Status Review of the Northeastern Pacific Population of White Sharks (*Carcharodon carcharias*) Under the Endangered Species Act

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EXECUTIVE SUMMARY

Background

In June and August of 2012, the National Marine Fisheries Service (NMFS) received two petitions requesting that the northeastern Pacific (NEP) population of white sharks (*Carcharodon carcharias*) be listed as endangered or threatened under the Endangered Species Act (ESA). In September 2012, NMFS published a 90-day finding (77 FR 59582) announcing that both petitions presented substantial scientific or commercial information indicating that the NEP white shark population may warrant listing under the ESA and that the agency would conduct an ESA status review of the NEP white shark population. NMFS formed a Biological Review Team (BRT) made up of scientists from the Southwest Fisheries Science Center having diverse backgrounds in population modeling, genetics, conservation biology, fisheries management, and shark biology and ecology to conduct the status review. The BRT considered a variety of scientific and technical information and conducted its own independent analyses. This document reports the results of its comprehensive status review of the NEP white shark population. This report was peer reviewed by three independent scientists.

Approach of the BRT

While recent years have seen a dramatic increase in the information available on the behaviors and movements of NEP white sharks, the BRT acknowledged that there is considerable uncertainty surrounding many aspects of abundance, trends in abundance, threats, life history, and habitat use for all age and sex classes. Such uncertainties are expected for a naturally uncommon highly migratory species. The BRT decided to treat the uncertainty explicitly by defining where it exists and using a process called Structured Expert Decision Making (SEDM) to weigh various plausible scenarios, taking into account all available data on the NEP white shark population but also considering information on white sharks from other regions and from similar species. For each SEDM exercise, the BRT developed a range of values, based on the best available data, across which plausibility points were allocated. Some of the results are provided in the "Executive Summary" and are identified as plausibility points. The exercises for which SEDM was used are presented in Appendix A. The BRT's objectives in taking this approach were to make plausible decisions in complex situations involving uncertainties, to insure that the decision making process was as transparent as possible and to provide assurance that the team was basing its decisions on a common understanding of the evidence.

Status of Northeastern Pacific Population of White Sharks

White sharks are globally distributed and are the largest of the predatory sharks. In the NEP, white sharks are most commonly observed as adults and subadults at near shore

aggregations that are typically associated with pinniped rookeries at Guadalupe Island (Domeier et al. 2012) and in central California (Klimley et al. 1992, Jorgensen et al. 2010). Recently, satellite tagging results have shown that they also exhibit an extensive offshore phase that differs by sex (Jorgensen et al. 2010, Domeier and Nasby-Lucas 2013). Early life history stages are more frequently encountered over the continental shelf in the southern California Bight and in Sebastián Vizcaíno Bay in Baja California Norte (Klimley 1985, Weng et al. 2007b, Domeier 2012, Santana-Morales et al. 2012). As with most of the large sharks, white sharks are relatively slow growing and have low overall fecundity, making them vulnerable to over exploitation. While they are not targeted in current fisheries they are taken incidentally in both artisanal and industrial fisheries (Lowe et al. 2012, Santana-Morales et al. 2012). Fisheries mortality, in addition to a historic reduction in the prey base across life history phases, may have reduced the number of white sharks in the NEP, although the data are insufficient to determine if and by how much the population may have declined. Low population numbers for subadult and adult white sharks in the NEP were recently reported by Chapple et al. (2011). The task of the BRT was to review the status of the white shark population in the NEP and assess its risk of extinction. To address these issues the BRT: 1) compiled the best available information relevant to the status review, 2) determined whether the NEP white shark represents a distinct population segment (DPS), and 3) examined the abundance estimates and current and projected population status. The BRT utilized the best available information to assess the risk of extinction to the population.

Determination of Distinct Population Segment

The BRT examined both the discreteness and significance of the NEP white shark population in relation to the global taxon to assess whether it met the criteria of a DPS. An examination of the behavioral and genetic data suggests that the NEP white shark population is markedly separated from other white shark populations. Photo identification (photo-ID) and electronic tagging data reveal a high level of site fidelity and no apparent movements outside the NEP (Domeier and Nasby-Lucas 2008, Jorgensen et al. 2010, Anderson et al. 2011, Nasby-Lucas and Domeier 2012). Genetic data reveal a monophyletic clade with no shared haplotypes between the NEP white shark population and those in Japan, Australia/New Zealand and South Africa (Jorgensen et al. 2010, Tanaka et al. 2011). This monophyletic pattern supports the inference that little movement occurs among these regions. Based on the genetic and tagging data, the BRT concluded that the NEP white shark population was discrete in relation to the global taxon.

The BRT also examined the significance of the NEP white shark population with respect to the global white shark taxon. The presence of a unique monophyletic clade with no shared haplotypes suggests that the NEP white shark is unique. In addition, the NEP represents a large portion of the global range of white sharks and its loss would result in a significant gap in the taxon's range. Thus, the BRT concluded that the NEP white shark population is significant to the global white shark taxon. Overall, the BRT concluded that the NEP white shark population meets the DPS criteria for discreteness and

significance and qualifies as a DPS. Some uncertainties exist, however. Of note is that the use of mtDNA data only assesses female-mediated gene flow and samples were only collected from central California. In addition, satellite tagging efforts have not been conducted in Japan, which could inform mixing across the North Pacific.

Risk Assessment

To determine whether the NEP white shark population is at risk of extinction either now or in the foreseeable future the BRT examined: 1) threats to the population, 2) direct and indirect information on population trends, 3) the population abundance and associated biases, and 4) the impacts of fishery mortality on the population under a range of estimated (or assumed) abundance and mortality levels. Finally, the BRT considered all of the available information related to extinction risk and assessed the overall extinction risk facing the NEP white shark population now and in the foreseeable future.

Threats: Key to assessing extinction risk for the NEP white shark population was to examine potential natural and anthropogenic threats to the population. In addition to current and future threats, the BRT examined historic threats which provided insight into the potential for past population declines. Threats included: 1) fisheries mortality in U.S., Mexican and international waters, 2) loss of prey due to overharvesting, 3) small population effects, 4) disease and predation, 5) habitat degradation linked to contaminants, and 6) global climate change. The BRT conducted SEDM assessment to characterize the severity of threats and the associated certainty.

The BRT examined fisheries mortality across a range of fisheries and in all white shark age classes. In the NEP, white sharks are and have been caught in a range of fisheries with varying risk over time. Offshore, the greatest threat was likely posed by the high seas drift net fisheries which overlapped, although minimally, with the habitat of adult white sharks in their offshore phase. These fisheries were banned in the early 1990s and were considered only a historic risk. Near shore, white sharks across all size classes have been taken primarily in net based fisheries. Similar to offshore, changes in regulations in U.S. coastal net fisheries have also reduced the overall catch of white sharks from peak values in the early 1980s. Current catch estimates for the last 10 years in the U.S. net based fisheries are ~28 young-of-the-year (YOY) and juvenile white sharks per year, of which ~50% are presumed mortalities. Additional catch of YOY and juvenile white sharks is documented in Mexico, primarily in Sebastián Vizcaíno Bay. The estimated annual white shark landings for Baja California Norte in 2011 was ~185 individuals all of which are assumed to be mortalities. New fisheries regulations banning shark fishing from May through July off the Pacific Coast of Mexico have reduced the reported take. This may, however, underrepresent the actual catch if there is non-reporting during the closure. Overall, fisheries bycatch and mortality were considered to be a moderate risk by the BRT.

Another potential risk to the NEP white shark population is linked to the reduction in their prey base due to overharvesting. For adult white sharks, pinniped populations were severely depleted in the 1800s and only started to recover in the mid-1900s (Gallo-

Reynoso et al. 2005, Carretta et al. 2011). For white sharks at early life history stages, the near shore net fisheries off California depleted a range of demersal species that are found in the diets of juvenile and YOY white sharks (Pondella and Allen 2008, Lowe et al. 2012). Since the near shore net ban was implemented by the State of California in 1994, a number of these species and other predators that have habitats and prey that overlap with white sharks have increased in abundance (Pondella and Allen 2008). The BRT considered prey loss to be a historic, but not a present risk to white shark populations across life history phases and the overall risk was considered to be low.

Very small populations are considered to be at increased risk of extinction due to Allee or other depensation effects which are tied to survival and reproduction via three mechanisms: ecological, genetic, and demographic stochasticity (Berec et al. 2007). The actual number at which populations would be considered at risk varies depending on the species and the risk being considered. The BRT examined small population effects with respect to the NEP white shark population. White sharks are thought to aggregate for mating (Domeier 2012, Jorgensen et al. 2012a) which will increase the local density and probability of finding mates. Throughout most of the year, the distribution of white sharks is very large and would not be vulnerable to local catastrophes like oil spills. Even during the months of aggregation, the range of the aggregation covers over a thousand miles and in any year about half of the adult females remain in remote pelagic habitat. Thus, white sharks are naturally buffered against catastrophic events and year-to-year variance in the productivity of local areas. The relatively high annual survival rates for large sharks should reduce the risk of demographic stochasticity which is linked to random variation on the annual number of births or deaths. The BRT concluded that the white shark population numbers were large enough (>50 mature adults) to avoid deleterious loss of genetic diversity. Overall the BRT considered the risk associated with small population size to be low.

One factor that influenced the BRT with regard to the number of mature adults in assessing small population risks, as well as in other decisions, was the relatively high haplotypic diversity observed in NEP white sharks. This level of genetic diversity either reflects a population that is larger than suggested by recent estimates, or a population that recently suffered a severe population decline (>90%). Modeling efforts to examine the feasibility of a severe decline over the last 2 generations indicated that even with mortality levels at the upper end of the estimated range, a severe decline was unlikely. The results of the genetic studies and the BRT's modeling efforts consequently supported the idea that the number of mature female white sharks in the NEP is in the hundreds to low 1000s.

The BRT also examined the threats associated with disease, predation, contaminants and global climate change. Predation on white sharks across life history stages is likely limited. There is no information on disease in white sharks. While contaminant loads are very high in juvenile white sharks, there is no data to suggest that there are deleterious physiological or population level effects (Mull et al. 2012). For climate change, recent modeling results suggest a potential increase in white shark habitat but this remains speculative and only encompasses adult life history phases (Hazen et al. 2012). Overall

the BRT ranked the extinction risk associated with disease, predation, contaminants and global climate change as being low. There was, however, a high degree of uncertainty given the lack of available information on a number of these factors.

Trends: One indicator of population status is information on trends. A declining trend would cause more concern than an increasing trend at a given population size. The BRT looked at direct and indirect evidence of population trends in the NEP. There were several examples that suggested increasing trends, but no examples that suggested declining trends. An analysis of catch-per-unit-effort over time in California net fisheries, factoring in changes in the fishery, showed an increase consistent with an increasing population. Photo-ID studies at Guadalupe Island showed a significant increase in the number of sharks from 2001-2011. Researchers working in central California have reported an increase in attacks on pinnipeds and suggested that this results from an increase in white shark populations (Klimley and Anderson 1996, Pyle et al. 1996). A dramatic increase in attacks on both California sea lions and sea otters has been observed in the Channel Islands and south of Monterey Bay. Based on these observations regarding trends, the BRT concluded using SEDM that it is likely that the NEP white shark population is stable or increasing (71% of plausibility points, see "Approach of BRT" above for explanation). The BRT did express considerable uncertainty (23% of plausibility points) given the lack of systematically collected data on abundance over time and uncertainty about the link between prey encounter rates and abundance.

Current abundance estimates: In addition to trends, the current abundance of a population is important for assessing population risk. Estimates of local population abundance were available from two studies and in both petitions these numbers were combined to provide an estimate of 339 adult and subadult white sharks for the NEP (Chapple et al. 2011, Sosa-Nishizaki et al. 2012). To characterize the demographics of the white sharks identified at the two sites, new data on size and sex from both sites were examined. Results revealed that the largest component of the population was mature males with fewer subadults and mature females at both sites. Sex ratios based on these analyses were highly skewed with a ratio of mature females: males of 0.2 at the central California sites and 0.6 at Guadalupe Island. The estimates at each site do not appear to represent all subadult and adult white sharks in the NEP. Prior to modeling efforts, the BRT more closely examined the potential for bias in 1) sex ratio and 2) abundance estimates in the studies at Guadalupe Island and in central California.

Sex ratio bias in population estimates: Given the concerns of the BRT that the recently published abundance estimates underestimated the population size for all subadult and adult white sharks in the NEP, the BRT undertook a number of steps to characterize the bias and estimate plausible population numbers. The first step was to characterize the bias in sex ratios at the two sites. Using SEDM the BRT concluded that, while the observed sex ratio was plausible (33% of plausibility points), the sex ratio at Guadalupe Island was likely to be higher (0.8 (48% of plausibility points) and 1.0 (20% of plausibility points)). There was very little support for the observed sex ratio at Central California (0.2 (4% of plausibility points)) representing the true sex ratio with a broad range of more likely scenarios (0.6 (43%), 0.8 (36% of plausibility points) and 1.0 (18% of plausibility

points)). The general conclusion that true sex ratios were higher than observed sex ratios was based on the timing of sampling efforts which may under-sample females, who appear arrive at the aggregation sites later than males. In addition, skewed sex ratios as a result of sexual segregation are common in lamnids (white, mako and salmon sharks) and so it is not surprising to observe this skew at the aggregation sites in the NEP. Also, there is no known source of mortality specific to juvenile female white sharks that would explain the skew. There was some concern about added mortality on mature females given their overlap with fisheries when they move into near shore waters to pup. These uncertainties are reflected in the BRT's conclusion that sex ratios of less than 1:1 are the most plausible.

Bias in population estimates: In addition to the bias in sex ratio, the BRT evaluated the bias associated with the fact that the photo-ID efforts at the two study sites may not capture all subadult and adult white sharks in the NEP. For the purpose of modeling population productivity, the BRT focused on the number of mature females and asked how many mature female white sharks may be missed by the two studies. Using SEDM, the BRT evaluated abundance bias factors of 1 (no bias), 1.2 (the abundance is 20% higher), 2 (the abundance is 2 times higher) and 10 (the abundance is 10 times higher). The BRT concluded that the actual female abundance in the population was likely 2-10 times the number of females that are observed at the two study sites (91% of plausibility points). The major factors that influenced the BRT's conclusion were: 1) the potential that mature females visited other, unsampled aggregation sites or do not visit the aggregation sites, 2) the haplotypic diversity suggesting a female population size of a few hundred to low thousands, and 3) the fact that low female abundance levels were not considered plausible in light of the estimated levels of fishery catch rates on YOY and juvenile white sharks.

Population abundance estimates: The BRT calculated 48 abundance estimates of mature females using all potential combinations of sex ratio and abundance bias. These 48 abundance estimates were then clustered into 4 abundance categories (60 – 125; 125 – 200; 200 – 400; and 400-1600 adult females) in addition to a fifth value representing the minimum estimate, based on the estimates of adult females from the two studies at Central California and Guadalupe Island (47 adult females). Based on the allocation of plausibility points from the SEDM on both types of bias, the BRT concluded that there is high support (>83% of plausibility points) for greater than 200 mature females in the NEP population. There was virtually no support (<<1% of plausibility points) for the observed number of adult females (47) representing the actual number of adult females. There was very low support (5% of plausibility points) for there being fewer than 125 adult females.

Population modeling and fisheries mortality: Fisheries mortality was the threat that the BRT felt posed a moderate level of risk. Therefore, the BRT conducted a population assessment to characterize the probability that fisheries mortality could lead to future population decline, especially to levels below certain thresholds reflecting extinction risk within relevant time periods. Population viability analyses (PVAs) were conducted for each female abundance category described above. Population growth rates were

calculated using life history information such as female fecundity, age at first reproduction and age specific survival rates. Data on fisheries mortality were taken from catch data for YOY and juveniles. Given the absence of good data on catch rates for mature females, the BRT used SEDM to determine the plausibility of different levels of adult female mortality (0, 1, 2, 5, and 10 females killed per year). The BRT concluded that the most plausible scenario would be 2 adult female mortalities (34% of plausibility points) from fisheries interactions each year but had considerable support for 1 and 5 (24% and 25 % plausibility points respectively) killed per year reflecting a good deal of uncertainty. For the purpose of the population modeling, the BRT considered all potential combinations of the 5 levels of female mortality and 4 adult female abundance categories, making for a total of 20 modeling scenarios.

Risk categories: To determine high risk we considered two categories:

1) the declining population category (where the population declines to near-extinction within a certain timeframe),
2) the small population category (where the population is dangerously small currently).

For each modeling scenario, the BRT evaluated the extinction risk to the population based on the following criteria regarding the abundance of mature females:

High Risk--a population is at high risk if it meets either condition:

A. the population has a 5% chance of falling below 50 mature individuals (25 mature females) in 60 years (~3 generations)
B. the current population is less than 250 mature individuals (125 mature females).

The scenarios that fell under this category included those with a present-day adult female abundance of 60 - 125 for all levels of adult female mortality and the scenario with a present-day adult female abundance of 125-200 and adult female bycatch mortality of 10 per year.

We related Medium risk to our High risk categories by adding 40 years (~2 generations) as our definition of 'foreseeable future'. The medium risk category is based on the idea of becoming a high risk population within ~ 2 generations.
Medium Risk--a population is at medium risk if it meets either condition:

A. the population has a 5% chance of falling below 50 mature individuals (25 mature females) in 100 years (~5 generations)
B. the population has a 5% chance of falling below 250 mature individuals (125 mature females) in 40 years (~2 generation times).

The scenarios that fell under this category included those with a present-day adult female abundance of 125-200 and adult female bycatch mortality levels of 0, 1, 2 or 5 per year.

Low and very low risk categories do not have the two categories like the high or medium risk categories.

Low risk-- The population does not meet the criteria for medium or high risk but the population has at least a 10% probability of showing a decline within 100 years (~5 generations).

The scenarios that fell under this category included those with a present-day adult female abundance of 200-400 with all levels of adult female bycatch mortality and the scenario with a present-day adult female abundance of >400 at an adult female bycatch mortality of 10 per year.

Very low risk-- The population does not meet any of the above criteria for high, medium, or low risk. The population has a high probability of being stable or increasing.

The scenarios that fell under this category included those with the highest present-day adult female abundance (>400) with adult female bycatch mortality of 0, 1, 2 or 5 per year.

In addition to the examination of extinction risk, the PVAs and associated estimates of fisheries catch and mortality rates also informed the plausibility of the different present-day adult female abundance categories (N_f). This was used to inform a second vote on the abundance bias as mentioned above. For the lowest two abundance scenarios (N_f = 60 – 125 and N_f = 125 – 200), the corresponding annual fishery catch rates for YOYs (means = 0.48 and 0.27) were extremely high. For the minimum estimate of 47 adult females (based on the photo-ID data), the mortality rate for YOY was essentially 100%. Population removal rates for sharks, even for selective gears (such as pelagic longlining), are probably less than 20% (Worm et al. 2013). Vaquita porpoises that are subjected to extremely intensive gill net exposure throughout their range have an estimated bycatch mortality rate ~0.20 based on the model used in Jaramillo-Legorreta et al. (2007). It would be exceptional if gill nets had the efficiency to capture greater than 20% of YOY white sharks each year, especially given that the available bycatch estimates come from small fishing fleets operating throughout a limited portion of known or suspected YOY nursery habitat (O. Sosa-Nishizaki, CICESE, pers. comm.). This information challenges the plausibility of the abundance scenarios with less than 200 adult females. It should be noted, however, that the model results are highly dependent on the inputs for fisheries mortality and there is some uncertainty with regard to the level of bycatch on juvenile and YOY white sharks.

<u>Overall Risk assessment:</u> Using SEDM, the BRT assessed the overall extinction risk to the NEP population, taking into consideration the broad range of factors evaluated in the status review, including all of the threats to the population, the population status and trends, and the model results. The BRT concluded that the NEP white shark population

was most likely at a low to very low risk of extinction (see Table below for summary of plausibility points). This is consistent with the BRT's analyses and conclusions that the present-day population most likely has greater than 200 mature females.

Table 1. Summary of the SEDM plausibility points for each risk category.

Risk Category	Average SEDM weight (SD)
Very low	50 (19.3)
Low	36 (12.8)
Medium	9 (8.3)
High	5 (5.6)

With a mature female population in the hundreds, it follows that the total population estimate, including mature males and subadults, would be considerably higher than the estimate of 339 subadults and adults provided by the petitions. While the BRT did not attempt to estimate the number of subadult and adults, the total population estimate of male and females across size classes was ~3000 at a mature female abundance level of 200.

The BRT's conclusions are based on the best available data. The BRT recognizes that there are a number of limitations to the analyses that will be informed as additional research results are published.

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List of Acronyms

BRT	Biological Review Team
CDFW	California Department of Fish and Wildlife
CICESE	Centro de Investigación Científica y de Educación Superior de Ensenada (Ensenada Center for Scientific Research and Higher Education)
CITES	Convention on International Trade in Endangered Species
CPUE	catch-per-unit-effort
CSULB	California State University, Long Beach
CV	coefficient of variation
DDT	dichlorodiphenyltrichloroethane
DGN	drift gillnet
DPS	discrete population segment
ESA	Endangered Species Act
FEMAT	Forest Ecosystem Management Assessment Team
FWS	U.S. Fish and Wildlife Service
IUCN	International Union for the Conservation of Nature
IUU	illegal, unreported and unregulated
MCSI	Marine Conservation Science Institute
MBA	Monterey Bay Aquarium
mtDNA	mitochondrial Deoxyribonucleic Acid (DNA)
NEP	northeastern Pacific
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
PacFIN	Pacific Fisheries Information Network
PCB	polychlorinated biphenyl
Photo-ID	photo identification
PSAT	popup satellite archival tag
PVA	Population viability analyses
SEDM	Structured Expert Decision Making
SOFA	Shared Offshore Foraging Area
SPOR	significant portion of its range
SPOT	Smart Position or Temperature Transmitting Tag
SWFSC	Southwest Fisheries Science Center
TL	total length
USGS-WERC	US Geological Survey Western Ecological Research Center
YOY	young-of-the-year

1. Introduction

1.1. Scope and intent of the status review

This document is the status review in response to two petitions[1,2] to list the northeastern Pacific (NEP) population of white shark (*Carcharodon carcharias*) as threatened or endangered under the Endangered Species Act (ESA). The petitions cite concerns regarding the small estimated population size of white sharks in the northeastern Pacific and threats to the population such as fisheries harvest and bycatch, coastal development, pollution, and ocean acidification. In addition, the petitioners requested the designation of critical habitat for the NEP population of white sharks.

Under the ESA, a status review shall be promptly commenced if a petition is found to present substantial scientific or commercial information that the petitioned action may be warranted (16 U.S.C. 1533(b)(3)(A)). The National Marine Fisheries Service (NMFS) decided that the petitions had sufficient merit for consideration and that a status review was warranted (77 FR 59582, September 28, 2012). The ESA stipulates that listing determinations should be made on the basis of the best scientific and commercial information available. As a result of this finding, NMFS committed to completing an ESA status review of the northeastern Pacific population of white shark (hereafter, NEP white sharks or NEP white shark population) and convened a Biological Review Team (BRT)[3] consisting of Federal biologists with diverse scientific backgrounds to conduct the review. The purposes of this Status Review were to determine whether the NEP white shark population is a distinct population segment (DPS) as defined under the ESA, assess its extinction risk, and identify and evaluate potentially significant threats to the population. To conduct this Status Review, the BRT considered a variety of scientific information from the literature, unpublished documents, and direct communications with researchers working on NEP white sharks, as well as technical information submitted to NMFS. All information not previously peer-reviewed was formally reviewed by the BRT. 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 status review.

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[1] WildEarth Guardians to U.S. Secretary of Commerce and the National Oceanic and Atmospheric Administration through the National Marine Fisheries Service, June 20, 2012, "Petition to list the northeastern Pacific ocean distinct population segment of great white shark (*Carcharodon carcharias*) under the U.S. endangered species act"

[2] Oceana, the Center for Biological Diversity, and Shark Stewards to U.S. Secretary of Commerce and the National Oceanic and Atmospheric Administration through the National Marine Fisheries Service, August 10, 2012, "Petition to list the northeastern Pacific population of white shark (*Carcharodon carcharias*) as threatened or endangered"

[3] The Biological Review Team for the NEP white shark consisted of the following members from the NMFS Southwest Fisheries Science Center (area of expertise in parentheses): Dr. Heidi Dewar (Team Leader; (ecology and movements of tunas, sharks and billfish), Dr. Tomoharu Eguchi (marine turtle population and ecological modeling), Dr. John Hyde (fisheries genetics), Dr. Doug Kinzey (population assessments of Antarctic krill), Dr. Suzanne Kohin (ecology and population dynamics of pelagic sharks), Dr. Jeff Moore (population dynamics/risk assessment), Dr. Barbara Taylor (genetics/risk assessment), and Dr. Russ Vetter (physiological ecologists and molecular geneticist).

This document reports the results of the BRT's comprehensive status review of NEP white sharks. These conclusions are subject to revision should important new information arise in the future. This document is a compilation of the best available scientific and commercial information and a description of past, present, and likely future threats to NEP white sharks. 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 and considering the information and conclusions presented in this Status Review, other 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 (http://www.nmfs.noaa.gov/pr/species/) and announced in the Federal Register.

