killer whales have been reported (Heithaus, 2001).

4.1.7. Parasitism

Various parasites have been documented in tissues of false killer whales, including nematodes (e.g., *Anisakis simplex*, *Stenurus globicephalae*), trematodes (e.g., *Nasitrema globicephalae*), acanthocephalans (e.g., *Bolbosoma capitatum*), amphipods (e.g., *Isocyamus delphinii*, *Syncyamus aequus*, *Syncyamus pseudorcae*) and crustaceans (e.g., *Xenobalanus globicipitus*) (Sedlak-Weinstein, 1991; Stacey et al., 1994; Andrade et al., 2001; Hernandez-Garcia, 2002; Zylber et al., 2002). In some cases, parasitic infections have been implicated as contributing to false killer whale strandings. For example, following two mass stranding events that occurred along the coasts of Japan and Florida, tissues of false killer whales were examined for parasites (Odell et al., 1980; Morimitsu et al., 1987). In both studies, evidence of parasitic infections (e.g., *Stenurus globicephalus*, *Nasitrema gondo*) were noted in the lungs, pterygoid sinus complexes, and tympanic cavities of the whales, and the authors postulated that these infections most likely contributed to the demise of the animals. Currently, no information is available on parasitism in Hawaii insular false killer whales. A pathogen survey conducted on 18 Hawaiian monk seals (*Monachus schauinslandi*) from the main Hawaiian Islands between January 2004 and November 2005 demonstrated that sera from two seals tested positive for the protozoan parasite *Toxoplasma gondii* and two tested positive for *Neospora caninum*, a coccidian parasite (Littnan et al., 2006). Discharge of raw or partially treated sewage effluent in to marine coastal waters can increase the risk of pathogen transmission to animals that reside in nearshore areas (Smith-Vaniz, 1999). In addition parasites, bacteria and viruses can enter Hawaiian coastal waters via freshwater runoff contaminated with these pathogens (Littnan et al., 2006).

4.1.8. Environmental contaminants

4.1.8.1. Persistent organic pollutants

Exposure to toxic chemicals, including persistent organic pollutants (POPs), was identified as one of the potential risk factors for the Hawaiian insular false killer whales (Fallon, 2009). "Legacy" POPs include a number of chemical classes [e.g., polychlorinated biphenyls (PCBs), DDTs, mirex, chlordane] that were used as industrial chemicals and pesticides in the United States and other countries around the world. Because many of these compounds are recalcitrant to metabolism and degrade slowly in the environment, they tend to biomagnify in marine ecosystems, especially in lipid-rich tissues of top-level predators (McFarland and Clarke, 1989). Starting in the early 1970s, PCBs, DDTs and other classes of POPs were banned in the United States from production and use because of toxic effects on wildlife and laboratory animals. As a result, the levels of these compounds in environmental samples in the United States have declined since the bans, including fish from Hawaii (Brasher and Wolff, 2004). However, some of these chemicals continue to be used in other regions of the world to control malaria-carrying mosquitoes and other pests and can be transported via atmospheric transport or ocean currents (Fiedler, 2008; van den Berg, 2009). In marine mammals, exposure to high levels of legacy POPs has been associated with immunosuppression (Ross et al., 1995; Beckmen et al., 2003), reproductive dysfunction (Helle et al., 1976; Subramanian et al., 1987), and morphological changes (Zakharov and Yablokov, 1990; Sonne et al., 2004). Even though these contaminants have been banned in the United States for more than 25 years, they continue to be measured in marine

animals from Hawaii (Hunter, 1995; Kimbrough et al., 2008; Ylitalo et al., 2009) and other portions of the United States (Kannan et al., 2004; Wells et al., 2005; Krahn et al., 2007; West et al., 2008; Krahn et al., 2009) and pose a potential risk to their health.

The polybrominated diphenyl ethers (PBDEs) are another group of POPs that have received much attention in the scientific literature, as well as in the news media. Three commercial mixtures of these compounds (i.e., pentaBDE, octaBDE, decaBDE) are used as flame retardants in manufactured goods including textiles, automobiles, and electrical components (de Wit, 2002). Similar to PCBs, these compounds are persistent and lipophilic, and they have been measured in samples of air, dust, sediments, and tissues of fish, marine mammals, and humans collected from many regions of the world (de Wit, 2002; Shaw and Kannan, 2009). Since 2004, PBDE manufacturers and importers in the United States have voluntarily phased out the production and use of the penta- and octaBDEs; however, decaBDE continues to be applied to manufactured goods. Some of the highest levels of these compounds have been reported in environmental samples from North America, particularly samples collected near urban areas (Ikonomou et al., 2002; Shaw and Kannan, 2009). Exposure to PBDEs has been associated with a number of biological effects (e.g., thyroid disruption, neurobehavioral effects, reproductive dysfunction) in laboratory animals (de Wit, 2002; Talsness, 2008). However, no threshold PBDE values have been established for toxic effects in marine mammals.

Contemporary concentrations of POPs measured in blubber of Hawaiian insular false killer whales were reported recently (Ylitalo et al., 2009). PCBs and DDTs were the most prevalent POPs measured, with adult females containing lower concentrations of POPs than those found in subadults and adult males. Previous marine mammal contaminant studies report much lower POP burdens in reproductive females than in males in the same age group (Aguilar and Borrell, 1988; Ross et al., 2000). These studies have shown that the POP concentrations in juvenile animals of both sexes increase until sexual maturity. Males continue to accumulate these lipophilic contaminants throughout their lives. In contrast, the levels of these contaminants decrease once a female reaches sexual maturity as a result of transfer of lipophilic POPs from the mother to her offspring during gestation and lactation (Aguilar and Borrell, 1994; Krahn et al., 2009). Three of the 9 Hawaiian insular false killer whales sampled in the study showed levels of summed PCBs that exceeded the 17,000 ng/g lipid weight threshold value associated with a number of biological effects in mammals (Kannan et al., 2000). An interesting finding in the false killer whale study was that subadult whales had higher levels of dieldrin, PBDEs, hexachlorocyclohexanes (HCHs) and hexachlorobenzene (HCB) in their blubber than adults. A similar finding was reported in SRKWs from Puget Sound, WA/British Columbia, with the youngest animals sampled showing the highest levels of PBDEs, HCHs and HCB (Krahn et al., 2007, 2009). These young, developing whales may be at higher risk of exposure to these toxic compounds than adults. First-born may be particularly at risk as they receive the highest doses of POPs from their mother compared to subsequent offspring (Lee et al., 1996; Ylitalo et al., 2001; Wells et al., 2005). Activities related to tourism, agriculture, defense, the principal economic contributors in the main Hawaiian Islands (State of Hawaii Department of Business, Economic Development and Tourism, 2008), as well as ongoing coastal development processes in the main Hawaiian Islands can be potential sources of POPs to nearshore coastal waters

in this region (Friedlander et al., 2008). Coastal pollutants may also enter marine waters of the main Hawaiian Islands via point source (e.g., sewage effluent) and nonpoint source (e.g., ocean current transport) inputs. However, it should be noted that toxic pollutants are not frequently measured in marine waters of Hawaii (Friedlander et al., 2004).

Limited POPs data are available for false killer whales from other regions (Table 4-3). Jarman et al. (1996) reported concentrations of PCBs, DDTs, and other classes of POPs measured in blubber of false killer whales stranded along the coast of British Columbia in 1987 and 1989. The levels of most of the POPs determined in the stranded whales were 5 to 50 times higher than those measured in the Hawaiian animals (Ylitalo et al., 2009). In contrast, the mean levels of PCBs measured in blubber of false whales from Taiwan and Japan (Chou et al., 2004; Haraguchi et al., 2006) collected from 2000 to 2003 were similar to those reported in the Hawaii insular false killer whales. Comparisons of summed DDTs and PBDEs among whales from the various geographical regions could not be made as no data were reported for the western Pacific whales or the whales that stranded in British Columbia.

Table 4-3. Concentrations of summed DDTs, PCBs, and PBDEs measured in blubber of false killer whales.

				ng / g lipid weight			
Collection region	**Collection year(s)**	**# of animals**	**% Lipid**	**ΣDDTs**	**ΣPCBs**	**ΣPBDEs**	**References**
Japan	2001-2003	4	NR	NR	19,500	NR	Haraguchi et al., 2006
Taiwan	2000-2001	1	22.3	NR	25,300	NR	Chou et al., 2004
British Columbia[1]	1987, 1989	2	91 ± 2.8	1,000,000 ± 1,400,000	45,000 ± 7500	NR	Jarman et al.,1996
Hawaii	2008	9	24 ± 3.8	20,000 ± 9700	13,00 ± 4800	1100 ± 390	Ylitalo et al., 2009

NR - not reported

[1]Includes adult males only

One issue of concern for Hawaiian insular false killer whales is that reduced prey quantity or quality could increase the risk associated with exposure to lipophilic POPs. It is suspected that body condition can influence POP burdens in the blubber of marine mammals even though the dynamics of blubber POPs during changes in physiological condition of these animals are complex and poorly understood (Aguilar et al., 1999). Marine mammals can lose weight during various stages of their life cycles as a result of stresses such as disease, migration or reduced prey abundance. The mobilization of lipid associated with weight loss could result in redistribution of POPs to other tissues, or to retention of these compounds in blubber that would result in a concentration increase (Aguilar et al., 1999). For example, a recent study on California sea lions demonstrated that during rehabilitation there were significant increases in blubber POP concentrations when animals were losing mass and decreases during mass gain (Hall et al., 2008). A basic mass balance model indicated that the changes did not conform to a simple concentrating and diluting pattern and a proportion of the contaminants were lost from the lipid pool. Thus, animals that are nutritionally challenged could be at higher risk as a

result of increased mobilization of these compounds to other organs where damage could result.

4.1.8.2. Metals

Heavy metals (e.g., mercury, cadmium, lead) have been shown to accumulate in marine mammals and, in some cases, may cause deleterious biological effects, including alterations in steroid synthesis and liver damage (O'Hara and O'Shea, 2001). Of greatest concern for marine mammals are mercury, lead, cadmium, and the organotins. These substances could pose risks to Hawaii insular false killer whales as they have been found in relatively high concentrations in false killer whales from other regions of the world. Kemper et al. (1994) provided a review of the concentrations of heavy metals and organochlorines in tissues of marine mammals from Australia. Of the 14 marine mammal species examined, false killer whales had the highest levels of mercury (ranging from 41 to 479 µg/g dry weight in liver and from 59143 µg/g dry weight in kidney). They also showed relatively high liver concentrations of lead and cadmium, with concentrations ranging from 0.05 to 0.47 µg/g, dry weight and from 0.05 to 75.8 µg/g, dry weight, respectively. High levels of mercury were also reported in muscle samples of false killer whales from Japan (2003; Endo et al., 2010). Contemporary concentrations of organotins were reported in false killer whales from Brazil and Japan, with sum butyltin concentrations in liver ranging from 0.341 to 4860 µg/g, wet weight (Harino et al., 2007; Dorneles et al., 2008). Currently, no data on toxic metal (e.g., mercury, cadmium, lead) data are available for Hawaiian nsular false killer whales so the risk associated with exposure to these substances is unknown.

4.1.8.3. Chemicals of emerging concern

With human population growth and increasing commercial development, there has been an increased demand for industrial chemicals, current-use pesticides, pharmaceuticals and personal care products. Many of these chemicals of emerging concern (CECs) are used in high volumes in various applications and, as a result, are capable of entering marine environments via various routes. Muir and Howard (2006) noted that 30,000 chemicals were in wide commercial use (> 1 ton per year) in the United States in 2005. Of these compounds, Muir and Howard (2006) indicated that those that are persistent, bioaccumulative, and toxic are of greatest concern for humans and wildlife. For example, in May 2009, the tetra-BDEs and penta-BDEs, hexa-BDEs and hepta-BDEs, as well as 7 other classes of POPs, were identified for global elimination under the Stockholm Convention. Unfortunately, an overwhelming number of high-volume industrial compounds are not being monitored currently in biota as few analytical methods have been developed to measure them in tissues. Atkinson et al. (2003) measured natural and synthetic estrogens (e.g., estrone) in sewage, injection-well water and tropical waters (both coastal and offshore) in the main Hawaiian Islands, Northwestern Hawaiian Islands, Florida Keys, and other regions of the world. They found that water samples collected near sewage sources had higher levels of estrone than samples collected at sites not associated with sewage effluent and that the mean levels at some sites, including Maalaea Bay in Maui, were above the threshold level (300 pg/L) determined for coral uptake. Currently, it is unclear what risk CECs pose to Hawaiian insular false killer

whales or their habitat as little is known about the current occurrence, fate, and transport of CECs in the main Hawaiian Island region.

4.1.8.4. Plastic and marine debris ingestion

Marine litter has become an increasing problem in the oceans, with plastic debris being the most abundant (Derraik, 2002). Plastics are manmade organic polymers that have been used in a wide range of manufactured goods because of their durability, resistance to degradation, and low production costs. Because they are lightweight and buoyant, plastics can be transported over long distances in the marine environment and have been found in the deep sea floor, along coastlines and in surface waters in densely populated areas, as well as remote regions of the world (Derraik, 2002). These compounds can be broken down into small pieces upon exposure to ultraviolet radiation in sunlight, thus making them available for ingestion by marine organisms (Rios et al., 2007). Ingestion of plastics can obstruct the esophagus, and the digestive or intestinal tracts, block gastric enzymatic secretions and cause other effects that could reduce an animal's ability to feed and ultimately its overall fitness (Derraik, 2002). Other risks linked to plastic debris include entanglement, exposure to environmental contaminants (e.g., PCBs, DDTs) contained in plastic resins and introduction of alien species (Derraik, 2002; Rios et al., 2007)

Ingestion of plastic debris poses a risk to Hawaiian insular false killer whales as plastics have been recovered in their habitat range. From November 2005 through April 2006, 35 tons of marine debris was removed from 9 miles of coastline in the Waiohinu-Ka Lae area in the main Hawaiian Islands as part of an effort to remove marine debris from coastlines and reefs of Hawaii. Eighty-six percent of the debris was made up of derelict fishing nets, whereas plastic was the primary component of the other debris items collected (NOAA, 2006). McDermid and McMullen (2004) determined the amounts of small plastic debris on remote beaches in the Northwestern and main Hawaiian Islands, including beaches in Oahu, Molokai and Hawaii. Small plastic particles (1–15 mm in size) were found at all beaches sampled, with plastic pieces 1–2.8 mm in size comprising 43% of the debris collected in the study. More recently, samples of plastic resins were collected from coastal sites in California and Hawaii and were analyzed for environmental contaminants (e.g., PCBs, DDTs, polycyclic aromatic hydrocarbons (PAHs) (Rios et al., 2007). Rios et al. (2007) measured a range of PCB, DDT, and PAH levels in particles collected along Hawaiian beaches, with sum PCBs and sum PAHs concentrations ranging from < limits of quantization (< LOQ) to 980 ng/g and < LOQ to 500 ng/g, respectively. Other studies have reported that plastics pellets collected from the coast of Japan contain appreciable levels of PCBs, DDE, and nonylphenol in these granules (Mato et al., 2001; Endo et al., 2005). Thus, ingestion of these particles could cause physiological problems as well as increase the levels of PCBs, DDTs and other toxic contaminants to which marine biota are exposed. Ingestion of chemical light sticks used on swordfish longlines in Hawaii may pose an additional risk to Hawaiian insular false killers.

Plastic ingestion by Hawaiian insular false killer whales has not been documented, as stomach content examination rarely occurs because of their infrequent stranding;

however, plastic debris has been found in organs of stranded harbor porpoise (*Phocoena phocoena*), Cuvier's beaked whale, killer whales, and many other species of cetaceans elsewhere in the world (Walker and Coe, 1990; Baird and Hooker, 2000; Pierrepont et al., 2005, Gomercic et al., 2006). In some cases, ingestion of these materials was considered the cause of death. Because many species of cetaceans live in waters far from shore, the bodies of animals that die from complications associated with plastic ingestion may sink and never be found (Baird and Hooker, 2000). Thus, deaths of whales and porpoises linked to plastic ingestion are likely underestimated. Despite national and international legislation to control ship sources of marine debris have been enacted, including the 1978 Prevention of Pollution from Ships (MARPOL) and the 1988 Annex V of MARPOL, it is difficult to enforce regulations over large expanses of ocean (Derraik, 2002). There is one documented case of ingestion of a net fragment by a false killer whale on the British Columbia coast (R. Baird, pers. comm.)

4.1.8.5. Oil spills

Oil is made up of thousands of different chemicals, including aliphatic, alicyclic and aromatic compounds (Clark and Brown, 1977). Some of the most toxic of these petroleum-related compounds are the PAHs. These compounds are prevalent in coastal waters, especially in urban embayments, and have been shown to alter normal physiological function in marine biota (Varanasi et al., 1989; Stein et al., 1993). Concerns have been raised over the effects of exposure to PAHs, alone or in combination with other toxic contaminants, on marine mammals because of the worldwide use of fossil fuels (Geraci and Aubin, 1990) and the occurrence of oil spills in areas that support marine mammal populations. Marine mammals can be exposed to oil by various routes, such as inhalation of volatile PAHs, direct ingestion of oil, and consumption of contaminated prey (O'Hara and O'Shea 2001).

Vertebrates, such as fish and cetaceans, rapidly take up PAHs present in the environment and quickly metabolize these compounds, most likely via induction of the mixed-function oxidases, to more polar compounds in their liver. The PAH metabolites are then concentrated in the bile for elimination (Varanasi et al., 1989). However, if a vertebrate (marine mammal) has been exposed to a large amount of petroleum (e.g., after an oil spill) and the liver enzyme system has been overwhelmed such that it cannot efficiently metabolize the PAHs, there is the possibility that petroleum-related PAHs pose a risk. After the *Exxon Valdez* oil spill in March 1989, several killer whales were observed to swim through oiled waters (Dahlheim and Matkin, 1994) in the region and 14 killer whales (33%) from the local AB pod disappeared between 1989 and 1991. There was no clear evidence to link the oil exposure to the disappearance (and presumable deaths) of these whales, but it is plausible (Matkin et al., 2008).

Oil spills have been reported in the main Hawaiian Islands. In May 1996, for example, an oil spill occurred in Pearl Harbor after a pipeline broke and spilled more than 25,000 gallons of oil (Honolulu Star Bulletin, 1996). The impact of this spill and other MHI oil spills (e.g., Barbers Point) on Hawaiian insular false killer whales and their prey are not known.

4.1.9. Harmful algal blooms

Single-cell organisms, such as dinoflagellates, constitute the base of the marine food web and certain of these organisms are responsible for the production of biotoxins associated with harmful algal blooms (HABs). From 1998 to 2008, HABs were documented in every coastal state of the United States and these events appear to be increasing in frequency and geographical distribution worldwide (Lopez et al., 2008). Biotoxins associated with blooms range from lipophilic chemicals (e.g., brevetoxin) to more water-soluble compounds (e.g., domoic acid and saxitoxin) and have been shown to accumulate in fish, shellfish, and top-level predators such as marine mammals and humans (Trainer, 2002).

Over the past 30 years, a number of marine mammal stranding events have been associated with HABs. Marine mammals can be exposed to biotoxins through ingestion of toxin-contaminated prey or seawater or through inhalation, with these compounds causing a variety of symptoms including muscular paralysis, confusion, seizures, memory loss and death (Trainer, 2002). The first well-documented HAB-associated deaths of marine mammals occurred in the spring of 1998 when more than 400 California sea lions (*Zalophus californianus*) died along the central California coast (Scholin et al., 2000). This large-scale mortality event coincided with a *Pseudo-nitzschia* diatom bloom in Monterey Bay. Clinical and histopathological examinations were conducted on live sea lions, with affected animals demonstrating seizures, head weaving, ataxia and abnormal scratching. The affected animals had lesions in the brain and heart that resembled those observed in mice and other laboratory animals exposed to domoic acid (DA), the biotoxin produced by *Pseudo-nitzschia*. Tissue samples collected from affected sea lions contained DA as did samples of *Pseudo-nitzschia* and northern anchovy, demonstrating that this biotoxin was transferred from the diatom to the sea lion via the anchovy. In addition, biotoxin exposure has been implicated in the mass strandings and deaths of humpback whales (saxitoxin) and manatees (*Trichechus manatus*) (brevetoxin) in the United States (Geraci et al., 1989; Oshea et al., 1991; Bossart et al., 1998; Flewelling et al., 2005). Recently, the first documented case of DA exposure in marine mammals from the southeast and mid-Atlantic region of the United States was reported in pygmy and dwarf sperm whales (*Kogia* spp.) that stranded from 1997 to 2008 (Fire et al., 2009). In the United States, this biotoxin is not commonly found along the southeast coastal states. Fifty-nine percent of the stranded *Kogia* contained measurable levels of DA, with feces containing the highest levels. A *Pseudo-nitzschia* bloom had not been reported in the region at the time of the strandings, and DA was not detected in feces and urine of other species of cetaceans that stranded in the same area in 2006 and 2008. In addition, it was unclear what the likely biotoxin vector was for the whales as very little is known about the feeding ecology of these pelagic cetaceans. The authors suggested that DA blooms occur in more offshore regions that are not monitored routinely for HABs and that the feeding behavior and habitat use of these pelagic whales should be further studied to help describe the movement of DA through their food web.

Ciguatoxin is one of the most commonly occurring biotoxins in subtropical and tropical regions of the world (Lehane and Lewis, 2000). In the main Hawaiian Islands, various reef fishes including kole (*Ctenochaetus strigosus*), roi (*Cephalopholis argus*), papio

(*Caranx* spp.) and po'ou (*Cheilinus rhodochrous*) have tested positive for this toxin (1987; Hokama et al., 1993), In addition, ciguatoxin symptoms have been reported in Hawaiian consumers after eating kole, kahala (*Seriola dumerili*) and ulua/papio (*Caranx* sp.) (Kodama and Hokama, 1989). However, information on ciguatoxin poisoning in marine mammals from the main Hawaiian Island/Northwestern Hawaiian Island region is scarce. For example, the only reported ciguatoxin-related mortality event involved the deaths of several Hawaiian monk seals (*Monachus schauinslandi*) from Laysan Island in 1978. Ciguatoxin and maitotoxin exposure was proposed to be the cause of the seal deaths as high levels of these biotoxins were measured in their livers (Gilmartin et al., 1980). No data on ciguatoxin levels in Hawaiian insular false killer whales or their primary prey are available. Thus, the potential risk posed by this biotoxin to the insular false killer whales from the main Hawaiian Islands is not known.

4.1.10. Ocean acidification and climate change

Ocean acidification can lead to changes in marine ecosystem productivity and community composition. However, impacts on false killer whales remain speculative. Species that are dependent on calcification would be directly affected by lowering of pH (Fabry et al., 2008). Many benthic, planktonic, and mesopelagic species have calcified body parts. In the pelagic environment one likely consequence is a gradual replacement of species with hard body parts (e.g., pteropods, heteropods) by those with soft body parts (e.g., gelatinous zooplankton). For example, jellyfish could become more prominent in marine ecosystems (Richardson et al., 2009). A relatively narrow suite of marine organisms feed directly on jellyfish (sea turtles, ocean sunfish (*Mola mola*), humans), and these species are not considered as prey for false killer whales. An increase in jellyfish would likely have a net negative impact on false killer whales unless there was a release from predation pressure or some kind of cascade in the trophic dynamics that led to increased prey abundance or decreased numbers of competitors. Prey-switching behavior to adjust to ecosystem changes is possible although ocean sunfish are rarely eaten by any marine predator. Acidification, per se, is unlikely to influence growth, mortality or reproduction in false killer whales given their well-insulated endoskeletons, physiological/biochemical modulation of pH, and the ability to simply relocate to other areas if acidification has strong spatial patterning. However, a large unknown is whether the insular population would remain in the same location if conditions became less favorable. Most motile organisms do exhibit varying degrees of kinesis, whereby movement can occur in response to a stimulus.

Climate change could be manifested in many ways, but three are briefly discussed here: sea level change, ocean temperature change, and expansion of low-productivity areas (i.e., "dead zones"). The first is unlikely to affect false killer whales directly. The primary effect of sea level increase will be the loss of terrestrial habitat which is critical for sea turtles and monk seals, for example (Baker et al., 2006), but not for false killer whales. In contrast, ocean temperature plays a key role in determining pelagic habitat for many species, and changes in this parameter would likely have a strong impact on false killer whales. Many prey species and competitor species have ranges closely linked to ocean temperature, both isotherms and gradients. Changes in temperature regimes could have severe impacts on pelagic ecosystems, in general. For false killer whales, specifically,

many of their forage species are migratory and/or mobile (i.e., few benthic species). The movement of other large predatory marine species ranges is likely to change, which could impact competition with false killer whales. However, a much better understanding is needed of prey preferences and predator-prey dynamics before speculating on the possible impacts of warming or cooling trends on insular false killer whales. Temperature may also have a direct linkage to productivity and growth rate but again it remains difficult to establish directionality of net effect. Increases in low-productivity areas (e.g., Polovina et al., 2008; Brewer and Peltzer, 2009) would probably have the strongest impacts on false killer whales. Lower productivity resulting in decreases in forage abundance would have a negative impact unless mobile forage species were concentrated into smaller regions that could then be exploited more easily. Again, presumed effects are large but net directionality is difficult to predict. As mentioned above regarding kinesis, one of the largest unknowns is whether the insular population would remain in the same location if conditions became less favorable.

4.2. Evaluating threats to Hawaiian insular false killer whales

Section 4(a)(1) of the ESA requires the agency to determine whether the species is **endangered** or **threatened** because of any of the following factors:

1) destruction or modification of habitat,
2) overutilization for commercial, recreational, scientific, or educational purposes,
3) disease or predation,
4) inadequacy of existing regulatory mechanisms, or
5) other natural or human factors.

The purpose of the Status Review is not to make the determination of whether a DPS is Endangered or Threatened, but to provide the best available scientific consensus to the agency for making that determination in light of these factors. To assist the agency in making that determination, the BRT organized its assessment of threats according to these Section 4(a)(1) factors using a Threat Table. The team evaluated the potential role that each factor listed in the Table may have played in the decline of Hawaiian insular false killer whales and the degree to which each factor is likely to limit population growth in the foreseeable future. Within the 5 categories, specific threats were ranked according to severity, geographic scope, the level of certainty that insular false killer whales are affected, and overall current and future (60 years) risk imposed. We have limited consideration of future threats to a 60 years period as this corresponds roughly to the life span of a false killer whale and represents a biologically relevant time horizon for projecting current conditions into the future.

The terms used in Table 4-4 are defined as follows:
Factors in decline – The specific historical factors responsible for a species' (or DPS) decline. The statutory ESA Section 4(a)(1)(A)-(E) factors encompass these historical factors. Factors in decline may or may not persist and limit the viability of the species.

Key limiting factors – The physical/biological/chemical features presently experienced by the population that result in the greatest reductions in the population's ability to recover compared to the conditions experienced prior to the onset of these threats. These

key limiting factors are the most significant natural and anthropogenic factors that are currently impeding the ability of the population to recover. Key limiting factors are those that, if improved, would have a marked favorable effect on the species' status.

Threats – Those human or natural events/actions that are responsible for, contribute to, or could contribute to the key limiting factors, inclusive of current and future threats. Future threats are activities that are likely to happen but are not currently occurring, or that may be existing threats that are likely to result in a mounting risk to the species. Examples of threats include those that alter or degrade habitat, actions that degrade water quality, and mixed-stock harvest management strategies that result in incidental mortality of the subject species. Threats can be described according to time frame, as follows:

- **Historic Threats** = threats that occurred in the past and may or may not be occurring presently.
- **Current Threats** = threats that are occurring now.
- **Future Threats** = threats that are likely to affect insular false killer whales over the next 60 years (duration chosen because it corresponds roughly to the life span of a false killer whale).

Key Ecological Attributes Affected – response of the organisms to the stressor, usually in the form of altered physiological processes (e.g., reduced reproductive potential or growth) or mortality.

The following factors were evaluated by the team to assess the overall level of risk for individual threats. Severity, geographic scope, and level of certainty were scored as a numeric value corresponding to high, medium, or low (as defined below) by each member, and the median value across the team member's point allocations are presented in Table 4-4. Threats were ranked using this high/medium/low scale because this better corresponds with how the threats may be evaluated, as opposed to a plausibility scale used previously.

Severity = the level of damage to the population that can reasonably be expected within 60 years under the current circumstances (i.e., given the continuation of the existing management/conservation situation). Specific rankings for this category are defined as follows:

- **3 = High:** The threat is *likely to seriously reduce or eliminate* the population over some portion of its range.
- **2 = Medium:** The threat is *likely to moderately reduce* population size over some portion of the population's range.
- **1 = Low:** The threat is *likely to only slightly reduce* the population size over some portion of the population's range.

Geographic Scope = the geographic scope of impact on the population that can reasonably be expected within 60 years under the current circumstances (i.e., given the continuation of the existing situation). Specific rankings for this category are defined as follows:

- **3 = Widespread:** The threat is likely to be widespread or pervasive in its scope and affect the population *throughout its range*.
- **2 = Moderate spread:** The threat is *likely to be localized* in its scope and affect the population at *some locations within its range*.
- **1 = Localized:** The threat is *likely to be very localized* in its scope and affect the population in a *limited portion of its range*.

Level of Certainty that Species is Affected = the level of certainty that the threat will affect the population with the severity and geographic scope ascribed in the next 60 years. Specific rankings for this category are defined as follows:

- **3 = High:** There is *definitive* published and unpublished data to support the conclusion that this threat is likely to affect the population with the severity and geographic scope ascribed.
- **2 = Medium:** There is *some* published and unpublished data to support the conclusion that this threat is likely to affect the population with the severity and geographic scope ascribed.
- **1 = Low:** There is *little* published and unpublished data to support the conclusion that this threat is likely to affect the population with the severity and geographic scope ascribed.

Overall Threat Level Currently and into the Future (over the next 60 years) = the current and future risk that this threat will contribute to the decline of the species over the next 60 years. Specific rankings for this category are defined as follows:

- **3 = High:** This threat is likely to *eliminate or seriously degrade* the Hawaiian insular false killer whale population *throughout its range*.
- **2 = Medium:** This threat is likely to *moderately degrade* the Hawaiian insular false killer whale population at *some locations within the species' range*.
- **1 = Low:** This threat is likely to *only slightly impair* the Hawaiian insular false killer whale population in a *limited portion of the species' range*.

Overall Ranking of Limiting Factor = the current and future risk that this limiting factor will contribute to the decline of the species over the next 60 years. Specific rankings for this category are defined as follows:

- **High:** This limiting factor encompasses *a high number* of threats that moderately or very likely to contribute to the decline of Hawaiian insular false killer whales, or contains some individual threats identified as very likely to contribute to the decline of the population throughout its range.
- **Medium:** This limiting factor encompasses *an intermediate number* of threats that are likely to contribute to the decline of Hawaiian insular false killer whale population or contains some individual threats identified as moderately likely to contribute to the decline of the population at many locations within its range.

- **Low:** This limiting factor encompasses *a low number* of threats that are likely to contribute to the decline of Hawaiian insular false killer whales at many locations within the species' range.

Because the severity and scope of individual threats may change through time, each threat was evaluated based on its historic impact and its current or future potential for impact. The period of historic impact may vary for each threat and is shown in the table. Hawaiian insular false killer whales are believed to have declined markedly during the 1990s, although their current population trajectory is unknown. Some potential causes for decline have been reduced in severity or geographic scope, while others have increased. The factors that are believed to have historically contributed the most (with a median score of 2 or higher) to the decline of Hawaiian insular false killer whales are: (1) the destruction, modification or curtailment of the population's habitat or range (i.e., reduced prey biomass and size, competition with commercial fisheries, and natural and anthropogenic contaminants); (2) overuse (i.e., live capture operations occurring prior to 1990); (3) disease and predation resulting from exposure to environmental contaminants; (4) inadequate regulatory mechanisms (i.e., lack of an observer program for nearshore fisheries); and (5) other natural and man-made factors (i.e., interactions with fisheries and reduced genetic diversity). Specific information on the severity, geographic scope, and the level of uncertainty about each threat is discussed previously in Section 2.3.5. Some potential causes of decline have been reduced in severity or geographic scope, and this should have reduced the overall threat level to Hawaiian insular false killer whales. Most notably, the reduced threats include live captures, halted in the early 1990s, and interactions with the commercial longline fishery, which have likely declined since the establishment of the 25–75 nmi longline exclusion boundary in 1991–1992.

Other threats have emerged over time or have increased in severity or geographic scope, making them more relevant to the current and/or future decline of the population. The overall levels of most threats considered under the destruction, modification or curtailment of habitat generally remain unchanged, with reduced prey biomass and size, competition with commercial fisheries, and accumulation of natural or anthropogenic contaminants remaining important potential problems. Other threats that may significantly affect Hawaiian insular false killer whales (median score of 2 or 3) include (1) disease and predation potentially caused by exposure to environmental contaminants or change increases in disease vectors as a result of short- and long-term climate change), (2) inadequate regulatory mechanisms (i.e., the continued lack of an observer program for nearshore fisheries), and (3) other natural and man-made factors (i.e., lack of information on level of interactions with troll, handline, shortline, and kaka-line fisheries, reduced genetic diversity, inbreeding depression, and Allee effects, and anthropogenic noise). Overall, the BRT ranked threats related to small population size and hooking, entanglement, or intentional harm by fishers as the highest threats to Hawaiian insular false killer whales.

Table 4-4. Historic and current and/or future threats to Hawaiian insular false killer whales organized according to the ESA (4)(a)(1) factors and population limiting factors.

ESA Factors for Decline	Key Limiting Factor	Threats	Key Ecological Attributes Affected	Historic Impact					Current and Future Impact				Overall Ranking of Limiting Factors
				Period of impact	Severity	Geographic scope	Level of certainty	Overall Threat Level	Severity	Geographic scope	Level of certainty	Overall Threat Level	
Habitat Destruction, Modification, or Curtailment	Reduced food quality and quantity	Reduced total prey biomass	Mortality, Foraging success	through early 1970s	2	3	2	2	2	3	1.5	2	Medium
		Competition with commercial deep-set and shallow-set longline fisheries	Mortality, Foraging success	through early 1990s	3	3	2	3	2	3	1.5	2	
		Competition with troll, handline, shortline, and kaka line fisheries	Mortality, Foraging success	before late 1980s	2	3	1	2	2	3	1	2	
		Competition with recreational fisheries	Mortality, Foraging success	before late 1980s	1	3	1	1	1	3	1	1	
		Reduced prey size	Mortality, Foraging success	through early 1970s	2	3	2	2	2	3	2	2	
		Ocean warming increasing low-productivity zones	Mortality, Foraging success	primarily current/future threat					1	3	1	1	
		Ocean acidification	Mortality, Foraging success	primarily current/future threat					1.5	3	1	1.5	
		Natural and anthropogenic contaminants	Mortality, Reproductive potential	through mid-1980s	2	3	2	2	1	3	1	1	
Overutilization	Low density	Live-capture	Reproductive potential	pre-1990	2	3	2	2	Primarily historic threat				No current threat
Disease and Predation	Disease	Harmful algal blooms	Mortality, Reproductive potential	primarily current/future threat					1	2	1	1	Medium
		Environmental contaminants	Mortality, Reproductive potential	through mid-1980s	2	3	1	2	2	3	2	2	
		Long and short-term climate change	Mortality, Reproductive potential	primarily current/future threat					2	3	1.5	2	

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Historic Impact					Current and Future Impact				
				Period of impact	Severity	Geographic scope	Level of certainty	Overall Threat Level	Severity	Geographic scope	Level of certainty	Overall Threat Level	Overall Ranking
Disease and Predation	**Disease**	**Parasitism**	Mortality, Reproductive potential, Foraging success	same threat level					1	3	1	1	Medium (w/above threats)
	Predation	Killer whales	Mortality	same threat level					1	3	1	1	Low
		Shark (inc. cookie-cutters)	Mortality	same threat level					1	3	1	1	
	Competition	Marlin, sharks, etc.	Foraging success, Behavior	through early-1970s	1	3	1	1	1	3	1	1	Low
Inadequate Regulatory Mechanisms	Incidental take	Lack of reporting/observing of nearshore fishery interactions	Mortality, Behavior	through late-1980s	3	3	2	3	2	3	1.5	2	Medium
Other Natural or Manmade Factors	Hooking, entanglement, or intentional harm (e.g. shooting, poisoning, explosives)	Interaction with commercial longline fisheries	Mortality, Behavior	through early-1990s	3	3	2	3	2	2	1	1	High
		Interactions with troll, handline, shortline, and kaka-line fisheries	Mortality, Behavior	pre mid-1970s	1	3	1	1	3	3	1	3	
		Interactions with aquaculture facilities	Mortality, Behavior	pre-2010	1	1	3	1	1	1	1	1	
	Environmental pollutants/toxins	Oil spills	Mortality, Reproductive potential, Foraging success, Behavior	same threat level					1.5	1	1	1	Low
		Plastic ingestion	Mortality, Foraging success	gradually increasing threat					1	2	1	1	
	Small population size	Reduced genetic diversity	Reproductive potential	through 1989	1.5	3	1	2	2	3	2	2	High
		Inbreeding depression	Reproductive potential	through 1989	1	3	1	1	2	3	1	2	
		other Allee effects	Reproductive potential, Mortality	through 1989	1	3	1	1	2	3	1	2	

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Historic Impact					Current and Future Impact				
				Period of impact	Severity	Geographic scope	Level of certainty	Overall Threat Level	Severity	Geographic scope	Level of certainty	Overall Threat Level	Overall Ranking
Other Natural or Manmade Factors	Other	Sonars and seismic exploration (including military, oceanographic, and fishing sonars)	Mortality, Reproductive potential, Behavior, Foraging success	HRC formation	1	2	1	1	2	2	1.5	2	Medium
		Vessel strikes	Mortality	gradually increasing threat					1	2	1	1	Low
		Whale/dolphin watching	Behavior, Reproductive potential	gradually increasing threat					1	2	1	1	Low

4.2.1. Justification of most significant key limiting factors

4.2.1.1. Reduced food quality and quantity

The impacts of reduced prey abundance appear to represent a medium risk for insular false killer whales both in the past and in the future. Although declines in prey biomass were more dramatic in the past when the insular false killer whale population may have been higher, the total prey abundance remains very low compared to the 1950s and 1960s as evidenced by CPUE data from Hawaii longline fisheries and biomass estimates from tuna stock assessments. A total of 2.9 to 3.9 millions of pounds per year is the estimated magnitude of prey required by the present-day Hawaiian insular false killer whale population (Section 4.1.2.2). Competition with longline fishermen for potential prey within the insular false killer whale habitat seems to have represented a higher risk prior to the early 1990s when the longline fishery was harvesting many millions of pounds of fish per year (Figure 4-8), and where reported catch locations (Figure 4-2) were almost all in what is now the longline exclusion zone. In the core nearshore habitat (< 40 km from shore) only the troll and handline fisheries now harvest as much (Figure 4-9 and 4-10) as is estimated to be consumed annually by the Hawaiian insular false killer whale population. Long-term declines in prey size also represent a medium threat from the earliest records to the future, and are related to measures of reduced total prey abundance which include prey size.

4.2.1.2. Interactions with commercial fisheries

The intense and increased fishing activity within the known range of insular false killer whales since the 1970s suggests a high level of risk of fisheries interactions, even though the extent of interactions with almost all of the fisheries is unquantified or unknown. The only fishery for which there are recent quantitative estimates of hooking and entanglement of false killer whales is the longline fishery. That fishery has been largely excluded from the known range of Hawaiian insular false killer whales since the early 1990s, suggesting a lower current and future risk from longlining (assuming the current restrictions remain in place) However, it is likely that unobserved interactions with the longline fishery represented a high risk through the early 1990s. For the other fisheries, a high level of current and future risk is assumed based on the scale and distribution of the troll and handline fisheries, and on anecdotal reports of interactions with cetaceans, although interactions specific to false killer whales are known only for the troll fishery. The troll fishery has by far the greatest participation and effort in fishing days of any fishery within the known range of insular false killer whales, followed by the handline fishery, with the kaka- and shortline fisheries a distant third. The kaka-line and shortline fishing methods have been implicated as a threat based on the similarity of these fishing methods to the longline method.

4.2.1.3. Disease

Disease-related impacts of individual threats, such as exposure to environmental contaminants, parasites and pathogens, harmful algal blooms (HABs) and, short and long-term climate change, pose a medium threat to Hawaiian insular false killer whales. Levels of persistent organic pollutants have decreased since their bans in the United States beginning in the 1970s but they continue to be measured in biota from the main Hawaiian Islands, including Hawaiian insular false killer whales. Recently, summed

PCBs measured in some of these whales were above a threshold value (17,000 ng/g, lipid) associated with deleterious health effects (e.g., thyroid dysfunction, immunosuppression). Although little is known about the occurrence of parasites in Hawaiian insular false killer whales, Hawaiian monk seals from the main Hawaiian Islands were exposed to protozoan and coccidian parasites. Discharge of raw or partially treated sewage effluent and contaminated freshwater runoff in to marine coastal waters can increase the risk of pathogen transmission to animals that reside in nearshore areas, such as Hawaiian insular false killer whales. Several Hawaiian monk seals died in the late 1970s, and these deaths were attributed to exposure to the marine biotoxins ciguatoxin and maitotoxin. HABs appear to be increasing in frequency and geographical distribution worldwide and pose a future threat to Hawaiian insular false killer whales. Climate change and ocean acidification could alter the productivity and composition of the main Hawaiian Islands ecosystem. Increased water temperature could change the composition of microbial communities in the main Hawaiian Islands and thus create an environment that could support new microbes not usually found in the region, thus exposing Hawaiian insular false killer whales to novel pathogens.

4.2.1.4. Small population size

Small population size represents a high risk because the current estimated number of breeding adults is approaching levels where inbreeding depression could have increasing negative effects on population growth rate and because other social factors (such as efficiency in group foraging and potential loss of knowledge needed to deal with unusual environmental events) may further compromise the ability of Hawaiian false killer whales to recover to healthy levels. SRKWs and Cook Inlet beluga whales have not made substantial recovery despite protection. These populations are at similar low levels and show that small population size poses a risk for reduced population growth for social odontocetes.

4.3. Population viability analysis

4.3.1. Description of the modeling exercise

The risk to this population was examined using a population viability analysis (PVA), which is a model type designed to estimate the probability of extinction over time for a population, given the current best understanding of the population's vital parameters and threats. PVA is an accepted format for quantitative risk analysis for endangered species and has been used in numerous marine mammal cases (MMC, 2007). Model results are used to evaluate our best estimate of future risks but are not intended to be predictive models, especially for species placed under the protection of the ESA where actions to reduce risk are expected.

For Hawaiian insular false killer whales, data are available on both trends and abundance that permit PVA modeling. The BRT chose to assess risk using three primary data sources: mark-recapture abundance estimates (Baird et al., unpublished PSRG report), relative trend data from aerial surveys (Mobley et al., 2000; Mobley, 2004; Baird, 2009), and an estimate of minimum abundance from 1989 (Reeves et al., 2009). The model uses the Baird abundance estimates (including uncertainty) but uses the Mobley time series only in a relative way to fit trends in abundance (see Appendix B for model details). The

reason that the Mobley counts were not used as measures of absolute abundance is that there is acknowledged negative bias because the aerial observers could not see directly below the plane and did not estimate the proportion of animals missed on the track line. This negative bias should have been constant over surveys and, therefore, have not biased use of the data for estimating trends in abundance. The model projects forward from 1989, the year of the Reeves et al. (2009) minimum count. The counts from Reeves et al. (2009) were used as the lower bound for plausible abundance in 1989 (see Appendix A). The team held an extensive discussion and concluded that the observers (Reeves and Leatherwood) were very experienced and would have correctly identified the species. However, uncertainty remains as to whether all the animals seen were Hawaiian insular false killer whales or whether pelagic false killer whales may have been present. The team took this uncertainty seriously, had lengthy discussions about the implications of the uncertainty, and finally expressed this uncertainty using the plausibility point system and verbal descriptions of different parameters (see Appendix A).

Recognizing the uncertainty in the model parameters, the BRT chose a Bayesian analysis that would include both the statistical uncertainty and also uncertainty in parameters for which there are no direct data (such as parameters for the upper limit on abundance in 1989 and the population rate of change).

After thorough discussions of each parameter in the model, the team composed text about the plausibility of different alternatives (Appendix A). In most cases, the BRT members independently allocated plausibility points among the alternatives. The group viewed the assembled tables of points and discussed any cases where the spread in points was large to assure that disparities were real differences in opinion as opposed to lack of clarity about the question. All BRT members agreed to make their point allocations known, which facilitated candid deliberation by the team. The allocation of points was iterative; after a first-round allocation, members had the opportunity to adjust heir allocation. Although adjustments were not uncommon, this iterative process enabled a robust discussion, and the in the end there were no large shifts in the total plausibility percentages. In some cases, notably the question of whether to use the encounter rate of "individuals" or "groups" in the Mobley time series, members reviewed the simulated population trajectories over the period with data (1989–2008) and changed their point allocation based on model fit. Plausibility scoring was done on the following parameters:

1) Lowest plausible abundance in 1989
2) Highest plausible abundance in 1989
3) Mobley time series represented in units of "group" or "individual"
4) Inclusion or exclusion of Kauai sightings in mark-recapture abundance estimates
5) Level of environmental variance
6) Autocorrelation in growth rates (reflecting series of good or bad years)
7) Functional form and magnitude of Allee effects
8) Annual probability of catastrophes
9) Magnitude of catastrophes
10) Variance in probability of catastrophes

The team also chose to explore model structure uncertainty by using two separate types of models: the one rate-of-change model and the two-rate-of-change model. The team considered two hypotheses for population growth rate: 1) the decline apparent in the Mobley series (which occurred after the implementation of the long-line exclusion area) results from ongoing mortality likely resulting from nearshore fisheries interactions, and 2) the decline apparent in the Mobley series results from a reduction in prey availability within the range of the insular population. These are logically, not mutually exclusive. However, the team chose to consider them according to the extreme of their consequences in hopes of bracketing the risk. The ongoing mortality hypothesis would result in growth rate that would be projected into the future. This represents the highest-risk interpretation of the data. The reduction-in-prey hypothesis results in the observed decline through the Mobley series but has a second growth rate for current and future growth based on a stable (but lower) prey base. This two-rate-of-change model is the lowest risk interpretation of that data. A model that allowed the decline in the Mobley series to result from both ongoing mortality and a reduced prey base would result in an intermediate level of risk. Arguments for and against these models are detailed in Appendix A.

Plausibility point assessments were taken on whether the one rate-of-change (single growth rate) model best represented current risk or whether there was a change between the earlier rate and the present rate that would be best represented by a two-rate-of-change model. For the two-rate-of-change model, there was discussion of whether the available data could support a model with 2 more parameters (the year of change and the slope of the second trend in abundance). Members were concerned that the estimate for the second rate would be poorly characterized because the year of change was likely to fall late in the Mobley time series but prior to the Baird estimates. Thus, in most cases only two abundance estimates that were close in time would be used to estimate the second rate of change, and the estimate would therefore be strongly influenced by the specified prior distribution. In Bayesian statistics, a distribution representing prior knowledge about the parameter of interest (the 'prior distribution') is specified for each estimated parameter, such as the current rate of change of the population. Because the team recognized that the paucity of data would result in the prior distribution being influential on the posterior distribution, three alternative prior distributions were explored. The first distribution represented a case with a very broad range (from an annual 30% decline to an annual 30% increase) to capture the results that one would expect from a non-Bayesian approach where prior knowledge is not used (such as from standard non-Bayesian methods, e.g., Maximum Likelihood estimation). The second prior distribution constrained positive rate of change to the maximum plausible for false killer whales and made the prior distribution symmetrical (from an annual 4% decline to an annual 4% increase) so that the prior distribution had an expectation of no growth. The third distribution also constrained growth to biologically plausible values but allowed declines to the levels indicated by the Mobley time series (from an annual 10% decline to an annual 4% increase), which has an expectation of a – 3% population decline. Members allotted plausibility votes among these alternatives conditional on their plausibility points for the two- rate-of-change model.

Based on results of the plausibility point allotments, the team chose a base case scenario with the following parameters that were expected to have the greatest influence on population trajectories (a specific explanation of each parameter is provided in Appendix A):

1) An upper limit for the prior distribution for 1989 abundance of 1392 (one standard error above the point-estimate for density at Palmyra Atoll, applied to the Hawaii insular false killer whale stock area within 140 km of the main Hawaiian Islands)
2) Environmental variance 0.02 (based on Northern Resident killer whales)
3) Allee effect (ongoing and affecting both reproduction and survival)
4) Autocorrelation (relatively strong)
5) Aerial encounter rate (group)
6) Probability and magnitude of catastrophe (10% killed on average every 50 years)
7) Include Kauai in Baird abundance estimate
8) Prior distribution minimum abundance estimate of 470 (Reeves minimum)
9) Use the aerial trend data

Models were run using the base case, but the second choices for each of these parameters were referred to as model "variants". The only model with multiple parameters that differed from the base model attempted to choose a set of parameters that would result in the most optimistic outcome for the population. The logic of examining this "rosy" case is that if even this scenario resulted in high risk, all other combinations would be at higher risk. In this case, there is nothing to be gained in understanding the risks facing Hawaiian insular false killer whales by running all the permutations of parameters that would all result in higher risk. All 11 models (the base-case plus 10 variants) were run for the one rate-of-change model and the 3 two-rate-of-change models.

The team also ran several sensitivity tests to increase transparency of the importance of different parts of the model (detailed in Appendix B). The six sensitivity trials were: (1) the aerial trend data were not used, (2) a higher upper bound on the prior distribution for 1989 abundance (3000), (3) a broader prior distribution for 1989 abundance (50 to 3000), (4) no Allee effect was used, (5) no mortality events ('catastrophes') occurred, and (6) small population effects only.

4.3.2. Model results

Complete results are given in Appendix B. The team chose to represent results from five models that give readers a comprehensive overview:

- One rate-of-change model
  - base-case
  - variant using individual aerial encounter rate (rather than group)
  - variant using maximum abundance of 769 (the point estimate for the highest false killer whale density measured anywhere, in the EEZ surrounding Palmyra, as applied to the insular population range)
- Two rate-of-change model
  - base-case where the second rate could be between an annual 4% decline and an annual 4% increase

- base-case for the two growth-rate model where the second growth rate could be between an annual 10% decline and an annual 4% increase.

For the favored one rate-of-change model, the base-case represents the scenario rated as most plausible by the team. The two variants show the most pessimistic case and the most optimistic case. The two-rate-of-change model, with the "uninformative" prior distribution that was between an annual decline of 30% and an annual increase of 30%, resulted in a posterior distribution for the current population growth rate that was primarily in the range above what is plausible for false killer whales (more than 4%/year). The team found this revealing because it suggested that the two-rate-of-change model is likely to be overparameterized given the paucity of the data.

Trajectories for the one rate-of-change model base case indicate nearly all simulated populations are extinct by 2050 (Figure 4-13). The strong decline fits the Mobley time series (black dots) and the simulations pass through the current abundance estimates (open circles). The lines indicate the best estimate (middle line) and an envelope that contains 95% of the trajectories.

Trajectories for the most plausible among the two-rate-of-change model, with the second growth rate constrained between an annual decline of 10% and an annual increase of 4%, reveal very different behavior (Figure 4-14).

Although these trajectories are shown running to complete extinction, the team preferred to represent extinction as a near-extinction value, below which recovery may not be possible. The value of 20 individuals was chosen because this is a number smaller than the average group size for Hawaiian insular false killer whales. There is also an empirical example suggesting that recovery is unlikely below this level in a group if killer whales called the AT1 transient stock. The group numbered 22 individuals when the last calf was born in 1984 (Craig Matkin, North Gulf Oceanic Society, pers. comm.). Only 7 individuals remained alive in 2008 (Allen and Angliss, 2010), and this group will almost certainly go extinct once the remaining individuals die. Because period of time between the last reproduction and the demise of the last individual could take decades, the near-extinction year better represents this time of remaining viability. Another reason that 20 individuals seems suitable for false killer whales is because they forage cooperatively, making successful recruitment unlikely once their numbers become fewer than one foraging group. Table 4-5 shows the probability of extinction for the five models chosen to cover the range of plausible risk

Of the 44 models run, only one had less than a 5% chance of reaching 20 individuals in 75 years. The model with that result (two-rate-of-change model with a distribution for the second growth rate between an annual decline of 30% and an annual increase of 30% that used individuals for the aerial survey) was not considered to be plausible by the team because the posterior distribution for population growth rate was largely above the range of values that are plausible for false killer whales.

With one exception, the sensitity tests differed little from the base case. The small population effect sensitivity test had a 1.7% chance of near extinction in 75 years. This model assumed level population growth (r = 0) and examined only Allee effects, catastrophes and variance in population growth rate. Therefore, the data indicating past trends or the potential for such trends to persist are influential and it is not small population size alone that makes Hawaiian false killer whales at high risk.

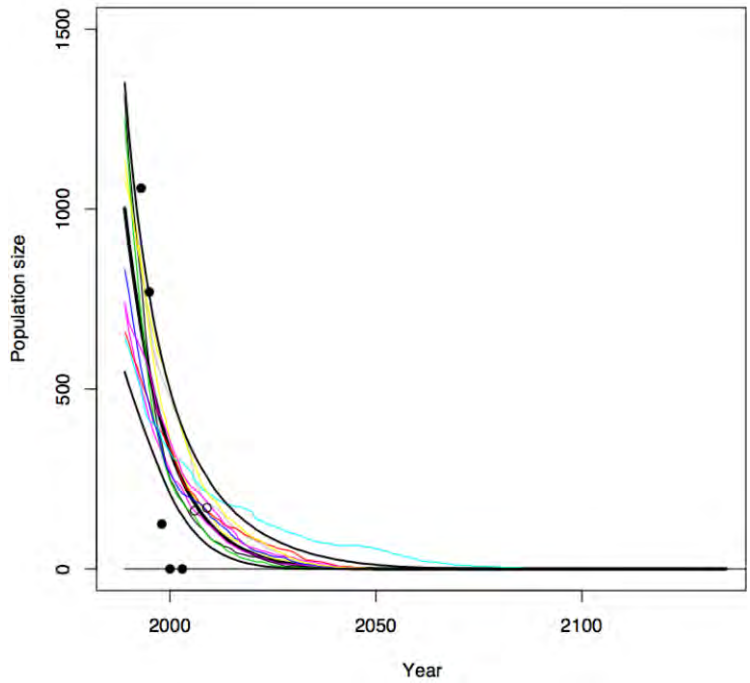


Figure 4-13. Trajectories for the one growth-rate base-case model with group numbers for the Mobley time series as black dots and Baird abundance estimates as open circles. The most likely trajectory is the middle line. Outer lines envelop 95% of the trajectories. Ten randomly chosen trajectories are shown in color to illustrate the representative variability.

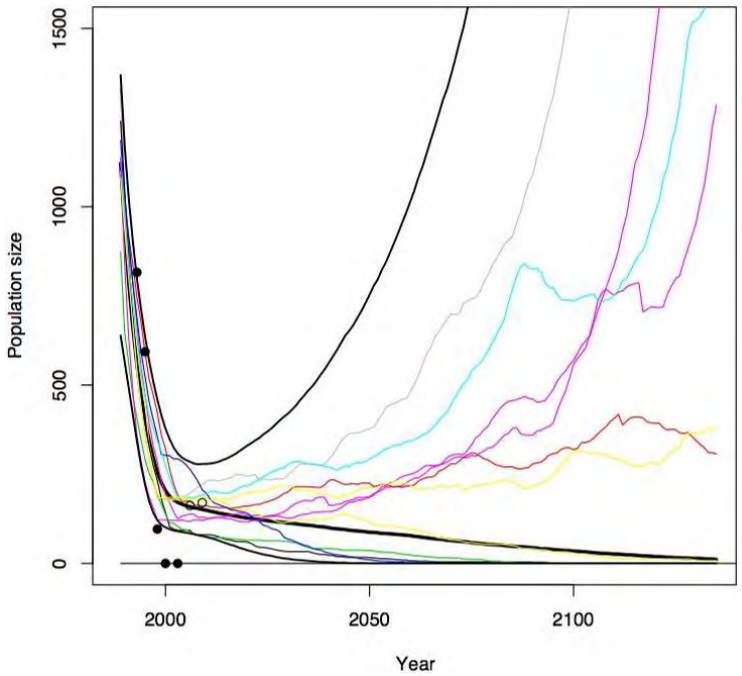


Figure 4-14. Trajectories for the two growth-rate base-case model with group encounter rate for the Mobley time series as black dots and Baird abundance estimates as open circles. The most likely trajectory is the middle line. Outer lines envelop 95% of the trajectories. Ten randomly chosen trajectories are shown in color to illustrate the representative variability.

Table 4-5. The percent chance of reaching near-extinction (fewer than 20 individuals) by 50 years (2 generations), 75 years (3 generations), and 125 years (5 generations) for 5 models covers a range of possible risks. The favored one-growth-rate model (using individual encounter rate from the aerial surveys) gives the highest risk model compared with the variant constraining the abundance to be fewer than 769 (using the point-estimate for the density from Palmyra) gives the lowest risk model.

	Probability of near-extinction (< 20)		
Model	**50 years (2 gen)**	**75 years (3 gen)**	**125 years (5 gen)**
1 growth-rate (base)	91%	99%	100%
1 growth-rate (individual encounter rate)	98%	100%	100%
1 growth-rate (769 animal maximum in 1989)	76%	98%	100%
2 growth-rate base (– 4% to 4%)	3%	18%	35%
2 growth-rate base (– 10% to 4%)	19%	36%	50%

4.4. Treatment of uncertainty

Any consideration of risk involves treatment of uncertainty. Uncertainty can be categorized into linguistic uncertainty, statistical uncertainty, and uncertainty about unknowns. Linguistic uncertainty results from language that means different things to different people (Regan et al. 2002). For example, the guidelines for assessing whether a group of individuals qualifies as a DPS use the word “markedly” in the significance criterion with the phrase “differs markedly from other populations in its genetic

characteristics." We dealt with linguistic uncertainty in two ways. Whenever possible, we gave our interpretation of words likely to be subject to linguistic uncertainty. We also discussed any cases when members had disparate point allotments to make sure that the differences were real differences of opinion as opposed to misunderstandings about the topic or question.

Statistical uncertainty results from sampling a "population" and making estimates based on inferences from the sample. The examples for Hawaiian insular false killer whales are abundance estimates made from aerial surveys or mark-recapture methods. In both cases, the sampling process results in an estimate of precision that is expressed in a range of possible values. The aerial surveys also have acknowledged negative bias that was not quantified (and thus cannot be corrected for). Bias results from a systematic factor that results in either overestimating or underestimating the value of interest. In the case of the aerial surveys, the estimate was a known underestimate because of two factors: 1) observers could not see directly below the plane, and 2) no correction was made for animals that were diving. Because the estimation method assumed that all animals on the track of the plane were seen, the resulting estimate violated this assumption and, therefore, is an underestimate.

Finally, there are uncertainties about unknowns. There are several important parameters for which we have no direct data for Hawaiian insular false killer whales. In some cases, data from false killer whales elsewhere allow researchers to place plausible bounds for the Hawaiian population. For example, plausible upper bounds for numbers of Hawaiian insular false killer whales in 1989 were based on densities of false killer whales in waters surrounding Palmyra Atoll, which has the highest known density for this species. We used those data to develop alternative values and then allotted plausibility points to indicate which value was favored. Arguments for each value were written out before members allotted their points.

Other uncertainties with no direct data are treated with data from other species deemed similar. For example, the variability in growth rate from year to year (often called environmental stochasticity) was based on killer whale case studies. Yet other uncertainties cover an even broader range of possibilities because there are no data even for similar species. For example, the frequency in catastrophes ranged from once in ten years to once in one hundred years. Although it may seem that treating variables with no data makes the scientific results questionable, denying even the possibility of a catastrophe would be unrealistic. We attempted to cover a range of possibilities so that we could see whether our conclusions were robust to these uncertainties. If the outcome was not substantially different over the range of plausible values, then the outcome was considered robust.

Our approach was to be as specific as possible on where the uncertainties existed and to make our treatment of options transparent by using the method of assigning points to different alternatives.

4.5. Overall assessment of risk

The BRT was asked to describe, to the extent possible, the links between demographic risks and the five causative factors (see Section 4.2, Potential Risk Factors). The BRT was also asked to illustrate the level of risk indicated by the collective body of information. Linkages between the causative factors and the observed trends in abundance are poorly understood. The risk table (Table 4-4) shows a number of factors causing a medium risk of extinction and only a few causing a high risk of extinction. Thus, the team felt that the decline had likely resulted from a number of factors acting together. For example, a reduction in prey base could make it difficult for individuals to find food, potentially leading to nutritional stress that may result in the metabolizing of fats laden with environmental contaminants and causing reduced immunocompetence that would increase the likelihood of diseases to cause additional mortalities.

The PVA models do not consider, or imply, causation but rather use the trend to embody the result of the threats acting in concert on the population. Projections into the future express whether those threats remain operative at the same level, decrease or cease. The spread of plausibility points between the one growth-rate model and the two growth-rate model express the team assessment as to whether the threats persist (one rate-of-change model) or potentially change (two-rate-of-change model). Both were considered plausible scenarios with more weight going towards a persisting decline because there was no direct evidence to indicate why population growth rates should have changed during the early 2000s. The team noted that the decline indicated by the Mobley time series occurred after the longline fishery had been excluded from the majority of the range of Hawaiian insular false killer whales.

We considered whether the threats associated with the five factors were generally occurring throughout all or a significant portion of the range of Hawaiian insular false killer whales. As seen in Table 4-4, most factors apply throughout the range because tagging results show the animals themselves move throughout the main Hawaiian Islands, where the threats are occurring.

The next consideration is the time period for considering risk. False killer whales live to more than 60 years of age, so considering a time span of less than the age of an individual would be biologically inappropriate. We also considered generation time (25 years) because this is a commonly used timeframe in conservation biology, and consequently the PVA is expressed in these terms. Generation length reflects the turnover rate of breeding individuals in a population. In general, time-based measures are scaled for the different rates at which taxa survive and reproduce, and generation length is used to provide this scaling.

Considering the above, the team agreed to the following categories of risk of extinction:

- *High risk*: Within 3 generations or the maximum age, whichever is the greater, there is at least a 5% chance of the population falling below a level where recovery is not likely. Because false killer whales are highly social animals this level was set at 20 animals, which is about the average group size.

- *Medium risk*: Within 5 generations there is at least a 5% chance of the population falling below a level where recovery is not likely. Because false killer whales are highly social animals this level was set at 20 animals, which is about the average group size.
- *Low risk*: Within 5 generations there is less than a 5% chance of the population falling below a level where recovery is not likely. Because false killer whales are highly social animals this level was set at 20 animals, which is about the average group size.

The team was in full agreement that Hawaii insular false killer whales were at high risk of extinction because all of the plausible PVA results met its agreed definition of high risk. Most PVA model results exceeded 5% chance of extinction in 75 years by a very wide margin, with most indicating a greater-than-90% chance of extinction within 3 generations.