1.2. Assessing plausibility of influential factors in the face of non-quantified uncertainty Change to SEDM

To conduct the status review, the BRT used the best available data and at times needed to make decisions in the face of different levels and sources of uncertainty. To deal with this uncertainty, the BRT adopted structured expert decision making (SEDM) as a formal method to make decisions or express opinions. These formal methods are important in a setting where quantitative measures of uncertainty derived from the empirical data are not available to address all sources of pertinent uncertainty. One example of an influential factor addressed in this manner is the sex ratio of adults at the sampling sites. SEDM employs a plausibility point method (although other methods were employed also, as described later), that is a variation of a method used by scientific teams evaluating options under the Northwest Forest Plan (Forest Ecosystem Management Assessment Team (FEMAT) 1993). This method allowed the BRT to quantify the team's best professional judgment regarding a decision, and thus improve transparency in how these decisions were made in the status review. Each BRT member was asked to distribute 100 plausibility points among the choices for various decisions throughout the status review process, reflecting his or her opinion about the plausibility of different scenarios (e.g., a range of possible sex ratios and proportions of adult population present at the sampling sites). If a BRT member was certain of a particular option, or felt it was the only plausible scenario, he or she could assign all 100 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 the status reviews of Southern Resident killer whales (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, Biological Review Team (BRT) 2005), black abalone (VanBlaricom et al. 2009), and Hawaiian false killer whales (Oleson et al. 2010). Alternative methods were occasionally used, such as when a particular question did not concern plausibility, but rather the degree of severity, or level of certainty of individual population threats.

1.3. Key questions in ESA evaluations

1.3.1. The "species" question

For the purpose of the ESA, the term "species" includes (ESA section 3(16))

> "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 (see 61 FR 4722: February 7, 1996). 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 petitioners

The petitions submitted by WildEarth Guardians and jointly by Oceana, the Center for Biological Diversity, and Shark Stewards assert that: 1) NEP white sharks are a genetically distinct population; 2) the population level is so low that NEP white sharks are at risk of extinction without regards to other risks; and 3) additional risks to NEP white sharks include pollution, global warming, ocean acidification, low levels of available prey resources, overfishing, and habitat destruction. The petitions assert that listing of the NEP white shark population as a threatened or endangered DPS under the ESA is needed to ensure that this population does not become extinct.

1.4.1. The DPS question: "Discreteness"

The petitioners presented the following arguments that NEP white sharks are "discrete" from other populations of white sharks.

1.4.1.1. The NEP white shark population differs markedly from other oceanic populations of the species because of physical factors

The petitioners stated that the NEP white shark population is reproductively isolated and that genetic analysis demonstrates that this population is genetically distinct. While the NEP white shark population has some genetic similarities with the Australian/New Zealand population of white shark, sampling shows that these populations have been separated for hundreds of thousands of years resulting in a clear genetic divergence between the populations (Jorgensen et al. 2010). The NEP white shark is thought to have initially been established approximately 200,000 years ago by individuals that migrated from Australia/New Zealand waters (Jorgensen et al. 2010).

1.4.1.2. The NEP white shark population differs markedly from other oceanic populations of the species because of behavioral factors

The petitioners state that the NEP white shark population is geographically separated from other white shark populations due to strong site fidelity and adherence to distinct migratory patterns within the NEP. Although white sharks are capable of long-distance dispersal, the NEP white shark population appears to remain within a fixed geographical range, with consistent use of three core areas within the NEP (i.e., coastal shelf waters of North America primarily from central California to Baja California; the slope and offshore waters of the Hawaiian archipelago; and offshore waters approximately halfway between California and Hawaii) (Boustany et al. 2002, Jorgensen et al. 2010, Domeier 2012). Tagging studies show no evidence of straying or spatial overlap with other white shark populations (Jorgensen et al. 2010).

1.4.1.3. The NEP white shark population should be considered discrete because 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

The petitioners state that the NEP white shark population inhabits the waters of the United States' Exclusive Economic Zone, but also transits the high seas and occurs in Mexican waters, with potentially infrequent migrations to the Exclusive Economic Zone of Canada. Thus, these white sharks are subject to exploitation outside U.S. waters by non-U.S. actors and may be impacted by differences in management between the United States, other nations, and the high seas.[4]

1.4.2. The DPS question: "Significance"

The petitioners presented the following arguments to demonstrate that the NEP white shark is "significant" to the taxon to which it belongs.

1.4.2.1. The NEP white shark population persists in a unique ecological setting for this species

The petitioners stated that the NEP white shark population persists in an ecological setting unique to the species, because it is the only white shark population in the California Current Large Marine Ecosystem. The petitioners describe the California Current Large Marine Ecosystem as a biologically rich and dynamic ocean environment with a coastal upwelling system that results in varying abundances of prey and predators, but that it is also highly sensitive to the impacts of the El Niño Southern Oscillation. The NEP white shark population is also the only white shark population to occupy the offshore area of the Pacific Ocean between Hawaii and California, which is hypothesized to be used for foraging and/or mating.

Additionally, the petitioners assert that the NEP white shark population plays an important ecological role that is essential for the health of the NEP ecosystem, as a top predator that may play an important top-down role in structuring the California Current Large Marine Ecosystem (Block et al. 2011). We note that in determining whether a discrete segment is significant, NMFS focuses on the biological and ecological significance of the population segment to the species, not to the ecosystem.

1.4.2.2. Loss of the NEP white shark population would result in a significant gap in the range of the taxon

The petitioners state that the loss of the NEP white shark population would result in a significant gap in the range of the species as recent data indicate that the NEP population does not mix with

[4] Note: The petitioners' arguments reflect a misunderstanding of this criterion. This criterion states that a population may be considered discrete if it is separated from other populations by international boundaries within which significant differences in regulatory mechanisms exist. That the NEP population crosses these international boundaries actually argues against considering this population as discrete from other white shark populations.

any other regional populations of white sharks and, if this population were lost, it would likely not be replaced by immigrating individuals from an outside population (Hueter et al. 2004, Jorgensen et al. 2010). The NEP white shark population persists only in its unique range. As described above, the NEP white shark does not interbreed with other breeding populations.

1.4.2.3. The NEP white shark population differs markedly from other populations of the species in its genetic characteristics

The petitioners state that the NEP white shark population is markedly different from other populations based on its genetic characteristics. Studies demonstrate that this population may have descended from the Australia/New Zealand population over 200,000 years ago (Jorgensen et al. 2010). Since then, however, this population has been isolated from other populations and breeds only with individuals within the population, resulting in genetic distinction from all other white shark populations (Jorgensen et al. 2010, Gubili et al. 2012).

1.4.3. Risk factors

The petitioners stated that the NEP white shark population qualifies as endangered or threatened under the ESA based on all five of the ESA factors to be considered (see Section 1.3.2 of this Status Review). The petitioners identified specific threats to the NEP white shark population to include: (1) their low population level; (2) incidental catch in fisheries; (3) high contaminant levels; (4) habitat alteration due to ocean acidification, ocean warming, and other stressors; and (5) paucity of scientific data on key population factors. The petitioners presented the following information to demonstrate that the NEP white shark population faces threats associated with all five ESA factors.

1.4.3.1. Present or threatened destruction, modification, or curtailment of the species' habitat or range

The petitioners asserted that white shark habitat in the NEP has been highly modified by human activity. Increasing human activity, especially when concentrated in coastal areas, may lead to degradation of important inshore feeding and reproductive habitat for white sharks. One concern cited by the petitioners was the depletion of prey resources for NEP white sharks. The petitioners stated that human exploitation massively depleted all populations of pinnipeds, an important prey resource for adult white sharks. While population trends for these pinnipeds are increasing, the petitioners contended that they were in a depleted state for a prolonged period of time and currently remain below historic levels. In addition, the petitioners stated that there have been and continue to be major commercial fisheries for most of the prey resources that are important for other life stages of white sharks, such as Pacific sardine (*Sardinops sagax*), salmon, white seabass (*Atractoscion nobilis*), black rockfish (*Sebastes melanops*), striped bass (*Morone saxatilis*), spiny dogfish (*Squalus acanthias*), crustaceans, cephalopods, cabezon (*Scorpaenichthys marmoratus)*, lingcod (*Ophiodon elongates)*, and Pacific mackerel (*Scomber japonicas*) (Klimley 1985, Ellis and McCosker 1995, California Department of Fish and Game 2010).

Another concern cited by the petitioners is the potential bioaccumulation of contaminants in white sharks in the NEP, resulting from the discharge of pollutants into coastal habitats. The petitioners stated that this is a particular concern in the Southern California Bight, an area that is believed to serve as a nursery area for white sharks and where young white sharks were found to have high levels of mercury and specific pollutants (Mull et al. 2012).

In addition, the petitioners cited a recent finding that ocean acidification is progressing rapidly in the California Current Large Marine Ecosystem and will grow more severe in the range of the NEP white shark population (Gruber et al. 2012). While severity of the effects on specific species or the overall ecosystem are uncertain, the study indicates that the California Current Large Marine Ecosystem is moving rapidly toward conditions that are well outside the natural range. The petitioners argued that this could adversely impact the marine food web, including white sharks. In combination with ocean warming, these effects could be cumulative and synergistic (Pörtner 2008).

1.4.3.2. Overutilization for commercial, recreational, scientific, or educational purposes

The petitioners stated that right now, the capture and killing of white sharks in fishing gear is the key direct threat to the NEP population. Worldwide, white sharks may be targeted or incidentally caught in commercial and sport fisheries and are highly valued for their fins (for use in shark fin soup) and in the curio trade. Off Southern California, reported white shark captures ranged from 2 to 25 sharks per year over the period of 1936 to 2010 and indicate an increasing trend in bycatch over the last decade (Lowe et al. 2012). The petitioners argued that despite the regulations curtailing near shore set gillnet effort off California, this fishery continues to account for the greatest number of white shark interactions of any U.S. West Coast fishery. The petitioners also expressed concern regarding bycatch of white sharks off Mexico, citing a study that documented bycatch of 111 juvenile white sharks over the course of 11 years (1999-2010) along the coast of Baja California, Mexico (Santana-Morales et al. 2012).

1.4.3.3. Disease or predation

One of the petitioners asserted that the introduction of mercury, organochlorines, and other pollutants to the ocean and the effects of these pollutants on NEP white sharks may be categorized as disease. The petitioner stated that although we do not know how these pollutants impact the population, potential effects may include behavioral changes, emaciation, cerebral lesions, and impaired sexual development (Mull et al. 2012). The petitioner added that all life stages of NEP white sharks may be vulnerable to high body burdens of pollutants (Domeier 2012).

1.4.3.4. Inadequacy of existing regulatory mechanisms

The petitioners stated that despite some protections, white shark populations are still declining globally. Although there are some prohibitions on the landing, targeting, and trading of NEP white sharks within the United States, the petitioners asserted that existing regulatory mechanisms are inadequate to address the continued incidental catch of white sharks by a suite

of fisheries on the U.S. West Coast. In particular, the petitioners argued that national protections for white sharks are insufficient when it comes to monitoring, control, and surveillance of this species, because it is captured in several fisheries that are not subject to limits on incidental take or adequate observer coverage (CITES 2004). At the international level, the petitioners cited the lack of international laws to protect live white sharks and the lack of trans-boundary management programs as deficiencies that need to be addressed for the conservation and management of a highly migratory species like the white shark (CITES 2004). Thus, the petitioners concluded that existing regulatory mechanisms are inadequate to provide the necessary protections for NEP white sharks.

1.4.3.5. Other natural or manmade factors

The petitioners cited several other natural or manmade factors that may be affecting the survival and recovery of NEP white shark. First, the petitioners stated that the bioaccumulation of high levels of mercury and organochlorines in the tissues of juveniles may result in physiological impairments and reduced fitness (Mull et al. 2012). Juvenile white sharks are particularly at risk of exposure to contaminants due to their use of nursery habitats next to urban areas in the Southern California Bight. Second, the petitioners stated that negative media attention is a threat to white sharks, because it generates general paranoia and encourages targeting of the species for sport or trophy hunting (IUCN 2009). Third, the petitioners considered the species' life history characteristics (e.g., slow growth, late maturation, long-life, long generation time, small litter size, and low reproductive capacity) and small estimated population size to be factors that increase the extinction risk of the NEP white shark population. Finally, the petitioners asserted that the synergistic effects of all of the threats facing the NEP white shark population and its habitat could cause a greater and faster decline in the population than might be expected.

1.5. Treatment of data and arguments presented in the petitions

The data and arguments presented in the petitions were considered by the BRT along with all other public comments submitted in response to the 90-day finding (77 FR 59582, September 28, 2012) requesting additional scientific and commercial data. The petitions and their arguments are occasionally referenced throughout this Status Review; however, the Status Review will not respond to each of the petitioners' arguments in turn.

2. Background Information on White Sharks

This review focuses on data from the NEP but draws on information from other areas across the species range where appropriate or where information in from the NEP was lacking or poor.

2.1. Life history and ecology

2.1.1. Taxonomy and distinctive characteristics

All white sharks belong to the family Lamnidae (Order Lamniformes) (Compagno 2001). White sharks are recognized by their spindle-shaped body and moderately long conical snout. Adults have large flat triangular serrated teeth in both their upper and lower jaw, long gill slits, a large first dorsal fin and small second dorsal and anal fins, and upper and lower lobes on the caudal fin which are approximately the same size. The white shark generally has a gray or brownish gray to blackish dorsal surface and white ventral surface. The margin between the dark dorsal and white ventral surfaces is sharply delimited and creates a distinct pattern that can be used to identify individuals. The iris of the eye is black.

2.1.2. Global range and habitat use

The white shark is a circumglobal species that lives in coastal regions as well as the open ocean (Compagno 2001). White sharks are most frequently observed in inshore temperate continental waters of the Western North Atlantic, Mediterranean Sea, southern Africa, southern and western Australia, and the northeastern Pacific (NEP). Young-of-the-year (YOY) and juvenile white sharks are thought to prefer shallow coastal waters (Dewar et al. 2004, Weng et al. 2007b, Bruce and Bradford 2012). Adults and subadults are most commonly observed near pinniped rookeries but also range far from shore spending protracted periods in pelagic habitats (Klimley 1985, Domeier and Nasby-Lucas 2008, Jorgensen et al. 2010). Range and habitat descriptions specific to the NEP population can be found below.

2.1.3. Reproduction and growth

Given the challenges associated with obtaining a large sample size of white sharks, a complete suite of values for life history parameters is not available for the NEP white shark population. Consequently, results from relevant white sharks studies from all ocean basins were surveyed to compile a summary of life history traits (Table 2.1). White sharks are categorized as YOYs, juveniles, subadults, and adults as defined in Bruce and Bradford (2012). YOY are sharks in their first year and range from approximately 1.2 - 1.75 m in total length (TL), with size at birth ranging from 1.2 -1.5 m TL (Francis 1996). Juveniles range approximately from 1.75 – 3.0 m TL, and subadults range from 3.0 m TL to the size at maturity which is reported to be 3.6 – 3.8 m TL for males and 4.5 – 5 m TL for females (Cailliet et al. 1985, Francis 1996, Wintner and Cliff 1999, Malcolm et al. 2001). A number of studies have used vertebral bands to construct von Bertalanffy growth curves, a model relating size to age that is used extensively in fisheries. Cailliet et al. (1985) used data from 20 white sharks sampled in the NEP (Table 2.1). Their results are similar to those of Winter and Cliff (1999) and Malcolm et al. (2001) for South Africa and Australia, respectively. A growth study on YOY in captivity found that under optimal

conditions in captivity, the sharks showed a high growth capacity with approximately twice the growth rate estimated from a von Betalanffy growth function as that determined by Cailliet et al. (1985) (Ezcurra et al. 2012). While very large maximum sizes are reported in the scientific literature, 6 m TL for females is the largest validated measurement (Castro 2012). Maximum age has also been difficult to obtain. In the NEP, the highest age estimate comes from Anderson et al. (2011) who saw the same individual for 22 years at a coastal aggregation site. Given that white sharks are thought to recruit to coastal aggregations at ~ 3 m TL or 5 years of age, the age estimate for this individual was 27-30 years of age. In recent study for Atlantic white sharks, vertebral band counts (validated using bomb radio carbon dating) provided maximum age estimate of ~70-100 years (Natanson and Skomal in review).

Data for reproductive parameters such as litter size are also limited. For example, no pregnant female white sharks have been caught in the NEP despite the fact that the southern California Bight is a nursery ground. Francis (1996) summarized the data from 10 pregnant white sharks from around the globe and reported that maturity ranged from 4.5-5 m, which is similar to that reported by others (Malcolm et al. (2001): 5 m; Domeier and Nasby-Lucas (2013): 4.6 m). The average litter size from 11 females was reported to be 8.9 (Mollet et al. 2000). Sex ratio in utero is 1:1 (Francis 1996). Gestation is thought to be longer than a year and estimated at 18 months (Francis 1996, Mollet et al. 2000), but the actual duration is unknown. Consistent with the long gestation period, pupping frequency has been suggested to range broadly between 2-3 years. As a part of the photo ID study at Guadalupe Island it was possible to quantify the visitation patterns of adult females over a period of 10 years. While most females returned every two years, which is presumably linked to their reproductive cycle, in some instances it was every three years. Taking an average over 23 females provided an estimate of a 2.2 years inter-birth interval (Nasby-Lucas and Domeier 2012, Nasby-Lucas unpublished data). Male maturity has been inferred typically from the length of the claspers, as well as the level of calcification and sometimes the presence of seminal fluid in the claspers. Estimates of the size at maturity for males typically fall between 3.6 and 3.8 m TL (Pratt 1996, Malcolm et al. 2001). The ages associated with the size at maturity (females: 12- 18 years; males: 8-10 years) are taken both from the von Bertalanffy growth curve and studies where age was determined from vertebral bands (Cailliet et al. 1985,Wintner and Cliff 1999, Malcolm et al. 2001). Additional information on most aspects of both growth and reproduction is needed.

2.1.4. Foraging Ecology

Information on foraging ecology in white sharks comes from stomach content analysis and visual observation of feeding events in larger sharks (Klimley 1985, Compagno et al. 1997, Skomal et al. 2012). YOY and juvenile white sharks are known to feed primarily on invertebrates, demersal teleosts and small elasmobranchs, as well as squid and epipelagic fish. Stomach contents of YOY off California were found to contain bony fishes (cabezon and lingcod), cartilaginous fishes (gray smooth-hound (*Mustelus californicus*), spiny dogfish and a dasyatid ray), and a crustacean (spot-bellied rock crab). Stomach contents for juveniles contained bony fishes (Pacific sardine, green sturgeon (*Acipenser medirostris*), king salmon (*Oncorhynchus tshawytscha)*, white seabass, black rockfish and striped bass), cartilaginous fishes (brown smooth-hound (*Mustelus henlei*), soupfin shark (*Galeorhinus galeus*) and bat ray (*Myliobatis californica*), and a crustacean (Klimley 1985).

Once white sharks reach about 3 m their diets expand to include marine mammals (Klimley 1985). The most important prey items include pinnipeds (including, seals, sea lions, and elephant seals) and fishes (including other sharks and rays). Less common items in stomach contents include marine reptiles (mostly sea turtles), larger cephalopods, gastropods, and crustaceans. White sharks are also observed to scavenge on large and small cetaceans (Compagno et al. 1997, Curtis et al. 2006, Dicken 2008). Recent isotope data suggests that not all sharks make the shift to marine mammal prey (Kim et al. 2012) and some may continue to forage at lower trophic levels. It is important to note that no diet data on shark feeding habits in the North Pacific during the period when they are outside coastal waters.

Table 2.1Compilation of *C. carcharias* life history characteristics from the published literature.

Parameter	Value	Location	References and notes
Max age	27-30 years	NE Pacific	(Anderson et al. 2011); saw same animal for 22 years.
Max (TL) (observed)	6 m	Western Australia	(Randall 1987, Castro 2012): all other estimates of max size were not verified. Males are smaller than females (max size probably around 540-550 cm).
Mature TL (male)	3.6-3.8 m	South Africa, Australia, Atlantic	(Pratt 1996, Wintner and Cliff 1999, Malcolm et al. 2001): based primarily on condition of claspers (both the length and level of calcification)
Mature age (male)	8-10 years	South Africa	(Wintner and Cliff 1999): counted 8 band pairs in a mature male. 360-380 cm TL corresponds to 7.5-8.5 years of age using Cailliet et al. (1985)
Mature TL (female)	4.5-5 m	Global	(Francis 1996, Uchida et al. 1996, Malcolm et al. 2001, Tanaka et al. 2011, Domeier and Nasby-Lucas 2013): Most put pregnant females at 450-480 cm TL
Mature age (female)	12-15 years	Global	This is the age range from the Cailliet et al. (1985) von Bertalanffy curve using a size a maturity of 4.5-5 m TL (using 4.6-4.8 m TL, the range is 12-13 years).
Gestation	~18 months	Global	(Cailliet et al. 1985, Francis 1996, Mollet et al. 2000): >12 months but actual number uncertain
Pupping frequency	2.2 years	NE Pacific	(Nasby-Lucas and Domeier 2012, Nasby-Lucas unpublished): using photo-ID data
Parturition	Spring/Summer	Global	(Francis 1996, Domeier 2012, Lowe et al. 2012)
Litter size	mean 8.9	Global	(Mollet et al. 2000): range 4-14 (n=11);
Size at birth	1.2-1.5 m	Global	Francis (1996) summarized values from a range of sources
L_∞ (m)[1]	7.64 m	NE Pacific	(Cailliet et al. 1985): size range of 1.29-5.08 m TL (n = 20)
t_0[2]	-3.53 years		(Cailliet et al. 1985)
K[3]	0.058		(Cailliet et al. 1985)
YOY TL	1.2-1.75 m	Global	(Cailliet et al. 1985, Francis 1996, Wintner and Cliff 1999, Malcolm et al. 2001)
Juveniles TL	1.75-3 m	Global	(Malcolm et al. 2001)
Subadults TL	3 m – Mature TL	Global	(Martin 2005, Bruce et al. 2006)
Adults TL	Mature to Max TL	Global	(Francis 1996, Pratt 1996)

[1]L_∞ = theoretical maximum length obtained
[2]t_0 = theoretical age at L = 0
[3]K = growth coefficient, proportional to rate at which L_∞ is reached

2.2. Distribution and Habitat use

Patterns of occurrence in space and time provide important insight into a species' ecology, life history, and stock structure and connectivity between regions, as well as potential sources of mortality from fisheries, loss of prey resources, or habitat destruction. As a consequence, considerable effort has been expended to understand the distributional patterns of white sharks. Efforts have advanced over the years from simple sighting, catch, and attack data to the use of electronic tags, photo-ID and genetics. All results reveal a habitat separation based on age classes categorized as YOY, juveniles, subadults and adults, as defined above. Larger sharks also show spatial segregation by sex. A summary of both historic and recent findings for different age classes are presented below.

2.2.1. YOY and juveniles

2.2.1.1. Distribution based on catch data

The most comprehensive historic review of patterns of NEP white shark distribution based on catch data is provided by Klimley (1985), who analyzed catch records from 1936 to 1984 over a broad size range. The catch records included 70 individuals that were less than 300 cm TL, providing insights into early life history. YOY white sharks were caught south of Point Conception (with one exception), whereas juveniles were caught both north and south of Point Conception. Given the distribution of YOY and the fact that adult white sharks were most frequently sighted and caught off central and northern California, Klimley (1985) hypothesized that the southern California Bight was a nursery for white sharks. A second more recent analysis of fishery interactions in Southern California between 1936 and 2009 examined catch data for 143 YOY and 77 juveniles and found a similar spatial pattern in YOY and juvenile distribution (Lowe et al. 2012). Both analyses also provide insight into temporal patterns. YOY first appear in incidental catch records in April, with an increase by June and a peak in August (Figure 2.1) (Klimley 1985, Domeier 2012, Lowe et al. 2012). Both YOY and juvenile white sharks are caught predominantly in near shore waters (<50m depth), and most commonly by entangling net fisheries (e.g., set and drift gillnets) (Klimley 1985, Lowe et al. 2012).

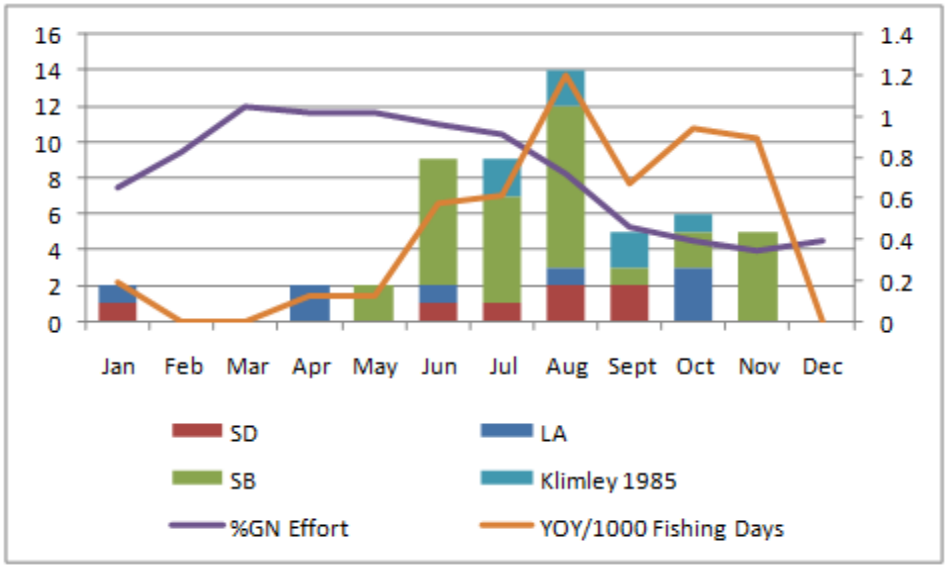

Figure 2.1 Seasonal pulse of YOY white sharks incidentally captured and landed in the set gillnet fishery, based on California Department of Fish and Wildlife (CDFW) set gillnet data for 1980 - 2000 (Figure 16.2 from Domeier, 2012). White sharks were landed in Santa Barbara (SB) (green bars), Los Angeles (LA) (blue bars) and San Diego (SD) (red bars). Light blue bars are from Klimley (1985) for which capture location data were missing. Set-gillnet fishing effort was plotted by month (purple line). The number of YOY caught per 1000 set gillnet fishing days are also shown (orange line).