4.6. Application of the International Union for the Conservation of Nature criteria

The International Union for the Conservation of Nature (IUCN) criteria do not apply within the U.S. ESA framework but provide a separate context for interpreting risk. Although the IUCN Redlist criteria were originally created to measure the global risk to all species, the widespread need for measures of risk on smaller scales prompted the IUCN to create guidelines for regional applications of their criteria. The guidelines (IUCN, 2001) suggest using the criteria unmodified if the group under consideration meets the following definition for subpopulations:

Geographically or otherwise distinct groups in the (global) population between which there is little demographic or genetic exchange (typically one successful migrant individual or gamete per year or less (IUCN, 2001); a subpopulation may or may not be restricted to a region.

By the IUCN criteria, Hawaiian insular false killer whales qualify as a subpopulation, given the strong differentiation indicated by the genetic data (see Section 2.3.2).

The IUCN "Redlist" criteria classify species into four different risk categories—critically endangered, endangered, vulnerable, and least concern—using five criteria:

A. magnitude of population reduction,
B. geographic range,
C. abundance and trends in abundance,
D. abundance alone (population size numbers fewer than 50 mature individuals), and
E. quantitative estimate of the probability of extinction.

Qualification for a risk category depends on meeting any one of these criteria. The Redlist guidelines base classification on the highest risk category for which the "species" can qualify. In this case, Hawaiian insular false killer whales would qualify as "critically endangered" under the following criteria:

- A4. An observed, estimated, inferred, projected or suspected population size reduction of more than 80% over any 10-year or 3-generation period, whichever is longer (up to a maximum of 100 years in the future), where the time period must include both the past and the future, and where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible, based on A1a (direct observation). The causes of the decline are not understood for Hawaiian insular false killer whales. The base-case model of the PVA (Appendix B—Fig. 4) has a 99% chance of the population declining to less than 20 individuals within 3 generations (75 years). From the current best estimate of around 150 individuals a decline to 20 individuals is an 87% decline.

- C1. Population size estimated to number fewer than 250 mature individuals and an estimated continuing decline of at least 25% within 3 years or 1 generation, whichever is longer, (up to a maximum of 100 years in the future).

- D. Hawaiian false killer whales are a borderline case as to whether there are fewer than 50 reproductive adults. The adult population of false killer whale adults is around 66 (the published point estimate of 123 times 43% mature from killer whales, Taylor et al. 2008). Pilot whales had the most similar life history (see Table 2-1, Section 2.1.4). The IUCN guidelines urge using a precautionary approach. Because the abundance has about a 50% chance of being less than the point estimate, it is very plausible that there are fewer than 50 mature individuals. The estimate of the effective population size (46) is also less than the threshold of 50 mature individuals. Because the number 50 was chosen to represent a threshold of high risk for inbreeding depression, the genetic estimate of effective population size is relevant.

- Quantitative analysis showing the probability of extinction in the wild is at least 50% within 10 years or three generations, whichever is the longer (up to a maximum of 100 years). The base-case PVA for Hawaiian false killer whales has a 99% chance of extinction in 3 generations (75 years).

4.7. Extinction risk—summary of BRT concerns

The Hawaiian insular false killer whale population is a relatively small, reproductively isolated population. Although its historic population size is unknown, data gathered since 1989 indicate a decline in abundance. To assess the risk, the BRT evaluated population size, population trends, social structure, and genetic diversity, as well as risk factors such as habitat degradation, death or serious injury in fishing gear, and the potential for catastrophic events. Based on the long life span of this species, the BRT believed that it is important to consider the risk of extinction using a relatively long time frame (e.g., several generations). Populations of long-lived animals can have considerable demographic inertia and, therefore, a nonviable population has the potential to persist for many decades, simply as a result of the long life spans of its existing members.

4.7.1. Population size and social structure

In comparison to most other marine mammal stocks in U.S. waters that have been assessed, the Hawaiian insular false killer whale population is small (Angliss and Lodge, 2002; Carretta et al., 2002). Table 4-6 summarizes the current estimates of abundance for the Hawaiian insular population.

Table 4-6. Current abundance estimate presented in the 2009 Stock Assessment Report (Carretta et al., 2010) and new estimates of abundance using more accurate mark-recapture models.

Time period of estimate	Population size	Source of estimate
2000–2004	123 (CV = 0.72)	Baird et al., 2005
2000–2004	162 (CV = 0.23)	Baird et al., ,unpublished (PSRG, 2009)
2006–2009 (w/out Kauai)	151 (CV = 0.20)	
2006–2009 (w/Kauai)	170 (CV = 0.21)	

False killer whales are typically found in tropical and subtropical waters and most often in groups. Although it is clear that false killer whales are highly social mammals with foraging strategies that depend on coordination and long-term relationships among the group members, many details about their social structure are unknown. Studies of groups of killer whales that died in drive fisheries or stranded suggest a very different social structure than that of SRKWs. Most of what we know about social structure of long-lived odontocetes results from studies of Resident killer whales, but false killer whales differ from resident killer whales in several ways. While both male and female Resident killer whales remain in their natal group for life, no subadult males were found in the groups of false killer whales either killed in drive fisheries or found stranded (Ferreira, 2008). At around the age of 18, the now sexually mature males must join (or at least visit – like sperm whales) a group. Robin Baird's work suggests that members of groups have long-term social bonds. Thus, these groups must regularly accept new members that are, at least in a social sense, strangers to them. For resident killer whales, the matrilineal group is the basic social unit and hence only one mtDNA haplotype is found in any pod. In contrast, six of seven groups of Hawaiian insular false killer whales contained females with different haplotypes (Chivers et al., 2010). Nevertheless, the social network indicated by photo-identification is intricate, with some individuals connected to nearly the entire network of photo-identified individuals. Movement data from satellite tags indicate that these false killer whales move rapidly to different areas and associate with different groups. Such social complexity is relevant to risks facing a small population. Negative aspects of social complexity are:

- Foraging success may depend on group size and certain key individuals. Disruption of group function can affect all group members making birth and death events correlated and increasing variance in population growth rate.

- Because individual false killer whales are usually near their group, any event that reduces survival (e.g., disease outbreak, oil spill) could adversely affect the entire group. Thus, the combination of small population size and their occurrence in socially cohesive groups puts the Hawaiian insular false killer whales at risk of a significant population decrease resulting from catastrophic events.

- Learned risky behavior, such as depredation on fishing gear, spreads quickly throughout the population.

Positive aspects of social complexity are:

- Foraging efficiency is optimized as groups can more readily locate prey even as prey densities decrease.

- Knowledge can be passed on inter-generationally and this may increase the group's ability to withstand environmental variability.

A small population could also experience a breakdown of social function. Although some aspects of the behavior and "culture" of Hawaiian insular false killer whales have been investigated or discussed, the mechanisms by which they might influence population viability are not well understood. The situation of this population could be analogous to those of other populations of large mammals in which females live well beyond their reproductive life spans (e.g., elephants, higher primates, and some other toothed cetaceans such as pilot whales) (McComb et al., 2001; Lahdenpera et al., 2004). The loss of only a few key individuals—such as the older, post-reproductive females — could result in a significant loss of inclusive fitness conveyed by "grandmothering" behavior (i.e., assistance in care of the young of other females in the pod). In addition, cultural knowledge (e.g., about how to cope with environmental changes occurring on decadal scales) could be lost, leading to reduced survival or fecundity of some or all age classes. Wade and Reeves (2010) document the special vulnerability of social odontocetes giving examples of killer whales, belugas, sperm whales, and dolphins in the eastern tropical Pacific.

A well-recognized factor that may have an adverse impact on an already small population is the inbreeding depression. Inbreeding depression is more likely to be manifested as the population size decreases. Based on Franklin's research on domestic animals (1980), the IUCN suggests that if the number of reproductive animals is fewer than 50, the potential for impacts associated with inbreeding depression increases substantially. Given that the adult population of Hawaiian insular false killer whale adults is probably around 65 (123 times 43% mature from killer whales, Taylor et al., 2008), it could be approaching the level at which the effects of inbreeding depression become a factor in determining whether the population is able to maintain itself or increase. The estimate of an effective population size of about 46 breeding adults (Chivers et al., 2010) makes the potential for loss of genetic diversity in the near future of concern.

4.7.2. Population trends

Aerial survey sightings since 1989 suggest that the insular population of Hawaiian false killer whales has declined over the last two decades. A 1989 survey, conducted in June and July on the leeward sides of Hawaii, Lanai, and Oahu, observed false killer whales on 14 occasions with 3 large groups (group sizes 470, 460, and 380) reported close to shore off the island of Hawaii on 3 different days (Reeves et al., 2009). The largest group seen in 1989 is more than three times larger than the current best estimate of the size of the insular population. Although the large groups seen in 1989 cannot be attributed to the

insular population with certainty, they occur within the defined insular population range and closer to shore than groups of pelagic false killer whales have been documented.

A regression of sightings during systematic aerial surveys conducted between 1993 and 2003 out to a maximum distance of about 46 km from shore (Mobley et al., 2000; Mobley, 2004) suggests a significant decline in the population size (Baird, 2009). The large groups sizes observed in 1989, together with the declining encounter rates from 1993 to 2003 suggest that Hawaiian insular false killer whales have declined significantly in recent decades. This decline occurred after the closure of the longline fishery close to the islands, suggesting continued decline even after possible interactions with the longline fishery were likely reduced due to the displacement of fishing effort to regions largely outside of the range of Hawaiian insular false killer whales.

The BRT found evidence for a substantial decline compelling. The cause or causes of the decline and whether that decline continues are unknown. The factors believed to have contributed most to the decline of Hawaiian insular false killer whales historically are: (1) the destruction, modification or curtailment of its habitat or range (i.e., reduced prey biomass and size, competition with fisheries, and accumulation of natural and anthropogenic contaminants); (2) overutilization (i.e., live capture operations occurring prior to 1990); (3) disease and predation because of exposure to environmental contaminants); (4) inadequate regulatory mechanisms (i.e., the lack of an observer program for nearshore fisheries); and (5) other natural and manmade factors (i.e., interactions with commercial longline fisheries and reduced genetic diversity). Uncertainty about the causes of the decline results in a higher level of concern among the BRT members, because mitigation is more difficult when causality is uncertain. A clear example of a case where causality was (and is) incompletely known is the western DPS of Steller sea lions. Declines persist in some areas despite decades of research and tens of millions of research dollars spent. Not knowing the cause or causes of decline will result in delays in any recovery actions which, in turn, argues for allowing a longer time for management to be effective for Hawaiian insular false killer whales than would be needed for a case with no uncertainty about the cause of decline.

4.7.3. Conclusions

Taken together, the small population size and evidence of a decline in Hawaiian insular false killer whales, combined with several factors that are likely to continue to have, or have the potential to adversely impact the population in the near future, describe a population that is at high risk of extinction. The small size of this false killer whale population makes it vulnerable to Allee effects (e.g., inbreeding depression and loss of social integrity). This description of risk and the level of concern are similar to the descriptions for other species of ESA-listed social odontocetes (e.g., SRKWs, Cook Inlet beluga whales). Uncertainty as to the causes of the recent decline, the current threats, and current viability of the population increases concern for this group of whales.

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Appendix A. DPS Designation and Risk Assessment Scores

To maximize transparency in their deliberative process, the BRT structured decisions on the following: (1) determine whether Hawaiian insular false killer whales are discrete and significant, and therefore qualify as a DPS under ESA, (2) parameterization of the PVA, (3) the plausibility of individual PVA models, and (4) the extent of threats to the population.

For each issue, detailed arguments for and against each option—and the resulting distribution of scores—are provided to fully document the team's decisions outlined in the text of this status review.

DPS - Determination of discreteness

1. Evaluation of DPS Discreteness Criteria—Are Hawaiian insular false killer whales markedly separated from other populations of the same taxon as a consequence of behavioral factors?

Arguments in favor of* marked separation *based on behavioral uniqueness

Throughout their distribution, false killer whales are considered a wide-ranging pelagic species not typically associated with coastal or island habitats. Hawaiian insular false killer whales are behaviorally unique because they are the only population of the species known to have movements restricted to the vicinity of an oceanic island group. This behavioral separation is supported by the following lines of evidence:

- Hawaiian insular false killer whales are linked through a tight social network, without any linkages to animals outside of the Hawaiian Islands.
    - All but 4 identified individuals encountered within 40 km of shore near the islands of Oahu, Maui, Lanai, and Hawaii are linked to all others through at least 2 distinct associations and often many more if they have been frequently sighted (Baird et al., 2005; Baird, 2009). None of these animals has ever been linked to any other false killer whales photographed in the North Pacific.
- Phylogeographic analysis indicates an isolated population with nearly exclusive haplotypes.
    - Genetic analysis indicates 80 of 81 individuals had haplotypes found exclusively within the Hawaiian population (Chivers et al., 2007; Chivers et al., 2010). Hawaiian insular false killer whales have highly significant differences in both mtDNA and nuclear DNA from all other geographic strata.
- Telemetry data show that all 20 satellite-linked telemetry tagged Hawaiian insular false killer whales remained within the main Hawaiian Islands (Baird et al., 2010, Baird et al., unpublished data), in contrast with a single tagged pelagic false killer whale, which ranged far from shore.
    - Although it is not unusual for false killer whales to be observed close to land, the long-term history of their exclusive use of a specific mainland or island system has not been documented. The short-term use of Gulfo

Dulce and Cocos Island off Costa Rica has been shown over a 2- to 3-year period for a small number of individuals; however, one interpretation of this pattern is that pelagic animals occasionally use nearshore waters for foraging.

  - Worldwide, the Hawaiian Archipelago is the most isolated extensive area of elevated productivity surrounded by oligotrophic ocean. Hawaiian insular false killer whales exploit this habitat exclusively. Other island groups are not as isolated, such that false killer whales occurring there could more easily cross into neighboring habitats, and be less dependent or specialized on the one isolated habitat. Further, the relative connectedness of other island groups may support larger networks of false killer whales that move widely among the islands.

Arguments against* marked separation *based on behavioral uniqueness

There are inadequate data on the potential occurrence of island-associated populations of false killer whales in other areas. Arguments against behavioral separation are:

- Repeat sightings of individual false killer whales have occurred at Cocos Island off Costa Rica (Acevedo-Gutierrez et al., 1997).
  - Although the authors attributed the resightings to local foraging by a pelagically distributed population, 8 of 50 identified individuals were seen in at least 2 separate years, and 17 others were seen more than once in any one year, a high resighting rate for occasional visitation by a pelagic population. In contrast, no individual from the pelagic population of Hawaiian false killer whales has ever been seen within 42 km of the main Hawaiian Islands and no individuals have ever been resighted. For this reason, resightings of 16% of identified animals may represent a population associated with Cocos Island, not unlike the Hawaiian insular false killer whales.
- The sample distribution is inadequate to characterize genetic separation.
  - Although the Hawaiian insular population is very well sampled with approximately half of the individuals represented, pelagic false killer whale genetics are poorly sampled with large sampling gaps to both the west and east of Hawaii. It is possible that mtDNA haplotypes found in Hawaii could be found elsewhere in these inadequately sampled areas.
- False killer whales are known to occur around tropical oceanic islands, particularly around the Cook Islands, Cocos Island, the Seychelles, and French Polynesia, but little to no survey effort has occurred that would allow us to assess whether there are island-associated populations at these island groups.

2. Evaluation of DPS Discreteness Criteria—Are Hawaiian insular false killer whales markedly separated from other populations of the same taxon as a consequence of ecological factors?

Arguments in favor of* marked separation *based on ecological discreteness:

False killer whales are generally reported to inhabit deep oceanic habitats, and are not typically associated with shallow waters near coastlines and islands. Hawaiian insular false killer whales inhabit a unique ecological setting because they are found primarily in island-associated waters that are relatively shallow and productive compared to surrounding oligotrophic waters. Evidence supporting marked separation based on ecological discreteness include:

- Movement data show that insular false killer whales have adapted to a different ecological habitat than their pelagic conspecifics.
    - Eleven insular Hawaiian false killer whales instrumented with satellite-linked telemetry tags remained closely associated with the Hawaiian islands, in waters that had a median depth of 575 m. This is substantially shallower than the global oceanic habitat, and it contrasts with one pelagic false killer whale satellite tagged about 42-km from the Hawaiian Islands that was tracked through waters with a median depth of 3844 m. Location data from all 23 insular false killer whales with satellite tags have not yet been fully analyzed, but their occurrence is primarily on the continental shelf or slope of the main Hawaiian Islands rather than the abyssal plain (Baird, unpublished data). This shallower near-island habitat supports enhanced productivity, and the tight association of insular false killer whales suggests they have adapted to a different ecological habitat than their oceanic conspecifics even when foraging on pelagic prey species.
- Contaminant data suggest that insular false killer whales may forage on island-associated prey.
    - Insular false killer whales have elevated levels of fire retardants (PBDEs), a contaminant primarily associated with urban environments. These elevated levels suggest that both insular false killer whales and their prey may be associated with the urban island environment.

Arguments against* marked separation *based on ecological discreteness:

The ecological setting in which false killer whales are found around the Hawaiian Islands is not unique. Arguments against marked separation based on ecological discreteness include:

- The ecological setting of Hawaiian insular false killer whales only appears unique because we lack data from other similar coastal or island habitats where undocumented discrete false killer whale populations may occur.
- False killer whale prey ranges widely in open-ocean habitats and areas near oceanic islands such as the Hawaiian Islands, such that false killer whale occurrence around Hawaii is not ecologically unique.

DPS – Determination of significance

1. Evaluation of DPS Significance Criteria—Do Hawaiian insular false killer whales persist in an ecological setting unusual or unique for the taxon?

Factor A:
Argument in favor of persistence in a significant ecological setting

The Hawaiian Islands represent a unique ecological setting and, therefore, false killer whales whose range is restricted to the islands are significant.

Argument against persistence in a significant ecological setting

The Hawaiian Islands represent a unique ecological setting; however, false killer whales restricted to the islands are not necessarily significant to the evolutionary persistence or potential of false killer whales.

Factor B:
Arguments in favor of persistence in a significant ecological setting

Hawaiian insular false killer whales persist in an ecological setting unusual or unique from other false killer whale populations. Lines of evidence in support of this significant ecological setting include:
Hawaiian insular false killer whales' utilization of prey associated with island habitat may require specialized knowledge of locations and seasonal conditions that aggregate prey or make them more vulnerable to predation. In an insular habitat, such foraging grounds may occur more regularly or in more predictable locations than on the high seas.

- Unlike a pelagic habitat, utilization of insular habitat may require
    - The shelf and slope waters surrounding the main Hawaiian Islands that insular false killer whales occupy differs from the surrounding oligotrophic waters by being relatively shallow and productive.
        - The mesopelagic boundary community supports a local food chain that may support insular false killer whales;
        - Island-mass effect, or regions of higher productivity attributed to land-based inputs such as nutrients and freshwater, and wake effects;
        - Insular false killer whales have been observed to consume threadfin jack and filefish, two species that are primarily found in coral reef habitats near the coast.
    - Contaminant levels are consistent with consuming nearshore prey contaminated by local urban run-off.
- Despite their small population size, the density of Hawaiian insular false killer whales is high density relative to other populations.
    - The density of the insular population is second only to waters surrounding Palmyra Atoll, but about 10 times higher than the adjacent pelagic population.

- Movement and photographic resighting data suggest Hawaiian insular false killer whales employ a unique foraging strategy compared to other false killer whales.
    - The home range of this population appears to be extremely small compared to other medium body-sized, piscivorous odontocetes, such as resident-type killer whales, which maintain larger home ranges in more productive habitats.
    - Pelagic false killer whales are at a lower population density in a lower productivity environment suggesting that they must occupy a larger home range to find sufficient prey.

Arguments against persistence in a significant ecological setting

Hawaiian insular false killer whales do not occupy an ecological setting unusual or unique from other false killer whale populations. Arguments against ecological significance include:

- There are insufficient data to conclude that the Hawaiian Islands are a unique ecological setting for false killer whales.
- There is insufficient evidence to conclude that Hawaiian insular false killer whales have a unique foraging strategy.
    - False killer whales are socially complex, adaptable animals, such that pelagic animals may be able to alter their foraging strategies to colonize insular Hawaiian waters.
    - Evidence for a unique foraging strategy by insular false killer whales is largely inferential and based on limited data on movements and prey selection. For example, it is unknown whether the Hawaiian insular false killer whales use the Northwestern Hawaiian Islands, which would represent a significantly larger range than has been estimated to date.
    - Home range is a poor indicator of foraging strategy because it can be influenced by other factors, such as social behavior.
- There are insufficient data to conclude specialization on localized resources.
    - There are too few observations of foraging by insular false killer whales to adequately quantify the use of island-associated prey.
    - Inferences from contaminant data are based on samples from few individuals and only from Hawaiian insular false killer whales, such that comparisons of contaminant levels with other populations are not possible.
    - Contaminant sources are unknown and may not be based solely on local prey.

2. Evaluation of DPS Significance Criteria—Is there evidence that Hawaiian insular false killer whales differ markedly from other populations of the species in their genetic characteristics?

Arguments in favor of marked genetic differentiation

Hawaiian insular false killer whales differ markedly from other populations of the species in their genetic characteristics. Evidence supporting marked differences are:

- Hawaiian insular false killer whales exhibit strong phylogeographic patterns that are consistent with local evolution of mitochondrial haplotypes nearly unique to the insular population.
    - Although other cetaceans in Hawaii have sufficient genetic differences to make them distinct and even to have demographically independent populations within the main Hawaiian Islands, only Hawaiian false killer whales have a pattern suggestive of a single colonization event with local evolution of a new haplotype.
- The magnitude of mtDNA differentiation is large enough to infer that time has been sufficient and gene flow low enough to allow adaptation to local Hawaiian habitat that would not be readily repopulated by pelagic whales without such adaptation.
    - Geneticists use one effective migrant per generation as a rule of thumb for the level of gene flow below which adaptation to local habitat is likely. Comparisons using mtDNA of the Hawaiian insular animals to those in all other geographic strata indicate less than one migrant per generation.
- Nuclear data (microsatellites) are consistent with little gene flow.
    - Nuclear data from this marker strongly suggest discreteness. Microsatellite data, in general, are weaker on the question of genetic differentiation on long-time scales because of the high mutation rate and mode of mutation that allows alleles of the same length to have different evolutionary histories.

Arguments against marked genetic differentiation

Hawaiian insular false killer whales might not differ markedly from other populations of the species in their genetic characteristics. Arguments against marked differentiation are:

- The sample distribution is inadequate to characterize genetic separation.
    - Although the Hawaiian insular population is very well sampled with approximately half of the individuals represented, pelagic false killer whale genetics are poorly sampled with large sampling gaps to both the west and east of Hawaii. It is possible that mtDNA haplotypes found in Hawaii could be found elsewhere in these inadequately sampled areas.
- Hawaiian insular false killer whales are not significantly different from nearby pelagic whales because of potential male-mediated gene flow. Results suggesting low nuclear gene flow are potentially biased as a result of inadequate sampling of nearby pelagic false killer whales.
    - Another plausible scenario that may reduce the significance of the Hawaiian insular false killer whales involves male-mediated gene flow. Inadequate sampling of nearby pelagic whales may not allow inference about gene flow from nearby males that could mate with insular females. Offspring would have the mtDNA of their mothers but half of their nuclear DNA from their pelagic fathers. If the level of male-mediated gene flow was high enough, then adaptation to local habitat is unlikely and insular animals could be easily replaced by pelagic animals if the habitat

became available due to the extirpation of Hawaiian insular false killer whales.

3. Evaluation of DPS Significance Criteria—Do insular Hawaiian false killer whales exhibit *other* factors that are significant for the taxon? Significance of cultural diversity:

Arguments in favor of significant cultural diversity

Culture (knowledge passed through learning from one generation to the next) is likely to play an important role in the evolutionary potential of false killer whales. The insular population contributes to cultural diversity that may enhance the ability of false killer whales to adapt to environmental change. Evidence in support of the significance of cultural diversity includes:

- Insular false killer whales may have unique knowledge of nearshore foraging areas that is transmitted through learning. Learning is a common feature of other social odontocetes. The movements of insular false killer whales demonstrate a clear preference for particular areas of the inshore waters.
- False killer whales are highly social mammals with long inter-birth intervals and reproductive senescence suggesting transfer of knowledge is important to successfully persist in this unique Hawaiian habitat.
- Learning to persist in this unique habitat may take many generations.

Arguments against significant cultural diversity

The cultural diversity of the insular population is not significant to the persistence of false killer whales. Arguments against the significance of cultural diversity include:

- There is little evidence that cultural transmission is important to the evolutionary potential of false killer whales.
- There is little evidence that learning is important to persistence in Hawaiian waters. Observations of movements and genetic discreteness could result from territorial behavior alone.

DPS – Plausibility-point allocation by team member

BRT members independently allocated plausibility points on the discreteness and significance criteria by distributing 10 points between the arguments for and against each factor. Allocating points in this manner allowed individual team members to express their level of certainty on each of the factors, such that placement of all 10 points either for or against a particular factor would indicate certainty in the arguments or evidence presented. Team members agreed to view resulting scores with names associated to facilitate discussion and assure that linguistic uncertainty was not responsible for any disparate votes. The BRT discussed the scores and, in some cases, adjusted scores when prior articulation of the arguments had been unclear. Anonymized individual final team member scores are shown below.

Discreteness Factors

Factor 1: Are Hawaiian insular false killer whales markedly separated based on behavioral factors?

Member	Yes	No	Total
A	8	2	10
B	8	2	10
C	8	2	10
D	8	2	10
E	10	0	10
F	9	1	10
G	7	3	10
H	7	3	10
TOTAL	**65**	**15**	**80**
	81%	**19%**	**100%**

Factor 2: Are Hawaiian insular false killer whale markedly separated based on ecological factors?

Member	Yes	No	Total
A	8	2	10
B	7	3	10
C	6	4	10
D	8	2	10
E	7	3	10
F	7	3	10
G	6	4	10
H	7	3	10
TOTAL	**56**	**24**	**80**
	70%	**30%**	**100%**

The BRT expressed strong support for a finding of marked separation based on behavioral and ecological factors, yet acknowledged that uncertainty remains, particularly based on ecological factors.

Significance Factors

Factor 1a: The Hawaiian Islands represent a unique ecological setting and, therefore, false killer whales whose range is restricted to the islands are significant.

Member	Yes	No	Total
A	4	6	10
B	4	6	10
C	1	9	10
D	4	6	10
E	3	7	10
F	2	8	10
G	5	5	10
H	8	2	10
TOTAL	**31**	**49**	**80**
	39%	**61%**	**100%**

Factor 1b: Hawaiian insular false killer whales are significant because they persist in an ecological setting unique for the taxon.

Member	Yes	No	Total
A	7	3	10
B	7	3	10
C	5	5	10
D	7	3	10
E	7	3	10
F	6	4	10
G	6	4	10
H	7	3	10
TOTAL	**52**	**28**	**80**
	65%	**35%**	**100%**

Factor 2: Hawaiian insular false killer whales differ markedly from other populations of the species in their genetic characteristics.

Member	Yes	No	Total
A	7	3	10
B	7	3	10
C	5	5	10
D	7	3	10
E	8	2	10
F	6	4	10
G	6	4	10
H	5	5	10
TOTAL	**51**	**29**	**80**
	64%	**36%**	**100%**

Factor 3: Hawaiian insular false killer whales significantly contribute to the cultural diversity of false killer whales.

Member	Yes	No	Total
A	7	3	10
B	6	4	10
C	4	6	10
D	8	2	10
E	7	3	10
F	6	4	10
G	7	3	10
H	6	4	10
TOTAL	**51**	**29**	**80**
	64%	**36%**	**100%**

The BRT found the Hawaiian insular false killer whale to be significant to the taxon in which it belongs. There was support for a finding of marked genetic differences, and ecological and cultural significance.

Significance: Hawaiian insular false killer whales are significant to the taxon because of genetic, ecological, and cultural factors taken together.

Member	Yes	No	Total
A	7	3	10
B	8	2	10
C	6	4	10
D	7	3	10
E	8	2	10
F	7	3	10
G	7	3	10
H	7	3	10
TOTAL	**57**	**23**	**80**
	71%	**29%**	**100%**

The BRT found there was stronger support for finding Hawaiian insular false killer whales to be significant to the taxon based on all factors taken together and the potential synergistic effects of those factors.

Population viability analysis (PVA) – initial model parameterization

There were a number of decisions to be made for conducting the PVA analysis. Prior to constructing any of the models, these options were broken down into 6 categories described below. The different options that could be chosen have been labeled A and B (and C, if necessary). The plausibility point system was used to determine which options to use for the "base-case" model, and whether other options should be run as variants. An option with little support was dropped (not run), but an option with some support was considered as a 'Variant Case'. In a few cases, the BRT asked for "Sensitivity Tests" for cases with little to no support but that promised to reveal how sensitive the model was to certain assumptions. An example would be not using the Reeves et al. (2009) minimum count of false killer whales seen during the 1989 aerial surveys in the modeling exercise. The results of this vote determined how the model was initially parameterized. Further exploration and evaluation of the models and model fit was necessary, and additional scores were required before a final suite of model parameters were chosen. Those later scores are described later in this Appendix.

(1) Absolute abundance: mark-recapture from Baird small boat surveys

Baird et al. unpublished (presentation to the Pacific Scientific Review Group in November 2009) made mark-recapture estimates of abundance for the small boat survey data for two time periods, 2000–2004, and for 2006–2009. He used the POPAN model and explored both constant and time-dependent survival and capture probability. These abundance estimates were preferable to use over the previous estimate from 2000 to 2004 (Baird et al., 2005) because they provide estimates from two time periods and, therefore, more information about potential trends.

Mark-recapture (MR) estimates and coefficients of variation (CV) from the best model were:

MR 2000–2004	162 (CV = 0.23)
MR 2006–2009	151 (CV = 0.20) without Kauai sightings
MR 2006–2009	170 (CV = 0.21) with Kauai sightings

Two groups sampled at Kauai have not (perhaps yet) been seen to associate with the social network of Hawaiian insular false killer whales seen at the other islands. Baird has information on eight groups encountered at Kauai by collaborators, but only three of the groups have photographs meeting the quality requirements. The three groups that have identifications from Kauai in Baird's catalog were all encountered by collaborators that were not equipped for biopsy sampling, so there are no genetic samples from Kauai to examine.

Options:
(1A) Use the 2006–2009 'without Kauai' estimate.
The three groups of whales photographed at Kauai have not been observed in association with any Hawaiian insular false killer whales from the other islands, so they are not connected into the social network. Further, none of the three groups encountered at Kauai are associated with each other, so each individual seen at Kauai has only been seen once. For the genetic analyses, the population is defined as the whales that have been connected in the social network, so by using that definition the Kauai whales are not part of the population. Although it is possible that the Kauai whales will be found to associate with other whales in the Hawaiian insular false killer whale social network in the future, it is also possible that they are part of the pelagic population and happened to be close to shore in those encounters. Additionally, there is an issue of comparability with the 2000–2004 mark-recapture estimate. The abundance estimate from 2000–2004 does not have any photo-identifications from Kauai. If there is some population substructure between the islands, meaning that whales do not freely mix among all islands without preference, then the 2006–09 estimate 'without' Kauai would be more comparable to the 2000–2004 estimate. This would potentially make any trend information that comes out of the difference between the two estimates more accurate, because if some whales had a higher probability of being at Kauai than the other islands, the 'with Kauai ' estimate would be higher and not exactly comparable to the other number. The 'without Kauai' value estimates the number of whales using all the islands except Kauai, whereas the 'with Kauai' value estimates the number of whales using all the islands including Kauai, which would be greater. Satellite-telemetry information for insular false killer whales suggests that individual whales move among all of the main Hawaiian Islands, but they show short-term patterns of movement potentially suggestive of core areas within their home ranges. However, the appearance of preferential use for certain areas may be an artifact of where individual animals were tagged and the duration of the signal from the transmitter.

(1B) Use the 2006–2009 'with Kauai' estimate
The encounter with false killer whales at Kauai was close to shore and well within the defined range of the Hawaii insular false killer whale stock as defined in the 2010 Stock Assessment Report (i.e., < 40 km from shore), so these whales should be considered part of the population and should be included in the estimate. If seen in that location on a line-transect survey (without photos) they would be considered part of the insular population abundance estimate. The location of the encounter was within the Mobley aerial survey study area as well, so if using the trend from the Mobley surveys, we also ought to be using an absolute abundance estimate from the same area. Additionally, the satellite

telemetry data show whales moving between Kauai and other islands; for example, one whale tagged at Oahu moved to Kauai and spent considerable time there. The fact that they have not associated with other whales in the Hawaiian insular social network may be a function of each individual having only been sighted once, and they may well be found to associate in the future. Small boat-based survey effort by Baird occurred in the years 2003 (24 days), 2005 (24 days), and 2008 (7 days), such that there was more dedicated effort around Kauai in 2000–2004 than there was in 2006–2009, even though none of the effort led to a detection of false killer whales. Because there is effort in Kauai in each time period, this indicates the two data sets are more or less estimating the population size over the same geographic area, and so are comparable. These arguments suggest that it would be acceptable to compare the 2000–2004 estimate to the 2006–2009 'with' estimate.

(2) Minimum absolute abundance from the 1989 aerial survey

The 1989 aerial survey (Reeves et al. 2009) covered relatively nearshore waters on the west side of Hawaii. The entire study area was within the study area of the Mobley et al. (2000) surveys. The 1989 group sizes could be used as a minimum for that year. The survey resulted in minimum counts of false killer whales on 3 separate days—460 individuals on 23 June (after 11 passes from the airplane), 470 individuals on 26 June (after 14 passes), and 380 individuals on 28 June (after 7 passes). From the air, factors such as dispersion of animals, size and arrangement of subgroups, and glare conditions can influence the accuracy of counts. For example, given the rapid travel speed of an aircraft, it is often easier to count animals dispersed in discrete, identifiable subgroups because each subgroup can be evaluated sequentially (vs. a single large group, which must be rapidly counted within the short time window that the aircraft is overhead). During the 1989 surveys, viewing conditions were rated as excellent, and the group of 460 seen on 23 June was reported to consist of 2 large subgroups and 6 to 8 small satellite groups. The BRT was asked whether the PVA should use the highest estimated count (460), an average of the 3 counts, or the lowest count (380) as the minimum population size in 1989.

Options:

(2A) Use 470 as a minimum population size for 1989.

On the aerial survey, the plane made multiple passes over the sighting until the observers were confident they had counted all subgroups and were sure there were at least that many whales there. They made 14 passes in the airplane over the sighting where they counted a minimum of 470 individuals. Two of the observers (Leatherwood and Reeves) were exceptionally experienced and can likely be trusted to have given an accurate conservative minimum count. Although it is known that group size estimates from a ship for large groups of dolphins (in the hundreds) can be imprecise and potentially biased, group size estimation from an airplane, while also potentially biased, offers some advantages. Given the relatively clear water and good sighting conditions, it would be possible to see whales underwater at shallow depths during a relatively fast pass from an airplane (i.e., the main problem with estimating a group from a ship is not being able to determine when a subgroup already seen surfaces for a second time), so the minimum count is likely of high reliability.

(2B) Use the mean of the counts on the 3 different survey days.
A group of cetaceans in the hundreds is difficult to count, and studies of dolphin surveys in the ETP where photographs from helicopters have been used to confirm the group size have shown observers to be relatively imprecise and sometimes biased in their estimation of group sizes (the bias is usually negative). If these group sizes were estimated with relatively low precision, the mean of the three surveys may represent the best conservative estimate of the number of whales there.

(2C) Use the lowest of the three counts (380 individuals).
Given that surveys on three different days (over a 6-day period) counted at least 380 whales on each survey, that number could be considered a reliable indicator of the minimum number of whales there. It would be sensible to err on the conservative side, acknowledging that the higher counts could have been a result of a counting error.

(3) Trend information: Mobley aerial surveys
Mobley flew five aerial surveys in the period of 1993–2003. Given the lack of correction factor, g(0), for animals that were on the track line but missed (e.g., because they were diving when the aircraft passed overhead), the encounter rate from these surveys was treated as a relative index of abundance through time that gives information about the trend of the population. Encounter rates can be measured in terms of groups per unit search effort or individuals per unit search effort. Baird et al. (2009) showed that there were no significant trends in encounter rates for other species during the five aerial surveys, providing some confirmation that detection probabilities had not declined through time (e.g., because of less experienced observers, poorer weather, or other issues). The group encounter rates (# of groups per 10 survey hours) for the five aerial surveys were 1.06 (1993, 8 groups), 0.975, (1995, 9 groups), 0.141 (1998, 1 group), 0.00 (2000, 0 groups), and 0.00 (2003, 0 groups). It is unclear whether small groups were circled during the surveys and, as such, whether these group size estimates may be more negatively-biased than those for larger groups. The number of false killer whale sightings was too small to estimate, f(0), the detection probability at zero distance from the survey track line, separately for each year, so this parameter was assumed to be constant among years. Variance estimates are not available for the group encounter rates, so the variance was estimated as an additional parameter during the model fitting process.

Options:
(3A) Use the individual encounter rate as the index.
The individual encounter rate is the group encounter rate multiplied by the mean group size in that year. We are interested in the trend of the population, and thus individuals, not in the trend of the number of groups. The average group size could change through time, so it is preferable to be evaluating the trend in terms of the number of individuals.

(3B) Use the group encounter rate as the index.
Although we want to use the trend of individuals, only one group was seen in 1998 and, therefore, the mean group size in 1998 is not reliably estimated. Mean group sizes were 6.1 (1993), 7.0 (1995), and 1.0 (1998). It is possible that the mean group size was

unchanged in 1998, and it was simply by chance that the only encounter involved a single animal. In 1992 and 1995 there were 2 and 3 single individuals recorded, respectively, out of the 8 and 9 total groups seen in those years.

(4) Environmental and demographic variance

The model is stochastic, using a single parameter to account for both environmental and demographic variance. The value of this parameter could be estimated from census data of Southern Resident killer whales (SRKWs), Northern Resident killer whales (NRKWs), or based on the variance in a more tropical species. The SRKW data are from a smaller population than Hawaiian insular false killer whales, so those data might over-estimate the variance and therefore it may be appropriate to use a lower value for the variance. In theory it would be possible to estimate demographic variance for SRKWs and false killer whales independently (via simulation or other means), subtract the SRKW demographic variance from the observed SRKW variance to get the SRKW environmental variance component, then add that back to the estimate of false killer whale demographic variance. Time constraints made that impossible for this review. Environmental variance may also be biased if conditions that affect birth and death rates are more or less variable for the tropical Hawaiian insular false killer whales than for the temperate SRKWs. Considering that temperate regimes tend to be more variable than tropical regimes, it is possible that SRKWs vital rates vary more than Hawaiian insular false killer whales rates. This would also argue for a lower total variance for Hawaiian insular false killer whales than that observed for SRKWs.

Assigning of uncertainty points indicates preference for base versus sensitivity cases. The category with the highest votes would be the base case. The exception would be if option 4B had the highest votes but later proved to have higher total variance than option 4A. Logically, this higher variance would be the result of sampling error not actual higher variance. In this case, if 4A had the second highest number of uncertainty points it would become the base case. Further, 4C may garner the greatest number of votes, but would not be used unless a lower variance can be justified based on another tropical or sub-tropical species with good census data (i.e., Sarasota bottlenose dolphins). It is unclear whether these data are readily available for this analysis. If they are not, the option with the second highest vote count would be used in the base-case.

Options:

(4A) Use of the SRKW variance has the advantage that population is fully counted every year, so the abundance in each year is known without error. Therefore, the variance in year-to-year population growth rates will represent only the environmental and demographic variance, because there is no observation error. One issue is that SRKW population has been smaller in number (~ 70–95) than the Hawaiian insular false killer whale population, so this might over-estimate the demographic variance of the Hawaiian insular false killer whale population.

(4B) Use of the NRKW variance has the advantage that the population is closer in size (~ 200) to the Hawaii insular false killer whale population. One disadvantage is that the population is not completely censused in each year, though estimates do come close to

the actual population size. Additionally, they are able to 'back-fill' population sizes in previous years for individuals not seen in one year that are seen in a subsequent year (as they are therefore known to have been alive in that previous year). The variance in population growth rate of NRKW will be more appropriate because the population size is similar so the demographic variance component should be more similar to Hawaii insular false killer whales. Because the NRKW population is not completely censused each year, there will be some sampling error added into the variance, so the estimated variance will over-estimate (perhaps not by much) the true demographic and environmental variance of the NRKW population. Given that the variance might be over-estimated, if the value from NRKW is lower than the value for SRKW, it would be a more appropriate number.

(4C) Use a smaller variance than that for killer whales to address the potential that environmental variance is less in tropical regimes. Resident killer whale vital rates have been shown to be linked to variability in salmon abundance which in turn varies due to variability in productivity in temperate waters. The tropical waters off Hawaii are less variable. The argument could be made that Hawaiian false killer whale prey are therefore also less variable resulting in reduced variability in vital rates compared to resident killer whales.

(5) Allee effects

Studying the demography of very small populations is difficult, but several studies have indicated that Allee effects do occur in a variety of species (Groom, 1998; Kuussaari et al., 1998; Courchamp et al., 1999) though one meta-analysis study found little evidence for depensation in fish species (Myers et al., 1995). Fowler and Baker (1991) concluded that depensation was likely to be a common phenomenon in animal population dynamics, especially in populations smaller than 10% of their original, pre-exploitation population sizes (0.1K). Potential mechanisms that cause Allee effects include inbreeding depression, decreased chances of fertilization (e.g., through decreased availability of mates), lack of predator saturation, decreased anti-predator vigilance or defense, decreased cooperative feeding, or consequences of a breakdown of social structure in social animals such as a loss of "cultural knowledge". It was proposed that SRKWs could experience an Allee effect because of a mating system where males always mated outside of their social group; a lack of males in a different social group could prevent mating. However, recent genetic evidence may suggest that the SRKW mating system is not as structured as previously thought, though this may be in response to a decline in the availability of suitable mates, which would increase the probability of inbreeding depression.

In many PVAs it is necessary to make assumptions about what kind of Allee effect would occur in a small population. In the absence of direct evidence of the level at which an Allee effect would take place, or what the mechanism of the effect would be, PVAs often include a pseudo-extinction level (a threshold set at a low population size, where it is assumed that extinction is inevitable if that threshold is reached). This PVA reports both actual extinction probability and pseudo-extinction levels, so the choice of including Allee effects is independent of the type of extinction level the Team chooses to use when judging extinction risk. Pseudo-extinction level results would serve as a knife-edge form

of an Allee effect, where once the population crosses the threshold, it's assumed to become extinct with certainty.

Options:
(5A) Do not include Allee effects
There is no direct evidence for Allee effects in Hawaiian insular false killer whales. Moreover, there are no specific data to guide setting the population level where Allee effects would take place, so any level specified would be based on speculation, not data.

(5B) Use Allee effects
There are many mechanisms (described above) that can potentially cause Allee effects in a population such as Hawaiian insular false killer whales. Ignoring the possibility of Allee effects could therefore underestimate the true extinction risk of the population. In general, Allee effects are thought to occur at higher population levels than the levels often used as 'pseudo-extinction' thresholds, so trying to use such thresholds as a substitute for Allee effects may not be appropriate. A true Allee effect will start weakly at a higher population size and become increasingly stronger as the population size decreases. This is likely more realistic than a 'knife-edge' Allee effect approximated by a 'pseudo-extinction' threshold.

(6) Mortality events ('catastrophes')
False killer whales worldwide are known to occasionally mass strand for unknown reasons. However, no mass strandings are known from Hawaii. There may be considerable concern about pathogens and their potential impacts on small populations, and there may be other forms of catastrophes, including naval sonar events, entanglement in aquaculture facilities, etc. Therefore, mortality events could be added to the PVA to represent episodic mortality factors that affect some fraction of the population.

Options:
(6A) Use mortality events in the model.
Even though not observed, mass strandings are well known for the species, and should not be ignored. If strandings do occur, ignoring mortality events in the model will under-estimate extinction risk.

(6B) Do not use mortality events in the model.
No strandings have been observed in Hawaii, and there are no data from which to specify the probability of an event, or its magnitude.

PVA—Model parameterization—initial plausibility-point allocation by team members

Table B. BRT member votes on the initial parameterization of the PVA model. The score represents the plausibility scores from each member following a discussion of the initial scoring outcome, and the opportunity for members to change their scores.

Option1: Abundance estimates from mark-recapture

Team member	A: MR w/Kauai	B: MR w/o Kauai	Total points assigned
A	7	3	10
B	6	4	10
C	5	5	10
D	5	5	10
E	6	4	10
F	6	4	10
G	7	3	10
H	5	5	10
	47	**33**	**80**
	59%	**41%**	

Choice: Run the PVA including the Kauai estimate as the base-case. Use w/o Kauai as a variant case. There was considerable uncertainty among the Team members on this choice.

Option 2: Minimum abundance from 1989 survey

Team member	A: Min count 470	B: Mean of 3 sightings	C: Min count 380	Total points assigned
A	5	3	2	10
B	4	6	0	10
C	8	1	1	10
D	10	0	0	10
E	3	5	2	10
F	6	2	2	10
G	0	10	0	10
H	8	2	0	10
	44	**29**	**7**	**80**
	55%	**36%**	**9%**	

Choice: Use 470 as base-case, use 380 as a variant case as it represents the other extreme. If results differ markedly, then re-run with mean as base-case.

Option 3: Trend information from Mobley surveys

Team member	A: Individual encounter rate	B: group encounter rate	Total points assigned
A	4	6	10
B	3	7	10
C	7	3	10
D	7	3	10
E	4	6	10
F	6	4	10
G	6	4	10
H	7	3	10
	44	**36**	**80**
	55%	**45%**	

Choice: Because these are the most influential data, both options should be run as full permutations in the PVA.

Option 4: Environmental and demographic variance

(assumes Sarasota dolphin data available)

Team member	A: SRKW variance	B: NRKW variance	C: lower tropical variance	Total points assigned
A	2	6	2	10
B	2	3	5	10
C	3	4	3	10
D	3	5	2	10
E	2	3	5	10
F	2	4	4	10
G	0	3	7	10
H	2	2	6	10
	16	**30**	**34**	**80**
	20%	**38%**	**43%**	

Choice: If Sarasota bottlenose dolphin (a tropical analog) demographic variance is available it should be used for the base case. SRKW variance will be used in a sensitivity analysis. If bottlenose data not available, use NRKW as base-case.

Option 5: Allee effects

Team member	Yes	No	Total points assigned
A	9	1	10
B	6	4	10
C	8	2	10
D	7	3	10
E	6	4	10
F	7	3	10
G	7	3	10
H	10	0	10
	60	**20**	**80**
	75%	**25%**	

Choice: Incorporate a mild negative slope into the base-case model, and then test sensitivity without an Allee effect and variants with more severe slopes.

Option 6: Mortality effects (i.e., catastrophes)

Team member	Use	Don't use	Total points assigned
A	6	4	10
B	6	4	10
C	6	4	10
D	7	3	10
E	6	4	10
F	7	3	10
G	8	2	10
H	9	1	10
	55	**25**	**80**
	69%	**31%**	

Choice: Include catastrophes causing 10% of the population to be lost every 50 years, and run variant cases with higher rates.

PVA—Final model parameterization

Following significant discussion about the impact of certain parameters on the model outputs, and after viewing model fits to the existing data given combinations of parameters, the Team re-evaluated the initial plausibility scores. Some additional parameters or combinations of parameters were evaluated in order to construct the final set of models. Arguments for various model parameters not previously discussed are described here.

Aerial Mobley data (Use it or not):
The Mobley data are from five aerial surveys (1993–2003), for which the protocols were consistent through time. The observers for each survey were mostly consistent through time, but with some changes. The survey area remained the same, and the effort had some variations due to weather. There were observations of groups of 8, 9, 1, 0, and 0 false killer whales during the five surveys, respectively. Since these were aerial surveys, the individuals were not confirmed to be Hawaiian insular individuals; however, the entire study area was within the range of the insular population. "No" votes primarily reflect questions over stock identity (since the survey couldn't confirm they were not pelagic individuals).

1989 prior upper limit (1392 vs. 769):
An upper limit for the 1989 abundance estimate should be specified, but without constraining the model too tightly. False killer whale density estimates for Palmyra Atoll are the highest of any region. Using the point-estimate of the density for U.S. EEZ waters surrounding Palmyra Atoll, applied to the area within 140 km of the main

Hawaiian Islands (the insular false killer whale stock range as described in the Draft 2010 Stock Assessment Report), the upper limit on the abundance of Hawaiian insular false killer whales would be 769 animals. Alternatively, using one standard error above the point-estimate of the density around Palmyra Atoll to account for uncertainty in that density estimate, the upper limit of the abundance of Hawaiian insular false killer whales could have reached 1392 animals.

Environmental Variance Autocorrelation (0 year vs. 2 year vs. 10 year cycles): Environmental variance is estimated within the model by sampling from a normal distribution. The selection is random if there is zero autocorrelation, or non-random if there is autocorrelation, with the value of environmental variance for each year influenced by previous years. Environmental variability is generally low in tropical ecosystems, but there may be decadal scale oscillations.

Allee Effect Level and Magnitude

Allee effects are quantitatively described by two parameters: 1) the population size (level) when half of the Allee effect would have acted on the population, and 2) the magnitude of the effect on the population growth rate (magnitude). A high Allee effect level would suggest that the population is experiencing Allee effects at relatively high population size, a plausible scenario for highly social odontocetes. A high magnitude would indicate a steep decline in growth rate as the population level falls. The following options were considered by the Team:

- No effect
- Strong Late: Births drop off when the population drops below 15 individuals, with large reductions in survival rates at lower population sizes. [level = 4, magnitude = – 0.09]
- Strong Early: Allee effects start early and are fairly strong. About 50% of reproductive capacity is lost at 75 individuals, and then at 40 individuals, all reproductive capacity is lost. Below 40 individuals the population declines very quickly. [level = 10, magnitude = -0.08].
- Birth Only: Similar to the Strong Early model until there are less than 75 individuals; below that population size the effect is lower than 'Strong Early' model because declines in population size are due only to a reduction in births, not the survival rate of adults. [level = 75, magnitude = – 0.02].

Mortality Events

Mortality events are rare catastrophes, but may represent disease die offs, strandings, etc. False killer whales do mass strand in other areas (Australia, South Africa), but have not stranded in large groups in Hawaii. Lack of strandings could reflect no mass mortality events or the geography of Hawaii where many coastlines are steep and would not allow live stranding. Mortality events are modeled to have relatively large impacts in which a large proportion of the population dies, as opposed to the smaller effects of environmental variation on the population. Anthropogenic events, including sonar testing events, oil spills, etc., are included here. Options are described in terms of the probability of a mortality events occurring (in 100 yrs), the magnitude of the event (proportion of the

population impacted), and the variation in that magnitude, expressed as a range of proportions (i.e., events may impact varying proportions of the population).

- No mortality events
- One mortality event every 100 yrs, 10% of the population is removed on average (range 4–16%)
- One mortality event every 50 yrs, 10% of the population is removed on average (range 4–16%)
- One mortality event every 10 yrs, 2% of the population is removed on average (range 1–3%)

Should a two rate-of-change model be considered?

Argument in favor of a two rate-of-change model

The base-case model fits all the data to a single growth rate (rate-of-change). Since the Mobley data indicate a strong decline, a one rate-of-change model will give strong weight to this decline and project this decline into the future. Using a two rate-of-change model allows the potential for two periods with different rates of change. There are a number of reasons to believe that false killer whale growth rates may have changed over the last decade:

- Reduction in prey due to competition with fisheries may have resulted in the reduction in false killer whales. If so, the current leveling off of fishing effort should result in a leveling off of false killer whale abundance. Competitive release of smaller fish may increase false killer whale abundance once the whales shift to these fish as their primary food source.
- Bycatch, shooting, and other incidental mortality in fisheries may have changed over time, as fishing regulations changed in 1991–1992, excluding commercial longline fisheries from nearshore waters. Although the Mobley decline occurred after the commercial longline exclusion, the false killer whale population may have been adjusting to losses in the earlier period.
- An oceanographic regime shift may have occurred
- Some contaminant levels peaked in the early to mid 1980s and have declined since

Argument against a two rate-of-change model

There is no reason, based on both the abundance and trend data or the fishing effort data, to assume population growth rate has changed to justify a more complex model.

- The cause of the decline is uncertain and therefore assuming the cause has changed is unwarranted.
- Even if the cause was known to be fishery effects (either direct fishery-related mortality or reduced survival and fecundity resulting from reduced food resources) there is insufficient evidence that such fishery effects have changed in insular waters. Much of the fishing effort in Hawaiian insular waters is poorly documented, and there are large uncertainties for both catch and bycatch.
- Contaminants do not decline in marine mammal tissues at the same rate they decline in the environment and are likely to have long-term effects

- Allowing two growth rates introduces three more parameters that must be estimated with few data. The prior distribution for the second growth rate will heavily influence the posterior distribution because the growth rate will be based largely on the last two data points which are close together in time and insufficiently precise to determine a trend. Given the data available, the model will be overparameterized.

PVA—Model parameterization—final plausibility-point allocation by team members

Table C. BRT member scores on the final parameterization of the PVA model. The scoring represents the plausibility scores from each member following review and discussion of individual model fits and further evaluation of specific parameters. Except where indicated, the choice with the highest score for each factor was chosen for the final model parameterization.

Use Mobley aerial survey data?			
Team member	Yes	No	Total points
A	8	2	10
B	8	2	10
C	10	0	10
D	8	2	10
G	9	1	10
F	8	2	10
E	8	2	10
H	8	2	10
	67	**13**	**80**
	84%	**16%**	

If Mobley data used, should it be considered as group or individual encounter rate?			
Team member	individual	group	Total points
A	3	7	10
B	3	7	10
C	3	7	10
D	6	4	10
G	3	7	10
F	4	6	10
E	6	4	10
H	7	3	10
	35	**45**	**80**
	44%	**56%**	

Time scale (yrs) of environmental autocorrelation				
Team member	0	2	10	Total points
A	0	4	6	10
B	0	3	7	10
C	2	4	4	10
D	0	5	5	10
G	0	6	4	10
F	0	5	5	10
E	0	4	6	10
H	0	4	6	10
	2	**35**	**43**	**78**
	3%	**45%**	**55%**	

Mark-recapture estimate with or without the Kauai animals?			
Voter	With Kauai	Without Kauai	Total points
A	7	3	10
B	6	4	10
C	5	5	10
D	5	5	10
G	7	3	10
F	6	4	10
E	7	3	10
H	5	5	10
	48	**32**	**80**
	60%	**40%**	

The lower limit of abundance based on the 1989 aerial survey				
Voter	470	mean	380	Total points
A	10	0	0	10
B	8	2	0	10
C	10	0	0	10
D	10	0	0	10
G	4	6	0	10
F	8	2	0	10
E	4	5	1	10
H	8	2	0	10
	62	**17**	**1**	80
	78%	**21%**	**1%**	

The upper limit on abundance for 1989			
Team member	1392	769	Total points
A	7	3	10
B	6	4	10
C	6	4	10
D	6	4	10
G	4	6	10
F	4	6	10
E	2	8	10
H	5	5	10
	40	**40**	**80**
	50%	**50%**	

Should a two growth-rate model be considered?			
Team member	Yes	No	Total points
A	3	7	10
B	6	4	10
C	2	8	10
D	7	3	10
G	6	4	10
F	5	5	10
E	8	2	10
H	3	7	10
	40	**40**	**80**
	50%	**50%**	

Frequency (yrs), magnitude of removal (%), and variance (range of population proportion) of catastrophes or unusual mortality events					
Team member	0 / 0 / 0	100 / 10 / 4-16	50 / 10 / 4-16	10 / 2 / 1-3	Total points
A	0	3	4	3	10
B	0	3	3	4	10
C	0	3	5	2	10
D	0	4	4	2	10
G	0	0	10	0	10
F	0	3	3	4	10
E	0	6	3	1	10
H	0	1	3	6	10
	0	**23**	**35**	**22**	**80**
	0%	**29%**	**44%**	**28%**	

Magnitude of environmental variation					
Team member	SRKW (high)	NRKW (medium)	tropical (low)	0	Total points
A	0	4	6	0	10
B	0	6	4	0	10
C	0	4	6	0	10
D	2	6	2	0	10
G	0	4	6	0	10
F	0	3	7	0	10
E	0	9	1	0	10
H	0	3	7	0	10
	2	**39**	**39**	**0**	**80**
	3%	**49%**	**49%**	**0%**	

Magnitude and level of Allee effects					
Team member	None	Birth only	Strong late	Strong early	Total points
A	0	5	1	4	10
B	0	2	5	3	10
C	0	4	1	5	10
D	0	4	0	6	10
G	0	4	2	4	10
F	0	4	2	4	10
E	0	1	7	2	10
H	0	5	0	5	10
	0	**29**	**18**	**33**	**80**
	0%	**36%**	**23%**	**41%**	

PVA—Model plausibility

Following construction of several PVA models given the above scored parameters, fully parameterized models were examined for their plausibility. This examination was aided through examination of model fits, with poor fits likely resulting in low plausibility scores. The evaluated models are described below. All scenarios refer to changes from the base case model (1), unless otherwise noted. Plausibility was examined on a 10-point scale. For example, if an individual team member felt the model was very plausible, that member could choose to allocate 10 points. Similarly, if that member found the model entirely implausible, then 0 points could be allocated. The total percentage of points awarded out of the total possible points (80) determined the overall plausibility of each model. The PVA outputs for many of these models are discussed in Appendix B.

1. Base-case: 1989 prior [470, 1392], Decadal environmental variation, Strong early Allee effects, mortality events occurring once every 50 yrs removing 10% of the population, recent mark-recapture abundance estimates including animals seen near Kauai, and using the group encounter rates from the Mobley aerial survey trend data.

2. Lower upper bound on 1989 abundance estimate: 769 individuals
3. Lower environmental variation
4. Birth-only Allee effect
5. 2-yr environmental autocorrelation
6. Mobley data with individual encounter rates
7. Less common (once in 100 yrs) mortality events
8. More common (once in 10 yrs) mortality events, but with lower magnitude (2% population removal)
9. Recent abundance estimate w/out Kauai animals
10. Plausible, but optimistic scenario
11. Base-case with two growth-rate model
12. Base-case with two growth-rate model and with recent abundance estimate without Kauai animals

The Team then evaluated the output of a number of two rate-of-change models to determine the plausibility of different constraints on the prior distribution of the second rate-of-change.

Option A—Uninformative prior: The distribution used for the second (current) growth rate should be uninformative (between an annual 30% decline and 30% increase).

- Yes, the second growth rate is unknown so the data should not be constrained with prior knowledge about growth rate.
- No, based on their life history, false killer whales cannot grow at more than 4%/year so there is prior knowledge about future growth rates that make a distribution that allows higher growth highly implausible. The model has utility in revealing the behavior of a two growth-rate model, but it is not plausible.

Option B—Symmetrical growth, limited: The distribution used for the current growth rate should be symmetrical but constrained by the maximum growth rate of false killer whales (between an annual 4% decline and 4% increase).

- Yes, using a symmetrical growth rate centered on no growth is consistent with the two most recent abundance estimates and does not bias the second growth rate towards a decline. This gives a fair indication of future uncertainty about population growth.
- No, Hawaiian insular false killer whales have evidence of a greater level of decline from the Mobley time series, and there is no reason to assume that the cause of that decline should have suddenly ceased after about the year 2000. It is entirely plausible that the previous level of decline (down to 10%/year) is continuing and should be a choice for the current growth rate.

Option C—Asymmetrical growth, limited plus constant decline possible: The distribution used for the current growth rate should be asymmetrical constrained by the maximum growth rate of false killer whales and by the level of decline consistent with the Mobley time series (between an annual 10% decline and 4% increase).

- Yes, using an asymmetrical growth rate using the maximum possible growth rate for false killer whales as an upper bound and the recently observed rate of decline as the lower bound best describes our current understanding of what is plausible. Hawaiian Insular false killer whales have evidence of a decline from the Mobley time series and there is no reason to assume that the cause of that decline should have suddenly ceased after about the year 2000. It is entirely plausible that the previous level of decline (down to 10%/year) is continuing and should be a choice for the current growth rate.
- No, using an asymmetrical growth distribution with more possibility of declining than increasing will result in a higher chance of heading towards extinction than towards recovery. The data are weak in altering the growth rate for the estimate of the second constant growth rate, so this prior distribution is very influential and should not be informative.

The PVA outputs for the most plausible two growth-rate models are compared with the results of the exponential models in Appendix B.

PVA—Model plausibility—point allocation by team members

BRT scores on the plausibility of specific PVA model parameterizations given examination of model fits and the combination of individual parameters used in each model. Individual member scores have been anonymized for presentation here. Model numbers are as described above.

	Team member									
Model number	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**TOTAL**	**%**
1	8	6	9	10	7	9	10	8	**67**	**84**
2	8	8	0	0	4	5	6	6	**37**	**46**
3	8	7	9	10	5	3	10	10	**62**	**78**
4	8	7	9	7	6	9	6	5	**57**	**71**
5	7	7	9	10	4	7	9	5	**58**	**73**
6	2	2	0	2	2	1	2	4	**15**	**19**
7	8	4	9	8	7	8	7	8	**59**	**74**
8	8	5	9	6	7	9	7	7	**58**	**73**
9	7	5	9	10	5	8	9	6	**59**	**74**
10	6	7	0	0	5	4	2	6	**30**	**38**
11	5	2	1	3	7	7	4	3	**32**	**40**
12	5	3	1	3	4	6	4	2	**28**	**35**

Plausibility-point allocation results for the prior distribution for the second rate-of-change in the two-rate-of-change model. This score is conditional on previous PVA model

plausibility scores, i.e., the results will be factored in based on the previous decisions on the one rate-of-change model versus the two-rate-of-change model.

	Options			
Team member	**A: uninformative prior – 0.3 to 0.3**	**B: symmetric growth prior – 0.04 to 0.04**	**C: asymmetric growth prior – 0.1 to 0.04**	**Total points allotted**
A	0	2	8	10
B	0	5	5	10
C	6	1	3	10
D	0	3	7	10
E	1	5	4	10
F	3	0	7	10
G	4	1	5	10
H	0	3	7	10
	14	**20**	**46**	80
	18%	**25%**	**58%**	

Threats table—categorization by team members

Individual threats are listed in Section 2.4 of the status review and the definitions of each of the column headings used here are described in the overall assessment of the threats in section 4.1. Team members assigned a score of 1 through 3 for each threat, with 1 representing lower threats and 3 higher threats. Scores are anonymized here for presentation. The Overall Threat Level ranking of limiting factors was completed through deliberation of the group, which produced a single consensus ranking for each factor (not shown here).

Individual BRT member threat table votes. Mean and median vote tallies are provided for each threat, with median values presented in the Threat table summary within the status review.