Juvenile and YOY white sharks have also been incidentally caught on the Pacific side of the Baja California peninsula, Mexico, also in near-shore habitats. Santana-Morales et al. (2012) compiled data from (1) surveys of artisanal-fishing camps, (2) surveys at sites where carcasses are discarded, (3) log books from a commercial drift gillnet vessel, and (4) observations onboard a commercial drift gillnet vessel, mostly from the northern and central portions of the Baja Peninsula. Between 1999 and 2010, 111 juvenile and YOY white sharks (ranging from 123 to 274 cm TL) were reported to have been incidentally caught along the Pacific coast of Baja California. Of these, 60% were caught near Sebastián Vizcaíno Bay, and 80% were YOY (Santana-Morales et al. 2012). Seasonal patterns of occurrence were similar to those observed in the southern California Bight with a peak in white shark catch in the spring and summer.

Additional catch data come from the Gulf of California. Incidental catch of 14 juveniles has been documented within the Gulf of California between 1981 and 2007. None of these were estimated to be YOY (Galván-Magaña et al. 2010) and to date only one YOY has been documented from the Gulf of California (O. Sosa-Nishizaki, Centro de Investigación Científica y de Educación Superior de Ensenada (CICESE), pers. comm.). The data for the Gulf of California came from (1) records from local newspapers; (2) white shark sightings made by scientists and reliable divers; and (3) shark parts or photographs examined by the authors.

2.2.1.2. Distribution information from electronic tagging data

In recent years, more detailed information on the movements of individual sharks has been obtained from electronic tags. The satellite tags used on YOY and juveniles are both the pop-up satellite archival tags (PSAT) and the Smart Position and Temperature (SPOT) tags (Wildlife Computers Redmond, WA). PSAT tags remain attached for a programmed period of time after which they release and transmit logged data to satellite. SPOT tags are generally attached to the dorsal fin and transmit to the satellites when the shark is at the surface. Similar to the catch data, a number of tagging studies reported that YOY stayed between Point Conception and Sebastián Vizcaíno Bay, while a 3-year old juvenile moved north to Point Reyes (Dewar et al. 2004, Weng et al. 2007b, Weng et al. 2012). The YOY showed seasonal movements within the apparent nursery region. Weng et al. (2007b) found that while YOY white sharks were in near shore waters off California in the summer, in the autumn months (October and November) they travelled south along the coast of Baja to Ensenada and Sebastián Vizcaíno Bay. It should be noted that tags were only deployed in the southern California Bight and only recorded data for 1-2 months; longer deployments may reveal more extensive movements. Weng et al. (2012) also released 5 tagged YOY and juveniles following a period of captivity at the Monterey Bay Aquarium (sizes at release were 137, 166, 176, 184, and 187 cm TL). These sharks were tracked for up to 4 months and travelled as far south as Cabo San Lucas and into the Gulf of California, Mexico.

2.2.1.3. Summary of information on YOY and juvenile distributions

Based on catch and electronic tagging data, YOY and juvenile white sharks in the NEP are found along the continental shelf in the near shore regions of Southern California, the Pacific coast of Baja, and into the Gulf of California (
Figure 2.2). YOY are documented primarily from Point Conception to Sebastián Vizcaíno Bay and exhibit a north-south seasonal movement in this region. This appears to be the key nursery region, within which there are likely preferred areas including the Ventura Flats (south of Santa Barbara, CA), Santa Monica Bay, San Pedro Shelf (south of Long Beach, CA) and Sebastián Vizcaíno Bay (offshore of Guerrero Negro, Baja California Sur). These areas all have a broad continental shelf. The range of juveniles extends both north of Point Conception and into the Gulf of California. Additional data are needed in the Gulf of California and along the southern Baja Peninsula to better define the differences in habitat use between YOY and juveniles.

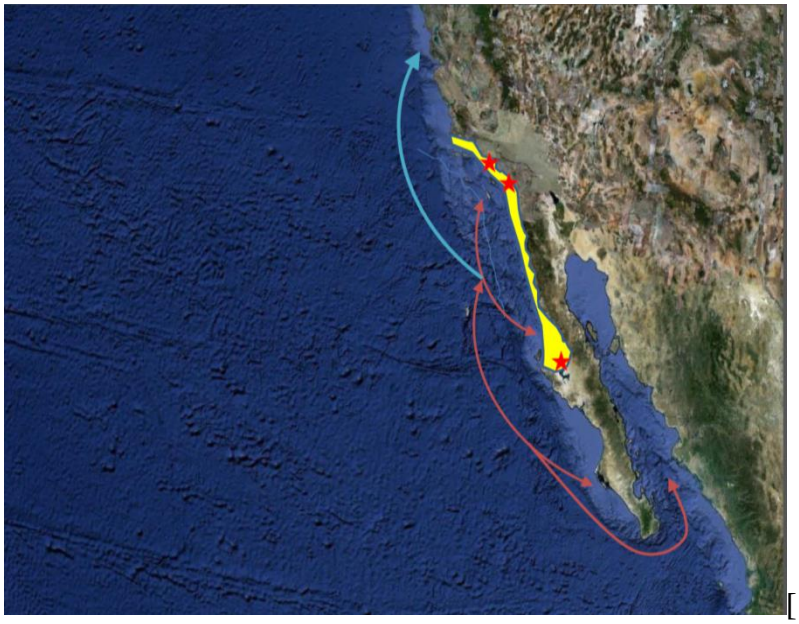

Figure 2.2 Distribution of YOY and juvenile white sharks in the northeastern Pacific Ocean. Young-of-the-year (YOY) white sharks are found in the nursery region (yellow) located from southern California to northern Baja California with several hot spots including Ventura Flats, San Pedro Shelf , Santa Monica Bay and Sebastián Vizcaíno Bay (red stars from north to south). Tagging data of YOY and juveniles in southern California have shown movements within this nursery region, with YOY in California waters during summer months and in northern Baja California in the fall and winter months, as well as movement by juveniles into the Gulf of California (pink arrows). Older juveniles and subadults begin to exploit habitats north of Point Conception (blue arrow). (Figure 16.10 from Domeier, 2012)

2.2.2. Subadults and adults

2.2.2.1. Distribution based on catch, sightings and attacks data

Insights into the range of white sharks and historical distribution of subadults and adults in the NEP are obtained from tagging and catch data as well as attacks on humans and marine mammals, strandings, and sightings. Again, Klimley (1985) provides the most comprehensive historic review of patterns in adult and subadult distribution. Between 1936 and 1984 along the west coast of North America, subadults and adults were caught predominantly north of Point Conception with the largest concentration being found off central California from Tomales Point to Monterey Bay (Klimley 1985) in regions also known for pinniped rookeries. The majority of attacks on humans and pinnipeds also occurred within these same regions of the coast, as well as in river mouths and harbors (McCosker and Lea 1996). In California, 14 mature white sharks greater than 460 cm TL were reported caught between 1959 and 1984 (9 female, 2 male, 3 unknown). Because a higher percentage of females were caught south of Point Conception than north, Klimley (1985) hypothesized that the females travelled south of Point Conception to give birth, consistent with the hypothesis that the area south of Point Conception is a nursery region.

An examination of attack data provides some insight into seasonal trends. Results from a number of studies reveal that attacks on both humans and marine mammals have been documented in all months indicating the presence of large sharks along the coast year around (Ainley et al. 1985, McCosker and Lea 1996). Patterns in attacks on marine mammals suggest an apparent seasonal shift in prey. Most predatory events closer to shore on harbor seals were reported to peak in June (Ainley et al. 1985), whereas predatory events for all pinnipeds were reported to peak in August (Long et al. 1996). At Southeast Farallon Island, the majority of predatory events are observed between September and November (Klimley et al. 1992).

While Klimley (1985) reported on records as far north as the southern end of Queen Charlotte Island off the Alaskan coast, the most comprehensive review of patterns at the northern extent of the white shark range comes from Martin (2004) (Figure 2.3). Between 1961 and 2004, there were 29 documented sightings, captures, and strandings of subadult and adult white sharks of both sexes in British Columbia and Alaska, most frequently in summer and autumn months (Martin 2004). Martin (2004) found that El Niño events did not have an impact on the frequency of white shark sightings, captures, and strandings in these areas, and that there was no discernible trend over the years. In the most recent year of the study (2004), four individuals were reported.

The southern extent of the range of NEP white sharks along the North American coast appears to be Mexico (Figure 2.3). The presence of subadult and adult white sharks has been documented as sightings and incidental catch within the Gulf of California (Galván-Magaña et al. 2010, Castro 2012). Between 1964 and 2012, Galván-Magaña et al. (2010), Castro (2012) and M. Domeier (Marine Conservation Science Institute (MCSI), pers. comm.) documented the killing of 8 mature white sharks in the Gulf of California (TL estimated between 450 and 600 cm) and 10 subadults (TL estimated between 300 and 400 cm). Galván-Magaña et al. (2010) noted that adults were most common within the Gulf of California from December to May and were less common from June to October. Beginning in the late 1990s, subadult and adult white sharks were observed in increasing numbers near Guadalupe Island, Mexico. While white sharks had been observed infrequently at Guadalupe Island prior to this time, the island has been a popular stop for sport fishing vessels for decades, by the early 2000s their occurrence was predictable enough in late summer and autumn months to support a new a cage diving industry.

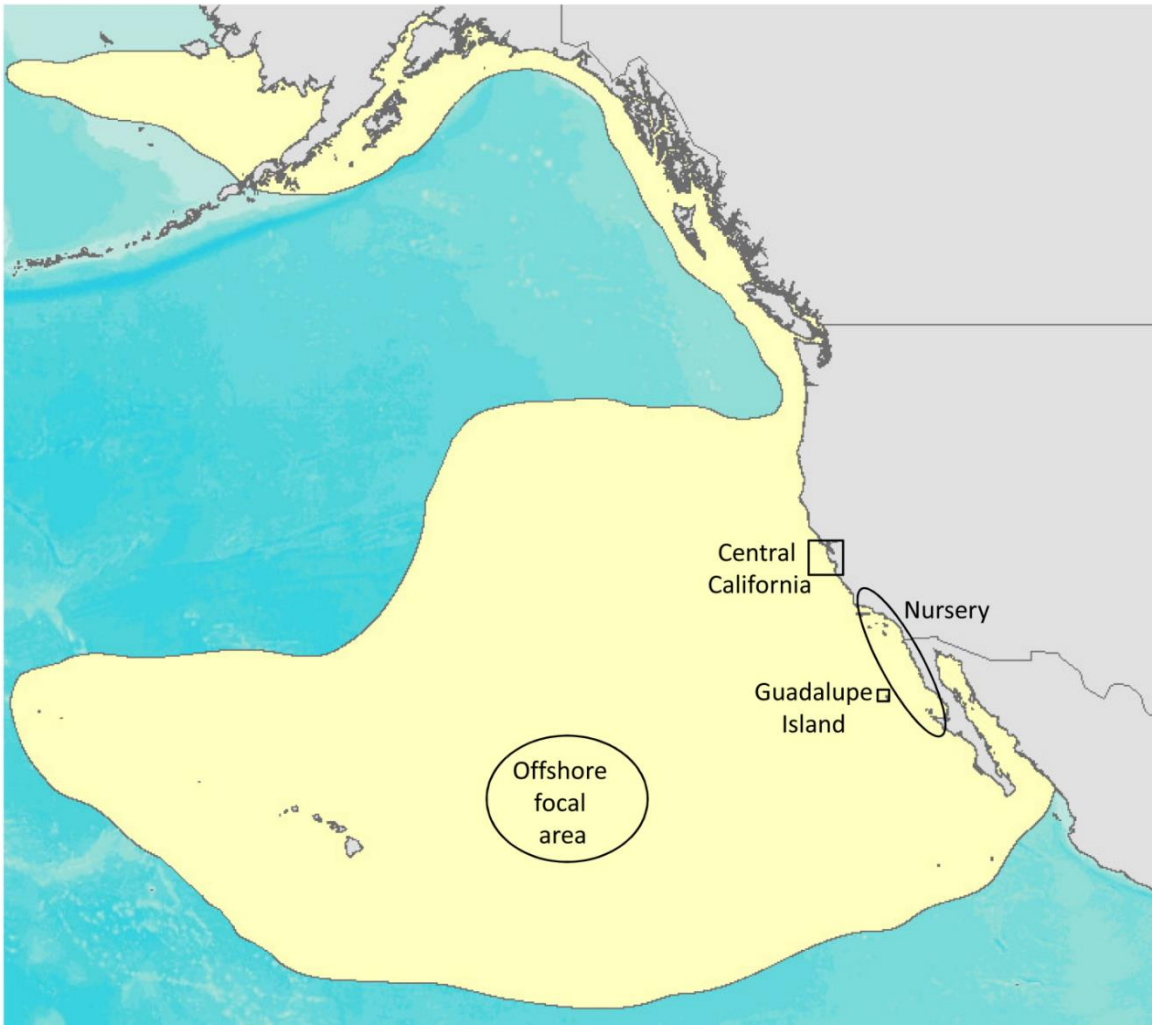

Figure 2.3. Known range of white sharks in the NEP. White sharks have been found as far north as Alaska, as far south as Mexico and as far west as the Hawaiian Islands. Data sources are from Klimley 1985, Martin 2004, Weng et al. 2007, Domeier and Nasby-lucas 2008, Galván-Magaña et al. 2010, Jorgensen et al. 2010, Domeier 2012.

The western extent of the white shark range in the NEP appears to be the Northwest Hawaiian Islands (Figure 2.3). White shark teeth are found in Hawaiian artifacts revealing a historic occurrence at the islands. However, white sharks are rarely caught or observed around the Hawaiian Islands. From 1926 to 2011 there were 14 confirmed observations of white sharks ranging from 3.3 to 4.5 m TL (Taylor 1985, Weng and Honebrink 2013). The fact that no YOY or juveniles sharks were caught indicates that Hawaii is probably not a nursery ground for white sharks. Additional insights into movements around Hawaii come from the electronic tagging data discussed below.

While white sharks have a broad distribution in the NEP, as in most oceans, the majority of adult white shark activity is observed near coastal sites and islands that serve as pinniped rookeries. In the NEP, Southeast Farallon Island has been one of the most predictable sites for observing white sharks. Southeast Farallon Island serves as a rookery for a number of different pinniped species including northern elephant seals (*Mirounga angustirostris*), California sea lions (*Zalophus californianus*), northern fur seals

(*Callorhinus ursinus*), Steller sea lions (*Eumetopias jubatus*), and harbor seals (*Phoca vitulina*). Other nearby sites where white sharks have been predictably observed include Tomales Point, Point Reyes and Año Nuevo Island (Klimley 1985, Jorgensen et al. 2010). Farther to the south, Guadalupe Island has more recently become an important aggregation site. The majority of white shark research in the NEP focuses on adults and subadults visiting these coastal aggregation sites.

2.2.2.2. Photo-Identification data

The consistent presence of adult and subadult white sharks at near shore aggregations sites has provided the opportunity for conducting photo-identification (photo-ID) studies. Photo-ID studies provide a valuable tool for studying the biology of adult and subadult white sharks for which limited fisheries data are available. White sharks are a good candidate for this approach given the ability to easily identify individual sharks using unique characteristics, and their propensity for repeat visits to near-shore locations where they can be photographed and identified (Cailliet 1996). Studies conducted to date at both central California and Guadalupe Island, Mexico have provided valuable data on demographics, clues to maximum age, visitation patterns over the time-series and regional abundance (Anderson et al. 2011, Chapple et al. 2011, Nasby-Lucas and Domeier 2012).

Concerns have been expressed regarding the violation of model assumptions when using photo-ID to estimate population sizes (Marshall and Pierce 2012, Delaney et al. 2012, Sosa et al. 2012) and some suggest that results are better used as an index of abundance (Sosa et al. 2012). Noted limitations include the potential for losing or changing marks over time (Marshall and Pierce 2012) and missing individuals that are actually at the site but not photographed (Delaney et al. 2012). Delaney compared acoustic data and photo-ID data and found that photo-ID had the lowest probability of detection. There are additional factors that violate the assumption of an equal probability of sighting such as dominance patterns where subordinate are less likely to approach the surface for photographing (Klimley 1994, Goldman and Anderson 1999, Sperone et al. 2010) and where sharks avoid baits and decoys once they learn that it does not lead to a food reward (Laroche et al. 2007). In addition, not all sharks in a given population may visit the study sites. The accuracy with which photo-ID studies estimate actual abundance will become clearer as more studies comparing methods become available.

Central California:
A photo-ID study was initiated at Southeast Farallon Island in 1987 (Anderson et al. 1996). In 1998, the study was expanded to include coastal areas near Tomales Point. High-resolution photographs and video were taken of each shark for comparison of dorsal fins. Only images with sufficient quality (based on angle, size, focus, and contrast) were added to the database. The data indicated that the same individuals return to the area in subsequent years, with males typically returning on an annual basis and females on a semi-annual basis (Anderson and Pyle 2003, Anderson et al. 2011). In this study a total of 364 individual dorsal fins were photographed between 1987 and 2008 and nearly twice as many confirmed males were sighted as confirmed females (Females:Males = 1:1.8). This

ratio however may be biased because it is easier to confirm the presence of claspers than the absence. One male has been identified at Southeast Farallon Island over a period of 22 years (Anderson et al. 2011). A subset of the photo-ID data collected at the Southeast Farallon Island and Tomales Point (2006-2008) were used in the study by Chapple et al. (2011) to estimate white shark abundance for central California. Sex and size data from this study are provided in Section 4.4.2. The BRT used these data from Chapple et al. (2011) to estimate abundance, sex ratio and size composition.

Guadalupe Island:
A photo-ID study was also initiated at Guadalupe Island in 2003 (including data from 2001 and 2002). This study used underwater photos and video taken by caged divers, beginning in 2001, from a protected anchorage at the northeastern corner of the island (Guadalupe Island is 41 km north/south and 15 km east/west). Sharks were identified by pigment patterns in the gill, pelvic fin and caudal fin region, along with dorsal fin notches. A shark was definitively identified once reference photos of both the left and right side were collected. As of 2011, the database consisted of 142 individual sharks. A recent abundance estimate is provided in Section 4.4.2.

Photo-ID has provided important insights into the demographics and dynamics of white sharks visiting Guadalupe Island. The overall sex ratio for sharks in the database was 1:1.6 (Females:Males). As in the central California region, adult males return annually and several males have been seen at Guadalupe Island every year since 2001. Most females, return semi-annually, being sighted every other year, whereas some return in consecutive years (Domeier and Nasby-Lucas 2007, Nasby-Lucas and Domeier 2012). White sharks are present at Guadalupe Island between late July and December (although effort past December was minimal over the study period) with differences in the arrival of males and females. Males arrive in late July to early August and sightings peak in late August and early September whereas females arrive September to October and sightings peak in November (Figure 2.4). As a result of the variation in arrival times, the sex ratio changes throughout the season. As with the data from Chapple et al. (2011), all Guadalupe Island photo-ID data were used by the BRT to estimate abundance, sex ratios and trends.

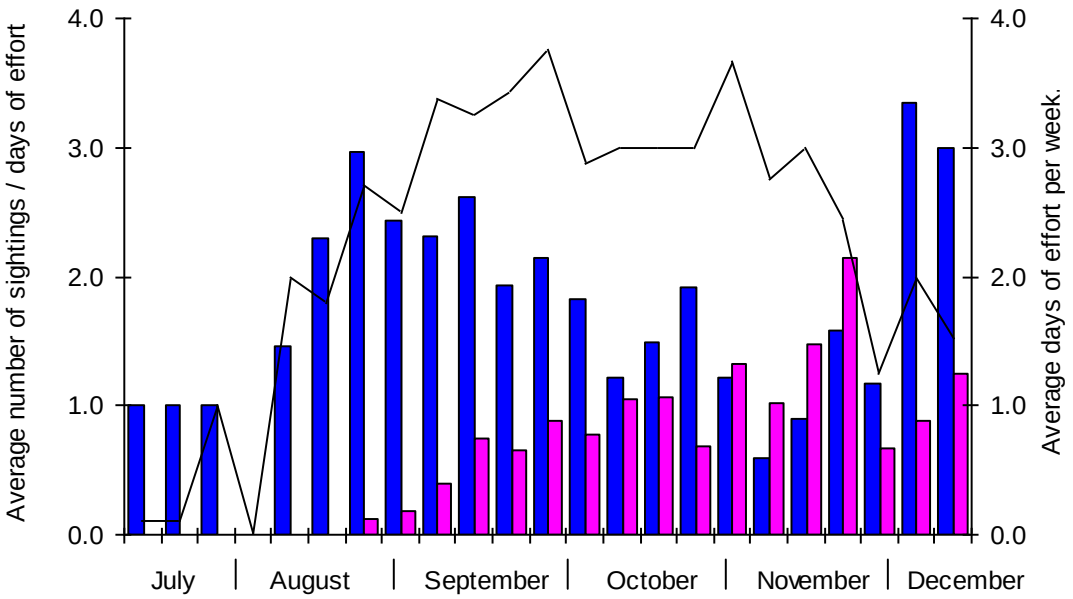

Figure 2.4 Average number of sharks sighted by week at Guadalupe Island adjusted by weekly sighting effort. Blue indicates male, pink indicates female, and the black line shows average number of days of effort for that week over all years of data (Figure 25.3 from Nasby-Lucas and Domeier 2012).

2.2.2.3. Electronic Tagging data

The development of satellite technologies has dramatically expanded our understanding of habitat use and distribution of subadult and adult white sharks. PSAT and SPOT tags have been used to track the movements of adult and subadult white sharks at both Southeast Farallon Island and Guadalupe Island. The PSAT tags can be deployed without handling the animal but the light and sea surface temperature-based geolocation estimates necessary to create an actual track, specifically for latitude, have large errors. Error estimates calculated by Teo et al. (2004) for latitude ranged from 1.16 to 1.89^{o} (>100 km). In addition, tracks typically span less than 12 months. For SPOT tags, sharks must be restrained to attach the tags but the position estimates are more accurate (ranging from 100s of m to several km (Patterson et al. 2010), and tracks spanning greater than one year are more common.

Studies using PSAT tags have shown that sharks from both central California and Guadalupe Island make long range migrations to the middle of the NEP (Boustany et al. 2002, Weng et al. 2007a, Domeier and Nasby-Lucas 2008). Some sharks move as far west as Hawaii (22% of tagged sharks from central California (Jorgensen et al. 2010) and 12% of tagged sharks from Guadalupe Island (Domeier and Nasby-Lucas 2008)).

Sharks from both central California and Guadalupe Island were found to travel to the same general offshore focal area and then return to the original area where they were tagged (Weng et al. 2007a, Domeier and Nasby-Lucas 2008, Jorgensen et al. 2010) (Figure 2.5). This offshore focal area has been termed the white shark “Café” or the Shared Offshore Foraging Area (SOFA) by the different research groups based on the

presumed role for the region. However, there is still a lack of consensus on the role of the offshore migrations, and whether this serves as a period of foraging or mating, as discussed below (Domeier 2012, Jorgensen et al. 2012a). Consequently the more neutral term "offshore focal area" is used here.