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Habitat Destruction, Modification, or Curtailment	Reduced food quality and quantity	Reduced total prey biomass from all causes	Mortality, Foraging success	A	through early 1970s*	2	3	2	3	3	2	3
				B		2	3	2	2	3	2	2
				C		3	3	2	2	3	2	2
				D		2	3	1	2	3	1	2
				E		2	3	1	2	3	1	2
				F		2	3	1	2	3	1	2
				G		3	3	3	3	3	3	3
				H		3	3	2	2	3	1	2
					Mean	**2.4**	**3**	**1.8**	**2.3**	**3**	**1.6**	**2.3**
					Median	**2.0**	**3.0**	**2.0**	**2.0**	**3.0**	**1.5**	**2.0**
		Competition with commercial deep-set and shallow-set longline fisheries	Mortality, Foraging success	A	through early 1990s	2	3	1	2	3	2	2
				B		3	3	2	1	3	2	1
				C		3	3	2	2	3	2	2
				D		3	3	2	2	3	1	2
				E		3	3	2	1	3	1	1
				F		3	3	2	2	3	1	2
				G		2	3	2	2	3	2	2
				H		2	3	1	1	3	1	1
					Mean	**2.6**	**3**	**1.8**	**1.6**	**3**	**1.5**	**1.6**
					Median	**3.0**	**3.0**	**2.0**	**2.0**	**3.0**	**1.5**	**2.0**
		Competition with troll, handline, shortline, and kaka-line fisheries	Mortality, Foraging success	A	before late 1980s	2	3	1	2	3	2	2
				B		2	2	1	2	2	1	2
				C		3	3	2	2	3	2	2
				D		2	3	1	2	3	1	2
				E		2	3	1	2	3	1	2
				F		2	3	1	2	3	1	2
				G		1	3	1	1	3	1	1
		Competition with troll, handline, shortline, and kaka-line fisheries	Mortality, Foraging success	H	before late 1980s	2	3	1	1	3	1	1
					Mean	**2.0**	**2.9**	**1.1**	**1.8**	**2.9**	**1.3**	**1.8**
					Median	**2.0**	**3.0**	**1.0**	**2.0**	**3.0**	**1.0**	**2.0**

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Habitat Destruction, Modification, or Curtailment (continued)	Reduced food quality and quantity (continued)	Competition with recreational fisheries	Mortality, Foraging success	A	before late 1980s	1	3	1	2	3	1	2
				B		1	2	1	1	2	1	1
				C		2	3	1	2	3	1	2
				D		1	3	1	1	3	1	1
				E		1	3	1	1	3	1	1
				F		1	3	1	1	3	1	1
				G		1	2	1	1	2	1	1
				H		1	3	1	1	3	1	1
					Mean	**1.1**	**2.8**	**1.0**	**1.3**	**2.8**	**1.0**	**1.3**
					Median	**1.0**	**3.0**	**1.0**	**1.0**	**3.0**	**1.0**	**1.0**
		Reduced prey size (whale energetics, nutritional quality)	Mortality, Foraging success	A	through early 1970s*	2	3	2	3	3	2	3
				B		2	3	2	2	3	2	2
				C		2	3	2	2	3	2	2
				D		2	3	1	2	3	1	2
				E		2	3	1	2	3	1	2
				F		2	3	2	2	3	2	2
				G		3	3	3	3	3	3	3
				H		2	3	1	2	3	1	2
					Mean	**2.1**	**3**	**1.8**	**2.3**	**3**	**1.8**	**2.3**
					Median	**2.0**	**3.0**	**2.0**	**2.0**	**3.0**	**2.0**	**2.0**
		Ocean warming increasing low-productivity zones	Mortality, Foraging success	A	Primarily current/future threat				2	3	1	2
				B					1	3	2	1
				C					1	3	2	1
				D					1	3	1	1
				E					1	3	1	1
				F					1	3	1	1
				G					2	3	2	2
				H					2	3	1	2
								Mean	**1.4**	**3**	**1.4**	**1.4**
								Median	**1**	**3**	**1**	**1**

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Habitat Destruction, Modification, or Curtailment (continued)	Reduced food quality and quantity (continued)	Ocean acidification	Mortality, Foraging success	A	Primarily current/future threat				2	3	1	2
				B					2	2	1	2
				C					2	3	1	2
				D					2	3	1	2
				E					1	3	1	1
				F					1	3	1	1
				G					1	3	1	1
				H					1	2	1	1
		Natural and anthropogenic contaminants (chemicals and particulates)	Mortality, Reproductive potential					Mean	**1.5**	**2.8**	**1.0**	**1.5**
								Median	**1.5**	**3**	**1**	**1.5**
				A	through mid-1980s	1	3	1	2	3	2	2
				B		2	3	2	1	3	1	1
				C		abstain						
				D		1	3	1	2	3	2	2
				E		2	3	2	1	3	1	1
				F		2	3	2	1	3	1	1
				G		2	2	2	2	2	2	2
				H		2	3	1	1	3	1	1
					Mean	**1.7**	**2.9**	**1.6**	**1.4**	**2.9**	**1.4**	**1.4**
					Median	**2.0**	**3.0**	**2.0**	**1.0**	**3.0**	**1.0**	**1.0**
Overutilization	Low density	Live-capture	Reproductive potential	A	Pre-1990	2	3	1	Primarily historic threat			
				B		2	3	2				
				C		2	2	2				
				D		2	3	2				
				E		2	3	2				
				F		2	3	2				
				G		2	2	2				
				H		2	2	2				
					Mean	2	2.6	1.9				
					Median	2	3	2				

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Inadequate Regulatory Mechanisms	Incidental take	Lack of reporting/observing of nearshore fishery interactions	Mortality, Behavior	A	Through late-1980s	3	3	2	2	3	1	2
				B		2	2	1	2	3	1	2
				C		3	3	2	2	2	2	2
				D		3	3	3	2	3	2	2
				E		2	3	2	2	3	1	2
				F		3	3	2	2	3	1	2
				G		2	2	2	2	2	2	2
				H		3	3	2	2	2	2	2
					Mean	**2.6**	**2.8**	**2**	**2**	**2.6**	**1.5**	**2**
					Median	**3**	**3**	**2**	**2**	**3**	**1.5**	**2**
Disease and Predation	Disease	Harmful algal blooms	Mortality, Reproductive potential	A	Primarily current/future threat				3	2	1	2
				B					2	2	2	2
				C					1	2	1	1
				D					2	3	1	1
				E					1	2	1	1
				F					1	2	1	1
				G					1	1	1	1
				H					1	1	1	1
								Mean	**1.5**	**1.9**	**1.1**	**1.3**
								Median	**1**	**2**	**1**	**1**
		Environmental contaminants	Mortality, Reproductive potential	A	through mid-1980s	2	2	1	2	2	2	2
				B		1	3	1	2	3	2	2
				C		2	2	1	2	2	2	2
				D		1	3	1	2	3	2	2
				E		1	3	1	2	3	2	2
				F		2	3	1	2	3	2	2
				G		2	2	2	2	3	2	2
				H		2	3	1	2	3	1	2
					Mean	**1.6**	**2.6**	**1.1**	**2**	**2.8**	**1.9**	**2**
					Median	**2.0**	**3.0**	**1.0**	**2.0**	**3.0**	**2.0**	**2.0**
		Long- and short-term climate change	Mortality, Reproductive potential	A	Primarily current/future threat				2	3	1	2
				B					2	2	2	2

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Disease and Predation (continued)	Disease (continued)	Long- and short-term climate change (continued)	Mortality, Reproductive potential	C	Primarily current/future threat				1	3	2	1
				D					2	3	2	2
				E					1	3	1	1
				F					2	3	1	2
				G					2	2	2	2
				H					1	3	1	1
		Parasitism	Mortality, Reproductive potential, Foraging success					Mean	**1.6**	**2.8**	**1.5**	**1.6**
								Median	**2**	**3**	**1.5**	**2**
				A	Same threat level				2	3	1	2
				B					1	3	1	1
				C					1	3	2	1
				D					1	3	1	1
				E					1	3	1	1
				F					1	3	1	1
				G					1	3	1	1
				H					1	3	1	1
								Mean	**1.1**	**3.0**	**1.1**	**1.1**
								Median	**1**	**3**	**1**	**1**
	Predation	Killer whales	Mortality	A	Same threat level				1	3	1	1
				B					1	3	1	1
				C					1	3	2	1
				D					1	3	1	1
				E					1	3	1	1
				F					1	3	1	1
				G					1	3	1	1
				H					1	3	1	1
		Shark (including cookie-cutter sharks)	Mortality					Mean	**1**	**3**	**1.1**	**1**
								Median	**1**	**3**	**1**	**1**
				A	Same threat level				1	3	1	1
				B					2	2	2	2
				C					1	3	2	1
				D					2	3	1	2

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Disease and Predation (continued)	Predation (continued)	Shark (including cookie-cutter sharks)	Mortality	E	Same threat level				1	3	1	1
				F					1	3	1	1
				G					1	3	1	1
				H					1	3	1	1
	Competition	Marlin, sharks, etc.	Foraging success, Behavior					Mean	**1.3**	**2.9**	**1.3**	**1.3**
								Median	**1**	**3**	**1**	**1**
				A	through early 1970s*	1	3	1	1	3	1	1
				B		1	3	1	1	3	1	1
				C		1	3	2	1	3	2	1
				D		1	3	1	1	3	1	1
				E		1	3	1	1	3	1	1
				F		1	3	1	1	3	1	1
				G		1	3	1	1	3	1	1
				H		1	3	1	1	3	1	1
					Mean	**1**	**3**	**1.1**	**1**	**3**	**1.1**	**1**
					Median	**1.0**	**3.0**	**1.0**	**1.0**	**3.0**	**1.0**	**1.0**
Other Natural or Manmade Factors	Hooking, entanglement or intentional harm (e.g. shooting, poisoning, explosives)	Interaction with commercial longline fisheries	Mortality, Behavior	A	Through the early 1990s	3	3	2	2	2	2	2
				B		3	3	2	2	2	1	1
				C		*3*	*3*	*2*	*2*	*2*	*2*	*2*
				D		3	3	3	2	2	1	1
				E		2	3	2	1	2	1	1
				F		3	3	2	2	2	1	2
				G		2	3	1	1	2	2	1
				H		2	3	2	1	1	1	1
					Mean	**2.6**	**3**	**2**	**1.6**	**1.9**	**1.4**	**1.4**
					Median	**3**	**3**	**2**	**2**	**2**	**1**	**1**
		Interactions with troll, handline, shortline, and kaka-line fisheries	Mortality, Behavior	A	pre mid-1970s	1	3	1	3	3	1	3
				B		1	3	1	3	3	1	3
				C		1	3	1	3	3	1	3
				D		1	3	1	3	3	1	3
				E		1	3	1	3	3	1	3
				F		1	3	1	2	3	1	3
				G		1	3	1	2	3	1	2

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Other Natural or Manmade Factors	Hooking, entanglement or intentional harm (e.g. shooting, poisoning, explosives)	Interactions…	Mortality…	H		1	3	1	2	3	1	2
					Mean	**1**	**3**	**1**	**2.6**	**3**	**1**	**2.8**
					Median	**1**	**3**	**1**	**3**	**3**	**1**	**3**
		Interactions with aquaculture facilities	Mortality, Behavior	A	pre-2010	1	1	2	2	1	1	1
				B		1	1	2	1	1	1	1
				C		1	1	2	1	1	2	1
				D		1	1	2	1	1	1	1
				E		1	1	2	1	1	1	1
				F		1	1	2	1	2	1	1
				G		1	1	2	1	1	2	1
				H		1	1	2	1	1	1	1
					Mean	**1**	**1**	**2**	**1.1**	**1.1**	**1.3**	**1**
					Median	**1**	**1**	**2**	**1**	**1**	**1**	**1**
	Hooking, entanglement or intentional harm (e.g. shooting, poisoning, explosives) (continued)	Oil spills	Mortality, Reproductive potential, Foraging success, Behavior	A	same threat level				2	1	2	1
				B					2	1	1	1
				C					1	2	1	1
				D					2	1	1	1
				E					1	1	1	1
				F					1	2	1	1
				G					1	1	1	1
				H					2	2	1	1
	Small population size	Plastic ingestion	Mortality, Foraging success					Mean	**1.5**	**1.4**	**1.1**	**1**
								Median	**1.5**	**1**	**1**	**1**
				A	gradually increasing threat level				1	3	1	1
				B					1	2	1	1
				C					1	3	1	1
				D					1	1	1	1
				E					1	1	1	1
				F					1	2	1	1
				G					1	1	1	1
				H					1	3	1	1
								Mean	**1**	**2**	**1**	**1**
								Median	**1**	**2**	**1**	**1**

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Other Natural or Manmade Factors (continued)	Small population size Small population size (continued)	Reduced genetic diversity	Reproductive potential	A	Through 1989	1	3	1	2	3	2	2
				B		2	3	1	2	3	2	2
				C		2	3	1	3	3	2	2
				D		1	3	1	2	3	2	2
				E		1	3	1	2	3	2	2
				F		1	3	1	2	3	2	2
				G		2	3	2	2	3	2	3
				H		2	3	1	3	3	3	3
					Mean	**1.5**	**3**	**1.1**	**2.3**	**3**	**2.1**	**2.3**
					Median	**1.5**	**3**	**1**	**2**	**3**	**2**	**2**
		Inbreeding depression	Reproductive potential	A	Through 1989	1	3	1	2	3	2	2
				B		1	3	1	2	3	1	2
				C		2	3	1	3	3	1	2
				D		1	3	1	2	3	1	2
				E		1	3	1	2	2	1	2
				F		1	3	1	2	3	1	2
				G		2	3	2	2	3	2	2
				H		2	3	1	3	3	3	3
					Mean	**1.4**	**3**	**1.1**	**2.3**	**2.9**	**1.5**	**2.1**
					Median	**1**	**3**	**1**	**2**	**3**	**1**	**2**
		Allee effects	Reproductive potential, Mortality	A	Through 1989	1	3	1	2	3	2	2
				B		1	3	1	2	3	1	2
				C		2	3	1	3	3	1	3
				D		2	3	1	2	3	1	2
				E		1	3	1	2	3	2	2
				F		1	3	1	2	3	1	2
				G		1	3	1	1	3	1	2
				H		2	3	1	3	3	3	3
					Mean	**1.4**	**3**	**1**	**2.1**	**3**	**1.5**	**2.3**
					Median	**1**	**3**	**1**	**2**	**3**	**1**	**2**

ESA Factor	Limiting Factor	Threats	Key Ecological Attribute	Team member	Period of historic impact	Historic Severity	Geographic scope	Level of certainty	Current / Future Severity	Geographic scope	Level of certainty	Overall Threat Level
Other Natural or Manmade Factors (continued)	Other	Sonar and seismic activities (military, oceanographic, and recreational fishing)	Mortality, Reproductive potential, Behavior, Foraging success	A	HRC formation	1	2	1	2	2	2	2
				B		1	2	2	2	2	2	2
				C		3	2	1	3	2	1	3
				D		1	2	1	2	3	1	2
				E		1	2	1	2	2	1	2
				F		1	2	1	2	3	2	2
				G		2	2	2	2	2	2	3
				H		1	3	1	2	3	1	2
					Mean	**1.4**	**2.1**	**1.3**	**2.1**	**2.4**	**1.5**	**2.3**
					Median	**1**	**2**	**1**	**2**	**2**	**1.5**	**2**
		Vessel strikes	Mortality	A	gradually increasing threat level				1	3	1	1
				B					1	2	1	1
				C					1	2	1	1
				D					1	2	1	1
				E					1	2	1	1
				F					1	2	1	1
				G					1	2	1	1
				H					1	2	1	1
		Whale/dolphin watching	Behavior, Reproductive potential					Mean	**1**	**2.1**	**1**	**1**
								Median	**1**	**2**	**1**	**1**
				A	gradually increasing threat level				1	2	1	1
				B					1	2	1	1
				C					1	1	1	1
				D					1	2	1	1
				E					1	2	1	1
				F					1	2	1	1
				G					1	2	1	1
				H					1	1	1	1
								Mean	**1**	**1.8**	**1**	**1**
								Median	**1**	**2**	**1**	**1**

Appendix B. Risk of Extinction of the Hawaiian Insular Population of False Killer Whales

Introduction

The Hawaiian insular (HI) false killer whale population is potentially at risk of extinction because the population is small (Baird, 2009) and estimated to be recently declining (Mobley et al., 2000; Mobley, pers. comm.; Baird, 2009). Therefore, we examine the risk of extinction of this population using a population viability analysis (PVA). PVA is a model type designed to estimate the probability of extinction over time for a population given the current best understanding of the population's size, dynamics, and threats. PVA is one of the most common methods used for quantitative risk analysis for endangered species (Morris and Doak, 2002). Model results are used to evaluate future risks and allow populations to be compared using a common risk "currency" conditional on a specified scenario. The utility of PVA as a tool for guiding conservation decision-making has been recognized by the Ecological Society of America (Carroll et al., 1996), The Nature Conservancy (Morris et al., 1999), and by a governmental science advisory board (National Research Council, 1995).

The processes that cause small populations to have a greater risk of extinction include genetic and behavioural problems, as well as chance processes like such as demographic and environmental stochasticity (Gilpen and Soule, 1986; Goodman, 1987a; Lande, 1993; Shaffer, 1981; Simberloff, 1988). A positive relationship between population size and per capita population growth, at low densities, is often referred to as the "Allee effect" or "depensation" (e.g., Allee, 1931; Allee et al., 1949; Dennis 1989, 2002; Fowler and Baker, 1991; Courchamp et al., 1999; Stephens and Sutherland, 1999; Stephens et al., 1999; Petersen and Levitan, 2001; Berec et al., 2007). In essence, as the number of individuals decreases there are costs consequences from possible factors such as a lack of predator saturation, impaired anti-predator vigilance or defence, a breakdown of cooperative feeding, an increased possibility of inbreeding depression or other genetic issues, decreased birth rates due as a result ofto the whales inability not to finding mates, or a combination of these effects. The Allee effect increases risk to small populations directly by contributing to the risk of extinction, and indirectly by decreasing the rate of recovery of exploited populations and therefore maintaining populations at a smaller size where extinction risk is higher for a variety of reasons (e.g., environmental or demographic stochasticity) (Dennis 1989, ; Stephens and Sutherland, 1999).

In addition, social odontocetes, such as false killer whales, may be particularly vulnerable to the effects of exploitation over and beyond the numerical loss of individuals to the population (Wade and Reeves, in press). Some of these effects may act in a similar fashion to Allee effects, or have a more pronounced effect at low population sizes. Survival and reproductive success may depend on such things as social cohesion and social organization, mutual aid in defense against predators and possible alloparental care such as 'babysitting' and communal nursing, sufficient opportunities for transfer of 'knowledge' (learned behavior) from one generation to the next, and leadership by older individuals that know where and when to find scarce prey resources and how to avoid high-risk circumstances (e.g., ice-entrapment, stranding, predation).

It is very difficult to detect Allee effects for practical and theoretical reasons (Lande, 2002), and little is known about these processes for most endangered species (Brook et al., 1997; Ralls et al., 2002). Likewise, the exact nature of the stochastic elements that affect a population's dynamics

is rarely known. Ralls et al. (2002) recommend incorporating several alternative forms of model dynamics in population viability analyses (PVA). The aim is to explore all plausible scenarios, such as recommended by Beissinger and Westphal (1998). The technical issue of how to weight the respective scenarios in a summary assessment can be difficult. However, if all or nearly all scenarios give qualitatively similar results, this can show that uncertainty as to which is the true scenario is unimportant.

PVA models estimate extinction risk by integrating the various risks a population may face. A population model was developed here to assess the extinction risk for Hawaiian insular false killer whales. PVAs usually involve relatively complex models, with many parameters that need to be specified. Generally, population models are used that include demographic and environmental variability and often include Allee effects. Another factor that is sometimes included is large mortality events (often referred to as 'catastrophes') that occur in a single year. Several papers have summarized current practices in the use of PVAs (Beissinger and Westphal, 1998; White, 2000). The model developed here contains all these effects – demographic and environmental variability, an Allee effect, and large mortality events.

For Hawaiian insular false killer whales there are data available to estimate abundance in some years and to estimate abundance trends in other years; these were sufficient to permit PVA modeling. We chose to assess risk using three primary data sources: mark-recapture abundance estimates (Baird, 2009), relative trend data from aerial surveys (Mobley et al., 2000; Baird, 2009), and an estimate of minimum abundance from 1989 (Reeves et al., 2009). The model projects forward from 1989, which was the year of the Reeves et al. (2009) minimum count. Bayesian statistical methods are used to fully incorporate uncertainty in the estimates of rate of change of the population, and to be able to specify prior distributions for parameters that are not fully specified by the data (Wade, 2002a).

One advantage of the PVA model used here is that it incorporates uncertainty in the recent abundance and rate of change of the population (1989 through 2009) into the forecast of future extinction risk. This is in contrast to many widely used PVA methods, particularly those using general software programs such as VORTEX (Lacy et al., 2005) and RAMAS (Akçakaya et al., 1999). PVAs that ignore uncertainty in classifying populations according to risk (such as under International Union for Conservation of Nature and Natural Resources [IUCN] criteria) or in estimating the probability of extinction can be misleading and inaccurate (Taylor, 1995; Ludwig 1996; Ralls and Taylor, 1997; Ludwig, 1999; White, 2000; Taylor et al., 2002; Wade, 2002a).

MATERIALS AND METHODS

Base model: single estimated population rate of change ("one- rate")

The base model was displayed in the form of a simple exponential model with two parameters to be estimated – a starting population size and a rate of change of the population. The model used individuals as the unit that could die or give birth each year, but age and sex of the individuals were not included. The model differed from standard exponential models by incorporating demographic variance due to births and deaths by sampling the change in population size each

year from a binomial distribution, using the specified population rate of change. For example, if the finite rate of change was 0.9 (i.e., a 10% decline each year) and there were 5 whales, a random number was drawn from a binomial distribution with probability 0.1 (i.e., 0.1 =1.0 –- 0.9) with 5 trials. The number of "successes" represents the net change in the number of whales, which can range from 0 to 5. This is one of the standard ways in which demographic variance is incorporated into an aggregated model for a PVA, as this approximates the process of random births and deaths. It exactly mimics the random nature of deaths, but it approximates the random nature of births by essentially reducing the expected number of deaths by the expected number of births. Specifically:

$$N_{t+1} = N_t + n_t^{\text{bin}} \qquad \text{(Eq. 1)}$$

where

N_t = the population size in year t

$n_t^{\text{bin}} \sim \text{binomial}(N_t, r_t)$ if $r_t > 0.0$

$\sim - -1[\text{binomial}(N_t, r_t)]$ if $r_t < 0.0$

and

r_t = the annual rate of change in year t.

Environmental variance was incorporated by treating r_t as a random variable with mean r and normally distributed process error (Morris and Doak, 2002):

$$r_t = r + \varepsilon_t \qquad \text{(Eq. 2)}$$

where:

r is the expected rate of change (i.e., the constant trend in the population),

ε_t is the environmental effect on r in year t

and auto-correlation is incorporated as

$$\varepsilon_{t+1} = (\rho * \varepsilon_t) + \eta_t \qquad \text{(Eq. 3)}$$

where

$\eta_t \sim \text{Normal}(0, [(1-\rho^2) * \sigma^2_{\text{env}}])$,

σ^2_{env} = the environmental variance for uncorrelated years

ρ = the autocorrelation parameter (between 0.0 and 1.0).

The Allee model used was that of Stephens and Sutherland (1999), referred to as the rectangular hyperbola model. It is a flexible model, containing both a location parameter and a scale parameter:

$$r_{t2} = r_t - \alpha * (\theta/(\theta + n))] \qquad \text{(Eq. 4)}$$

Where

r_{t2} is the rate of change after incorporating the Allee effect,

θ is the population size where half of the Allee effect has taken place,

α is a parameter that scales the Allee effect (e.g., if $\alpha=2r_t$, then r_{t2} is – $-r_t$ at $n = 0$).

In this case, α sets the magnitude of the maximum decrease in the rate of change. When r is considered to be the intrinsic rate of increase, setting α to $2r$ will give a function that is symmetric in the sense that the population increases at r when at a higher population size (away from Allee effects) and the population decreases at $-r$ when the population approaches zero. In this application, r is not confined to be positive (such as an intrinsic rate of increase). Instead, r is the current rate of change of the population (which could be negative due to, for example, human caused mortality of some kind). Therefore, in this case α is set to be an absolute decline in r, such that a positive r can become a negative r after Allee effects, or a negative r can become a more negative r after Allee effects.

Large mortality events ('catastrophes') were also incorporated by changing r_t in a year where a mortality event occurs:

$$r_{t3} = r_{t2} - r_{me} \qquad \text{(Eq. 5)}$$

where

r_{me} = 0.0 if $u_t > p_{me}$

= ~ Normal(m_{me}, sd_{me}) if $u_t < p_{me}$

and

p_{me} = probability of occurrence of a mortality event in any given year,

m_{me} = magnitude of the mortality event,

sd_{me} = standard deviation of the magnitude of the mortality event,

u_t. = ~ Uniform(0, 1).

A uniform random number between 0 and 1 was generated, and if it was below p_{me} (0.01, for example), a catastrophe would occur in that year. The magnitude of the event was randomly drawn from a normal distribution with mean of m_{me} and standard deviation of sd_{me}. This represents the percentage of the population that dies from the mortality event in that year. Potential sources of such mortality events are mass strandings, disease, and oil spills.

Alternative model: two estimated rates of population change ("two- rate")

Following additional discussion of some of the limitations of the single rate model, the team felt that there had potentially been changes in the level of impact of the factors that were most likely affecting survivorship in the population. In particular, incidental mortality from longline fisheries, which may have contributed a great deal to mortality in the population, may have been significantly reduced as a result of this fishery being largely excluded from the population's primary range (based on recent telemetry locations) in the early 1990s. Secondly, the heavy fishing pressure within this population's range has likely reduced the standing stock of many of their preferred prey species, such that the nearshore ecosystem now has a lower carrying capacity. However, recent estimates of fish stocks suggest that these remain stable but far lower than historic levels. In addition, the two Hawaiian insular false killer whale population estimates from the 2000s (which were based on a relatively high resight rate (from photo-ID), while only 5 years apart do not indicate a reduction of the population in the 2000s.

This type of a scenario, where a major source of mortality has been removed while an additional factor that reduces survivorship but allows the population to persist on a lower plane, may be very similar to that which has been observed in southern Southern Resident killer whales, also a small population of social odontocetes. That population was reduced by a third (primarily juveniles) in just a few years in the late 1960s because of live-capture for the oceanarium trade. Simultaneously, their prey base, of salmon, continued to be significantly reduced during the 1970s and 80s, hitting its nadir in the mid-1990s. Correlations in killer whale survival and fecundity have been found with the precipitous decline in salmon in the 1990s (Ford et al., 2010; Ward et al., 2010). The limited growth in this population following cessation of live captures, while in part due to a lack of juvenile animals, may have been confounded by an eroding prey base, and the low and erratic pattern of growth that this population has exhibited over the past 30 years is in sharp contrast to the adjoining Northern Resident killer whale population.

To account for the possibility of a new lower carrying capacity, within which the Hawaiian insular false killer whale population is now maintained without further decline due to competition for prey or interactions with longline fishing, a second model was considered that allowed for two different rates-of-change to operate over two different time periods. This model attempted to reflect the possibility that some mechanism that might have affected the rate of change of the population was not in operation for the entire time period, or necessarily into the future. This model had two additional parameters, which are the second rate of change, and the year in which the rate-of-change occurred. Otherwise, all the other parameters of the model were constant across the two time periods.

Available data and values of specified parameters

Absolute abundance estimates from mark-recapture data
Baird et al. (2009) made mark-recapture estimates of abundance using identification photographs taken from small boats. They made estimates for two time periods: 2000–-2004, and for 2006–-2009. The POPAN form of the Jolly-Seber open population model was used, and they explored both constant and time-dependent survival and capture probability. These estimates are preferable to use over the previous estimate from 2000–-2004 (Baird et al., 2005) because these two estimates use all the data through 2009, and provide two estimates for two different time points, to provide more information about trends in abundance. POPAN provides an estimate of abundance for the end of the given time period, so the estimate from the 2000–-2004 data was assigned to 2004, and the estimate from the 2006–-2009 data was assigned to 2009.

Estimates from the best model were:

2000–-2004	162 (CV = 0.23)
2006–-2009	151 (CV = 0.20) without Kauai identifications
2006–-2009	170 (CV = 0.21) with Kauai identifications

Baird et al. (2009) calculated two different estimates from the 2006–-2009 data, one using a small number of identifications from Kauai and one that did not use those data. Use of the data from Kauai raised two possible issues related to potential bias. No identifications were obtained from Kauai during 2000–04, so the overall effort and sampling was examined to determine what was the most comparable estimate for 2006–2009. Additionally, the whales identified at Kauai are the only individuals that have not been linked through association into the social network that encompasses all other whales in the study, raising questions as to whether they are members of the Hawaiian insular false killer whale population.

In terms of survey effort in the two periods, photo-identification effort at Kauai by Baird occurred only in the years 2003 (24 days), 2005 (24 days), and 2008 (7 days) (R. Baird, pers. comm.), so there was actually more dedicated effort in 2000–2004 then there was in 2006–2009. Baird's surveys did not encounter false killer whales in any of the years. Baird has information submitted by collaborators (primarily whale watching vessels) on eight groups encountered at Kauai, and the submission of photographs from collaborators was relatively constant from 2000 through 2009 (R. Baird, pers. comm.). Only three of the groups have photographs that meet the quality requirements for use in the mark-recapture study. Those three encounters (representing a small number of identifications) were all from the 2006–2009 period. Because there was survey effort in Kauai in each time period, the two data sets should be comparable when including the Kauai data, indicating the 2006–2009 estimate including Kauai identifications should be used.

The three groups that have identifications from Kauai were all encountered by collaborators that did not conduct biopsy sampling, so there are no genetic samples from Kauai to examine. All the encounters with false killer whales at Kauai were close to shore and well within the defined range of the Hawaiian insular stock (i.e., < 40 km from shore), so these whales would be considered part of the insular population if seen in that location on a line-transect survey (i.e, without identification photos). The locations of the encounters were within the Mobley aerial survey study area, so including those whales in the abundance estimate is consistent with the

other data on abundance and trends. Additionally, the satellite tagging data shows whales moving freely between Kauai and other islands; for example, one whale tagged at Oahu moved to Kauai and spent considerable time there (Baird, 2009). The fact that they have not associated with other whales that are part of the Hawaiian insular false killer whale social network may just be a function of each individual having only been sighted once; they may well be found to associate in the future, though we have no way of determining that now. Therefore, the 2006–2009 abundance estimate including Kauai identifications (170) was used in the base PVA analysis. A variant was specified using the estimates made without the Kauai identifications.

Trend information from aerial surveys

Five aerial surveys for cetaceans were flown during the period of 1993–2003 (Mobley et al., 2000; Mobley, 2004; Baird, 2009). Baird (2009) reported the group encounter rate (#/ 10 survey hours) for each year of the aerial surveys as 1.06 (1993, 8 groups), 0.975 (1995, 9 groups), 0.141 (1998, 1 group), 0.00 (2000, 0 groups), and 0.00 (2003, 0 groups). There were not sufficient data from false killer whales to estimate detection probability separately for each year, so it is not possible to estimate absolute abundance for each year. Baird (2009) showed there were no significant trends in other species from those surveys, providing some evidence that detection probabilities had likely not declined through time (e.g., because of some different observers, poorer weather, or other issues).

Given the lack of correction for detection probability, an abundance index from these surveys was treated as a relative index of abundance through time that gave information about the trend of the population. Although it would make most sense to use the trend of individuals (rather than the trend in encounters with groups), mean group sizes were 6.1 (1993), 7.0 (1995), and 1.0 (1998), with no groups seen in 2000 and 2003. Given that only one group was seen in 1998, mean group size is not reliably estimated in that year. The single encounter in 1998 was of a single individual. In 1992 and 1995 there were 2 and 3 singletons seen, respectively, out of the 8 and 9 total groups seen in those years. Therefore, it is possible that mean group size had not actually changed in 1998. The group-encounter rate declines rapidly over the five surveys, and the individual encounter rate declines at an even greater rate. Given that group size could not be reliably estimated over the last three surveys, the model was fit to the group encounter rate, not the individual encounter rate. This makes the assumption that group size has not changed across years. One trial was specified where the model was fit to the individual encounter rate for comparison.

Minimum absolute abundance from the 1989 aerial survey

A minimum count of false killer whales from the main Hawaiian Islands was obtained from a 1989 aerial survey (Reeves et al., 2009). The survey covered relatively nearshore waters and the entire study area was within the study area of the Mobley et al. (2000) surveys. The survey encountered groups of false killer whales larger than 300 on 3 separate days resulting in minimum counts of 460 on 23 June (after 11 passes from the airplane), 470 on 26 June (after 14 passes), and 380 on 28 June (after 7 passes).

From the air, factors such as dispersion of animals, size and arrangement of subgroups, and glare conditions can influence the accuracy of counts. For example, given the rapid travel speed of an aircraft, it is often easier to count animals dispersed in discrete, identifiable subgroups because

each subgroup can be evaluated sequentially (vs. a single large group, which must be rapidly counted within the short time window that the aircraft is overhead). During the 1989 surveys, viewing conditions were rated as excellent. The whales were not actually in a single large group; for example, the group of 460 seen on 23 June was reported to consist of two large subgroups and six to eight small satellite groups.

The protocol on the survey was to make multiple passes over the sighting until the observers were confident they had counted all subgroups and were sure there were at least that many whales there. They made 14 passes in the airplane over the sighting where they counted a minimum of 470 whales. The total minimum count was a sum of the counts of subgroups spread over a considerable area. Two of the observers (Leatherwood and Reeves) were exceptionally experienced and can likely be trusted to have given an accurate conservative minimum count, and to correctly have identified the species. Further confirmation of species identification comes from the fact that on the survey the observers identified sightings of other species that could possibly be confused with false killer whales. Although it is known that group size estimates of dolphins of large groups in the hundreds can be imprecise and potentially biased, group size estimation from an airplane is different in nature. Given the relatively clear water conditions, it would be possible to see whales underwater at shallow depths during a relatively fast pass from an airplane (i.e., the main problem with estimating a group from a ship is not being able to determine when a subgroup already seen surfaces for a second time), so the minimum count is likely of to be highly reliabilityreliable.

As in the Mobley surveys, given the count occurred from an aerial survey means that there was no information such as genetics or photographs that could link these sightings into the HI population. However, the encounters were relatively close to shore, well within the HI defined stock area, and closer to shore than any pelagic False false killer whales have been seen from boat surveys. Therefore 470 was used as a minimum population size for the HI population in 1989 by using this as the lower bound of a prior distribution for abundance in that year (as this was the year when the model started began projecting from).

Environmental variance

The model incorporates environmental variance. The value of this parameter was estimated from northern Northern Resident killer whale census data (Olesiuk et al., 2005), as this is a similar species at a similar population size. The change in population size for each year was calculated, and the standard deviation of those changes in abundance was calculated to be 0.029. That value would include both environmental and demographic variation in the population from year to year. To approximate the amount of demographic variability that would have occurred, the population model was run for a population the size of the Northern Resident killer whale population with no stochastic elements other than demographic variability. The standard deviation of the annual growth rate was found to be approximately 0.010. This was subtracted from the observed standard deviation of 0.029 to give an estimate of 0.019 for environmental variance in Northern Resident killer whales. Therefore, a value of 0.020 was used for environmental variance.

Northern Resident killer whales are a more temperate population and may have more variability than found in tropical regimes, so it is possible that Hawaii insular false killer whales have a

lower amount of environmental variability. Therefore, a trial was run using a lower rate of 0.010. Information from Southern Resident and Northern Resident killer whales indicates a high degree of auto-correlation in population growth rates through time. Therefore, a value of 0.7 was specified for the auto-correlation parameter, as this value was found to mimic ~ 5–-9 year cycles in good and bad population rates of change. A trial with a much lower value of 0.2 was also specified.

Allee effects

Studying the demography of very small populations is difficult, but several studies have indicated that Allee effects do occur in a variety of species (Groom, 1998; Kuussaari et al., 1998; Courchamp et al., 1999), though one meta-analysis study found little evidence for depensation in fish species (Myers et al., 1995). Fowler and Baker (1991) concluded that depensation was likely to be a common phenomenon in animal population dynamics, especially in populations smaller than 10% of their original, pre-exploitation populations (0.1 K). Unfortunately, few empirical data are available about possible Allee effects in marine mammals (Fowler and Baker, 1991, ; Wade, 2009). There are many mechanisms (described above) that can potentially cause Allee effects in a population such as Hawaiian insular false killer whales. Ignoring the possibility of Allee effects could therefore underestimate the true extinction risk of the population. Two versions of the Allee model were used. In the first θ =10 and α = – -0.08, and in the second θ =25 and α=-0.02 (Fig. 1). The first was designed to simulate an Allee effect on both reproduction and survival, and the second was designed to simulate an Allee effect on just reproduction.

Mortality events ('catastrophes')

Mortality events (often termed 'catastrophes' in the conservation biology literature) were added to examine the potential influence of extreme mortality events. False killer whales worldwide are known to occasionally mass strand, for unknown reasons. However, no mass strandings are known from Hawaii. Even though not observed in Hawaii, mass strandings are well known for the species, and therefore the potential effect of mortality events on extinction risk should not be ignored. If mass strandings do occur, ignoring mortality events in the model would under-estimate extinction risk. A mortality event annual probability of 0.02 was specified with a magnitude of 0.10. This represents a 1 in 50-years chance of a 10% decline in the population. A second trial was specified with a 1 in 100-year chance of a 10% decline in the population. A third trial was specified with a more frequent but small in magnitude effect, with a 1 in 10-year chance of a 2% decline in the population.

To put these magnitudes in context, over the 4 years following the *Exxon Valdez* oil spill, the AB pod of killer whales from Prince William Sound lost 40% of its members. The PVA model did not use a magnitude this high (40%), because the AB pod is only one of many pods that belong to that killer whale population.

<u>Prior distributions</u>

For the estimated parameters of the model, uniform ("non-informative") prior distributions were specified as much as possible. The prior distribution for the starting population size in 1989 was a uniform distribution ranging from 470 to 1392. The lower bound of 470 was (as discussed above) from Reeves et al. (2009). The upper bound was calculated from a presumed pristine state based on the maximum observed density of false killer whales extrapolated to the stock area for

the Hawaiian insular population. An upper bound of 769 was specified in one trial to examine the effect of that assumption.

Prior distributions were also established for the rate-of-change parameters. For the one-rate model this was relatively easy to do, as it was relatively easy to specify a broad uniform distribution for the rate-of-change, and the posterior distribution was precise enough it did not range into implausibly high positive values because the distribution was entirely negative; for example, in the base case the 95% probability interval ranges from – -0.047 to – -0.123.

The prior distribution that is specified for the second rate of change in the two-rate model influences the results because the data are too limited to estimate two rates of change well. The data are sufficient to give a reasonably informative estimate of the population rate of change in the one-rate model. However, the data are insufficient to give precise estimates of the two rates of change in the two-rate model. For the second rate of change, the posterior distribution is broad and ranges from large negative to large positive numbers. For example, if broad prior distributions for the rates are specified, in the base-case the 95% probability intervals become – -0.270 to –-0.073 for the first rate and – -0.094 to 0.20 for the second rate. Though the posterior distribution for the first rate is broad, it is entirely negative and therefore the prior distribution does not cause any problems. However, for the second rate the data cannot rule out either a large decline or large increase. Specifying a broad prior distribution is an objective way of not putting any prior knowledge into the analysis, but it allows for biologically unreasonable rates of increase to occur, as an upper range to what is biologically reasonable for false killer whales, given their life history, is likely in the range of 0.03 to 0.04. Allowing a rate of increase as high as 0.20 is clearly unreasonable.

Limiting the upper range to biologically reasonable rates makes sense, but; however, if the prior distribution is not symmetric around 0.0, the expectation (mean) of the prior distribution is negative; this is equivalent to specifying *a priori* that the overall expectation is that the population is declining. Therefore, the effect of two different prior distributions for the rates of change were was examined. The first used a prior distribution for the second rate of change from – 0.04 to 0.04. This is set on the upper end by what is biologically realistic, and the lower end is set to make the distribution symmetric. A second prior distribution for the second rate of change was set from – 0.10 to 0.04. This was set on the upper end by what is biologically realistic, and the lower end is set by roughly encompassing the point estimate of decline for the one-rate model.

The year at which the second rate of change occurred was set by requiring that there be two estimates of abundance or relative abundance between the change-point year and either end of the abundance time series. This resulted in a prior distribution for the year of change that was uniform from 1993 (year 4) to 2006 (year 17).

Estimation of parameters

Parameter uncertainty was explicitly incorporated by fitting the model to available data on trends and abundance of the Hawaiian false killer whale population. Bayesian statistical methodology was used to estimate the parameters of the model. The Baird et al. (2009) absolute abundance

estimates were assumed to have a log-normal distribution, so the likelihood function for those estimates was the standard log-normal likelihood.

$$– \text{-logLik} = \sum\{ [\ln(N_t^{obs}) - \ln(N_t^{model})]^2/cv_t\}$$

where:

N_t^{model} is the model population size in year t,

N_t^{obs} is the abundance estimate in year t,

cv_t is the coeffecient of variation of the abundance estimate in year t,

t is either 2004 or 2009.

The Mobley aerial survey data were sparse for observations of false killer whales in Hawaii, with 5 surveys that yielded counts of 9, 8, 1, 0, and 0 groups in years 1993, 1995, 1998, 2000, and 2003. The zero observations cause problems in fitting to those data under an assumption of log-normal error structure. Therefore, a poisson likelihood function was used, as follows:

$$\text{-logLik} = \sum [q\, N_y^{model}\, E_y - n_y \ln(q\, N_y^{model}\, E_y)]$$

where:

q is the detectability coeffecient,

E_t is the effort in year t,

n_t is the number of observations from the aerial survey in year t.

and q is defined in this way:

$$q = \sum n_t \;/\; [\sum N_t\, E_t].$$

This derives from the formula for a Poisson process with expectation μ, where the probability of n occurrences is $[\mu^n/n!]*\exp(-\mu)$, after taking the natural log of each side and dropping the constant. The estimate for q is derived from setting the partial derivatives of the negative log-likelihood with respect to q to zero.

The Sampling-Importance-Sampling routine (Rubin, 1988; Smith and Gelfand, 1992; Wade, 2002b) was used to generate a random sample of 10,000 sets of parameter values from the posterior distribution, which are then used to approximate the posterior distribution. These 10,000 sets of parameter values were then used to project simulated populations forward for 125 years, each time keeping track of the fate of the projected population. The number of times a simulated population reaching the 'near-extinction' level, defined to be less than 20 whales, was tabulated after 50, 75, and 125 years.

<u>Sensitivity trials</u>

A number of sensitivity trials were run changing values of some of the key input data and parameters, for the purpose of having a better understanding of the behavior of the PVA model. None of these models were given a high likelihood by the Biological Review Team compared to the Base model and its variations that are described above. The sensitivity trials examined the influence of the aerial trend data, the prior distribution on 1989 abundance, the Allee effect, and mortality events ('catastrophes'). All trials were identical to the Base model except for changing one component. Details are in the Results section.

RESULTS

The one-rate models generally provide reasonable fits to the data (Fig. 2). The trend data suggested a steeper rate of decline than allowed by the prior distribution on abundance in 1989, as the first two aerial survey estimates (1993 and 1995) tend to be above the trajectories, and the last three aerial survey estimates (1998, 2000, 2003) tend to be below the trajectories. Model 2 (with a lower upper limit for the prior distribution for 1989 abundance) provides a poorer fit to the data, and it is limited to a more moderate rate of decline by the prior distribution. The posterior distributions for the initial population size in 1989 favors higher population sizes (Fig. 3). The posterior distributions for the rate of change are entirely negative, with a point estimate of an ~ 9% decline, with 95% probability intervals ranging from ~ – 5% to – 12.5%. The probability of near-extinction is greater than 0.75 for all models at 50 years, and increases to nearly 1.00 for all models at 75 years and beyond (Fig. 4).

The trajectories from the two-rate models with a prior distribution for the second rate of change of – 0.04 to 0.04 show a greater range of uncertainty in the future projection of the population than do the one-rate models (Fig. 5). In general, the first rate of change is estimated to be similar to the value estimated for the one-rate models (Fig. 6). In contrast, the second rate of change is estimated imprecisely, and mostly reflects the prior distribution specified for that parameter, indicating that the parameter estimate was not informed much by data. The change-point year was estimated to be approximately years 10–12 (1999–-2001). Even though the posterior distribution for the second rate of change includes positive values (so some trajectories increase and do not go to near-extinction), a large percentage of the trajectories decline and go to near-extinction (Fig. 5). The probability of near-extinction is between only 1–-3% for all models at 50 years, but increases to ~ 0.20 for most models at 75 years, and increases to ~ 0.35 at 125 years (Fig. 7).

The two-rate models with a prior distribution for the second rate of change of – 0.10 to 0.04 are similar to the other two-rate model results, but with a higher probability of a decline (Figs. 8, 9). The probability of near-extinction is ~ 0.20 for most models at 50 years, and increases to ~ 0.40 for most models at 75 years, and increases to ~ 0.50 at 125 years (Fig. 10).

Sensitivity trials

Sensitivity 1: no aerial trend data

In this trial the aerial trend data were not used in the estimation of the model parameters. Therefore, the estimated trend of the population would be solely due to the prior distribution on

1989 abundance and the mark-recapture absolute abundance estimates from 2000–-2004 and 2006–2009.

This trial decreases the probability of near-extinction in 50 years from 0.91 to 0.80, because the population is estimated to be declining at a lower rate than in the Base model (Table 2). However, the trial still estimates a declining population, and by 75 years the probability of near-extinction is nearly the same as the Base model.

Sensitivity 2: higher upper bound on the prior distribution for 1989 abundance
This trial was run with a prior distribution on 1989 abundance that ranged from 470 to 3000 (rather than 470 to 1392 in the Base model), to examine the influence of the specified upper bound on the prior distribution.

This trial increases the probability of near-extinction (Table 2) because the higher upper bound on 1989 abundance allows the model to fit the aerial trend data more closely and therefore estimate a steeper rate of decline.

Sensitivity 3: broader prior distribution for 1989 abundance
This trial was run with a prior distribution on 1989 abundance that ranged from 50 to 3000 (rather than 470 to 1392 in the Base model). This provides a broad prior distribution for 1989 abundance that influences the results less than in the base model, so that the estimated rate of decline is due based primarily to the aerial trend data and the mark-recapture abundance estimates, rather than onto the prior distribution for 1989 abundance.

As in Trial 2, this trial increases the probability of near-extinction (Table 2) because the higher upper bound on 1989 abundance prior allows the model to fit the aerial trend data more closely and estimates a steeper rate of decline. The probability of near-extinction is slightly less than in trial 2, which is a result of a small increase of 1989 abundance values less than 470.

Sensitivity 4: no Allee effect
This trial was run with no Allee effect, so the underlying rate-of-change remained constant with population size.

This trial lowered the probability of near-extinction in 50 years from 0.91 in the Base model to 0.86 (Table 2). The probability of near-extinction at 75 and 125 years in is essentially identical to the Base model. This confirms that in this PVA application (where the overall trend is declining) the Allee effect essentially hastens the demise of populations that would go extinct anyway at a later time without Allee effects.

Sensitivity 5: no mortality events
This trial was run with no mortality events to examine the effect these events have on the probability of near-extinction.

The probability of near-extinction in this trial was essentially identical to the Base model (Table 2), indicating that the mortality events did not substantially increase the risk of near-extinction.

Sensitivity 6: Small Population effects
This trial was run to assess the potential fate of the population based solely on its being very small. No aerial survey data were used. The prior distribution for the 1989 abundance was very broad (20-3000 individuals). Population growth was set at zero ($r = 0$). Thus, the projection shows the effect of Allee effects, catastrophes and variance in population growth rate.

The probability of near extinction was much lower than for any other model: 0.001 in 50 years, 0.013 in 75 years and 0.174 in 125 years. This trial indicates that the probability of near-extinction is substantially lower based on the population being subject to small population effects alone.

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Table 1. Specifications of Models 1–-9. See the text for further information about the parameters.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9
Diff from Base	Base	769 prior for 1989	Low env variance	Birth only Allee	2-yr Auto-corr in Env	Ind enc rate trend	UME 1 in 100 yrs @10%	UME 1 in 10 yrs @2%	Without Kauai Mark-Recapture
1989 Prior upper bound	470, 1392	470, 769	470, 1392	470, 1392	470, 1392	470, 1392	470, 1392	470, 1392	470, 1392
Environmental variance σ_{env}	0.02	0.02	0.01	0.02	0.02	0.02	0.02	0.02	0.02
Auto-correlation ρ	0.7	0.7	0.7	0.7	0.2	0.7	0.7	0.7	0.7
Allee θ	10	10	10	25	10	10	10	10	10
Allee α	-0.08	-0.08	-0.08	-0.02	-0.08	-0.08	-0.08	-0.08	-0.08
Aerial survey group or ind	Group	Group	Group	Group	Group	Ind	Group	Group	Group
Mortality event annual probability p_{me}	0.02	0.02	0.02	0.02	0.02	0.02	0.01	0.10	0.02
Magnitude of mortality event m_{me}	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.02	0.10
Mortality event sd_{me}	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.005	0.03
2009 Abundance	170	170	170	170	170	170	170	170	151

Table 2. Probability of near-extinction (< 20 individuals) in 50, 75, and 125 years, for six sensitivity trials, compared to values for the Base model, using the one-rate-of-change model: (1) the aerial trend data were not used, (2) a higher upper bound on the prior distribution for 1989 abundance (3000), (3) a broader prior distribution for 1989 abundance (50 to 3000), (4) no Allee effect was used, (5) no mortality events ('catastrophes') occurred, and (6) small population effects.

Model	50 years	75 years	125 years
Base model	0.91	0.99	1.00
Sensitivity 1	0.80	0.98	1.00
Sensitivity 2	0.97	1.00	1.00
Sensitivity 3	0.95	0.99	1.00
Sensitivity 4	0.86	0.99	1.00
Sensitivity 5	0.92	1.00	1.00
Sensitivity 6	0.00	0.01	0.17

Figure 1. The form of the two Allee models used in the simulations. In the first θ =10 and α =-= – 0.08, and in the second θ = 25 and α =-= – 0.02.

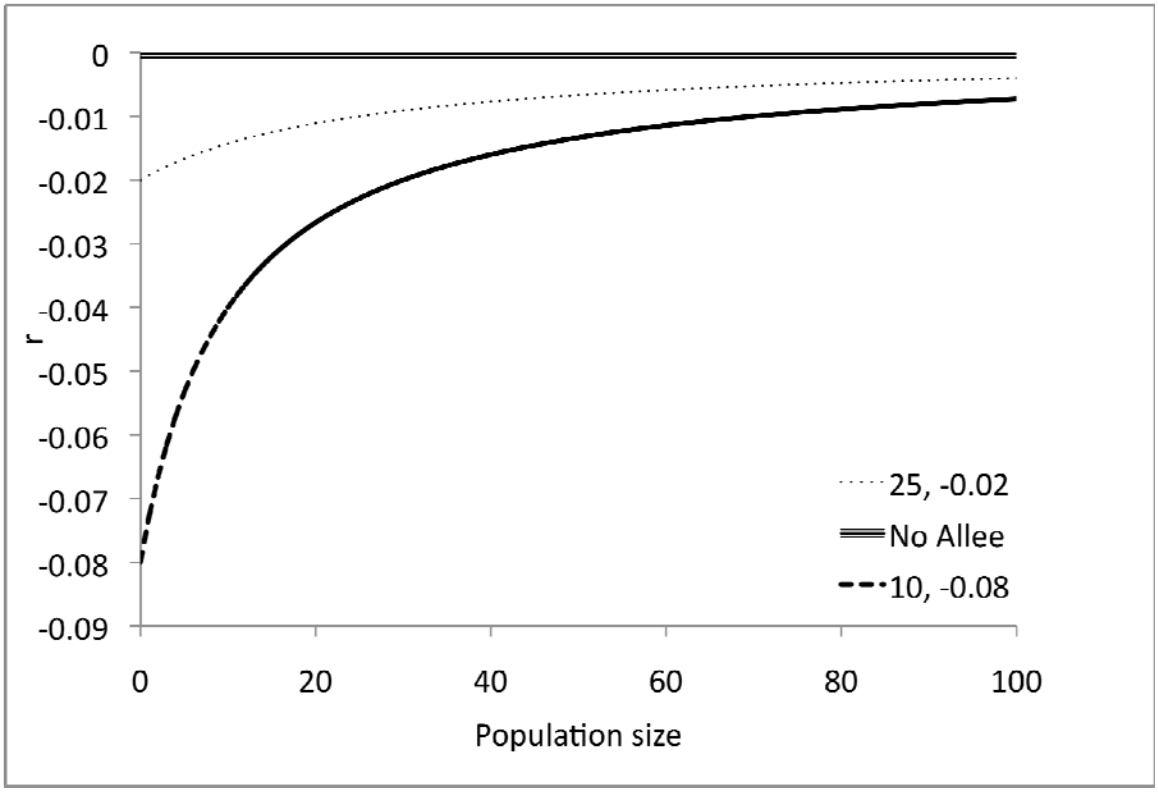


Figure 2. Trajectories for the one-rate models. Models 1–9 as defined in Table 1. The absolute abundance estimates (from Baird et al., 2009) are shown as open circles. The scaled trend estimates (from Mobley et al., 2000; Mobley, 2004) are shown as filled circles. The median trajectory is the thick black middle line. The thick outer lines envelop 95% of the trajectories. Ten randomly chosen trajectories are shown in color to illustrate the representative variability. Three plots are shown for each model to show different aspects of the trajectories: (A) the left panel shows the fit to the abundance and trend data, (B) the middle panel shows the entire trajectory, and (C) the right panel shows a finer resolution of the main period where trajectories go extinct, if they do.

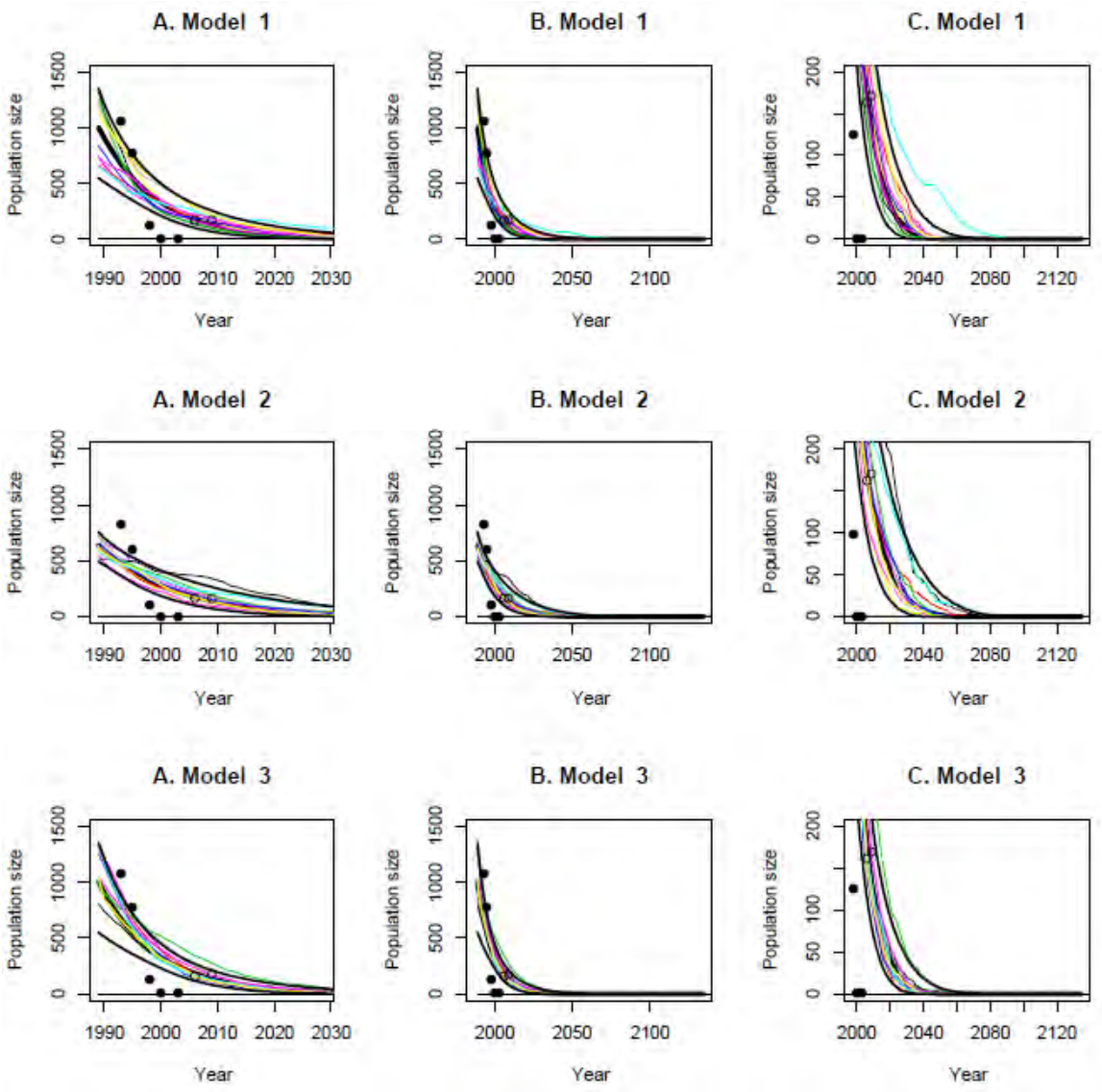
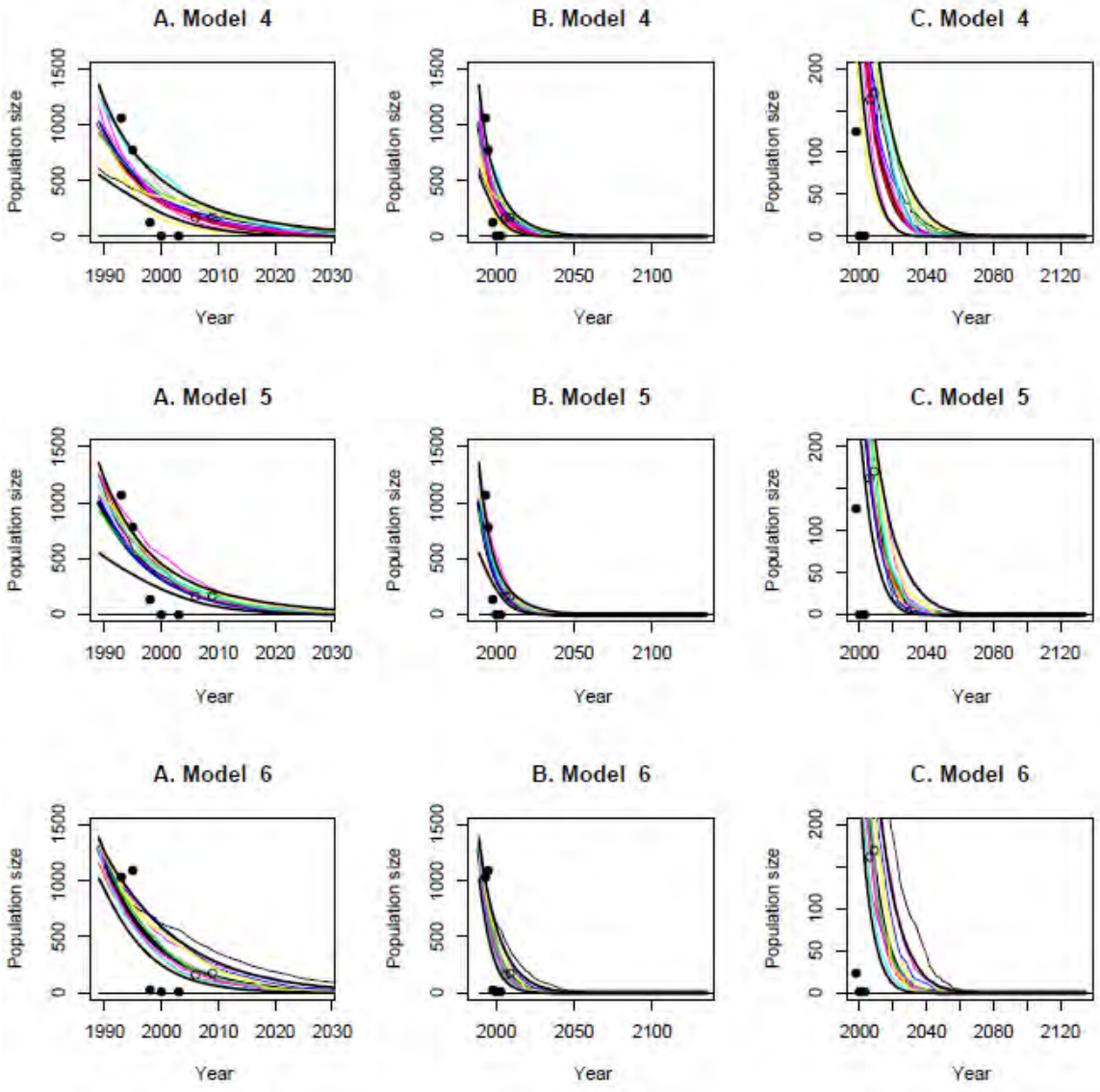


Figure 2 (Continued)



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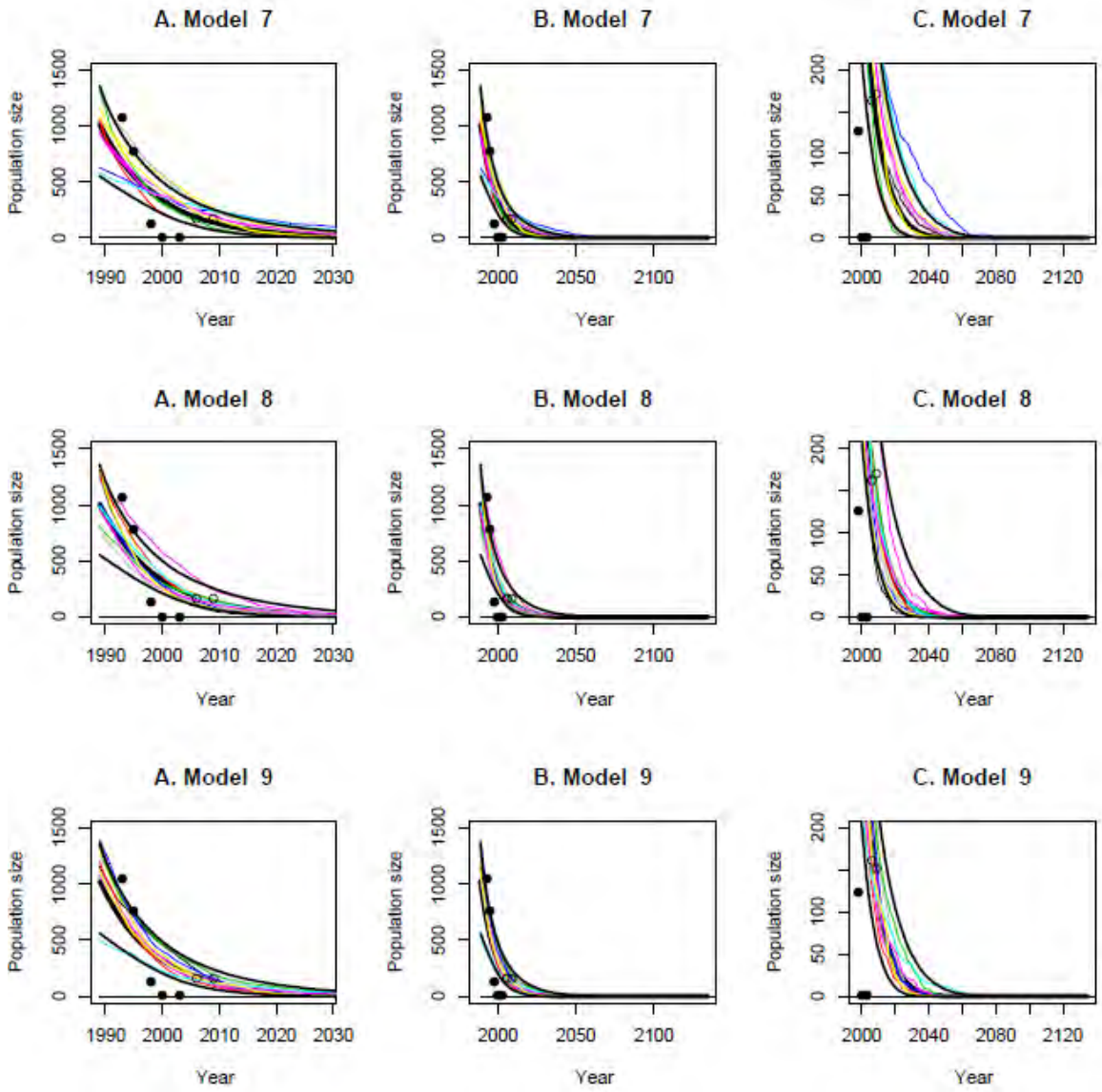


Figure 3. Prior (straight line) and posterior (curved line) distributions for the two parameters of the one-rate models. N_init is the intital population size in 1989. rl is the rate of change in the population. Models 1–9 as defined in Table 1.