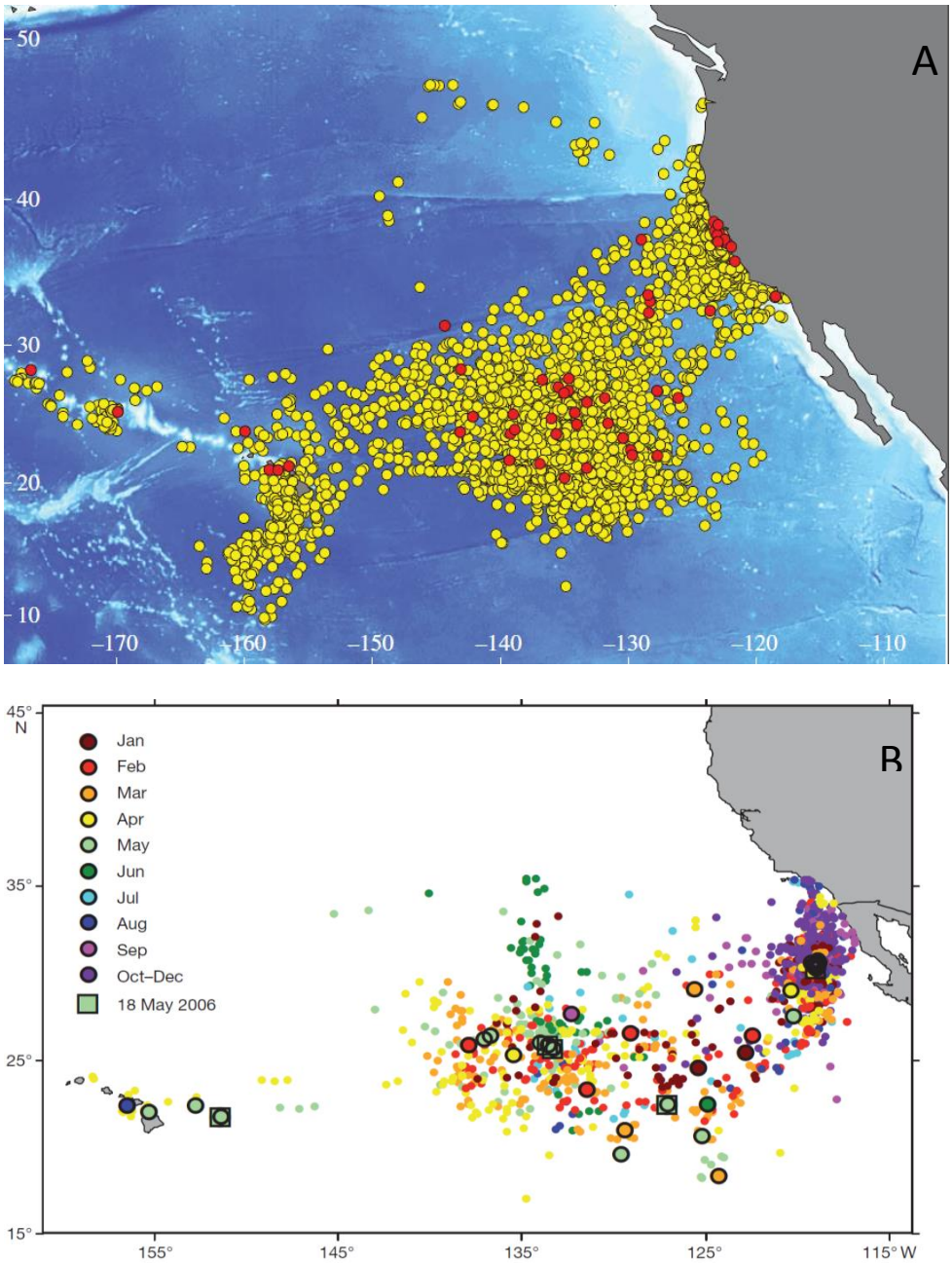

Figure 2.5 PSAT tracking data for white sharks tagged at the Farallon Islands (A) from Jorgensen et al. 2010, and at Guadalupe Island (B) from Domeier and Nasby-Lucas 2008. Satellite tag end points are indicated by (A) red circles and (B) larger bold circles. The remaining geolocation estimates were calculated using light data. The spread in the points reflects some uncertainty associated with errors in latitude estimates.

The tagging data provide greater detail than the sightings data mentioned above, and confirm that the two aggregation sites show different temporal patterns of occupancy. Sharks from central California depart for offshore areas earlier than those at Guadalupe Island. Sharks from Southeast Farallon Island began their offshore migrations between mid-November and the end of March (Weng et al. 2007a) and sharks from Guadalupe Island left between late December and early May (Domeier and Nasby-Lucas 2008). At Guadalupe Island, the return to coastal aggregation sites is staggered with males arriving in late July to early August and females arriving from mid-September to early October. Acoustic data should provide insight as to whether the same pattern occurs at central California. Consistent with the photo-ID observations (Anderson and Pyle 2003, Nasby-Lucas and Domeier 2012), satellite tagged males from both regions have a 1-year migration cycle (Weng et al. 2007a, Domeier and Nasby-Lucas 2008). Because the PSAT tags typically only last for 12 months or less, the semi-annual pattern of occupancy (returning every other year) apparent for females in the photo-ID data (Anderson and Pyle 2003, Nasby-Lucas and Domeier 2012) cannot be examined using this technology.

The long-term movement patterns of females tagged at Guadalupe Island have been documented using SPOT tags for which deployment durations greater than 2 years are not uncommon. Results confirm that females typically do not return to aggregations sites on a yearly cycle (Domeier and Nasby-Lucas 2012). Mature females remain offshore for around 15 months, which is presumed to be associated with an 18 month gestation cycle (Domeier and Nasby-Lucas 2012). Following 15 months offshore, four females were tracked to the coast between the months of April and August (Figure 2.6). This time period corresponds with the seasonal presence of YOY and suggests that these females travelled to the coast to give birth. Two of the females were tracked into the Gulf of California (April through July) and 2 were tracked to the Pacific coast of Baja California, Mexico, near Sebastián Vizcaíno Bay (June through August) (Domeier and Nasby-Lucas 2013). Three of the four females then returned to Guadalupe Island in late September to early October, after the males would have returned (the fourth tag stopped transmitting once the shark exited the Gulf of California and she has not since been resighted at Guadalupe Island).

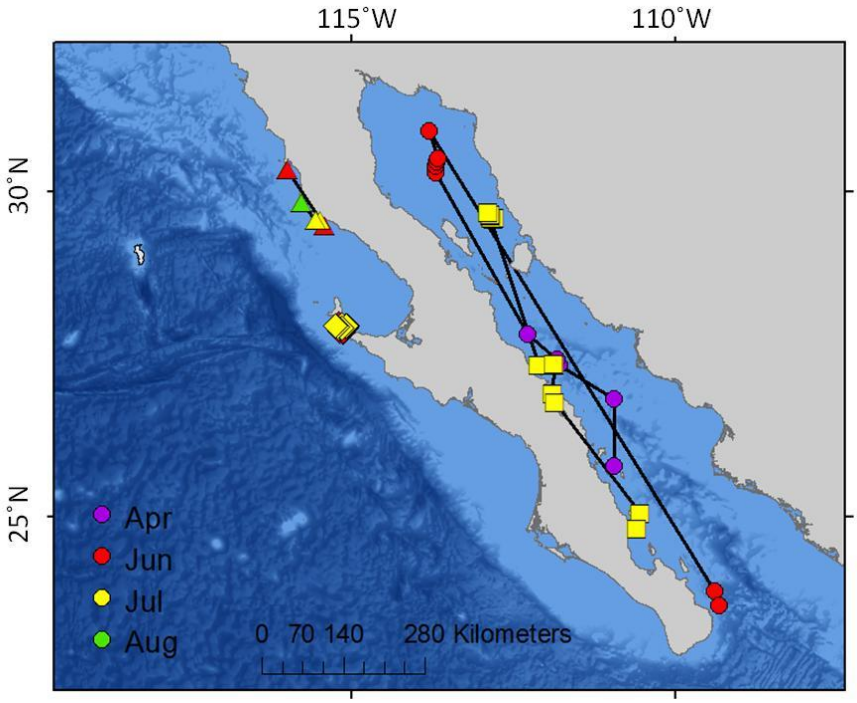

Figure 2.6 Coastal location data for four SPOT-tagged Guadalupe Island female white sharks during the pupping season following 15 months offshore. Location data for each shark is indicated by shape (shark1: triangle; shark 2: square; shark 3: circle; and shark 4: diamond) and by color for each month. (Figure 2 from Domeier and Nasby-Lucas, 2013).

Using the results from both tag types and both aggregation sites provides some insight into the overlap between males and females in the offshore focal area. There appears to be sexual segregation, with males from both regions consistently using the relatively confined offshore focal area (Figure 2.7a) while females cover a larger pelagic realm and spend less time in the offshore focal area (Figure 2.7b) (Jorgensen et al. 2010, Domeier and Nasby-Lucas 2012) resulting in limited overlap in offshore waters (Figure 2.8). For example, one tagged adult female white shark spent 8 months at least 600 km to the east of the core offshore focal area (i.e., the 50% density contour of the offshore area used by males, Domeier and Nasby-Lucas 2012) and only moved further to the west once the males had returned to Guadalupe Island.

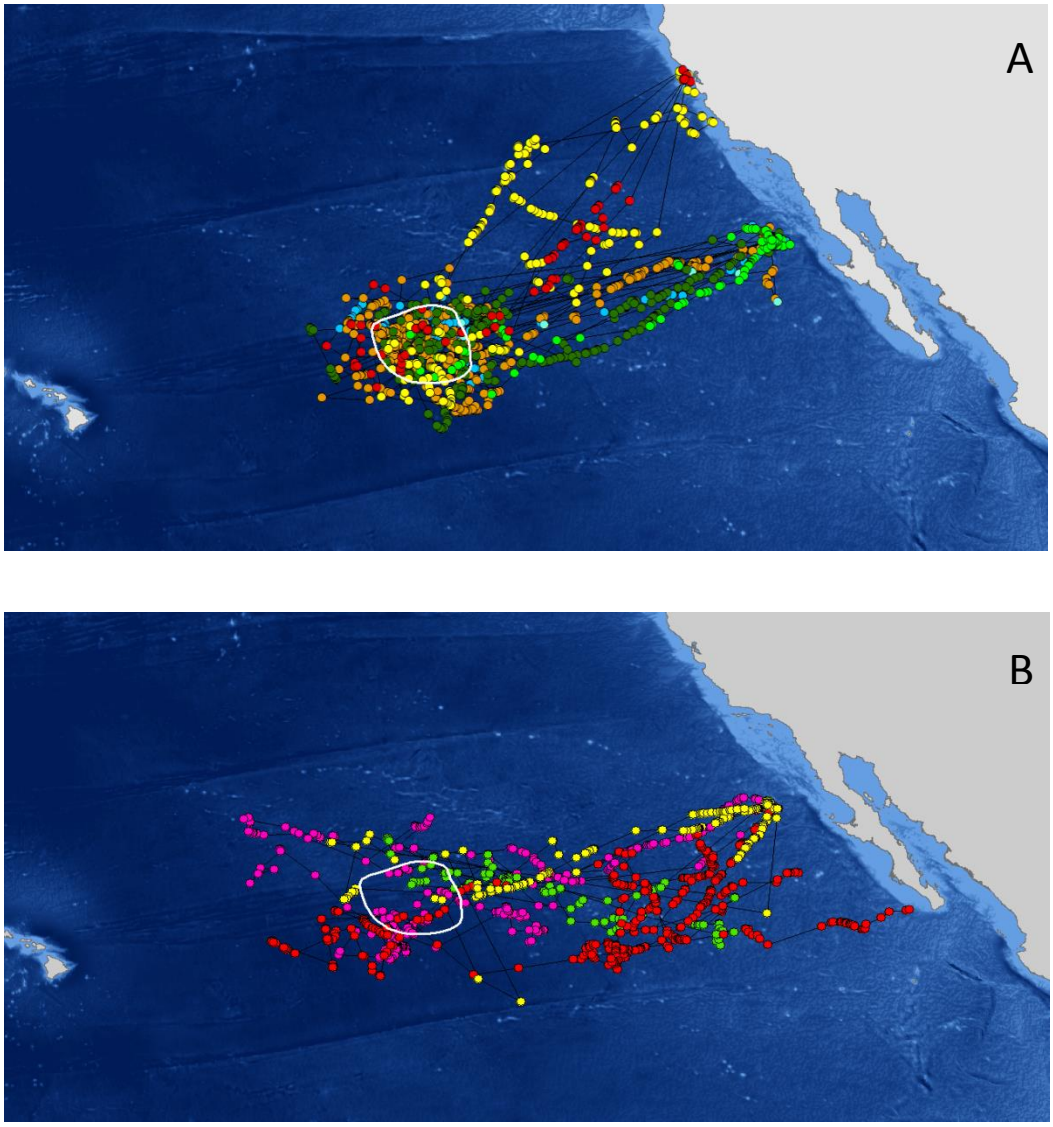

Figure 2.7 Multi-year tracks from (A) eight SPOT-tagged male white sharks tagged at Guadalupe Island and the Farallon Islands and (B) four females SPOT-tagged at Guadalupe Island. The white contour represents the 50% density contour, or "core," for the offshore focal area used by adult males (Domeier and Nasby-Lucas 2012).

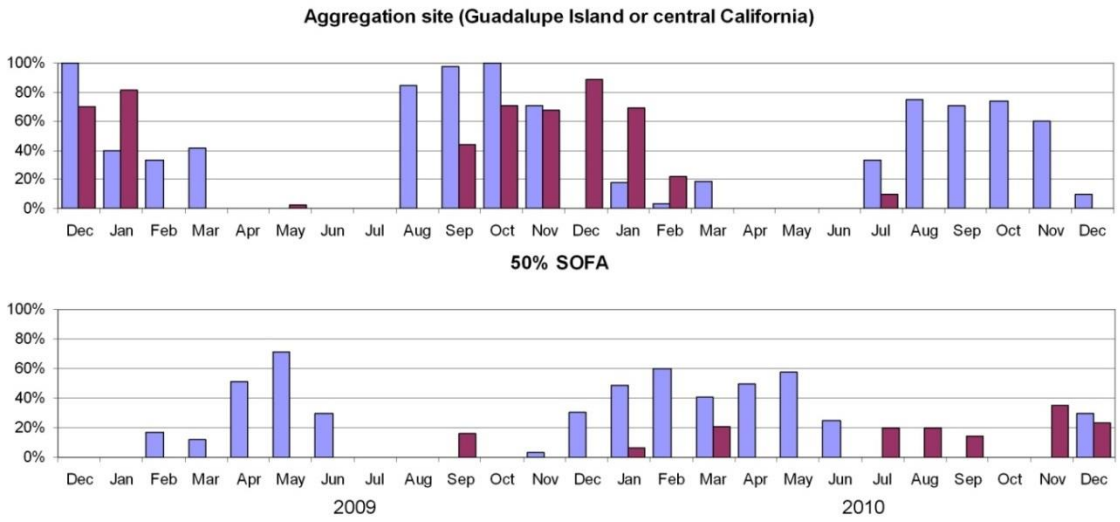

Figure 2.8 Comparison of temporal overlap while in the offshore focal area and at the coastal aggregation site for male and female white sharks tagged at Guadalupe Island Percentage of hits for SPOT tagged males and females that were at the coastal aggregation sites (top panel) or within the 50% density contour of the offshore focal area (bottom), calculated by month for each year. Males are colored blue and females are dark red (Domeier and Nasby-Lucas 2012).

The debate about the habitat function of both the offshore focal area and the coastal aggregation sites centers around whether the areas are used primarily for foraging or mating (Nasby-Lucas et al. 2009, Jorgensen et al. 2010, Domeier 2012, Jorgensen et al. 2012a, Domeier and Nasby-Lucas 2013).

The following observations support the view that the offshore focal area serves as a mating area (Jorgensen et al. 2010, 2012)

1) Jorgensen et al. (2012a) identified a unique dive pattern in the offshore focal area and proposed that the vertical and geographic movements supported a lek-style mating strategy with both sexes aggregating to facilitate female mate choice. The unique dive pattern is different from what would be consistent with feeding on the deep scattering layer, is dominant in the core of the offshore focal area, and increases at the time when mating is proposed to occur.
2) Carlisle et al. (2012) report that foraging is limited in the focal area, contributing about 25% of an individual's total dietary input based on isotope data.
3) Mating scars are rarely seen at the central California coastal aggregation sites and any seen are considered to be partially healed and having not occurred at the coastal aggregation sites.
4) No mating behavior has ever been observed at the central California coastal aggregation sites.
5) The coastal aggregation sites are thought to be productive foraging areas with energy rich prey indicating its potential as a foraging site, not that foraging and mating are mutually exclusive.

The following observations support the view that the coastal aggregation sites serve as a mating area (Domeier 2012, Domeier and Nasby-Lucas 2013):

1) No mating behavior has been observed at Guadalupe Island, but wounds around the head and gills are frequently observed on females at Guadalupe Island just prior to the beginning of the 2-year migration cycle for females. This suggests that the wounds are linked to mating activity and that the wounds are obtained prior to their movement offshore.
2) During the 2-year migration cycle for females, some females that were presumed to have been pregnant, avoided the offshore focal area after leaving Guadalupe Island and prior to visiting the nursery grounds.
3) The presence of seminal fluid was documented in the claspers of three males at Guadalupe Island and two males tagged at the central California coastal aggregation sites.
4) In one location off New Zealand, mating behavior was observed at a coastal aggregation site (Francis 1996) although this was in the spring rather than the fall or winter.
5) The offshore focal area has an abundance of squid and possibly other squid predators (i.e., potential prey for white sharks), indicating its potential as a foraging site, not that foraging and mating are mutually exclusive (Domeier et al. 2012).
6) The density of males and females is highest at the coastal aggregation sites. Increased density should enhance mating success. In the offshore focal area the 50% quartile covers an area with a radius of 250 km encompassing ~200,000 km^2.
7) White sharks depart the NEP coastal aggregation areas prior to the timing of peak pinniped use and breeding. This seems counterintuitive if these areas are primarily used for feeding on pinnipeds.

Given that the gestation period is not known, working back from the pupping period to estimate the timing of mating is not helpful. A gestation period of 15 months places mating in the offshore focal area, whereas a gestation period of 18 months places mating at the coastal aggregation sites (Domeier 2012). Therefore, additional information is needed to definitively determine when and where white sharks are mating. Understanding reproductive patterns is important for characterizing habitat use, better understanding the stock structure of the NEP white shark population, and examining whether there are significant subunits within that population.

2.2.2.4. Passive acoustic tracking

To complement the data obtained from the PSAT tags, researchers in central California used an acoustic array to document the coastal movements of white sharks in and around coastal aggregation sites. Acoustic tracking data for white sharks tagged in central California showed that upon their return to the coast following their offshore migration, tagged white sharks were detected by receivers at a number of central California locations. Movements between the different listening stations varied but occurred between all coastal aggregation sites. Four of these listening stations accounted for a disproportionately high number of detections during the coastal aggregation phase

(approximately Aug through February), revealing a preference for a limited number of key hotspots (Figure 2.9). Some sharks showed a distinct preference for certain sites.

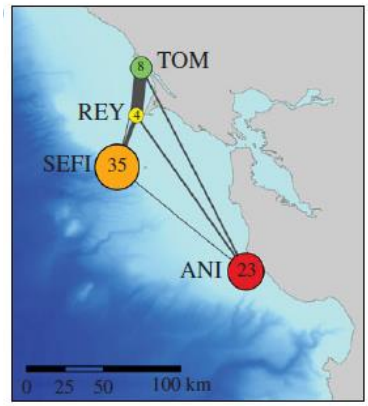

Figure 2.9 Detections of white sharks tagged with acoustic transmitters along the central California coast, with receiver locations scaled in size by the mean residence time of all individuals detected at that location (number of days in text). From north to south, the receiver locations include: Tomales Point (TOM; green), Point Reyes (REY; yellow), South East Farallon Island (SEFI; orange), and Año Nuevo Island (ANI; red). Lines connecting locations are scaled in thickness relative to the number of transits between adjoining sites. (Figure from Jorgensen et al., 2010; caption modified)

Despite their long-range migration patterns, satellite tagged white sharks from central California have not been tracked to Guadalupe Island, or vice versa. However, a female white shark SPOT tagged at Guadalupe Island migrated offshore and returned back to the coast to an area just off Point Conception (M. Domeier, MCSI, pers. comm.). In addition, out of 110 acoustic tags deployed at central California and 31 at Guadalupe Island, two white sharks from central California have been detected at Guadalupe Island and one Guadalupe Island white shark was detected on the central California coast ((Jorgensen et al. 2012b); S. Jorgensen, Monterey Bay Aquarium (MBA), pers. comm.). In addition, 10% of acoustically tagged sharks from central California have been detected at receivers located around Hawaii.

2.2.3. Overall distribution patterns

Diverse data sources, new and old, provide an overall picture of the distribution of white sharks in the NEP. The more recent efforts using electronic tagging technology and photo-ID complement previous work focusing largely on catch data and provide a clearer picture of temporal and spatial patterns in movements, especially for those sharks that are found at Guadalupe Island, Southeast Farallon Island and Tomales Point. The range of white sharks appears to be bounded by Mexico to the south, Alaska to the north, the coast of North America to the east and the Hawaiian Islands to the west (Figure 2.3). YOY and juvenile white sharks are restricted primarily to the coastal regions from Mexico to central California remaining largely over the continental shelf. As they make the transition from juveniles to subadults (at about 3 m TL in size), their diet expands to include marine mammals and their subsequent distribution in near shore waters is

strongly influenced by the distribution of their new prey. Also at around this size, white sharks also begin making large-scale, seasonal offshore migrations that differ for males and females. Most males move to the offshore focal area in the winter and spring where they remain until the following summer when they return to the near shore coast. Females, on the other hand, are more dispersed while offshore and arrive later at the coastal aggregation sites. Some adult females return the following year while others return to the coastal aggregation sites every other year after spending ~15 months offshore. Females appear to pup off of California or Mexico between April and August.

While the new tagging technologies and studies conducted at Guadalupe Island and off central California have provided a wealth of information about the movements and distributions of subadult and adult white sharks in the NEP, the picture is likely more complex. Deploying electronic tags only at aggregation sites may bias results, since individuals that might use other areas would not be available for tagging. Several gaps in our understanding have yet to be addressed. These include:

1) Recent data using isotopes to characterize diet across life history phases suggest that not all white sharks make the transition to marine mammal prey as adults and consequently would not necessarily be found at coastal aggregation sites, which are associated with pinniped rookeries (Kim et al. 2012).
2) In addition, there are white sharks documented in all months along the coast (Ainley et al. 1985, Klimley 1985) when the tagging data would indicate they should be offshore. Of the white sharks that have been tagged, very few seem to stay in or return to coastal waters during the offshore migratory phase (Jorgensen et al. 2010, N. Nasby-Lucas, MCSI, pers. comm.). This suggests that there may be a subset of the white shark population that remains near the coast during the offshore migratory phase. If these sharks do not visit the aggregation sites when tagging occurs, this pattern would not be documented.
3) Subadults appear to be underrepresented at the coastal aggregation sites (Section 4.4.2). These smaller sharks can be displaced by larger sharks and are sometimes observed at the aggregation sites before or after the peak fall season (Goldman and Anderson 1999, Sperone et al. 2010, Domeier 2012). Smaller sharks may use different areas due to a dominance hierarchy.
4) There are also white sharks observed far away from known coastal aggregations, in areas off California, Oregon, Washington, Mexico, Canada and Alaska (Klimley 1985, Martin 2004). Where these sharks fit into the overall picture remains to be determined.
5) Due largely to weather patterns, the majority of tagging efforts and observations both at Guadalupe Island and along the central California coast occurs in the fall. Thus, white sharks that may arrive later in the year would be underrepresented in the tagging and photo-ID studies.
6) The use of PSAT tags at central California limits tracks to a year or less. Consequently, there are no long-term tracking data (>12 months) for mature female white sharks off central California, and the full scope of their movements and reproductive patterns is not known.

2.3. Inferences on stock structure and population size using genetic data

In addition to the tagging data, genetic data also provide insights into stock structure and connectivity among groups along with estimates of historical abundance. Mitochondrial DNA (mtDNA) from white sharks sampled in central California and compared with white sharks from South Africa and Australia/New Zealand showed strong clustering of the California sharks with those from Australia/New Zealand. Furthermore, the data show that the California sharks form a unique monophyletic clade (i.e., a group developed from a single common ancestral form) of relatively recently derived lineages (Figure 2.10). It has been hypothesized that the NEP population was founded by Australia/New Zealand migrants during the Late Pleistocene (~150,000 years ago (150 kya)) and that subsequent strong homing behavior and reproductive site fidelity has maintained the separation of the population (Jorgensen et al. 2010). More recent analyses of mtDNA of only 7 white sharks caught off Japan suggests that Japanese white sharks also form a monophyletic clade and are separate from the populations off central California, Australia/New Zealand and South Africa (Tanaka et al. 2011).

The patterns of genetic diversity (short branch lengths between multiple closely related haplotypes) suggest a rapid demographic expansion upon colonization of the NEP (Figure 2.10). Though the overall sample size for which genetic data are available is relatively low in all regions (California: n = 59; South Africa: n = 43; and Australia/New Zealand: n = 52), the observation that the NEP lineage is monophyletic and that no shared haplotypes have been observed between regions strongly supports that the NEP is genetically distinct from the other studied regions (i.e., there are no South Africa or Australia haplotypes in the California group and no California haplotypes in the South Africa /Australia groups; Figure 2.10). However, as only mitochondrial data are presently available and this locus is inherited maternally, these results only represent genetic patterns attributed to female behavior (Pardini et al. 2001). The future application of nuclear DNA markers will help us understand whether male-mediated gene flow follows similar patterns.

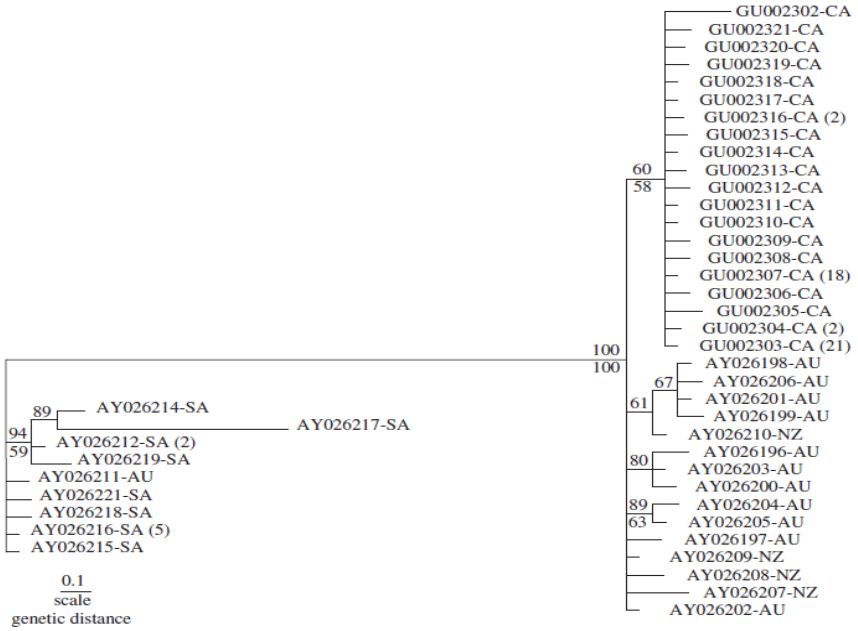

Figure 2.10 Phylogeny of unique haplotypes found in Indo-Pacific white sharks inferred from comparisons of mitochondrial DNA control region sequences (1109 base pairs) of 59 individuals from the central California coast with previously published sequences from South Africa (SA) and Australia/New Zealand (AU/NZ). Bayesian tree branch lengths reflect substitutions per site. The number of samples comprising each haplotype is given in parentheses if greater than 1. GenBank accession numbers are indicated for each haplotype followed by the location of the sample. Bayesian posterior probabilities are indicated above nodes while likelihood bootstrap values are shown below. NEP sharks from California (CA) form a monophyletic clade (bootstrap =58%, Bayesian posterior probability = 60%). Individuals from the two dominant lineages within this clade had similar migratory patterns. (Figure from Jorgensen et al., 2010)

The number of haplotypes expected in a given population depends, among other things, on the effective population size. For populations that are naturally at low abundance, the number of haplotypes is expected to be low and normally there would be no truly rare haplotypes (defined here as haplotypes found at frequencies equal to or less than 5%). As an example, for shark and cetacean populations with 1-5 haplotypes, the abundance of females in the population is in the low hundreds or less (Table 2.2). Higher haplotypic diversity is consistent with a population that is currently large or a population that was larger but suffered a dramatic population decline (>90%) in the last few generations (Hoelzel et al. 1993). After only a few generations genetic drift will reduce haplotypic diversity.

This relationship gives some context to the pattern of haplotypes observed in NEP white sharks. The number of haplotypes (n=20 from 59 individuals, Jorgensen et al. 2010) and the number of low frequency haplotypes for central California (n=18 of 20 haplotypes) are both high. It should be noted that five of the 20 haplotypes reported in Jorgensen et al. (2010) contained at least one ambiguous nucleotide and of these, two haplotypes were differentiated from other haplotypes solely by an ambiguity. Therefore, the true number of haplotypes from this study may be 15-18. This, however, is still relatively high.

The haplotypic diversity for white sharks and other species are shown in Table 2.2. Haplotypic diversity was characterized as the proportion of rare haplotypes among the haplotypes sampled, where 'rare' is defined as occurring in equal to or less than 5% of the sampled individuals. For direct comparison to other available white shark data (Blower et al. 2012) and to resolve many of these ambiguities, the data from Jorgensen et al. (2010) was trimmed to 842bp in length and the two haplotypes defined solely by an ambiguity were removed. This reduced the total number of haplotypes to 9 with 66.7% of the haplotypes being rare. For comparison to NEP white sharks, species were chosen that were long-lived, slow reproducers and not characterized by strong social structure. The haplotypic diversity of NEP white sharks is within the range of species where populations of adult females are in the high hundreds to low 1000s. This suggests that either a greater number of animals exist than are accounted for in current samples or that white sharks recently suffered a dramatic decline. The potential for a recent population decline is addressed by the BRT in Section 4.3 and Appendix B to determine whether the haplotypic diversity could be used to inform the current population size of NEP white sharks.

Table 2.2 Comparison of the haplotypic diversity for a range of long-lived and slow reproducing species. Abundance categories for all life stages were defined as low (in the low thousands or less), medium (in the low tens of thousands), and high (greater than the low tens of thousands). The term 'rare' is defined as occurring in equal to or less than 5% of the sampled individuals. Adult female abundance was extrapolated based upon life history parameters and predicted to be roughly 25% of the total population for each species. Listed are the relative level of abundance, n = sample size, #hap = number of haplotypes, %n rare = number of samples with rare haplotypes/ number of samples, %hap rare = number of rare haplotypes/ number of haplotypes. Abbreviations used in the table are: NEP (northeastern Pacific), BCB (Bering-Chukchi-Beaufort).

Abundance category	Population	n	#hap	%n rare	%hap rare	Notes on total abundance and adult female abundance levels
Low	vaquita (*Phocoena sinus*)	43	1	0.0	0.0	Total abundance naturally in low 1000s, adult females naturally in high 100s
Low	Okhotsk bowhead whales (*Balaena mysticetus*)	20	4	5.0	25.0	Total abundance naturally in low 1000s, reduced to 100s, some gene flow possible. Adult females naturally in high 100s
Low	North Atlantic right whales (*Eubalaena japonica)*	430	5	1.2	20.0	Total abundance reduced to low 100s many generations ago. Adult females likely <100.
Low	East Australian sandtiger shark (*Carcharias Taurus*)	65	1	0	0	Total abundance less than 500 individuals. Adult females likely <100. Probable decrease of 80-90% in the past 40 years (Ahonen et al. 2009)
Medium	Eastern gray whales (*Eschrichtius robustus*)	103	32	58.3	84.4	Total abundance reduced a few generations ago but quickly recovered to about 20,000. Adult females likely in mid 1000s.
Medium	NEP blue whales (*Balaenoptera musculus*)	50	13	22.0	61.5	Total abundance reduced a few generations ago probably from low 10,000s but recovered to low 1000s. Adult females likely reduced from

						high 1000s to high 100s
Medium	BCB bowheads (*Balaena mysticetus*)	343	68	69.4	97.1	Total abundance reduced from 10,000s to low 1000s a few generations ago but recovered to over 10,000. Adult females likely in low 1000s.
Medium	S Africa sandtiger shark	77	5	6.5	40	Total abundance likely stable in the 10,000s. Adult females likely in high 1000s (Ahonen et al. 2009, Dicken 2006)
High	Antarctic blue whales	78	36	53.8	80.6	Total abundance reduced from 100,000s to 100s a few generations ago.
High	NEP mako shark	382	35	30.9	88.6	Unknown population size or history but considerably larger than white shark population size. (Michaud et al. in prep)
High	SWP mako shark	66	23	40.9	78.3	Unknown population size presently or historically but considerably larger than white shark population size. (Michaud et al. in prep)
Unknown	SE Australia white sharks	61	13	18	53.8	Adult abundance likely less than 10,000 animals in Australian waters, adult females likely in low 1000s to high 100s. Unknown historical population size (Blower et al. 2012, Malcolm et al. 2001)
Unknown	SW Australia white sharks	30	9	10	33.3	Adult abundance likely less than 10,000 animals in Australian waters, adult females likely in low 1000s to high 100s. Unknown historical population size (Blower et al. 2012, Malcolm et al. 2001)
Unknown	NEP white sharks	57	9	10.5	66.7	Unknown population size or history. (Jorgensen et al. 2010)

3. DPS Determination

3.1. ESA discreteness and significance criteria

Joint National Oceanic and Atmospheric Administration (NOAA)/ U.S. Fish and Wildlife Service (FWS) policy defines a population to be a distinct population segment (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, NMFS will consider available scientific evidence for this significance. 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 NEP white shark population's behavior, geographic range and genetics that relate to both the discreteness and significance criteria. The BRT considered the best available biological and ecological information in addressing the discreteness criterion and did not consider whether or not the international boundary criterion was applicable because other information indicated

the population met the markedly separate condition. If the population is determined to be markedly separate, it is not necessary to address the international boundary factor since both conditions do not need to apply. Some discreteness and significance factors as defined by the DPS guidance were not considered in detail as the BRT determined there was no evidence to suggest that those factors were applicable to NEP white sharks. In particular, there is no evidence to suggest that NEP white sharks are discrete from other white shark populations based on ecological characteristics, or marked physical or physiological separation. The team also dismissed the significance criterion based on the discrete segment representing the only surviving natural occurrence of the species within its historical range, as white sharks are known to occur in large areas of both the Atlantic and Pacific Oceans.

Each remaining discreteness and significance criterion relevant to NEP white sharks was discussed separately and the arguments for and against each factor were examined in detail. As noted in the Introduction, the BRT used the Structured Expert Decision Making (SEDM) method to evaluate each discreteness and significance factor potentially relevant to NEP white sharks, with each team member asked to allocate 100 points to the arguments for and against each factor. Point allocations from all team members were combined to produce percentage plausibility for each argument, reflecting the BRT's opinion regarding the plausibility for the arguments for or against each factor. A score greater than 50% indicates that the arguments in favor of that factor carry more weight than the arguments against, and higher values indicated 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 C.

3.1.1. BRT determination of ESA discreteness

The BRT expressed strong support for a finding that the white sharks in the NEP are discrete from other white sharks found in the Atlantic and Pacific Oceans (Table 3.1). The team found that NEP white sharks are markedly separated from other white sharks based on behavioral (spatial) factors as supported by tagging efforts, photo-ID and genetics data. In particular, the team felt that the site fidelity, lack of apparent movement outside the NEP (Domeier and Nasby-Lucas 2008, Jorgensen et al. 2010) and lack of shared haplotypes with other white sharks from Australia/New Zealand and South Africa (Jorgensen et al. 2010), supported discreteness. All tagging and photo-ID data from the two NEP aggregation sites show consistent patterns of pelagic and near shore phases with adults and subadults, both male and female, moving between regions within the NEP. One individual has reportedly returned to the same site in central California intermittently for 22 years (Anderson et al. 2011). Tagging data for YOY and juveniles also show movements restricted to the coastal waters of North America (Weng et al. 2007b). Results from genetic studies using mtDNA reveal no shared haplotypes among the NEP white sharks and those studied from other regions (Jorgensen et al. 2010). This monophyletic pattern supports the inference that little movement occurs among these regions. To maintain genetic separation requires that less than one individual per generation moves between groups. Of note, however, is that the use of mtDNA only assesses female-mediated gene flow. While some uncertainty exists regarding sampling bias associated with the tagging and genetics, the BRT concluded that the weight of the

evidence did support recognition of NEP white sharks as behaviorally discrete from other white sharks in the taxon (93% of plausibility points: Table 3.1).

3.1.2. BRT determination of ESA significance

The BRT discussed the significance of the NEP white sharks with respect to the global taxon. In particular, the BRT focused on the following two factors:

- genetic information for NEP white sharks and differences from other white sharks in the Pacific and Atlantic Oceans; and
- the loss of the NEP white sharks resulting in a significant gap in their range.

The BRT concluded that NEP white sharks are significant to the taxon in which they belong (Table 3.1), based on the following: marked genetic differences, and significant gap in range if lost.

3.1.2.1. Genetics

The BRT evaluated the genetic significance of the NEP population by examining mtDNA haplotype data from white sharks sampled in central California, Japan, New Zealand/Australia, and South Africa (Jorgensen et al. 2010, Tanaka et al. 2011). Comparisons across studies revealed that no NEP haplotypes are shared with any other regions supporting the conclusion that NEP white sharks make up a unique monophyletic clade. The magnitude of mtDNA differentiation suggests that migration between sites is less than one migrant per generation and that enough time has passed to potentially allow adaptation to NEP habitat. While the mtDNA data only provide data regarding female gene flow, many of the central California white sharks included in the genetic study were adult males with photographic histories supporting fidelity to NEP aggregation sites. All of these males had haplotypes indicating an NEP origin. These observations are consistent with a closed population with little evidence for different patterns of male-mediated gene flow. However, this cannot be known with certainty until multi-locus nuclear DNA data are evaluated (Pardini et al. 2010, Gubili et al. 2012).

Potential issues with the genetic data include: 1) the small sample sizes for most studies, 2) the large number of unique haplotypes among the NEP white sharks sampled, suggesting that diversity is not fully characterized, 3) the use of only maternally inherited markers (mtDNA), and 4) bias associated with the sample collection in time and space (e.g., the NEP study only included samples from central California). Despite these potential issues, the BRT concluded that NEP white sharks are significant to the taxon based on marked genetic differences from other populations of the species (81% of plausibility points Table 3.1).

3.1.2.2. Significant gap in the range of the taxon

The BRT compared the range of NEP white sharks to the global distribution of white sharks to assess whether the loss of NEP white sharks would result in a significant gap in the range of the taxon. The BRT felt it is highly likely that loss of the NEP white sharks

would result in the loss of white sharks from half the North Pacific as well as along the west coast of North America and potentially around Hawaii. The BRT felt strongly that this represented a significant portion of the range of the taxon (96% of plausibility points) (Table 3.1).

3.1.3. Conclusions for DPS determination

The BRT concluded that the NEP white shark population met the DPS discreteness and significance criteria. Uncertainty expressed by the BRT, as measured by the minority of points assigned to arguments against individual discreteness and significance factors, primarily reflected an unknown degree of sampling bias in the available datasets, limitations of the genetic data and a lack of complete data for white sharks in other regions of the Pacific. However, the information that is presently available and the application of the best-available science standard, supports the BRT's conclusion that this population meets the DPS discreteness and significance criteria.

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 C for a complete list of point allocations by team member). One hundred points were assigned to arguments for and against each criterion by each team member, such that the total point allocation for each criterion is 800 points.

Criteria	Percentage (%) 'For'	Percentage (%) "Against'
Discreteness	93	7
Significance		
Gap in range	96	4
Genetics	81	19

3.2. Significant Portion of Species Range:

In addition to deciding whether the NEP white shark population met the DPS criteria, the BRT was asked to identify whether there are any specific portions of the species' geographic range within the NEP that are significant in terms of the population's overall viability, and which if lost would significantly increase the population's risk of extinction. This approach was taken to be consistent with the NMFS-FWS draft proposed policy on interpreting the phrase "significant portion of a species range" in the ESA definitions of threatened and endangered species (76 FR 76987; December 9, 2011). Under this draft proposed policy, a portion of the range of a species is "significant" if its contribution to the viability of the species is so important that, without that portion, the species would be in danger of extinction. The BRT used the SEDM method to determine the plausibility that there was a significant portion of its range (SPOR) within which NEP white sharks were at high risk.

To examine the potential for a SPOR, the BRT assessed what were considered to be the greatest known threats to the population and whether these were localized to a significant portion of the range or affecting the whole DPS. The only threat considered by the BRT

to be moderate in its severity (others were considered low risk) was the threat from fisheries bycatch (see Table 4.5). The largest known current threat is the bycatch of YOY and potentially juveniles in gillnets. Currently, the majority of bycatch occurs in Mexican waters, with the highest documented catch rates occurring between Sebastián Vizcaíno Bay and the U.S. - Mexico border, and some bycatch of YOY and juveniles in the southern California Bight. There is no evidence to suggest that the YOY and juveniles caught in fisheries off Mexico are distinct from animals caught in other locations (e.g., in U.S. waters). YOY and juvenile white sharks caught and tagged in the southern California Bight have been tracked to Mexican waters including Sebastián Vizcaíno Bay and the Gulf of California, indicating connectivity between Mexico and U.S. habitats. With respect to later life history stages, subadult and adult white sharks are found seasonally at known coastal aggregation sites (central California and Guadalupe Island, Mexico) but make a seasonal offshore migration. Males from both aggregation sites migrate to the same offshore habitat whereas females distribute more broadly when they leave the coastal aggregations. Consequently, potential areas of risk were considered to be throughout the range of the NEP white shark population with no portion of the range being separable. The BRT assigned 100% of plausibility points that the entire NEP white shark population should be examined for extinction risk and there was no portion of the range that should be considered separately.

4. Assessment of NEP White Shark Population Extinction Risk

After concluding that the NEP white shark population met the DPS criteria, the BRT evaluated the biological status of the population to determine whether it is at risk of extinction using the best available biological information. In assessing extinction risk of this population, the BRT evaluated both its present risk of extinction and the risk of extinction it may face in the foreseeable future to the extent it can be predicted. For the purposes of this extinction risk assessment, the BRT defined foreseeable future as the timeframe over which threats and other factors can be reliably predicted to impact the biological status of the NEP white shark population. To assess extinction risk, the BRT examined threats to the population, direct and indirect information on population trends, abundance of the population and associated biases with current estimates, and developed a white shark population model to assess the impacts of fishery mortality on the population under a range of estimated (or assumed) abundance and mortality levels. Finally, the BRT considered all of the available information related to extinction risk and assessed the overall extinction risk facing the NEP white shark population now and in the foreseeable future.

4.1. Potential Threats to the NEP White Shark Population

A key element to understanding current and future population status is assessing threats to the populations. The BRT considered a range of threats both natural and anthropogenic and evaluated both the historic and current/future risk associated with each threat. Historic risk covered a period considered to be appropriate for each risk. Consideration of future threats was limited to 60 years, a period that corresponds roughly to 3 generation times, and represents a biologically relevant time horizon for projecting conditions into the future. Threats included: 1) fisheries mortality in U.S., Mexican and international waters 2), loss of prey due to overharvesting, 3) small population effects, 4) disease and predation, 5) habitat degradation linked to contaminants, and 6) global climate change. Based on the information provided, the team conducted SEDM to characterize the severity of threats and the associate certainty. The SEDM results were then used to guide the inclusion of threats into the subsequent risk analyses.

4.1.1. Interactions with commercial and recreational fisheries

4.1.1.1. High Seas Driftnet Fishery

From the 1970s to 1992 there were a number of large-scale fisheries on the high seas that had significant amounts of shark bycatch. Large-scale drift gillnet (DGN) fisheries in the Pacific included the salmon, the flying squid and the large-mesh DGN fishery for tuna and billfish. The salmon fishery was located west of 180° W and is not likely to have interacted with the NEP white shark population. The area fished by the flying squid and large mesh fisheries included the waters around Hawaii, although these fisheries are centered in the western Pacific and both had an eastern bound of between 142° and 145° W (Bonfil 1994). The easternmost areas of these fisheries overlapped with the western

portion of pelagic habitat used by subadult and adult white sharks around Hawaii, but they were west of the core region of the offshore focal area, which is between approximately 140° and 134° W (Domeier and Nasby-Lucas 2012).

Catch of white sharks was reported in both the flying squid and large mesh DGN fisheries, although data are scarce. Using data collected from observer programs in 1990, Bonfil (1994) reported 7 white sharks observed captured in the flying squid fishery and 35 in the large mesh DGN fishery. Where these sharks were caught was not reported. Accounting for the percentage of the fishery that was observed, he used these numbers to estimate total white shark catch of 156 and 564 individuals, respectively, for the two fleets in 1990. There is no way of determining how many of these sharks may have been from the NEP white shark population as opposed to from a western Pacific population. However, based on the information available to the author, his conclusion was that very few large (mature or subadult) white sharks associated with the NEP DPS were likely to have been caught in the high-seas fisheries (Bonfil, Instituto Nacional de la Pesca Progreso, pers. comm.) given the smaller size of animals typically caught and the geographic distribution of effort. According to Bonfil (pers. comm.), the size data reported for these catches in Bonfil (1994) (suggesting captured white sharks averaged about 50 kg in weight per individual) are now considered unreliable and associated instead with mako sharks.

Due to concerns about bycatch of many taxa in these fisheries, the high seas drift net fisheries were phased out in 1992 following a United Nations resolution banning their use. A survey of NOAA personnel involved in international affairs and Illegal, Unreported and Unregulated (IUU) fishing yielded no information to indicate that these fisheries are still operating in waters east of Hawaii. However, by their nature these fisheries are difficult to document and uncertainty exists in assessing the presence of IUU fishing.

4.1.1.2. Hawaii Longline Fisheries:

Longline fisheries based out of Hawaii for swordfish and bigeye tuna operate in waters that can overlap with NEP white shark habitat. Observer data for the shallow set swordfish fishery records from 1997 – 2008 show 7 records of white shark catches. These records were not verifiable and were considered suspect by NOAA personnel familiar with the database (Bill Walsh, NMFS, pers. comm.). Only one record had written notes and the description provided was consistent with a salmon shark (*Lamna ditropis*). The condition of four sharks was reported with three returned to the water alive and one dead.

4.1.1.3. U. S. West Coast Fisheries

Juvenile white sharks have historically been taken in the near-shore net fisheries operating off the coast of California, Oregon and Washington. These fisheries include the coastal set-net fishery for halibut, other shark species and white sea bass, the large-mesh drift gillnet fishery for sharks and swordfish, and the small-mesh drift net fishery for seabass and yellowtail. Klimley (1985) summarized the catch data for 109 white sharks

caught in a range of gear types from 1935-1984. Lowe et al. (2012) characterized 369 catch records from 1936 to 2009, and examined trends over time in relation to fisheries effort. These studies indicated that most white sharks were caught off the California coast in bottom set nets both north and south of Point Conception. The majority of the catch consisted of juvenile and YOY white sharks; however, catch of subadults and adults was also recorded in these fisheries. Catches were sporadic throughout the 1970s, followed by an increase in the 1980s as the small and large mesh net fisheries expanded. Catches then decreased, reaching a low in 1994 when California adopted protections for white sharks and the Marine Resources Protection Act of 1990 phased out gill and trammel nets within three miles of the mainland and one mile of the offshore islands (Lowe et al. 2012).

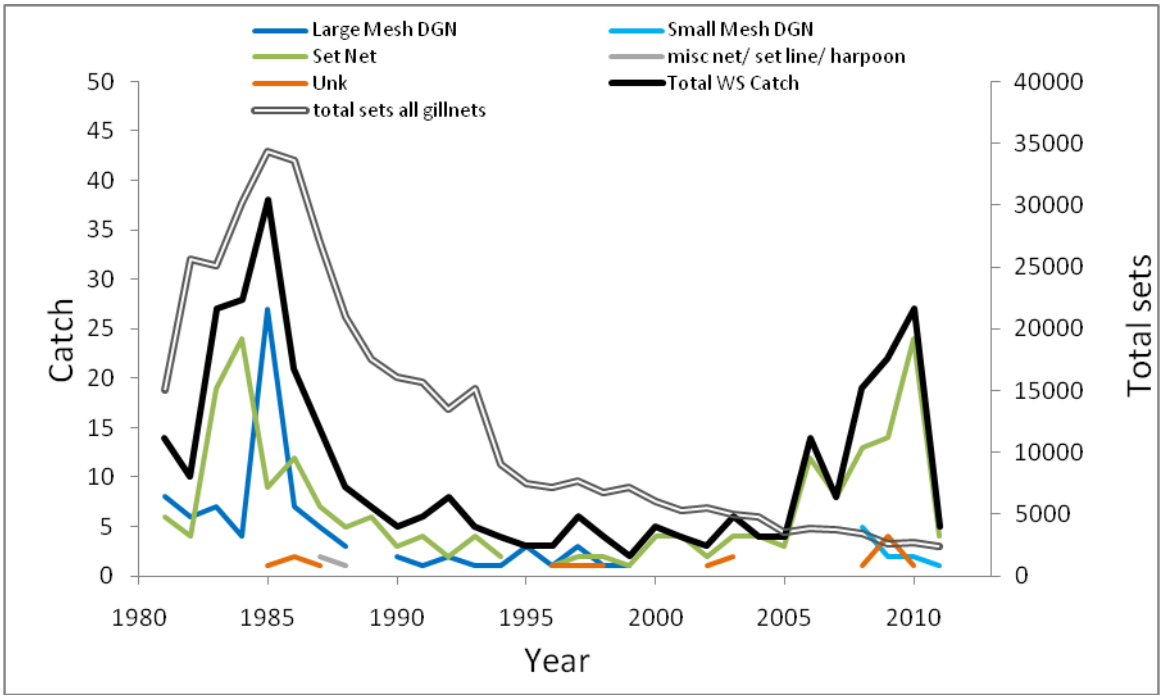

Figure 4.1 White shark reported catch data and total number of gillnet sets from the different fisheries operating off the West Coast from 1981-2011. Catch data are from logbooks, the MBA sampling program the NMFS observer program and PacFIN.

In addition to these published summaries, the BRT compiled U.S. fisheries effort and catch data (Figure 4.1) from PacFIN (Pacific Fisheries Information Network, http://pacfin.psmfc.org/), logbooks, fisheries observer programs, and the Monterey Bay Aquarium (MBA) scientific white shark collection program, which since 2002 has financially incentivized fishermen to report their white shark catches (Appendix D). In the last 10 years the majority of catch data for U.S. fisheries has come through the MBA program.

Prior to the mid-1990s, most reported catches occurred in the coastal set net and large-mesh drift net fishery. Reported catch numbers peaked during the mid-1980s and declined steadily thereafter as a result of decreasing fishing effort and increasingly restrictive fishing regulations. The most important regulatory change was likely the near shore net ban mentioned above. Because these fisheries operated primarily over the continental shelf, the 1994 ban dramatically reduced the areas in which they could operate to a few regions including the Ventura Flats, Channel Islands, Huntington Flats, and Oceanside where the continental shelf extends beyond the 3 nmi near-shore closure. Other relevant regulatory changes to the large-mesh DGN fleet include the delay in the fishing season start date from spring to mid-August and the increase in the dropper length on the nets to 12 m to move them out of the surface waters benefitting small white sharks which spend the majority of their time in the top 10m of the water column (Weng et al. 2007b). In addition, in 2001 a time-area closure essentially eliminated the large-mesh DGN fleet from near-shore waters north of Morro Bay. Since 1999 only one white shark has been reportedly caught in the large-mesh DGN fishery. Most current catch occurs in the set net fishery operating in the southern California Bight. The management regimen for fisheries interacting with white sharks has been similar since the 1994 closure of state waters. Reported catches in this area, primarily of YOY white sharks, has been increasing through the 2000s. The difference in catch distribution across size classes associated with the major regulatory change in 1994 is shown in Figure 4.2.