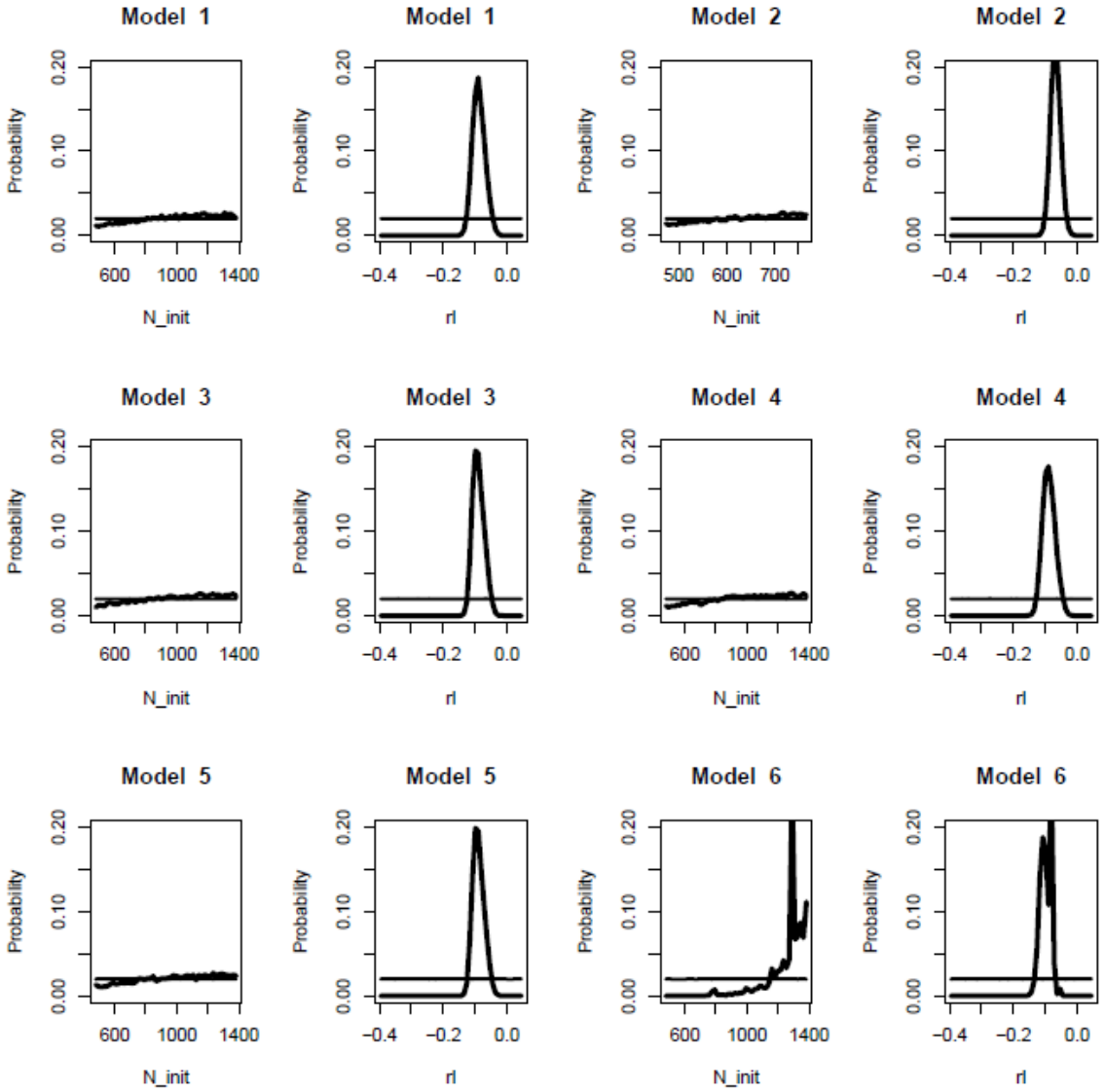


Figure 3 (Continued)

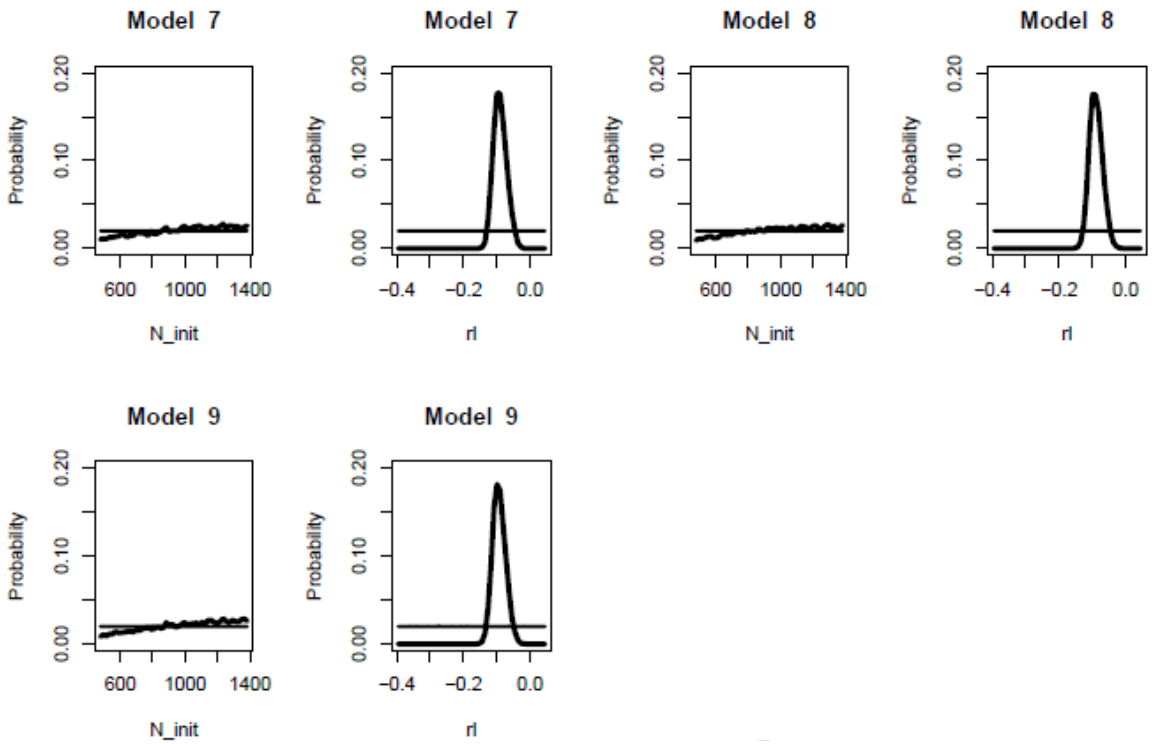


Figure 4. Probability of near extinction (< 20 individuals) at 50, 75, and 125 years for the ten 10 different one-rate-of-change models. Models 1–9 as defined in Table 1.

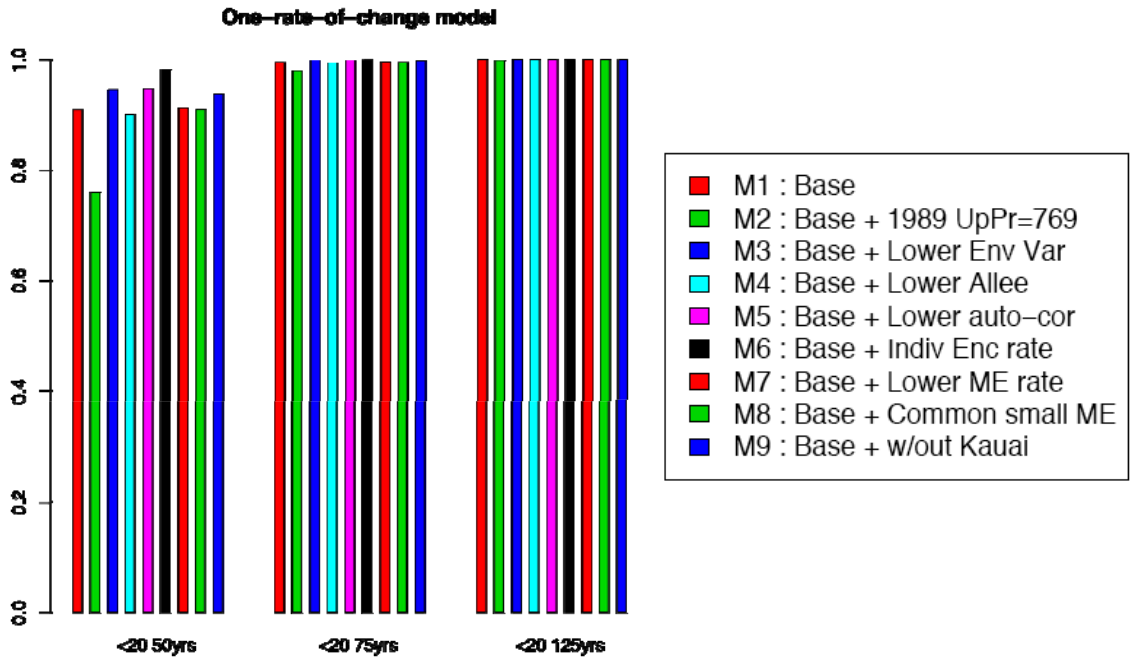
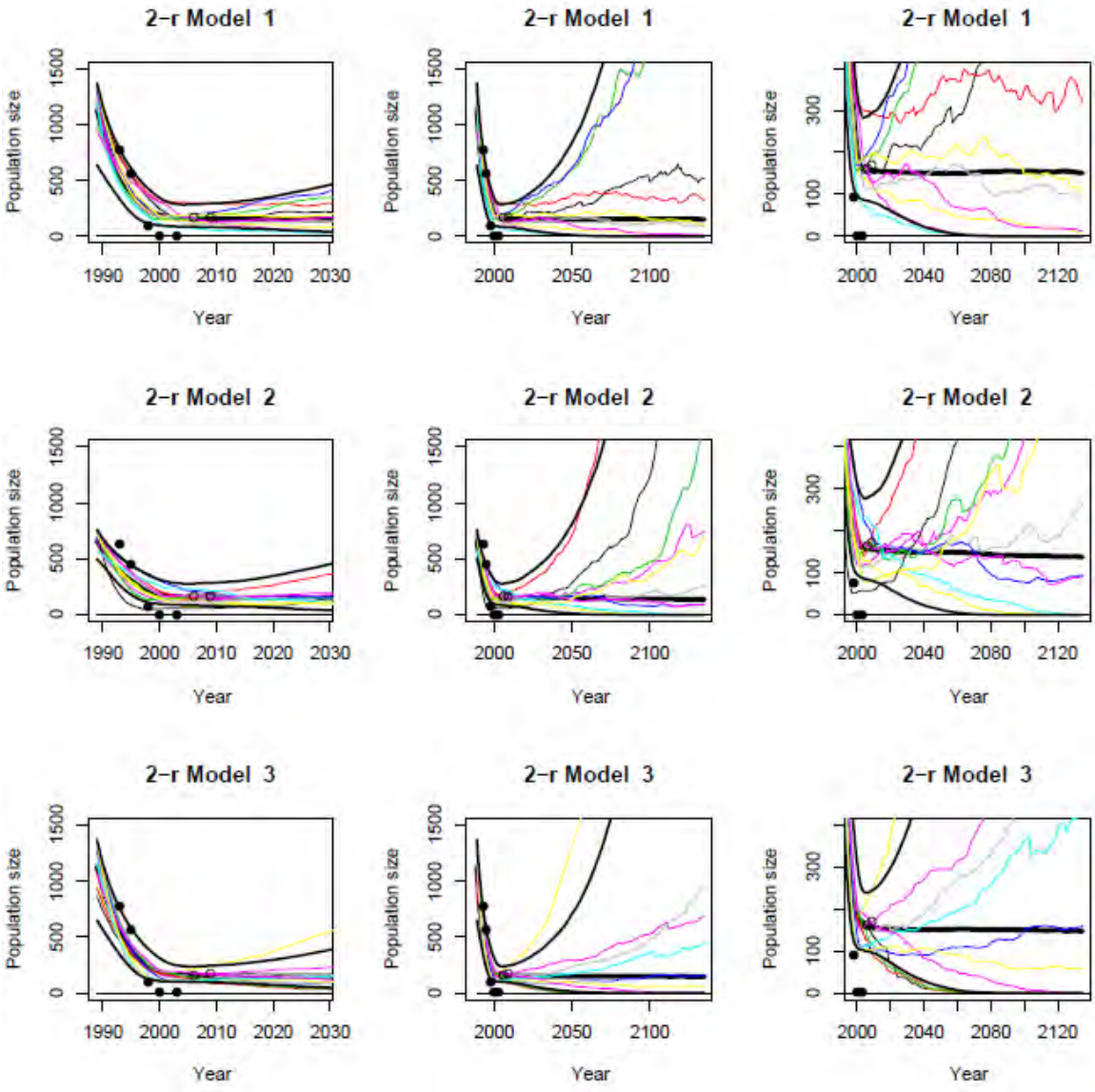


Figure 5. Trajectories for the two-rate ("2-r Model") with a prior distribution for the second rate of change of – 0.04 to 0.04. Models 1–9 as defined in Table 1. The absolute abundance estimates (from Baird et al., 2009) are shown as open circles. The scaled trend estimates (from Mobley et al., 2000; Mobley, 2004) are shown as filled circles. The median trajectory is the thick black middle line. The thick outer lines envelop 95% of the trajectories. Ten randomly chosen trajectories are shown in color to illustrate the representative variability. Three plots are shown for each model to show different aspects of the trajectories: the left panel shows the fit to the abundance and trend data, the middle panel shows the entire trajectory, and the right panel shows a finer resolution of the main period where trajectories go extinct, if they do.



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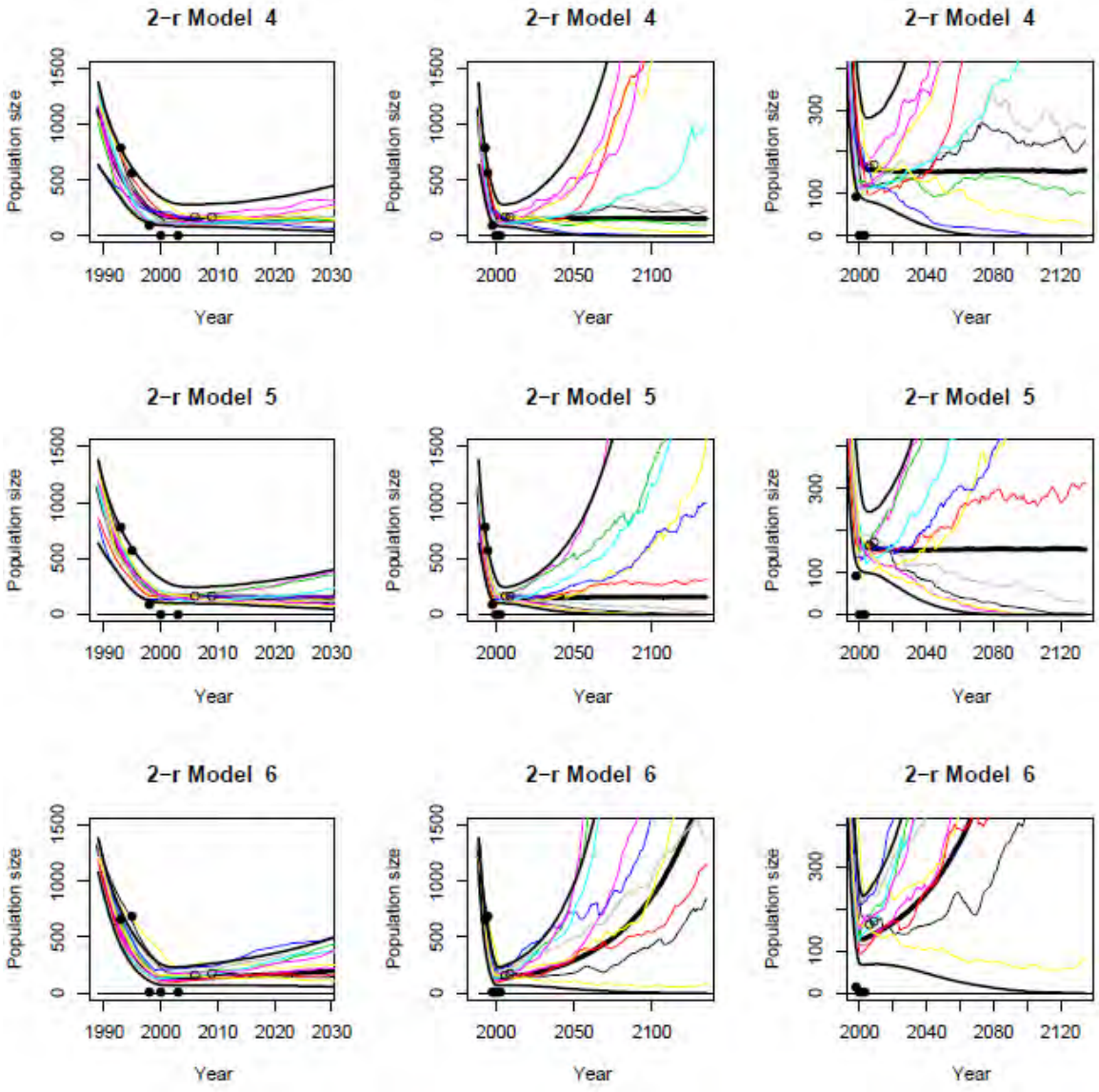


Figure 5 (Continued)

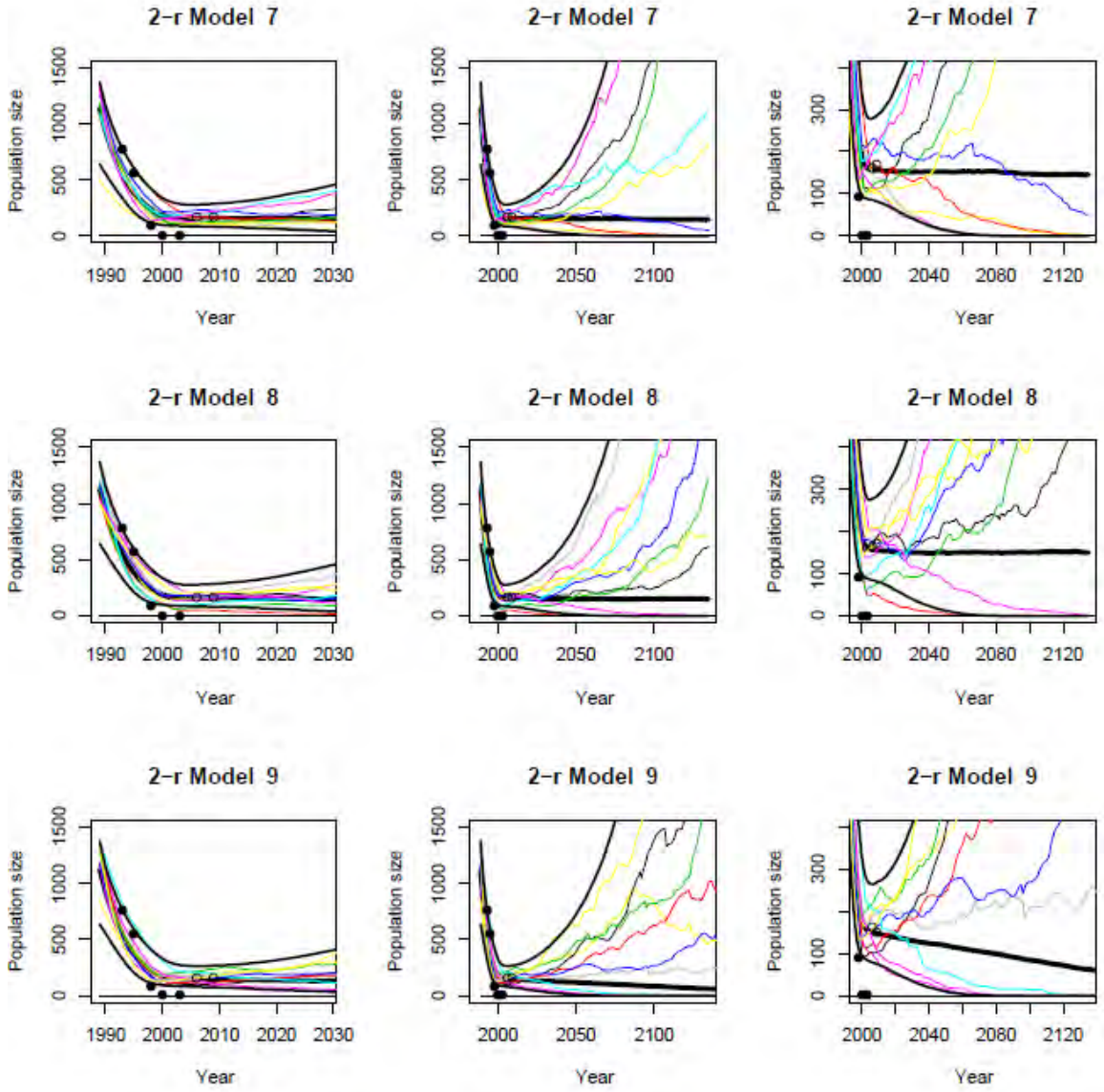
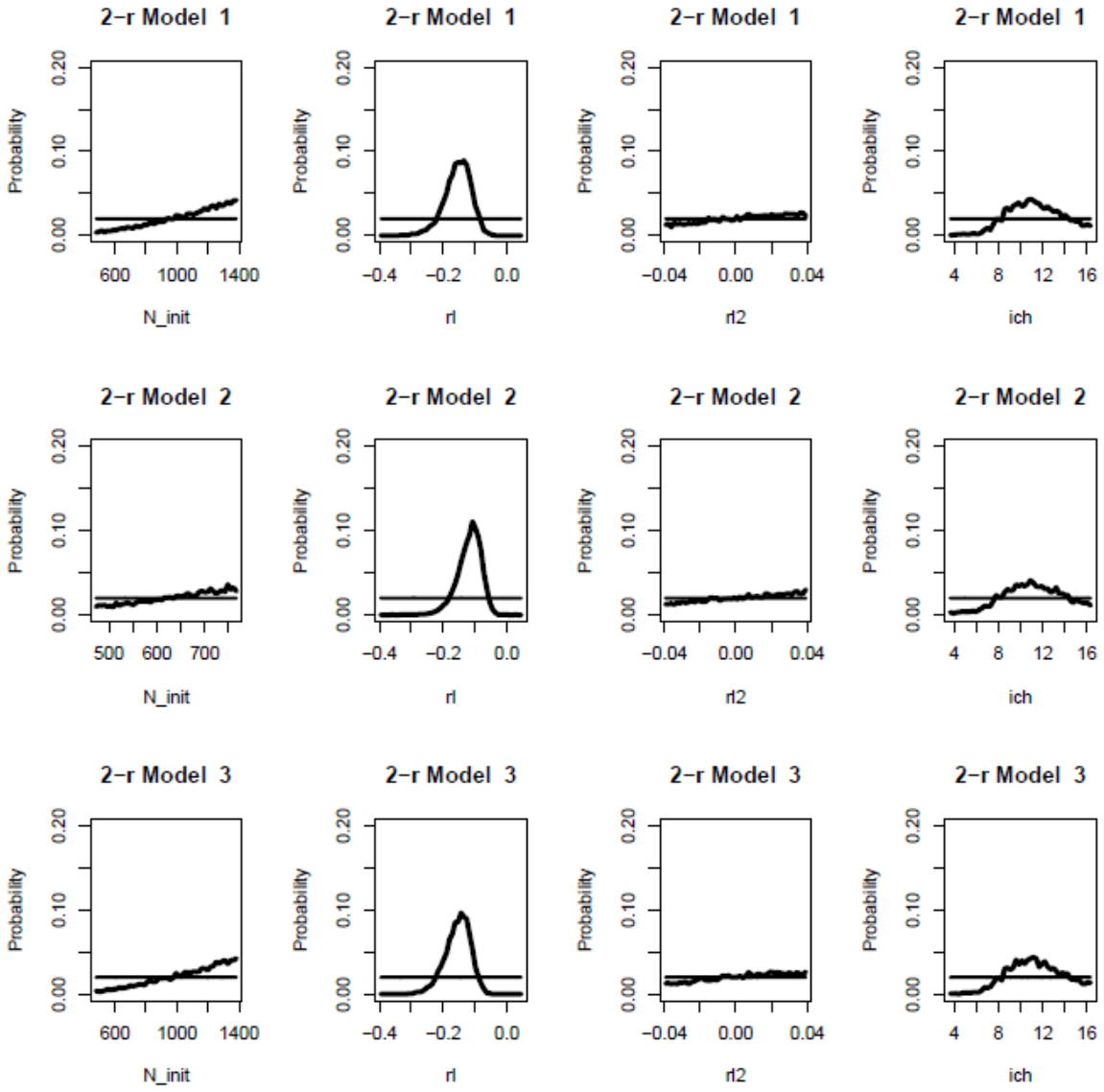


Figure 6. Prior (straight line) and posterior (curved line) distributions for the four parameters of the two-rate models with – 0.04 to 0.04 prior distribution for the second rate of change. N_init is the intital population size in 1989. rl is the first rate of change in the population, rl2 is the second rate of change in the population, and ich is the year of change to rl2, where year 1 is 1989. Models 1–9 as defined in Table 1.



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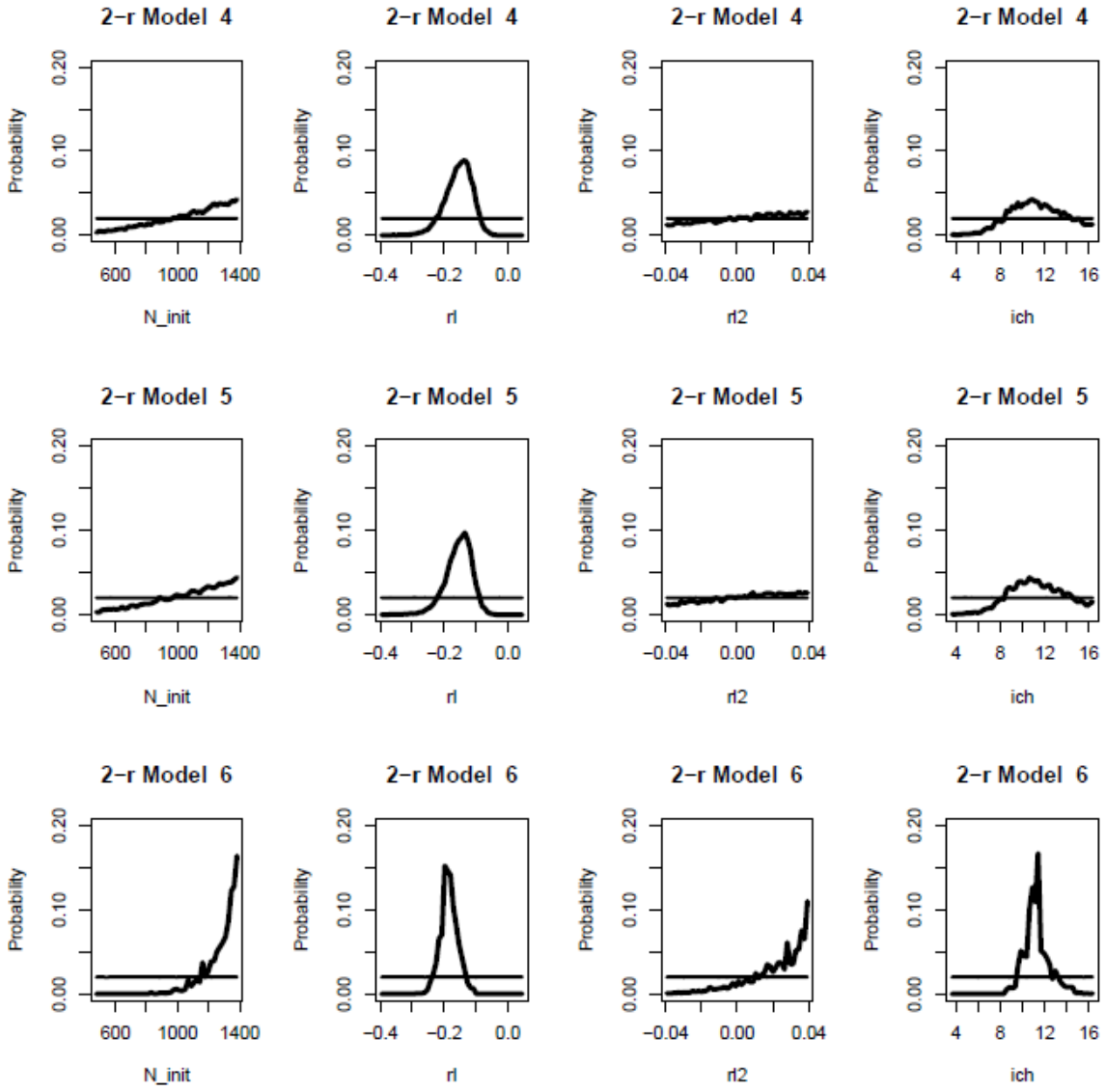

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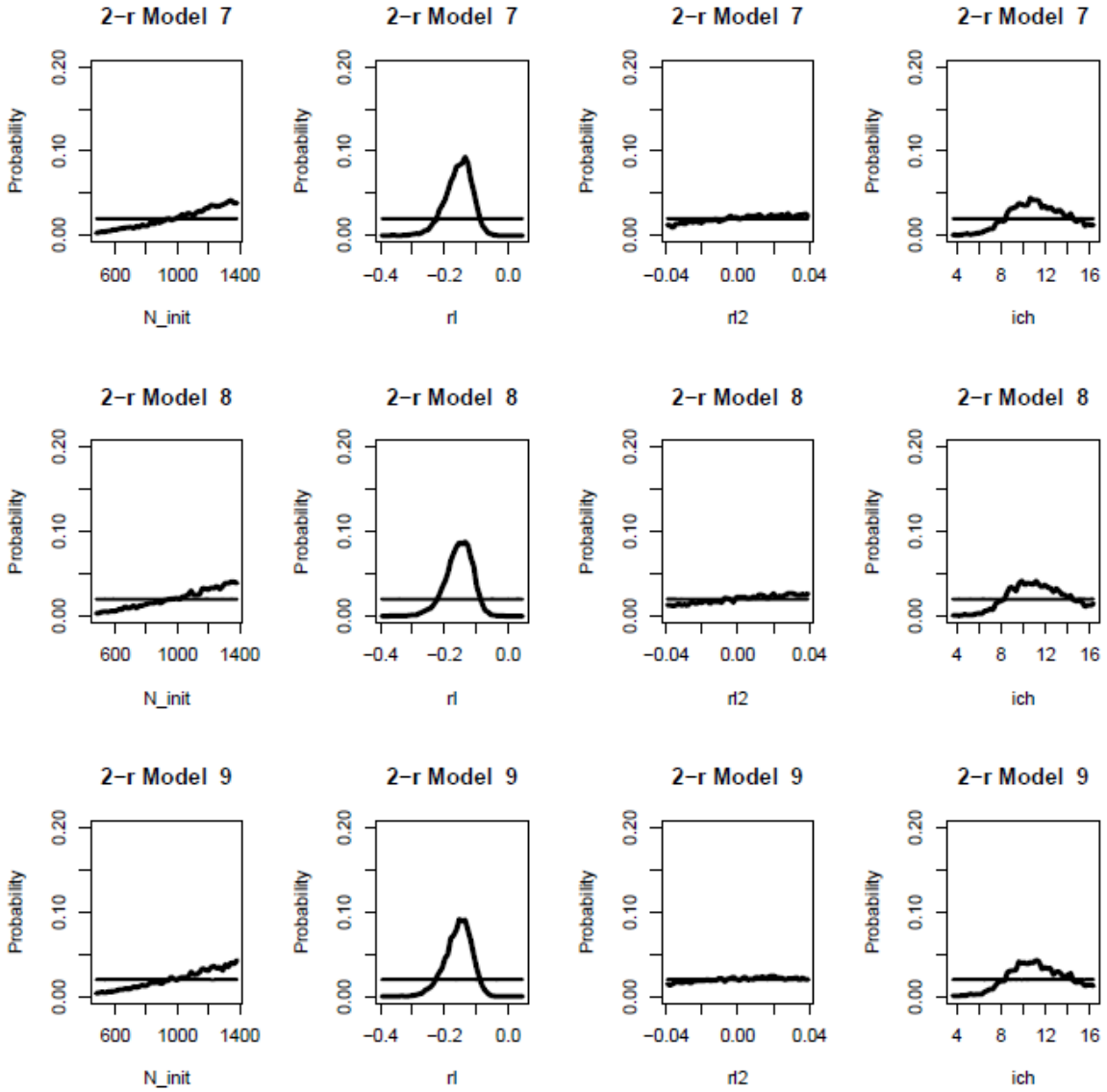


Figure 7. Probability of near extinction (< 20 individuals) at 50, 75, and 125 years for the 10 different two-rate-of-change models with prior distribution for the second rate of change of – 0.04 to 0.04. Models 1–9 as defined in Table 1.