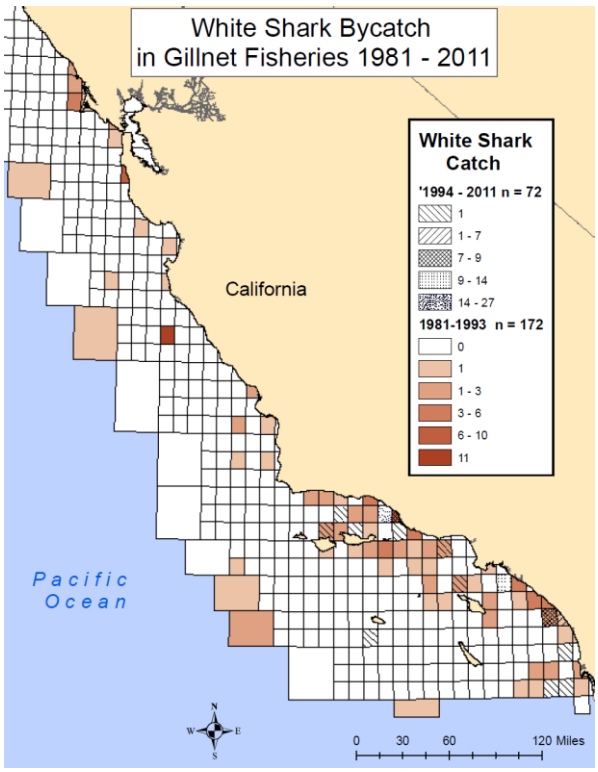

Figure 4.2 The distribution and amount of gillnet catch along the California Coast in relation to CDFW fishing blocks showing the periods from 1981-1993 and 1994 - 2011.

The BRT used the fisheries catch and effort data to examine both historic and current levels of catch. By comparing the calculated catch per unit effort (CPUE calculated as the

number of sharks caught per fishing set) from logbook data to CPUE calculated from fisheries observer programs and the MBA program, it is possible to estimate the level of under-reporting by logbook information and thus calculate an approximate corrected time series of white shark catch. Most catch data recorded as part of observer or MBA programs are from the set net fishery. The number of catches recorded for the large-mesh or small-mesh drift net fisheries are too few to reliably estimate under-reporting rates for these fisheries independently. Therefore it was assumed that under-reporting rates would be similar across all net fisheries, and a single correction factor was estimated based on information combined across fisheries. Table 4.1 summarizes effort and catch data for each fishery, for years when observer programs or the MBA program were active. Across all these years, total CPUE based on data from these programs was 0.00339 sharks per set, whereas CPUE based on logbook data for these years was 0.00062. The logbook CPUE is 18.3% of the CPUE calculated from the scientific data, yielding a correction factor of 1/0.183 = 5.6. The correction can be alternatively calculated by noting that out of 12 catches recorded by NMFS observer programs between 1990 and 2011, 3 (25%) of these were also recorded in logbooks (Table 4.2), yielding a correction factor of 1/0.25 = 4. For purposes of the analysis, a correction factor of 5 was used as an approximation. The logbook-reported catches were multiplied by this correction factor to estimate total catches in each fishery per year from 1981 to 2011 (Table 4.3).

Table 4.1 Comparison of effort (sets) and catch data summaries for U.S. west coast net fisheries, for years when scientific observer programs were active. Data from the Monterey Bay Aquarium scientific collection program are also included for the set net fishery. (lgDGN and smDGN = large- and small-mesh drift gill net; CPUE = catches per set).

		Logbook data		Scientific data		CPUE	
Fishery	Years	Sets	Catch	Obs. sets	Obs. catch	Logbook	Obs.
set net	1990 – 1994	45857	8	8614	9	0.00017	0.001
set net	2002 – 2011*	25909	52	8456	74	0.0020	0.0088
lgDGN	1990 – 2011	47099	14	8225	3	0.00030	0.00037
smDGN	2002 – 2004, 2010 – 2011	1233	1	81	0	0.00081	0
TOTAL		**120098**	**75**	**25376**	**86**	**0.00062**	**0.00339**

*MBA program

Table 4.2 White shark catches recorded as part of the scientific observer program for U.S. west coast net fisheries, or from the MBA scientific collection program, and the number of those that were also recorded in fishermen's' logbooks. Note that these are only the sharks that are recorded in both datasets. (lgDGN and smDGN = large- and small-mesh drift gill net).

Fishery	years	catch	# also reported in logbooks	Data program
Set net	1990 – 1994	9	2	NMFS observer program
Set net	2002 – 2011	74	40	MBA program
lgDGN	1990 – 2011	3	1	NMFS observer program
smDGN	2006 – 2011	11	2	MBA program
TOTALS		**97**	**45**	
Totals, NMFS data only		**12**	**3**	

There is some belief that logbook reporting rates throughout the set net fishery may have increased with initiation of the MBA program (and compare proportion of MBA vs. observer-documented catches that were reported in logbooks, Table 4.2). If so, then increased reports (and calculated catches and CPUE) of white sharks during the 2000s could simply reflect increased reporting (not increasing CPUE), and using the same correction factor for under-reporting across all years (as in Table 4.3) might over-estimate catch and CPUE in the 2000s. Potential changes in reporting rate can be evaluated for the set net fishery. For 1990-1994, the ratio of the CPUE based on observer data to the CPUE based on logbook data was 5.9, whereas for 2002 – 2011, the ratio of the CPUE based on MBA data to the CPUE based on logbook data was 4.4. This corresponds to a 34% increase in mean logbook reporting rate since 2002, assuming that all catches from vessels participating in the MBA program are reported to the MBA. However, given limited sample sizes from these programs, it is difficult to know how much reporting has truly changed (vs. these differences being an artifact of sampling error). Moreover, it is not known if 100% of the white sharks captured during the 2000s were reported to the MBA. A reporting rate of less than 100% would bias downward the correction factor for 2002-2011. In short, it appears that logbook reporting rates for the set net fishery could have increased during the 2000s, but the apparent increase seems relatively modest and uncertain and would not fully explain the increased number of catches in the logbook data through this time period. The logbook-based CPUE was 7 times higher in the period from 2002-2011 than from 1990-2001 (based on the data in Table 4.3). These results are consistent with a true increase in the white shark CPUE in the set net fishery during the 2000s, as suggested by Lowe et al. (2012). This is further explored in Section 4.3.

Table 4.3 White shark catches recorded in logbooks from 1981 – 2011 for three U.S. west coast net fisheries, and approximate corrected estimates of total catch (lgDGN and smDGN = large- and small-mesh drift gill net).

Year	Effort (sets)			Logbook reported catches			Estimated catches (logbook x 5)			
	lgDGN	smDGN	set net	lgDGN	smDGN	set net	lgDGN	smDGN	set net	TOTAL
1981	5594	477	8982	8	0	6	40	0	30	70
1982	10292	308	15053	6	0	4	30	0	20	50
1983	9883	187	14525	7	0	19	35	0	95	130
1984	9800	155	20093	4	0	24	20	0	120	140
1985	9618	321	22799	27	0	9	135	0	45	180
1986	10392	623	20577	7	0	12	35	0	60	95
1987	8280	461	17876	5	0	7	25	0	35	60
1988	5760	496	14697	3	0	5	15	0	25	40
1989	5620	361	11601	0	0	6	0	0	30	30
1990	4163	372	11593	2	0	3	10	0	15	25
1991	4232	512	10975	1	0	2	5	0	10	15
1992	3736	357	9367	2	1	0	10	5	0	15
1993	5040	504	9586	1	0	1	5	0	5	10
1994	4076	614	4336	1	0	2	5	0	10	15
1995	3373	478	3667	3	0	0	15	0	0	15
1996	3143	500	3571	0	0	1	0	0	5	5
1997	2678	376	4627	1	0	2	5	0	10	15
1998	2652	205	3883	1	0	2	5	0	10	15
1999	2389	260	4495	1	0	1	5	0	5	10
2000	1193	251	4577	0	0	4	0	0	20	20
2001	1410	217	3651	0	0	3	0	0	15	15
2002	1410	252	3861	0	0	2	0	0	10	10
2003	1171	255	3522	0	0	4	0	0	20	20
2004	932	186	3620	0	0	4	0	0	20	20
2005	817	255	2508	1	0	3	5	0	15	20
2006	1409	291	2167	0	1	8	0	5	40	45
2007	1172	276	2318	0	0	3	0	0	15	15
2008	963	311	2174	0	1	8	0	5	40	45
2009	606	232	1756	0	0	8	0	0	40	40
2010	377	285	1998	0	1	11	0	5	55	60
2011	157	255	1985	0	0	1	0	0	5	5

Size and sex information for white sharks caught in the fisheries were obtained from the scientific observer records and the MBA programs. Based on 97 records between 1991 and 2012, an estimated 82% of white sharks caught were < 175 cm TL and considered to be YOYs, and the sex ratio of YOY catches was close to parity (49 females: 48 males) (Figure 4.3). Larger sharks were taken primarily in the 1980s before implementation of the observer program and thus are only reported in the logbook data.

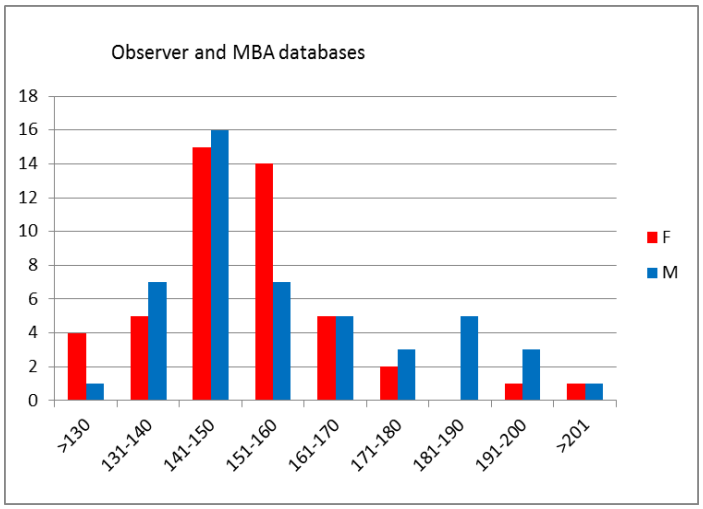

Figure 4.3 Sex and size composition of white shark catch from the observer and MBA programs covering the period from 1991-2012 (n=97).

The expected mortality of white sharks captured in the set net fishery can be estimated from the number of live and dead sharks reported in the MBA program. Overall, 49% of white sharks sampled by the MBA program were brought in dead. For the sharks brought in alive that were subsequently tagged with satellite tags, survival was 98% when soak times (the amount of time the set net was left in the water to fish) were 24 hours or less (C. Lowe, California State University Long Beach (CSULB), pers. comm.).

4.1.1.4. Mexican Fisheries

Because the NEP white shark population occupies Mexican waters, it is important to examine potential fisheries interactions in Mexico as well. A number of researchers have examined white shark catch rates both along the Pacific side of the Baja Peninsula and in the Gulf of California. A range of approaches have been used to document catch including surveys of artisanal fishing camps, surveys of carcass discard sites, examination of the logbook data from one commercial drift gillnet vessel and the associated observer data, and examination of purchase records of trunks (after removal of head and innards) by local buyers.

Pacific Coast of Baja Peninsula:
Catch along the Pacific coast of the Baja Peninsula is documented primarily by researchers working out of Centro de Investigación Científica y de Educación Superior de Ensenada (CICESE) in Ensenada, Mexico. Santana-Morales et al. (2012) summarized the results from 111 catch records of white sharks between 1999 and 2010. The size of the sharks was estimated to range between 123 to 274 cm TL with 80% of the catch being YOY (<175cm). Most white sharks were caught in the summer months (60%), and most YOY were caught inside Sebastián Vizcaíno Bay.

More recent efforts to quantify catch of white sharks have been conducted by CICESE and involve working with the local distributors in Sebastián Vizcaíno Bay that purchase fish from the camps in the area where the majority of white shark bycatch occurs (O.

Sosa-Nishizaki, CICESE, pers. comm.). The advantage of this approach is that it targets the point of contact for all fishermen. The disadvantage is that only trunks are brought to the local distributors, which raises concerns about the accuracy of reported species identifications and yields no information on size or sex. According to the local distributors, white sharks have a lower monetary value because of their (visibly) lower meat quality. Because the color pattern of a white shark's body trunk differs from that of other species (e.g., more strongly bicolored in salmon than white shark), white sharks trunks are considered to be distinguished easily from other similar shark species like mako sharks (*Isurus oxyrinchus*) and salmon sharks. Also, mako and salmon sharks tend to be caught in pelagic longline fisheries whereas as white sharks are primarily caught in near shore bottom set nets for halibut (O. Santana, CICESE. pers. comm.). The accuracy of the species identification, however, has not yet been independently validated.

In 2011, ~ 186 white sharks were caught in Baja California Norte (located between the U.S.-Mexican border and in Sebastián Vizcaíno Bay) (O. Sosa-Nishizaki, CICESE, pers. comm.). The vast majority of sharks were caught in Sebastián Vizcaíno Bay and documented through the distributor (~170). In 2012, Mexico adopted new regulations prohibiting all targeted shark fishing along the Pacific coast of Mexico from June 1 – July 31. As a consequence, there was no reporting of white sharks being caught during those two months and the total number of white sharks reported caught for the year dropped to 85. It is possible that white sharks were still caught and landed during the closed period; however, it is impossible to know how many given the current lack of reporting. Beginning in 2013, the shark fishing closure will include the month of May as well. It is uncertain how effectively these regulations will be enforced, but some level of enforcement was in place during 2012. If the regulations are fully enforced, we would expect a 60% reduction in white shark catches given the closure of fishing in May, June and July. The projected annual catch would be approximately 68 white sharks. If there is no or limited enforcement during this period or no reduction in catch, overall catch levels for juvenile and YOY white sharks on the Pacific coast of Baja California Norte may still be around 186 per year. One caveat is that it is impossible to predict how fishing effort might shift in response to the new regulations and how this will impact on white shark bycatch.

White sharks are also caught along the Pacific coast of the southern portion of the Baja California peninsula (Baja California Sur) but have not been quantified, except for the reporting that 6 juvenile white sharks were caught by 3 boats fishing out of Isla Magdalena (near Magdalena Bay), documented by IEMANYA (Investigacion, Educacion, Manejo y Asesoria) (H. Peckham, Grupo Tortuguero, pers. comm.).

Gulf of California:

Although white sharks are known to be caught in fishing gear in the Gulf of California, Mexico, incidental catch is not well quantified. Galván-Magaña et al. (2010) reported 30 records of white shark catch between 1964 and 2010. These included a mixture of adult, subadult and juvenile sharks, but no YOY. To date there is only one record of a YOY white shark captured in the Gulf of California, in the northern region (O. Sosa-Nishizaki, CICESE, pers. comm.), although large females are documented to come into this area,

presumably to have pups (Domeier and Nasby-Lucas 2013). A second study conducted in 1998 and 1999 documented elasmobranch catch from 83 artisanal camps along the eastern and western coasts of the Gulf of California with most effort focused from March-November (Bizzarro et al. 2009). This is the most comprehensive survey conducted to date of artisanal fisheries in the Gulf of California. These researchers did not see any white sharks over the course of their survey. Records of two additional individual catches of adult female white sharks in the Gulf of California are reported for 2004 (Castro 2012) and 2012 (M. Domeier, MCSI, pers. comm.).

4.1.1.5. Recreational Fisheries

White shark interactions with recreational fisheries are relatively rare off California. Marine recreational fishery survey data collected in California [including Marine Recreational Fisheries Statistics Survey (MRFSS) and the California Recreational Fisheries Survey (CRFS)] from 1980 through the present contain only one record. Logbook records for Commercial Passenger Fishing Vessels (CPFV) operating out of California ports indicate that from 1980 through 2011, seven white sharks were caught. They are also occasionally caught off fishing piers. In 2012, two fishing citations were given for the illegal take of juvenile white sharks.

4.1.1.6. Summary of YOY and juvenile bycatch estimates

Current bycatch of YOY and juvenile white sharks is the sum of bycatch in both U.S. and Mexican fisheries. In the U.S., the average annual bycatch from 2001 – 2011 was approximately 28 (Table 4.3). Given an estimated mortality rate of 0.49, the estimated number of individuals killed per year was 16. Of these, 82% are estimated to be YOYs and the other 18% are estimated to be juveniles at least one year in age. In Mexican fisheries, 100% of the YOY and juveniles captured white sharks are assumed to be killed (i.e., retained and sold) given their market value. The estimated mortality in 2011, therefore, was 186 white sharks in Baja California Norte and an additional 6 white sharks in Baja California Sur. Of these, 77% are considered to be YOYs (O. Sosa-Nishizaki, CICESE, pers. comm.), similar to the estimate of 80% by Santa-Morales et al. (2012). The remaining 23% are juveniles at least 1 year in age. The total estimated catch mortality for U.S. and Mexico is therefore 192 + 16 = 208 white sharks per year, with approximately 77% being YOYs and 23% being juveniles greater than one year in age. There is some level of uncertainty in these estimates since bycatch estimates are not available for all fisheries in Mexico and the bycatch estimates from Baja California Norte have not been scientifically verified.

Bycatch for subadult or adult white sharks has not been quantified. Based on the best available information, the BRT developed plausible bycatch estimates for subadult/adult white sharks in the U.S. and Mexico (see Section 4.4.4).

4.1.2. Prey Availability

Another proposed threat to white shark populations is the over utilization of their prey by humans which could have impacted the carrying capacity for white sharks in the NEP. Declines in the major prey species across life history stages have been documented over different time periods.

4.1.2.1. Subadult and adult white sharks

Adult white sharks are documented to forage on pinnipeds. Populations of pinnipeds in the NEP were dramatically reduced in the 1800s due to hunting. Northern elephant seals and Guadalupe fur seals (*Arctocephalus townsendi*) were thought to have been driven to extinction, and census numbers in the early 1900s were close to zero (Townsend 1924, Huey 1930, Hubbs 1956). Similar declines were documented for harbor seals (*Phoca vitulina richardsi*) and California sea lions (Bonnot 1928). Population levels for these four main species started increasing in the 1950s and 1960s and have rebounded dramatically (Gallo-Reynoso et al. 2005, Carretta et al. 2011) (Figure 4.4). The BRT considered the period of historic risk to white sharks due to over utilization of pinniped prey to be from the 1800s through 1980, when significant recoveries were noted in all of the pinniped species listed above, although the number of California sea lions at Guadalupe Island remains low but stable (estimated at less than 1200 animals) (Gallo-Reynoso et al. 2005).

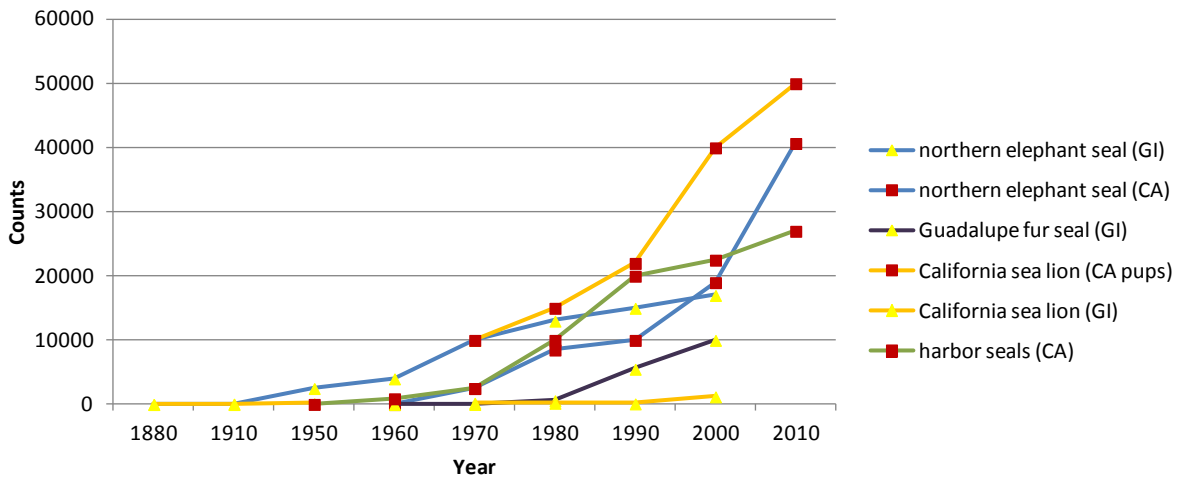

Figure 4.4 Estimated population abundance counts (y-axis) for northern elephant seals, Guadalupe fur seals, California sea lion and harbor seals from 1880 to 2010 for Guadalupe Island and California. Note that the numbers for northern elephant seals in California indicate the estimated number of elephant seal births for that year. (Gallo-Reynoso et al. 2005; Carretta et al. 2011; M. Lowry, NMFS, pers. comm.). Note that for California sea lions in California, the number indicates pups rather than adults which are currently at an abundance of ~150,000.

In addition to pinnipeds, subadult and adult white sharks have been observed feeding on carcasses of cetaceans (Dudley et al. 2000, Skomal et al. 2012, Fallows et al. 2013). Like pinnipeds, large whales also suffered severe population declines due to overharvesting up

through the mid 1980s. While whale recoveries have not been as dramatic or as consistent across species as for many pinnipeds, many of the larger baleen whales such as fin (*Balaenoptera physalus*), humpback and gray whales (*Eschrichtius robustus*) have greatly increased (Carretta et al. 2011, Moore and Barlow 2011). The recovery of large whales and pinnipeds was aided by the closure of commercial whaling in the late 1980s and passage of the 1972 Marine Mammal Protection Act.

4.1.2.2. YOY and Juvenile white sharks

Juvenile and YOY white sharks forage on a range of demersal and epipelagic fish, squid and crustaceans over the continental shelf in near shore waters. In the 1970s, fisheries targeting a range of fish and sharks expanded in the shelf waters along the U.S. West Coast. These fisheries targeted primarily white seabass, Pacific angel shark (*Squatina californica*) and California halibut (*Paralichthys californicus*), but caught a range of other species including fish, sharks, birds and marine mammals. Due to concerns in the 1980s and 1990s about unsustainable levels of bycatch of already depleted or protected species (including marine turtles, marine mammals, birds and some fish), the set gillnet fisheries were prohibited inside California state waters (within 3 nmi of shore) in 1994 in accordance with the provisions of the California Marine Resource Protection Act of 1990. White seabass, for example, had declined to 10% of historic levels by 1981 (Pondella and Allen 2008). Following the near shore gillnet ban in 1994, there has been an increase in the abundance of many species including white seabass, and leopard (*Triakis semifasciata*) and soupfin sharks. The BRT considered the period of historic risk associated with over exploitation of the YOY and juvenile white shark prey base to extend from the 1970s until 2000, providing some time for prey populations to recover after the gillnet closure in 1994.

To evaluate possible threats to juvenile and YOY white sharks as a result of reduced prey resources off California, the BRT examined the status of 23 fish and invertebrate species (

Table 4.4) that have either been documented in the stomach contents of juvenile white sharks or are potential prey because they fit in the category of standard prey types (i.e. demersal fish) and are found in the same habitat as white sharks and their prey. Official stock assessments were not available for most species. Consequently, we interpreted the language in the CDFW semiannual status reports and the Pacific Fishery Management Council's Stock Assessment and Fishery Evaluation (SAFE) reports as well as the results in Pondella and Allen (2008) to determine whether there is currently concern about the populations. Of the 23 species examined, a broad range including crustaceans, squid, and demersal and epipelagic fish are considered to be healthy, with many having recovered from earlier overfishing. White seabass populations, for example, began to rebound in 1997 and are now considered healthy. Similar patterns were seen in both soupfin and leopard shark (Pondella and Allen 2008). For some prey items, such as bat rays, there is no available information on stock status. Concerns remain regarding some species, including six species and some runs of Chinook salmon (*Oncorhynchus tshawytscha*) populations that are considered depleted. However, white sharks are generalist feeders, and generalists are considered more resilient to the loss of some prey items than a dietary specialist. The fact that there is currently sufficient prey in the California Current System is supported by the increase in other species that have overlapping diets including pinnipeds, leopard and soupfin sharks, and giant sea bass (*Stereolepis gigas*).

Table 4.4 List of likely and possible prey items for white shark including whether there is concern about the population status due to fishing and associated comments.