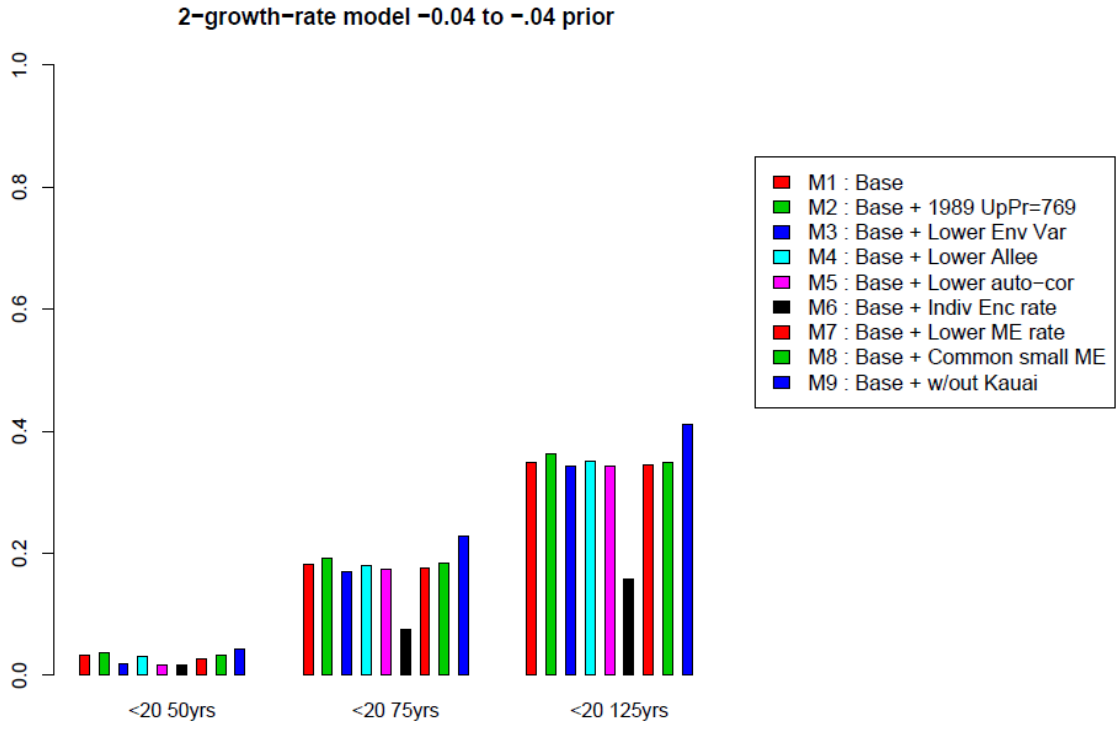


Figure 8. Trajectories for the two-rate models with a prior distribution for the second rate of change of – 0.10 to 0.04. Models 1–9 as defined in Table 1. The absolute abundance estimates (from Baird et al., 2009) are shown as open circles. The scaled trend estimates (from Mobley et al., 2000; Mobley, 2004) are shown as filled circles. The median trajectory is the thick, black middle line. The thick outer lines envelop 95% of the trajectories. Ten randomly chosen trajectories are shown in color to illustrate the representative variability. Three plots are shown for each model to show different aspects of the trajectories: the left panel shows the fit to the abundance and trend data, the middle panel shows the entire trajectory, and the right panel shows a finer resolution of the main period where trajectories go extinct, if they do.

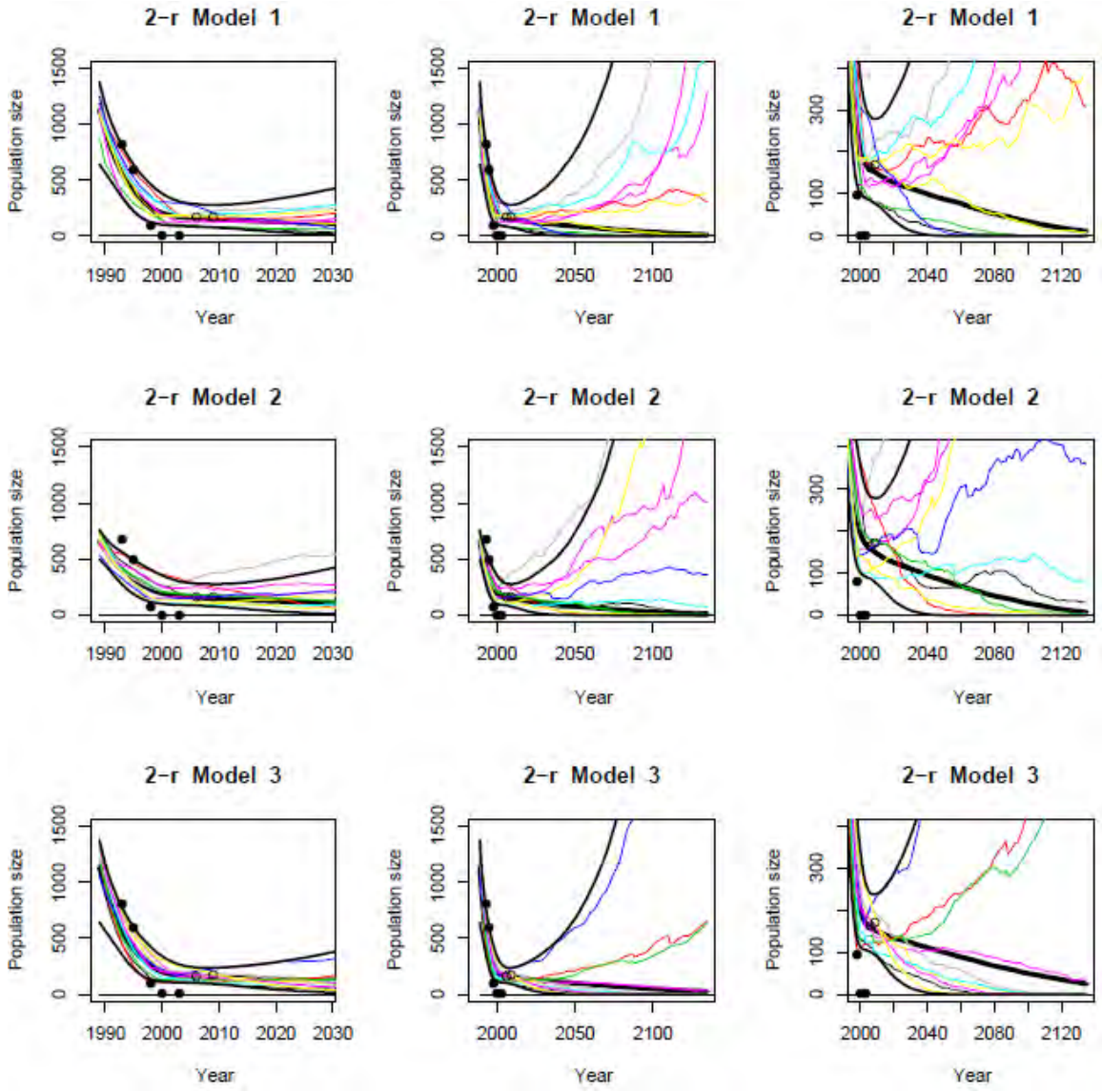
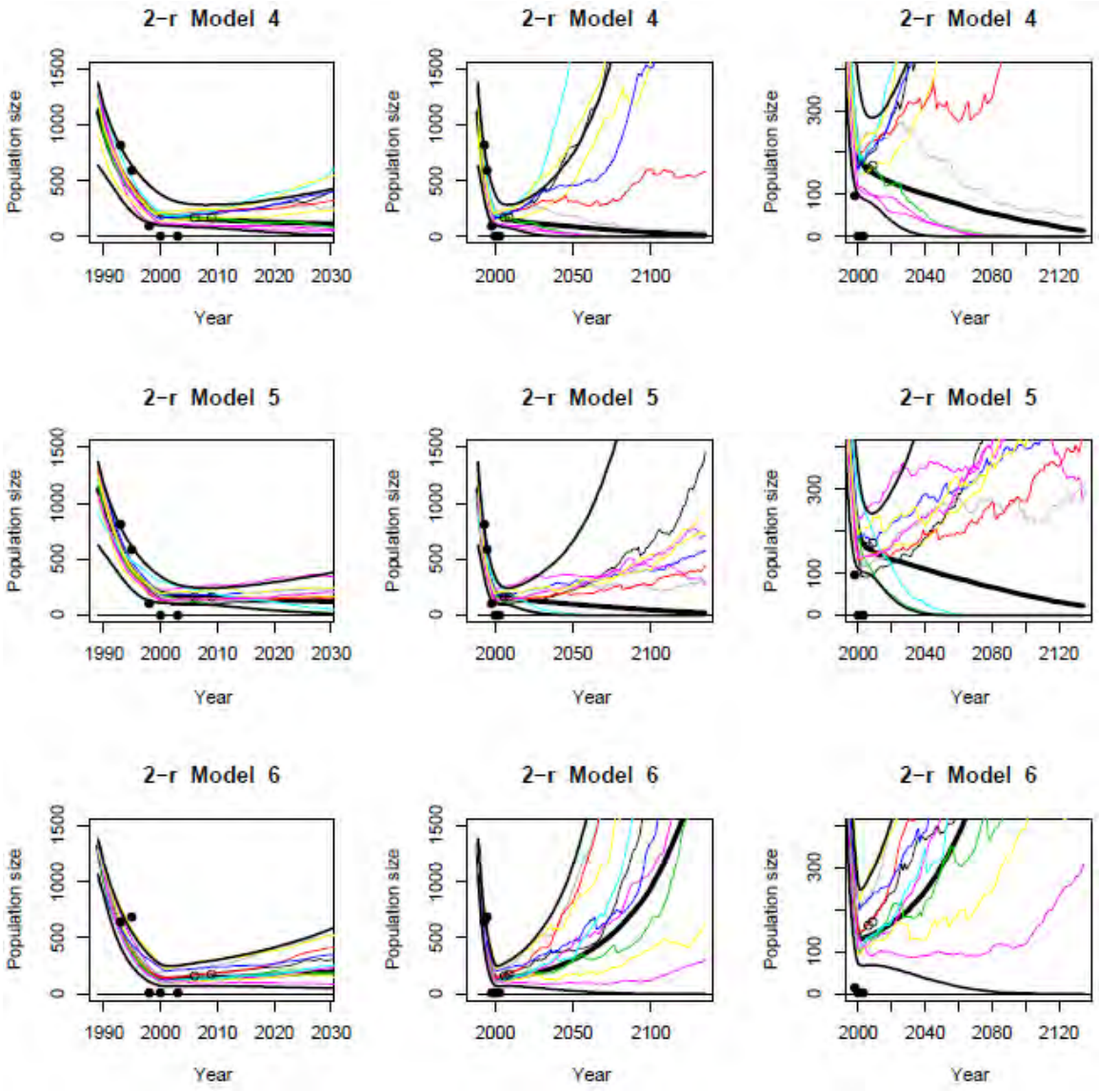


Figure 8 (Continued)



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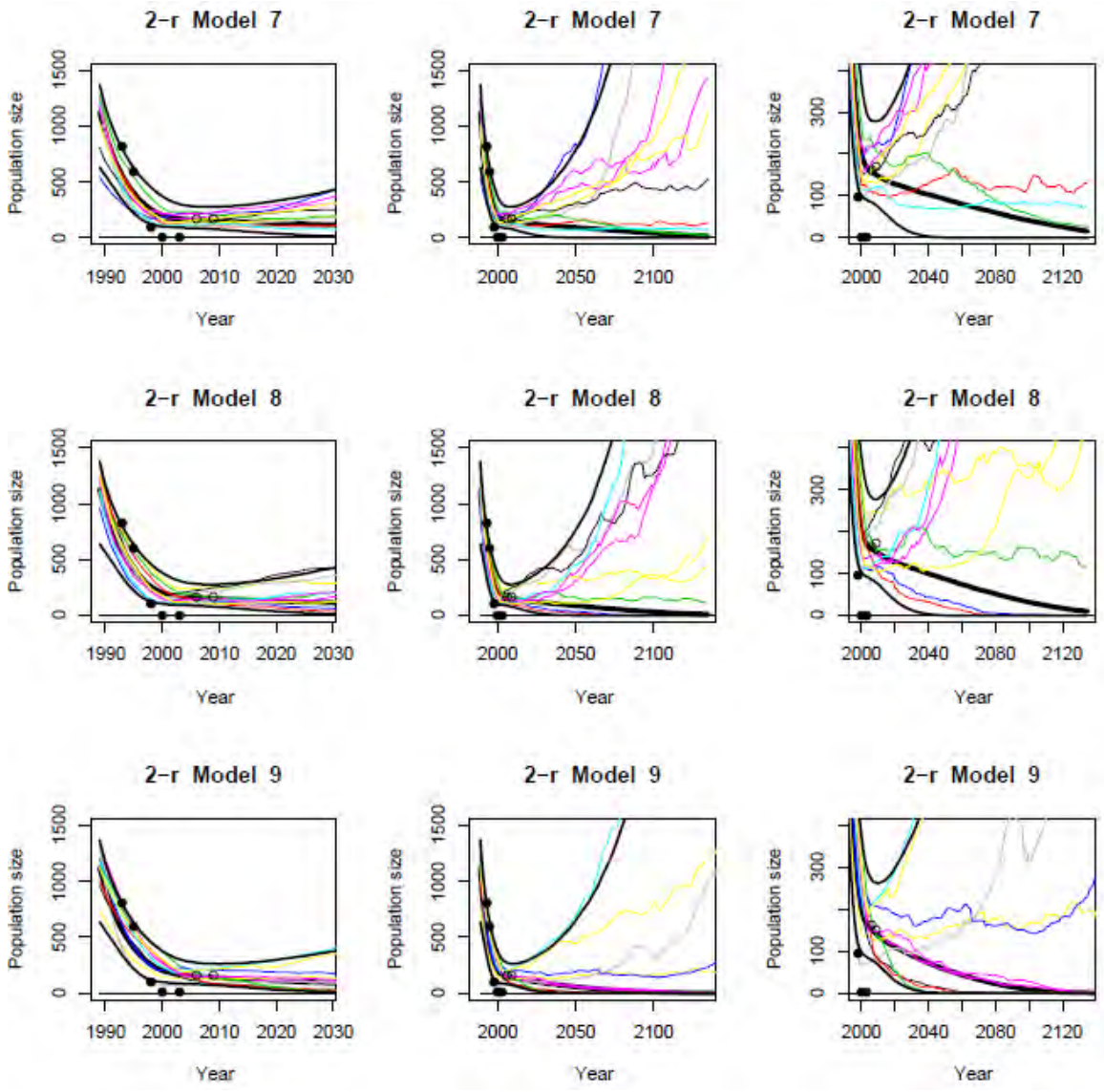
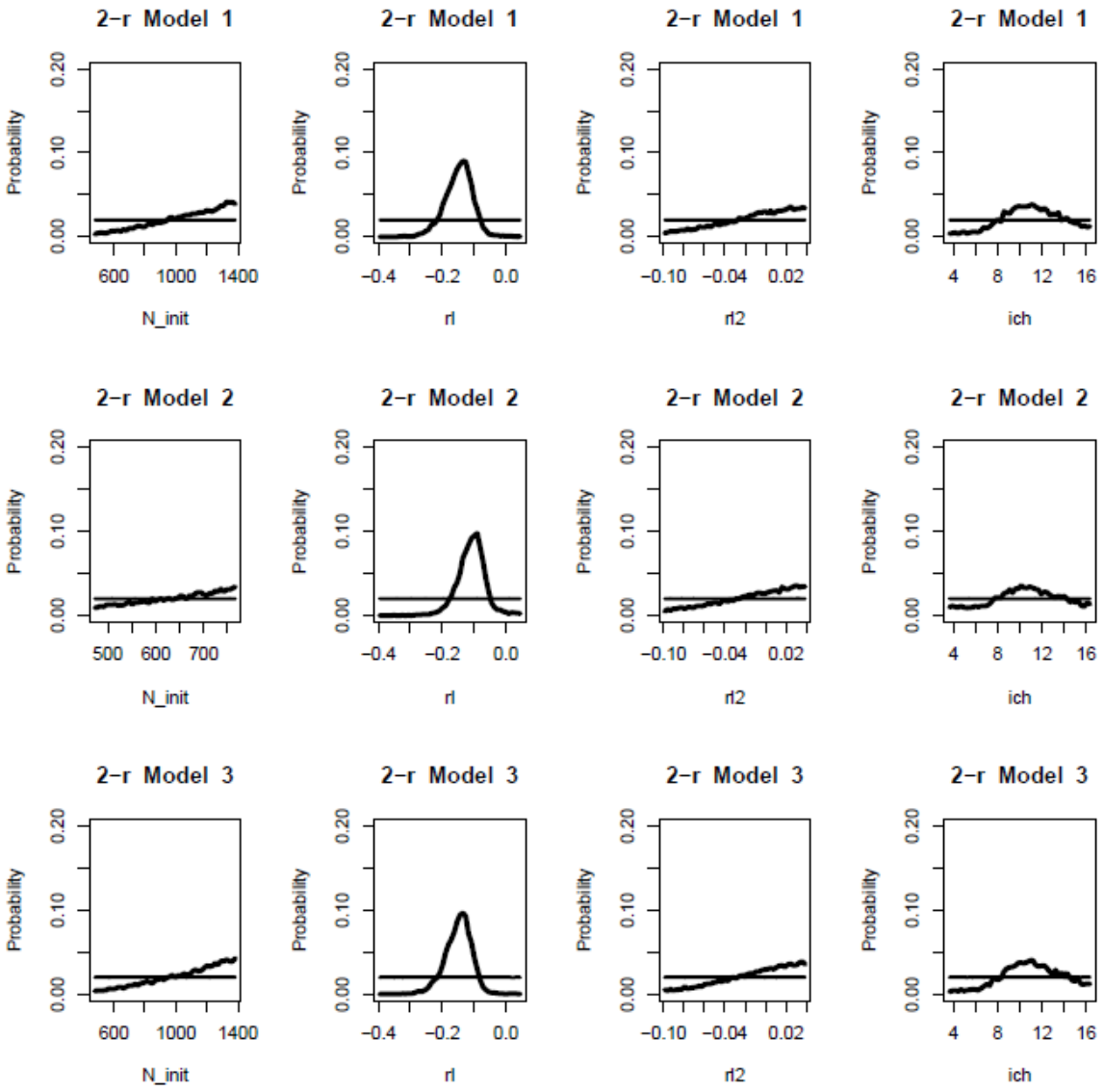


Figure 9. Prior (straight line) and posterior (curved line) distributions for the four parameters of the two-rate models with – 0.10 to 0.04 prior distribution for the second rate of change. N_init is the initial population size in 1989. rl is the first rate of change in the population, rl2 is the second rate of change in the population, and ich is the year of change to rl2, where year 1 is 1989. Models 1–9 as defined in Table 1.



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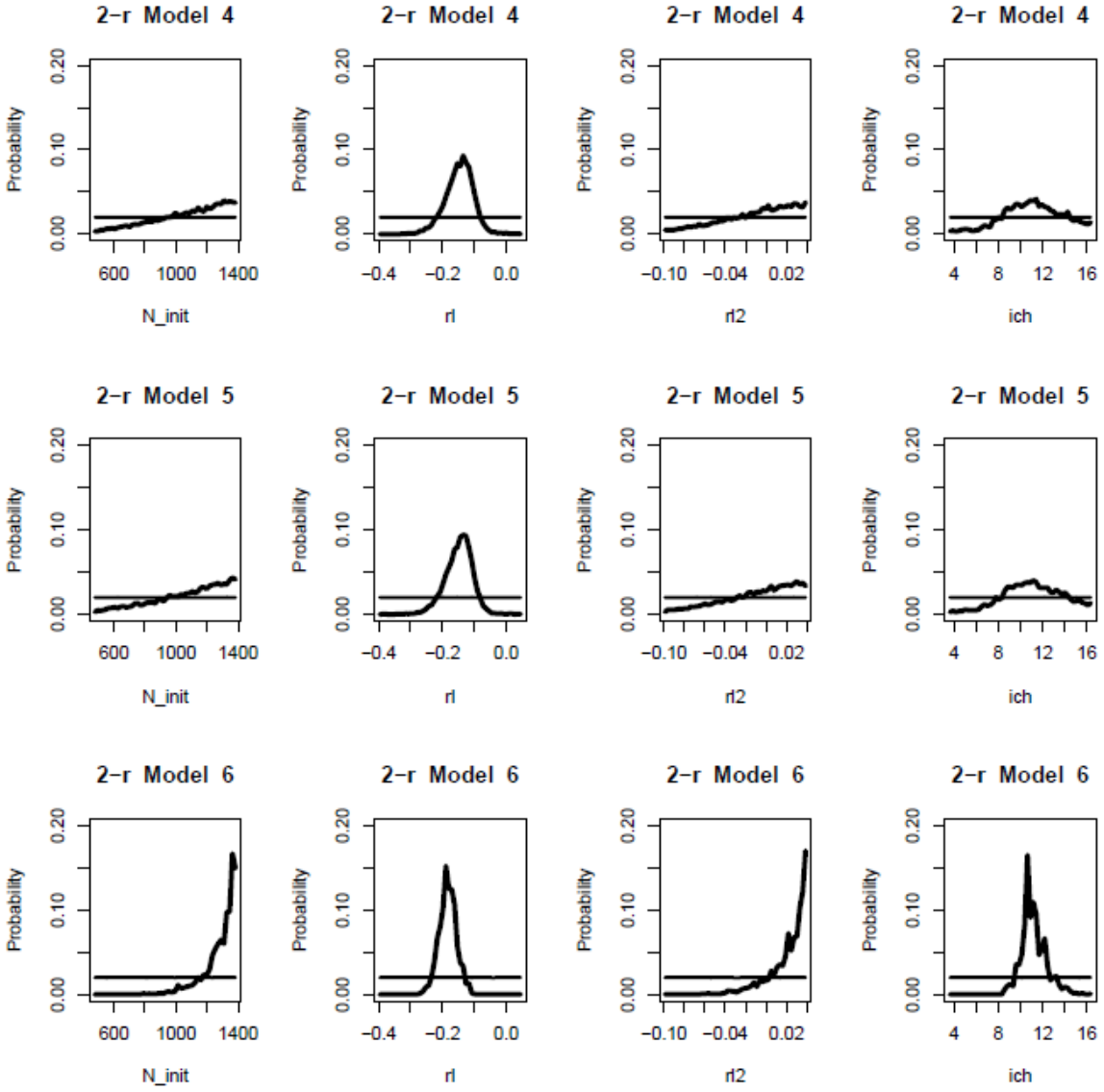

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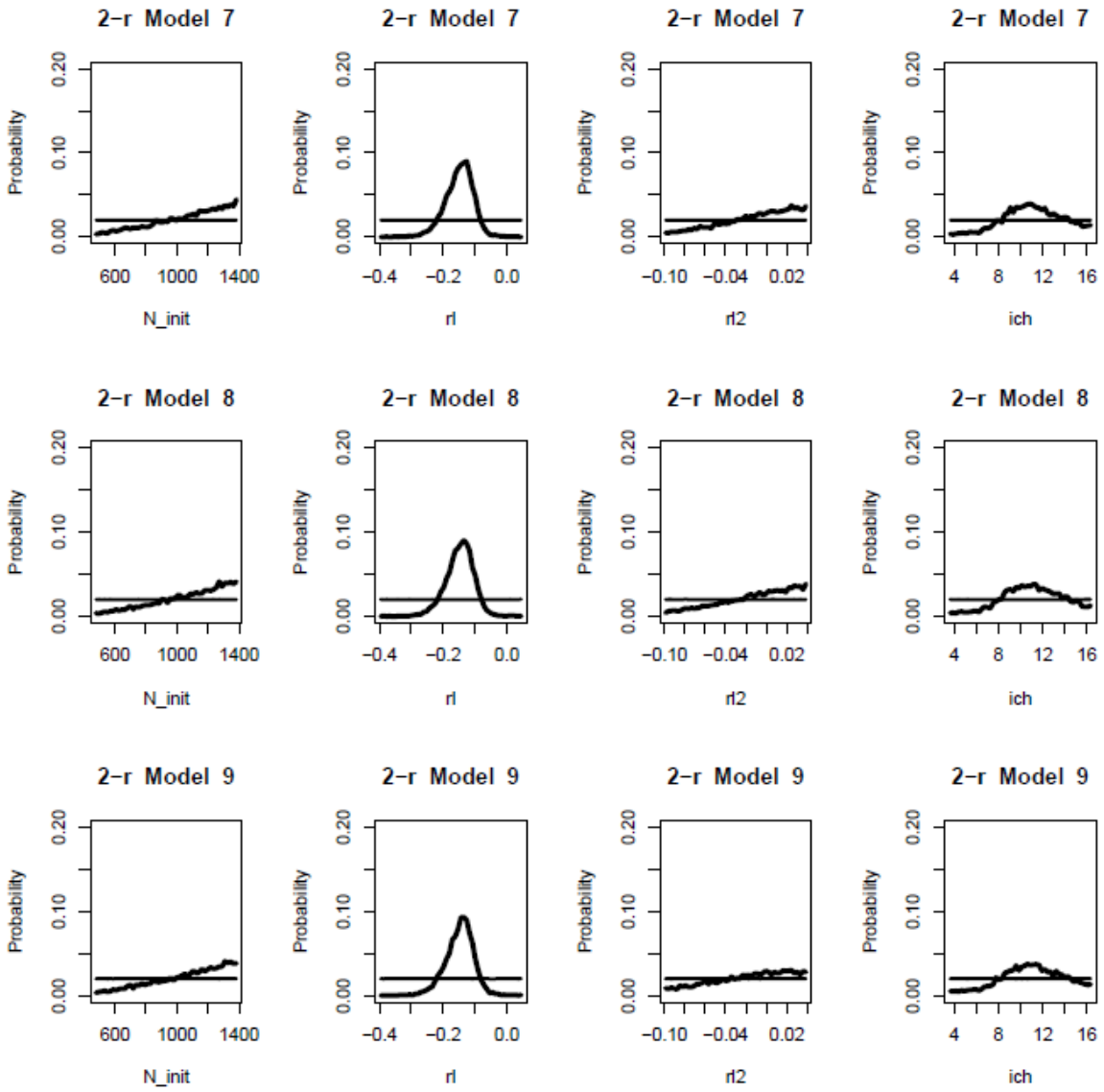
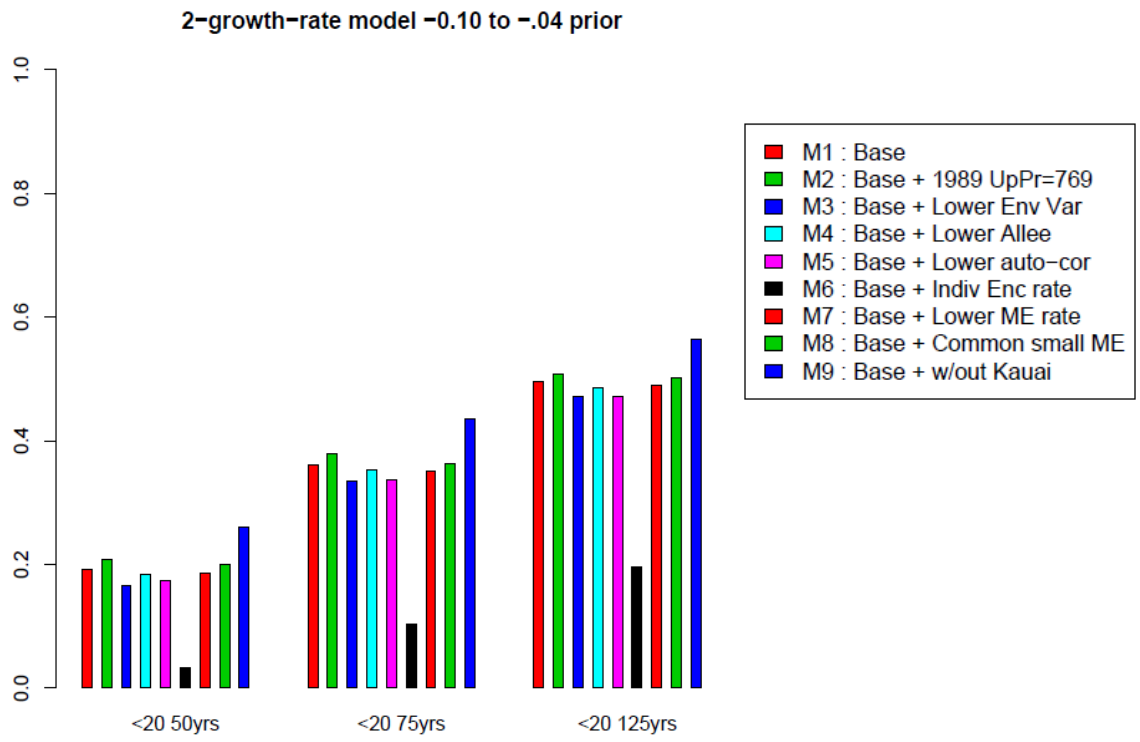


Figure 10. Probability of near extinction (< 20 individuals) at 50, 75, and 125 years for the 10 different two-rate-of-change models with prior distribution for the second rate of change of – 0.10 to 0.04. Models 1–10 as defined in Table 1.



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