Prey	Current Concern?	Comments
Chinook salmon[1]	YES/NO	some runs healthy, others not
Lingcod[2]	YES	southern stock at 24% of unfished levels in 2005
Cabezon[3]	YES	28% of unfished level in 2004
Green sturgeon[4]	YES	listed as species of concern
CA Halibut[5]	YES	specifically south of Point Conception
Petrale Sole[5] (*Eopsetta jordani*)	YES	recent assessment puts at 18% of unfished biomass
White Croaker[5] (*Genyonemus lineatus*)	YES	catch declines evident, may be linked to warming water temperatures
Market squid[6] (*Loligo opalescens*)	NO	highly productive and variable population levels, with relatively high catch levels
White Seabass[7]	NO	Stock assessment likely to be conducted in 2013
Black rockfish[2]	NO	
Pacific Mackerel[8]	NO	relatively low population levels and very low catch
Sardine[5]	NO	although current populations levels may be low due to natural population variability
Long Nosed Skate[5] (*Dipturus oxyrinchus*)	NO	although some model uncertainty
Jack mackerel[9] (*Trachurus symmetricus*)	NO	relatively low population levels and very low catch
CA spiny Lobster[5] (*Panulirus interruptus*)	NO	
Dungeness Crab[5] (*Metacarcinus magister*)	NO	
California Corbina[5] (*Menticirrhus undulates*)	NO	Sustainable under current recreational harvest levels
Spotfin Croaker[5] (*Roncador stearnsii*)	NO	
Yellowfin Croaker[5] (*Umbrina roncador*)	NO	
Soupfin shark[7]	NO	population increasing
Leopard shark[7]	NO	population increasing
Spiny dogfish[10]	LIKELY FINE	very effective competitor in an ecosystem, stock assessment not available but no fishery since the 1930s
Northern anchovy[9] (*Engraulis mordax*)	NO	relatively low population levels and very low catch

[1]Myers et al. (1998), [2]Larinto (2010), [3]Cope et al. (2004), [4]Adams et al. (2007), [5]Larinto (2013), [6]Rogers-Bennett (2001), [7]Pondella and Allen (2008), [8]Crone et al. (2011), [9]Crone, SWFSC, pers. comm., [10]Gertseva and Taylor (2012).

4.1.3. Small population effects

Recent studies estimate that the abundance of NEP white shark adults and subadults is in the low hundreds (Chapple et al. 2011, Sosa-Nishizaki et al. 2012). While the BRT felt that the numbers were biased and an underestimate, we addressed the potential threats associated with a small population size because they are cited in the listing petitions.

If a population is very small there are a number of inherent risks that may increase the potential for the population to decline further and be at an increased risk of extinction. These risks typically include Allee or other depensation effects which are tied to survival and reproduction via three mechanisms: ecological (mate limitation, cooperative defense, cooperative feeding, and environmental conditioning), genetic (inbreeding and genetic drift), and demographic stochasticity (i.e. individual variability in survival and recruitment) (Berec et al. 2007). The actual number at which populations would be considered critically low and at risk due to low population levels varies depending on the species and the risk being considered. Below we address the potential for such risks to impact the NEP white shark population.

4.1.3.1. Reproduction

If a population is critically small in size, individuals may have difficulty finding a mate. However, the probability of finding a mate depends largely on density rather than absolute abundance. Individual white sharks in the NEP are not randomly distributed throughout their range, but instead exhibit regular seasonal migration patterns which include aggregating at certain near shore sites and offshore areas (Weng et al. 2007a, Domeier and Nasby-Lucas 2008). This aggregation behavior is thought to be associated with mating (Domeier 2012, Jorgensen et al. 2012a) and serves to increase their local density and increase the probability of individuals finding each other for mating. This mating strategy will reduce the effects of small population size on finding mates over other strategies that do not concentrate the remaining individuals.

4.1.3.2. Demographic Stochasticity

If a population is critically small in size chance variations in the annual number of births and deaths can put the population at added risk of extinction. Demographic stochasticity refers to the variability of annual population change arising from random birth and death events at the individual level. When populations are very small (e.g., < 100 individuals), chance demographic events can have a large impact on the population. Species with low mean annual survival rates are generally at greater population risk from demographic stochasticity than those that are long lived and have high mean annual survival rates. Or in other words, species that are long lived and have high annual survival rates have lower "safe" abundance thresholds, above which the risk of extinction due to chance demographic processes becomes negligible. The bias in abundance estimates, haplotypic diversity and inferences about abundance from YOY and juvenile catch and mortality rates (see section 4.5) are all consistent with a number of adult females in the hundreds (see Sections 2.3, 4.1.4 and Appendix B), with a total population abundance across sexes

and age classes being much greater. Consequently they should be at low risk from demographic stochasticity.

4.1.3.3. Genetics

If a population is critically small in size, Allee effects can act upon genetic diversity to reduce the prevalence of beneficial alleles through genetic drift. This may lower the population's fitness by reducing adaptive potential and increasing the accumulation of deleterious alleles due to increased levels of inbreeding. Population genetic theory typically sets a threshold of 50 individuals (i.e. 25 males, 25 females) below which irreversible loss of genetic diversity is likely to occur in the near future. This value, however, is not necessarily based upon the number of individuals present in the population (N_C) but rather on the effective population size (N_E), which is typically less than N_C. In extreme cases N_E may be much (e.g. 10 – 10,000 times) smaller, typically for species that experience high variance in reproductive success (e.g., sweepstakes recruitment events). N_E may also be reduced in populations that deviate from a 1:1 sex ratio.

With respect to considerations of N_E in white shark populations, the following points are relevant. Mature white sharks in the NEP are known to seasonally aggregate, suggesting increased mating opportunities even at low population size. As live-bearers with large precocious young, there is the expectation for consistently high survival of offspring across females and broods (i.e., no sweepstake recruitment events). As multiple paternity has been found in white shark broods (Gubili et al. 2012), this polyandrous mating system is likely to increase N_E. Multiple paternity also has the potential to increase the number of viable offspring in a single brood (Newcomer et al. 1999) and increase individual fitness (Neff and Pitcher 2005). Together these aspects of white shark life history suggest that N_E is likely to be close to the number of mature individuals in the population, which is greater than the threshold of 50 mature animals that is typically thought to be sufficient to avoid deleterious loss of genetic diversity in the short term (IUCN 2001). In addition, although there are no available data for nuclear DNA diversity in NEP white sharks, the relatively high number of unique mitochondrial DNA haplotypes (Jorgensen et al. 2010) can be used as a proxy for evidence of high levels of overall genetic diversity currently within the population (Section 2.3, 4.1.4 and Appendix B).

4.1.3.4. K selected Life History parameters

K selected species (i.e. those with slower population growth rates) generally tolerate a lower level of additive mortality than r selected species. In general, sharks are k-selected due to their life history characteristics, and therefore their potential for growth and recovery falls between those of marine mammals and teleosts (Hutchings et al. 2012). White sharks in particular reach maturity at a relatively older age, have smaller litter size than most sharks, and exhibit a semiannual reproductive cycle (Table 2.1). Comparing across lamnid sharks, Hutchings et al. (2012) estimated a higher rate of population growth potential for white sharks than for mako sharks or salmon sharks. The fact that a

species has a k-selected life history is not in itself a risk factor, but it does affect the level of removal that a population can sustain and the rate at which it can recover. Examples of recovery trends for long-lived sharks are white sharks in the Atlantic Ocean (Tobey Curtis, NERO (Northeast Regional Office, pers comm) and common thresher sharks in the NEP (PFMC June 2010, http://www.pcouncil.org/wp-content/uploads/E2b_SUP_HMSMT_JUNE2010BB.pdf).

4.1.3.5. Stochastic and Catastrophic Events

Animals that are highly mobile with a large range are less susceptible to stochastic and catastrophic events (such as oil spills) than those that occur in concentrated areas across life history stages. White sharks in the NEP are likely to be resilient to catastrophic and stochastic events for the following reasons: 1) they are highly migratory, 2) there is a large degree of spatial separation between life history stages, 3) while they are found in near-shore aggregation sites, adult white sharks are dispersed over a very large area extending from the Gulf of California in the south, the Hawaiian Islands to the west, and Alaska to the north, 4) adult males and females exhibit a large degree of sexual segregation, particularly during the estimated 18 month gestation period for females, and 5) although male white sharks utilize a common offshore focal area, this area is large, with the 50% density contour covering ~ 200,000 km^2 (Domeier and Nasby-Lucas 2012).

4.1.4. Depletion from loss of prey or direct mortality

There are at least two historic risks that could have reduced the NEP white shark population either through direct mortality or through a reduction in the carrying capacity. The first of these is the major reduction in marine mammal population numbers that began in the 1800s and continued through 1970. The second is white shark catch and mortality in regional and high seas fisheries from the 1970s through the early 1990s (see Section 4.1.1 of this report). However, the virtual lack of pinniped prey for nearly 100 years also suggests that white sharks were capable of switching to other prey resources. It is unknown, for example, what white sharks feed upon during the majority of the year when they are in offshore waters, but it is unlikely to be marine mammals. Flexibility in diet argues against strong depletion based on prey alone. A review of the impact of fisheries mortality on the NEP white shark population starting in the 1970s does not suggest that white sharks have been severely depleted (defined by the BRT as a 90% reduction in the population) over the last 2 generations. Furthermore, the level of genetic diversity found in the NEP white shark population is consistent with a population of adult females ranging from a few hundred to low thousands (Section 2.3 and Appendix B).

The petitions suggest that white sharks in the NEP are severely depleted. They cite heavily depleted populations in other ocean basins as support for this determination. Reports of white shark population declines are common for all oceans and have resulted in the listing of the global population under CITES (the Convention on International Trade in Endangered Species of Wild Fauna and Flora) Appendix II and by the IUCN as vulnerable (Hilton-Taylor 2000). Initial reports of white shark population declines are as high as 80% (Baum et al. 2003). More recent analyses of some of these datasets (FAO

2004) however, indicate that while population declines are evident in some areas, declines are exaggerated or unsubstantiated in others. For example, Baum et al. (2003) used logbook data to examine trends in the Northwest Atlantic and reported an 80% decline in white shark populations there. A reexamination of the logbook data however, revealed that catch patterns were linked to a change in reporting requirements where fishers targeting sharks were no longer required to report their catch in pelagic longline logbooks (FAO 2004, Burgess et al. 2005). One of the challenges in most of the studies is the lack of systematically collected data, making documentation of trends difficult. Similar challenges occur in the NEP.

4.1.5. Disease and predation

4.1.5.1. Disease

As with most large predatory sharks, there is no information available on disease in white sharks.

4.1.5.2. Predation

Predation typically varies with size, and large size is considered to provide some level of refuge from predation. Given the large size of white sharks at birth, their predation risk is likely to be relatively low and be restricted to killer whales and larger sharks. Their shallow near-shore habitat will likely also provide some level of protection. As adults, the only known predators are killer whales, with the first documented occurrence at Southeast Farallon Island in 1997 (Pyle et al. 1999). The overall level of predation on adult white sharks is likely very low.

4.1.6. Environmental contaminants

4.1.6.1. Contaminants

Exposure to toxic chemicals, including PCB, DDT, and mercury, was identified by the petitioners as a potential risk factor for NEP white sharks. The vast majority of mercury contamination comes from urban runoff, although a significant amount of mercury is introduced through atmospheric deposition from local power plants, oil refineries, and motor vehicles (Schiff et al. 2000). PCB and DDT are included in a group of persistent organic contaminants that were used as industrial chemicals and pesticides in the United States and other countries around the world. From the 1950s to the 1970s, large amounts of organic contaminants were introduced to the marine environment through waste water treatment system outfalls off Los Angeles, California, from residential and industrial sources (Lee et al. 2002). Other Pacific coastal areas that have had relatively high levels of organic and inorganic contaminants over the past 50 years include Puget Sound and San Francisco Bay (Brown et al. 1998). Starting in the early 1970s, the production and use of PCBs, DDTs and other classes of persistent organic contaminants were banned in the United States, because of toxic effects on wildlife and laboratory animals. By the early 1980s, there was evidence that the concentrations of organic contaminants in

sampled fish along the U.S. West Coast had declined (Mearns et al. 1988). Despite this ban, evidence indicates that a high level of contamination still remains throughout the entire southern California Bight (Schiff et al. 2000).

The contaminant loads found in NEP white sharks are high. In a study by Mull et al. (2012), the average liver concentrations of DDT in YOY white sharks found in southern California were 50-times higher than those observed in juvenile white sharks from South Africa's east coast. DDT and PCB liver concentrations in YOY white sharks were also much greater when compared to other larger, older shark species, such as the Greenland shark (*Somniosus microcephalus*) and the smooth hammerhead (*Sphyrna zygaena*) and comparable to local marine mammals. Mull et al. (2012) also observed elevated levels of mercury in YOY and juvenile white sharks from the southern California Bight with higher concentrations of total mercury in the muscle than the liver. The high levels of pollutants are thought to be linked to maternal offloading (Mull et al. 2013).

Even though elemental and organic contaminant levels are high in NEP white sharks, no deleterious physiological effects have been documented in elasmobranchs (Mull et al. 2012). In addition, no hepatic lesions or other visible effects have been observed with white sharks (K. Lyons, CSULB, pers. comm.). Indications that high tissue contaminant levels are not causing problems at a population level are the apparent increase in other predators that have similarly high contaminant levels including the coastal stock of bottlenose dolphins, California sea lions and harbor seals (Section 4.1.2). For example, pollutant levels, especially DDT residues, found in Southern California coastal bottlenose dolphins have been found to be among the highest of any cetacean examined (O'Shea et al. 1980, Schafer et al. 1984). The population has remained stable between 1996 and 2005 (Dudzik et al. 2006).

4.1.6.2. Plastic and marine debris

Marine litter has become an increasing problem in the oceans, with plastic debris being the most abundant (Derraik 2002). The main danger to sharks includes entanglement and ingestion. Because they are lightweight and buoyant, plastics can be transported over long distances and in the North Pacific, massive volumes of plastic have accumulated in the North Pacific Gyre in an area termed the Great Pacific Garbage Patch (Rios et al. 2010). This region is north of Hawaii and northwest of the offshore focal area where male sharks aggregate when offshore. While there is apparently limited direct overlap with the Great Pacific Garbage Patch and the off shore focal area, white sharks likely encounter marine debris both near shore and offshore. Given the low catch rates of white sharks, incidence of entanglement may not be apparent. However, no sharks at Guadalupe Island or caught in the southern California Bight have been reported to be entangled in marine debris. As for the risk of ingestion, sharks are capable of evacuating their stomachs, which may help reduce the impact associated with ingesting plastics. A shortfin mako shark caught on a longline was observed to evert and then retract its stomach (Brunnschweiler et al. 2011, S. Kohin, NMFS, pers. comm.). In addition, numerous satellite tags have been consumed by what are thought to be lamnid sharks (mako,

salmon, or white sharks) given the warm internal temperatures, and then regurgitated (Kerstetter et al. 2004).

4.1.7. Global Climate Change

Climate change could be manifested in many ways, but the two ways that have the greatest potential to impact white sharks are ocean acidification and warming ocean temperatures.

4.1.7.1. Ocean Acidification

The anthropogenic increase in atmospheric concentrations of carbon dioxide (CO_2) is causing a reduction in pH in the waters of the NEP. This has already been documented in coastal ecosystems. While the severity of the effects on specific species or the overall ecosystem are uncertain, changes to CO_2 levels in the California Current marine ecosystem could adversely impact the marine food web. A number of direct impacts of ocean acidification have been documented for teleosts. For example, at elevated CO_2 levels and reduced pH, metabolic depression has been observed (Cruz-Neto and Steffensen 1997) as well as a reduced ability to detect olfactory cues, hindering predator avoidance and habitat selection in marine fish larvae (Munday et al. 2009, Dixson et al. 2010). There are no data available on the direct impacts of reduced pH on sharks. The impacts of ocean acidification on NEP white sharks remain speculative.

4.1.7.2. Ocean warming

Ocean temperature plays a key role in determining the pelagic habitat for many species. The exact impact of global warming is likely to be species-specific, and model results suggest a range of responses to predicted warming patterns. Hazen et al. (2012) examined the temperature preferences of 23 marine species, inferred from their current thermal habitat-use patterns, to estimate whether there would be changes in habitat associated with projected warming through 2100 in the NEP. Model results suggest an increase in white shark habitat of approximately 7%. For northern elephant seals, the model predicted a 5% increase in habitat, and for California sea lions a decrease of approximately 0.5% was predicted. Impacts are highly uncertain and likely more complicated, and the actual impact of global warming will also depend on how the foodweb changes. In addition, results focused on adult and subadult white sharks. If and how the habitat of juvenile and YOY white sharks would shift is not known. As with ocean acidification, the impacts of global warming on NEP white sharks remain speculative.

4.2. Evaluation of Threats to the NEP White Shark Population

The BRT evaluated the threats discussed in Section 4.1 in terms of their severity and certainty. Each threat was evaluated in terms of its current and future risk to the NEP white shark population and where appropriate, in terms of its historic risk over a time period considered to be appropriate by the BRT. The threats were grouped into five key

categories (e.g., habitat destruction, modification or curtailment, overutilization, etc.). Consideration of the future risk of threats was for a period of 60 years, which corresponds to 3 generations of white sharks. The BRT also felt that for assessing threats, 60 years represented a biologically relevant time horizon for projecting current conditions into the future.

The terms used in Table 4.5 are defined as follows:

Threat Categories– Categories of threats corresponding to the statutory ESA Section 4(a)(1)(A)-(E) factors considered in listing determinations.

Threats – Those specific human or natural events/actions that may affect the past, present or future status of a species or population. Threats were defined by 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 result in a mounting risk to the species in the future. These threats may or may not be occurring currently.

The BRT evaluated the severity and certainty associated with each threat. Severity and level of certainty were scored as a numeric value corresponding to high, medium, low or none (as defined below) by each BRT member. The mean value and standard deviation for the BRT are presented for each threat in Table 4.5. The BRT viewed this method as a more effective way to rank the threats than the plausibility point method used previously.

Severity: Historic = the degree to which the threat is likely to have contributed to the decline of the population historically (over the prescribed time period). Specific rankings for this category are defined as follows:

- **3 = High:** The threat is *likely to have seriously reduced* the NEP population size.
- **2 = Moderate:** The threat is *likely to have moderately reduced* the NEP population size.
- **1 = Low:** The threat is *likely to have only slightly reduced* the NEP population size.
- **0= None:** The threat is not likely to have reduced the NEP population size.

Severity: Currently and Future (over the next 60 years) = the level of risk that this threat is likely to contribute to the decline of the population currently and in the future (over the next 60 years). Specific rankings for this category are defined as follows:

- **3 = High:** The threat is likely to *eliminate or seriously degrade* the NEP white shark population.
- **2 = Moderate:** The threat is likely to *moderately degrade* the NEP white shark population.
- **1 = Low:** The threat is likely to *only slightly impair* the NEP white shark population.

- **0 = None:** The threat is *not likely to impact* the NEP white shark population.

Level of Certainty: Historic and Current/ Future Risk Assessment = the level of certainty that the threat did affect, is affecting, or is likely to affect white shark populations. 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 did affect, is affecting, or is likely to affect the NEP population with the severity ascribed.
- **2 = Moderate:** There is *some* published and unpublished data to support the conclusion that this threat did affect, is affecting, or is likely to affect the NEP population with the severity ascribed.
- **1 = Low:** There is *little* published and unpublished data to support the conclusion that this threat did affect, is affecting, or is likely to affect the NEP population with the severity ascribed.
- **0 = None:** There is no published or unpublished data to support the conclusion that this threat did affect, is affecting, or is likely to affect the NEP population with the severity ascribed.

Current and Future Ranking of Threat Categories = The BRT's overall ranking for each category of threats. Specific rankings for this category are defined as follows:

- **High:** This threat category includes *a high number* of threats that are moderately or very likely to contribute to the decline of the NEP white shark population, or contains some individual threats identified as very likely to contribute to the decline of the NEP white shark population.
- **Moderate:** This threat category includes *an intermediate number* of threats that are likely to contribute to the decline of the NEP white shark population, or contains some individual threats identified as moderately likely to contribute to the decline of the NEP white shark population.
- **Low:** This threat category includes *a low number* of threats that are likely to contribute to the decline of the NEP white shark population.
- **None:** This threat category does not include any threats that are likely to contribute to the decline of the NEP white shark population.

Table 4.5 Summary of the BRT Threats Evaluation. For each threat, the BRT ranked the historic, current, and future severity and certainty (3= high, 2= moderate, 1= low, and 0= none). The average rankings and standard deviation are provided, as well as the BRT's overall ranking for each factor.

		Historic Threats			Current and Future Threats		
Threat Categories	**Specific Threats**	**Period of impact**	**Severity**	**Certainty**	**Severity**	**Certainty**	**Current and Future Threat Ranking**
Habitat destruction, modification or curtailment	Loss of prey (subadults and adults)	1800-1980	2.4 (0.5)	1.9 (0.4)	0.6 (0.5)	2.4 (0.5)	**LOW**
	Loss of prey (YOY and juveniles)	1970-2000	1.8 (0.7)	1.7 (0.5)	0.9 (0.4)	2.3 (0.5)	
	Environmental contaminants	1950-2012	1.6 (0.5)	1.7 (0.4)	1.5 (0.5)	1.8 (0.4)	
	Marine debris	Considered as Current/Future Threat			0.6 (0.5)	1.8 (1.0)	
Overutilization: commercial, recreational or education	High seas fisheries	1970-1992	2.1 (0.6)	1.5 (0.5)	1.1 (0.4)	1.8 (0.7)	**MODERATE**
	Mexico's fisheries	1990-2011	2.1 (0.4)	1.6 (0.5)	2.1 (0.4)	1.5 (0.5)	
	Coastal net fisheries (off California)	1970-1994	2.0 (0.6)	2.0 (0)	1.0 (0)	2.4 (0.5)	
	Recreational fisheries (off California)	Up to 1994	0.6 (0.5)	2.3 (0.9)	0.6 (0.5)	2.4 (0.7)	
Small population concerns	Allee effects: reproductive, genetic and demographic stochasticity	Considered as Current/Future Threat			0.9 (0.4)	1.9 (0.6)	**LOW**
	K- selected life history strategy				1 (0)	2.4 (0.5)	
	Stochastic and catastrophic events				1 (0)	2.1(0.6)	
	Heavily depleted				1.4 (0.5)	1.8 (0.7)	
Disease or predation	Disease	Considered as Current/Future Threat			1.0 (0.5)	1.1 (1)	**LOW**
	Predation				0.5 (0.5)	2.1 (0.8)	
Inadequacy of existing regulatory mechanisms		Considered as Current/Future Threat			1.3 (0.5)	2.3 (0.7)	**LOW**
Other	Ocean acidification	Considered only as a Current Threat			1 (0)	0.8 (0.7)	**LOW**
	Global warning				1 (0)	1.0 (0.5)	

4.2.1. Summary of threats assessment

Table 4.5 summarizes the BRT's assessment on the severity of individual threats to the NEP white shark population, based on the team's evaluation of the best available information regarding these threats. This assessment helped the BRT identify which threats should be incorporated into subsequent population analyses and informed the final assessment of extinction risk. The ranking indicates that the BRT considers the overall risks associated with climate change, disease and predation, habitat degradation, and small population size constitute low level threats to the NEP white shark population both now and within the next 60 years. Conversely, the BRT concluded that overutilization of white sharks as a result of fisheries bycatch is a moderate threat to the population, particularly as a result of bycatch in coastal fisheries off Mexico. The BRT felt that high-seas drift net fisheries and coastal gillnet fisheries presented a moderate threat to the population in the past (i.e., these fisheries were likely to have moderately reduced the population size of NEP white sharks). Since the early 1990s, however, changes to U.S. and international fisheries and their management have reduced the severity of impacts and the level of perceived threat. Changes have also been made to fisheries regulations in Mexico for the protection of white sharks, but their effectiveness is uncertain. Thus, the BRT continued to rank as a moderate threat to the population currently and into the future. For many threats (e.g. disease and global warming), the BRT felt there was only limited information to inform its evaluation and so expressed a relatively high degree of uncertainty associated with its evaluation.

4.3. Trend Information for the NEP White Shark Population

Trend information is considered to be highly informative in assessing the extinction risk facing a population (Musick et al. 2000, IUCN 2001). Below we summarize the available trend information for the NEP white shark population from a variety of different sources. Direct indictors of trends include analyses of fisheries catch and effort data and the photo-ID study at Guadalupe Island. Indirect indicators of trends include reports of white shark attack frequency on marine mammals and considerations regarding possible range expansion of the population.

4.3.1. Fisheries data

Catch data for U.S. fisheries provide an opportunity to examine trends in the NEP white shark population (see Section 4.1.1). Population trends can be evaluated by examining trends in CPUE which commonly serves as an index of relative abundance. For the CPUE analysis, the BRT focused on the California set net fishery which has accounted for a majority of the white shark bycatch off California since the early 1980s (see Section 4.1.1.3). Across the entire time series of available logbook data (1981 to 2011), CPUE in the set net fishery (calculated as the reported catch divided by the number of sets per year) appears to have declined from the early 1980s through the mid-1990s and generally increased since the mid-1990s (Figure 4.5, [can also refer to Table 4.3 of bycatch estimates in the fisheries bycatch section (Section 4.1.1)]). The period of increasing CPUE has coincided with a steady trend in decreasing fishing effort. It is possible that increasing CPUE during the 2000s could be due to increased reporting rates associated with the MBA program (see Section 4.1.1.3 and Appendix D), however, an increase in reporting rates

does not seem to fully account for the trend (see Section 4.1.1.3). Average soak time per set in recent years has also increased, further complicating the interpretation of CPUE trends.

The CPUE in 2011 was low in comparison to the previous years. The low number of sharks caught in 2011 (n=5) may be linked to cooler waters reported that year but a more in-depth analysis of environmental conditions across years is required. In 2012 the catch of sharks again increased and 12 sharks were reported in the MBA program (Winkler pers com.). These data were not included in analyses conducted in the status review because of the BRT's concerns about a reduction in involvement due to fishermen's concerns about the ESA listing petitions. Also in 2012 the vessel with the highest catch rate (average of 5.75 white sharks per year in comparison to 0.7 sharks per year for the rest of the fleet) left the fishery and consequently reported no catch.

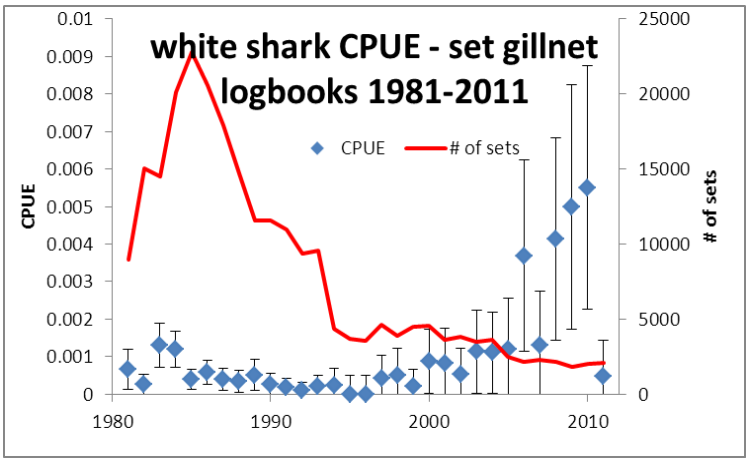

Figure 4.5 White shark CPUE data (+/- SD) for the California set net fishery from 1981-2011 as well as the total number of sets in each year compiled from fishery logbook data.

Over the course of the time series soak times increased in the set net fishery. To examine the potential impact of soak time on trends, the BRT used multiple linear regression to examine the impacts of both year and soak time on CPUE. To be conservative in light of potentially increased reporting rates since 2002, analysis was limited to data from1994-2001. Over this period the regulatory framework impacting set net fisheries catching white sharks did not change. Based on this analysis the BRT found there was a significant increase in CPUE over time and that soak time was not a significant factor (Figure 4.6).

It should be noted that currently the fishery is operating in only a subset of the white shark nursery habitat in the southern California Bight. A comparison of the distribution of fishing effort and YOY white shark habitat in the southern California Bight shows that heavy fishing now occurs in 28% of the CDFW spatial blocks (spatial blocks are as shown in Figure 4.2) that overlap with YOY white shark habitat based on information from SPOT tagging (C. Lowe, CSULB, pers. comm.).

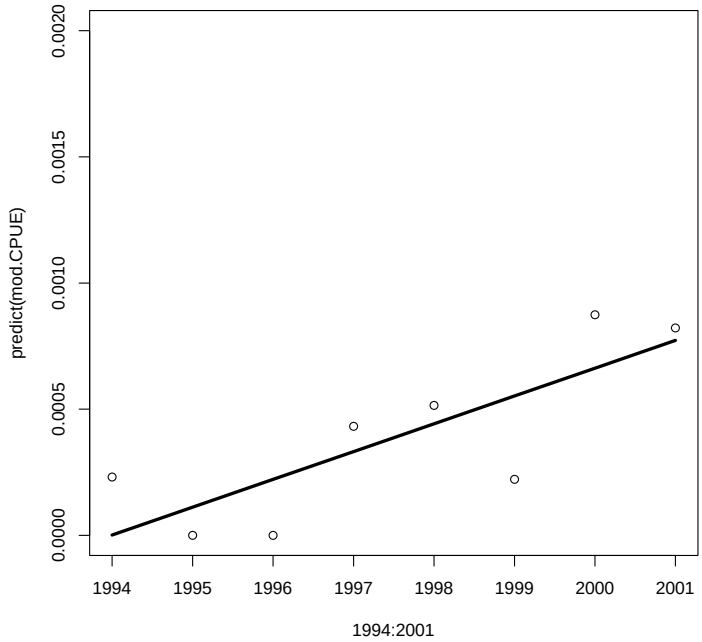

Figure 4.6 Regression analysis fit of white shark CPUE by year for the California set net fishery. The increase in CPUE with year from 1994-2001 is significant (p-value 0.017).

4.3.2. Guadalupe Island Photo-ID

The white shark photo-ID study conducted at Guadalupe Island provided the opportunity to examine trends in abundance over a period of 11 years. There were only sporadic sightings of white sharks at Guadalupe Island in the early 1990s (Domeier, MCSI pers comm) despite the regular presence of sport fishers. The photo-ID study was only initiated after white sharks sightings were sufficiently consistent to support a cage diving industry (Section 2.2.2.1). Thus, the photo-ID study was not initiated until after the number of white sharks at Guadalupe Island had begun to increase. As noted in Section 4.4.2 there was a substantial increase in estimated abundance at the site from 2001 – 2011 (Figure 4.11). Annual abundance estimates increased from ~ 40 in 2001 to over 90 in 2011. For males, this pattern occurred throughout the duration of the time series. For females, estimated abundance increased for the first several years before the estimates stabilized.

4.3.3. Trends in attacks on marine mammals and humans

Observations of white shark attacks on marine mammals have been documented at Southeast Farallon Island since the 1980s providing a relatively long time series of information. Over the last 30 years researchers working at Southeast Farallon Island have published a number of papers reporting an increase in white shark abundance around the islands based on the incidence of attacks on pinnipeds. Ainley et al. (1985) suggested that white shark populations were increasing in association with the increase in northern elephant seals at Southeast Farallon Island, and they

also reported an increase in the size of white sharks. Elephant seals were first seen at Southeast Farallon Island in the 1970s after which their presence at the island increased (Stewart et al., 1994). McCosker (1985) reported an increased incidence of white shark attacks on humans that they linked to an increase in white shark populations. These data were, however, confounded by the fact that more humans were likely spending more time in the water. At a 1996 white shark symposium Pyle et al. (1996) and Klimley and Anderson (1996) all concluded that white shark populations at Southeast Farallon Island were increasing given the increased number of attacks on pinnipeds even after accounting for the increase in pinniped populations that occurred during the 1970s and 1980s. In a more recent paper, Brown et al. (2010) found that variation in white shark attacks on northern elephant seal was correlated with the number of elephant seals present during the autumn haul-out. That their estimated white shark abundance index explained very little annual variation in attacks could indicate a stable white shark population or that their index does not accurately reflect annual variation in white shark abundance.

White shark attacks on marine mammals in other locations have also increased. At San Miguel Island, annual surveys of pinniped populations have been ongoing for decades (Jeff Harris, SWFSC pers. comm.). The Channel Islands are home to more than 100,000 California sea lions, a larger population than is found off central California. It is only in the last couple years that there has been evidence of attacks by white sharks on pinnipeds at these islands. While not all bite marks have been examined, a preliminary analyses shows that marks are consistent with a white shark bite. The increase in shark inflicted wounds is dramatic. In 2010 and in prior decades there were essentially no observed wounds on California sea lions. However, in 2011 there were around 136 recorded bite marks, and in 2012 there were over 300 bite marks recorded (Jeff Harris, SWFSC, pers. comm.). The wounds are observed primarily in June-August (Figure 4.7), although the occurrence of scars early in the year suggest that attacks may occur year around. Interestingly, wounds were primarily observed on juveniles and females. Given the known differences in their foraging range, with males migrating to areas north of Monterey Bay (Melin et al. 2010), this suggests that the attacks are occurring closer to the Channel Islands, within the foraging range of a lactating female. While the Channel Islands also produce 80% of the elephant seal pups in California, very few lesions were noted on elephant seals.

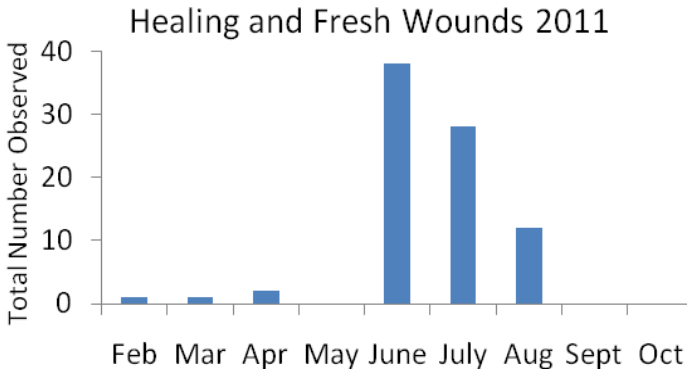

Figure 4.7 Total number of white shark-inflicted wounds (healing and fresh wounds) observed on California sea lions at San Miguel Island in 2011. Data for 2012 have not yet been fully analyzed (Jeff Harris, SWFSC, pers. comm.).

In addition to pinnipeds, white shark bite marks have been observed on southern sea otters (*Enhydra lutris nereis*) in central California. Researchers at the U.S. Geological Survey Western Ecological Research Center (USGS-WERC) report a dramatic increase in the number of mortalities linked to white shark bites over the past 5 years, particularly in the region between Estero Bay and Pismo Beach, although increases have also been seen in Monterey Bay and areas north of Santa Cruz. Overall the proportion of beach-cast sea otter carcasses for which shark bites were the primary cause of has increased 3-4 fold over the past 5 years, compared to the long-term average (1983-2000). Shark-bite trauma has now become the single most frequently observed cause of death (USGS-WERC, unpublished data, (http://www.werc.usgs.gov/ProjectSubWebPage.aspx?SubWebPageID=22&ProjectID=91). Although definitive evidence for the species of shark responsible for the trauma is only available for 10-20% of carcasses (i.e. where tooth fragments or tooth scrapes on bone are found), in all such cases the evidence indicates white sharks as the species responsible for the trauma, with no evidence found as of yet to indicate other shark species. While other factors are likely impacting sea otter population trends in California, the increased level of shark-bite mortality is thought to be linked to sea otter population declines in some regions.

For both sea otters and sea lions it is difficult to differentiate whether the increase in encounters is associated with an increase in the population or a shift in habitat or foraging strategy. It could also be a combination of these factors. One factor that could provide some insight is whether the rate of increase is too fast to be explained by an increase in abundance. This is made difficult by the lack of information on the frequency of successful and failed predatory events that would provide some insight into how encounters may translate into numbers of sharks. One important point is that there is no noted decrease in white shark interactions in other locations. The uncertainty in how the increase in encounters translates into a change in abundance is reflected in the SEDM voting process on population trends.

Reports of shark attacks on humans are archived by a number of organizations. Data from the International Shark Attack File (http://www.flmnh.ufl.edu/fish/sharks/ISAF/ISAF.htm), the Global Shark Attack File (http://www.sharkattackfile.net/incidentlog.htm), and a published book on shark attacks of the North America Pacific Coast (Collier 2003) have been summarized in Figure 4.8. The BRT received spreadsheets of the data directly from the database manager of the International Shark Attack File and from R. Collier, the author of the published book, and downloaded data from the Global Shark Attack File website. The data were filtered to include only unprovoked attacks that were confirmed attributed to white sharks along the coasts of Alaska, Washington, Oregon and California. As the 3 sources are presumably making an effort to tabulate all attacks, the majority are expected to be represented in each of the databases. In fact, the databases do cross reference each other's listings for a large number of the documented attacks. Although there are some differences, perhaps due to the criterion used to distinguish a provoked vs. an unprovoked attack, or the confidence with which a definitive identification of the species could be made, the data show a slightly increasing trend in the number of attacks by year. When examined by decade, the trend becomes more apparent. However, it is important to note that the increase in white shark attacks cannot alone be used to indicate an increase in the number of white sharks present along the U.S. West Coast. Since the 1950s, there has been marked population growth with greater numbers of people utilizing the ocean for recreation, as well as enhanced media coverage through the internet and social media networks. The BRT did

not consider the increase in the number of shark attacks to be an indicator of an increase in the white shark population.

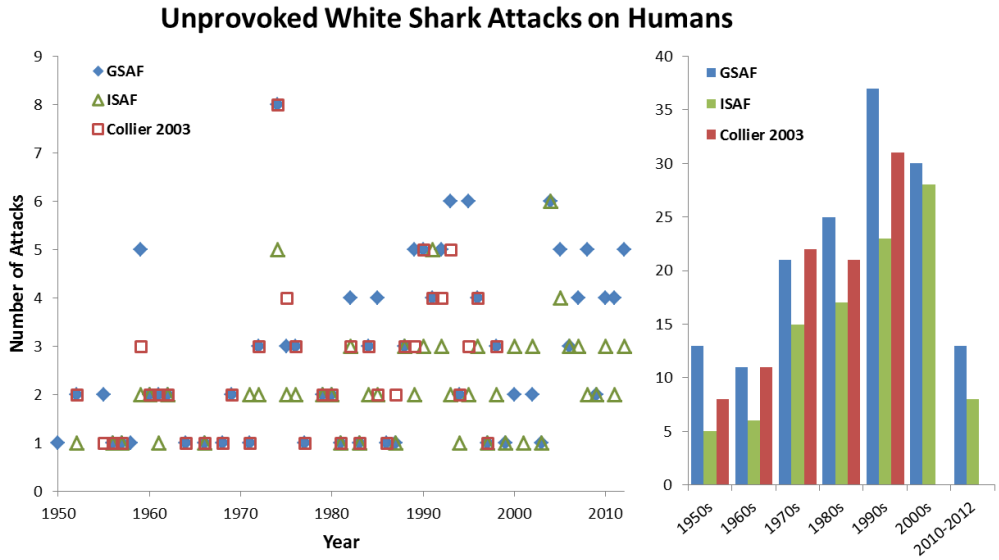

Figure 4.8 The number of unprovoked white shark attacks along the U.S. West Coast recorded since 1950 by year (left) and decade (right). The data are from three sources with a high degree of overlap: the Global Shark Attack File (GSAF), International Shark Attack File (ISAF), and Collier, 2013.

4.3.4. Range

In addition to evaluating trends in abundance and other indicators, information on range expansion or contraction can provide insight into the status of a population. Range expansion is typically thought of as when a population uses new areas or sites outside of its current range, but can also include an expansion in high use of areas or sites within a population's current range. The increase abundance of white sharks at Guadalupe Island over the past 20 years suggests there may have been a geographic expansion of the core near-shore aggregation sites. In addition, the increase in white shark bite marks on sea lions and sea otters south of Monterey Bay suggests an increased presence of white sharks in this region. While the coastal waters from the Channel Islands to Monterey Bay are clearly within the historical range of white sharks along the coast of California, the majority of white shark activity in the last 10 years has been reported at Guadalupe Island and central California sites.

4.3.5. Evaluation of Trend Information

The BRT used the SEDM process to evaluate whether or not the available trend information indicates the NEP white shark population abundance is increasing, decreasing, or stable while also assessing the BRT's level of uncertainty. Overall, the BRT concluded strongly that the available information supports that the NEP white shark population is either increasing or stable (71% of plausibility points, Table 4.6) rather than decreasing (6% of plausibility points). Other information the BRT felt was consistent with their determination that the population is increasing included: recent increases in marine mammal and fish prey resource for all life stages and changes in near-shore and high seas fisheries that have reduced fisheries impacts. As mentioned,

the human attack data did not influence the assessment of trends. The BRT expressed a moderate level of uncertainty (23% of plausibility points) about whether the available information supports a determination that the population is increasing due to the lack of historic information on abundance, uncertainty about female mortality levels, and uncertainty about whether there have been changes in the white shark range or foraging strategies rather than an increase in the overall population size. Individual plausibility point allocations and arguments for the difference categories are in Appendix E.

Table 4.6 SEDM voting results regarding whether the NEP white shark population is increasing, decreasing, or stable. The average plausibility points out of 100 per category across all BRT members are shown.

	Increasing	Stable	Decreasing	Uncertain
Avg %	42	29	6	23
Stdev	6.5	9.4	8.6	16

4.4. Population Abundance and Assessment of Extinction Risk

4.4.1. Overview of Approach used by BRT

A critical element of this status review is an assessment of the current and future extinction risk of the NEP white shark population. Information needed to conduct a quantitative assessment of extinction risk include estimates of adult female abundance, population vital rates corresponding to maximum potential population growth, and assessments of human-caused mortality on population growth. The BRT's assessment of threats to the NEP white shark population concluded that fishery bycatch was the most highly ranked threat, and therefore, the focus of the quantitative risk assessment was on evaluating impacts of fishery bycatch on the population over time.

Few of the parameters necessary for a risk assessment have been well-quantified for white sharks, so the BRT undertook the following process to obtain plausible estimates (also see Figure 4.9):

1) The BRT's first objective was to define plausible ranges of the true NEP white shark abundance, with an emphasis on estimating adult female numbers. As discussed below (Sections 4.4 and 4.5), the BRT concluded that the available published estimates for NEP white shark abundance necessitated some level of correction to account for potential sampling biases that likely resulted in underestimating both the NEP population and the total abundance of adult females. The BRT therefore re-analyzed the original (but since updated) abundance data to re-estimate abundance for the adult female component of the sampled population. Based on this re-analysis and a consideration of other information, the BRT then defined several plausible types and levels of sampling-bias which were used to develop a range of true adult female abundance estimates.

2) To develop estimates of true female abundance, the BRT first used a SEDM evaluation process to express its uncertainty regarding specific types and levels of sampling bias associated with the current white shark abundance estimates (Section 4.4.3). The results

of the SEDM process were used to calculate 48 alternative abundance estimates each having different weights based upon the BRTs assessment of bias. These abundance values were ranked and then grouped into four female abundance categories defined by an upper and lower limit (60-125; 125 – 200; 200 – 400; 400-1600). The sum of weights for abundance values within each category provided a total weight associated with that abundance range and these weights represented the BRT's collective view about which abundance category most likely included the true abundance of female white sharks in the NEP.

3) Population viability analyses (PVAs) were conducted for each female abundance category described above. In addition to using female abundance inputs, these analyses required information about demographic vital rates and estimates of fisheries bycatch mortality. Empirical bycatch mortality estimates were available for YOY and juvenile white sharks (see Section 4.1.1) but not for larger subadults and adults, and therefore, the BRT conducted a SEDM assessment to define and assign weights to 5 plausible levels of annual adult female bycatch in fisheries (Section 4.4.4). Combining these 5 bycatch scenarios with the 4 abundance categories described above resulted in 20 population viability models that were used to conduct PVA. The PVA results provided the probability of adult female abundance falling below specific thresholds in specific timeframes as a function of female population size and fishery bycatch level. The BRT defined four extinction risk levels (High, Medium, Low and Very Low) that incorporated different abundance thresholds and timeframes and then evaluated the PVA results against these risk levels. Each outcome from the 20 models corresponded to High, Medium, Low, or Very Low population extinction risk (as defined by the BRT in Section 4.5.2.3).

4) Results of the PVA provided important information to the BRT about the feasibility of different scenarios for abundance bias. Specifically, existing estimates of YOY and juvenile bycatch were incompatible with some of the lowest abundance scenarios that the BRT initially considered in the modeling exercise. In light of the new insights provided by the model results, BRT members re-evaluated bias regarding the proportion of the NEP population that is available for photo-ID at the aggregation sites. The new weights were propagated through the PVAs, resulting in revised weights for the 20 PVA models and four risk categories.

5) The results of the PVA exercise reflected the BRTs conclusion about risk to the NEP white shark population associated with fisheries-related mortality, but did not consider any other information that might influence perception of risk. The BRT therefore conducted a final SEDM assessment (Section 4.6) to evaluate overall extinction risk taking into consideration the PVA modeling outcomes, trend data or the influence of other threats on extinction risk. This assessment resulted in the BRT's final conclusion about the likelihood of the NEP population being at risk of extinction currently or in the foreseeable future.

These steps are described in detail below (also see Figure 4.9).

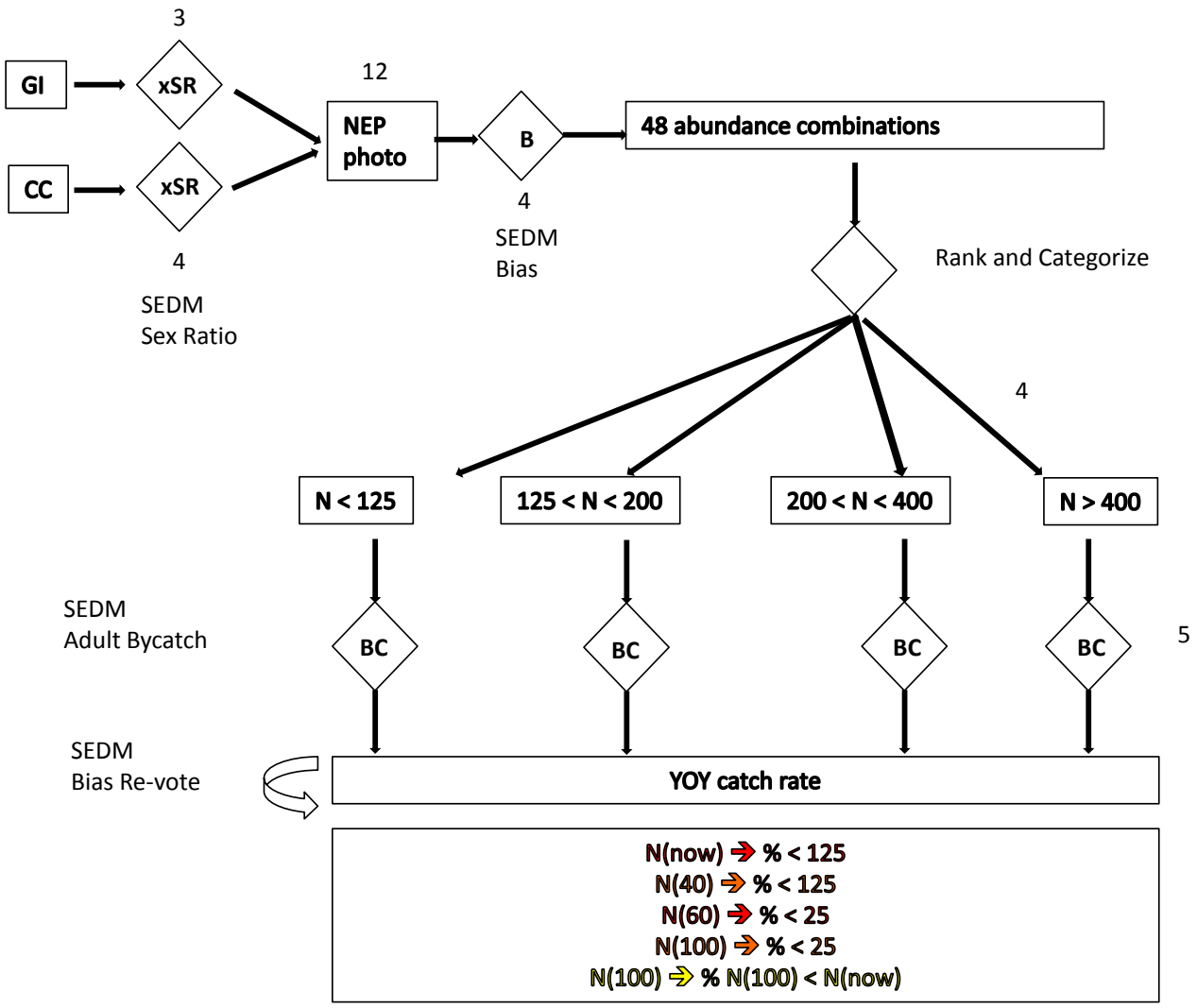

Figure 4.9 Flow diagram for the process used to examine bias in sex ratio and population abundance and to ultimately evaluate the plausibility of different risk models. Starting from the upper left, the number of adult females from each study site (GI = Guadalupe Island, CC = central California) is multiplied by the sex ratio multiplier options (3 options for GI symbolized by the '3' above the 'xSR' diamond, and 4 options for CC) to result in 12 plausible total abundance estimates for the study sites (NEP photo). The weights for each of these options results from the SEDM Sex Ratio plausibility point allotment. The NEP abundance estimates were then multiplied by a bias factor to account for the potential number of sharks that do not come to either study site (diamond 'B' for bias with plausibility points resulting from the SEDM bias allotment). The 4 bias multipliers resulted in 48 abundance combinations that were then ranked from lowest to highest and broken into 4 categories to represent ranges of possible abundance (N). Population Viability Analyses were run for the 4 abundance categories and for 5 plausible levels of annual adult bycatch (ranging from 0 to 10) to a total of 20 models. For each model the following output was used in evaluating risk: YOY catch rate (now), and N at the following times: now, 40, 60 and 100 years. Following the initial analysis the BRT re-voted on the bias options. This re-vote did not change the models that were run but rather the weight assigned to each of the 20 models. The percent of the runs that met various criteria determined whether that particular model would qualify as high (red), medium (orange), low (yellow) or very low (none of the above and not shown).

4.4.2. Mark-recapture analyses

4.4.2.1. Background

The white shark listing petitions submitted to NMFS cites an estimate of approximately 339 subadults and adults in the NEP. This estimate is based on using mark-recapture methods in published photo-ID studies at two sites in central California (the Farallon Islands and Tomales Point) (Chapple et al. 2011) and one site at Guadalupe Island (Sosa-Nishizaki et al. 2012). Concern that the number of adults and subadults is only 339 individuals was a primary motivation for both listing petitions. Consequently, the BRT re-examined the Guadalupe Island and central California datasets as well as additional data provided by the researchers at each study site.

Using mark-recapture data to estimate the population size of white sharks in the NEP relies on a number of assumptions: 1) animals do not lose their marks with time; 2) sampling time is negligible compared to mixing time; and 3) all individuals modeled as a single category (i.e., gender in models that include it as a factor) have the same survival probability (Cooch and White 2011). The most important final assumption 4) is that all members of the population being estimated (all subadult and adult sharks in the NEP) have equal probabilities of being photographed at the study sites. To address the validity of this assumption requires information on demographic, temporal and spatial patterns throughout the population's range relative to the area where sampling occurs. This information is challenging to obtain given that the vast majority of studies on NEP white sharks have occurred in just two study areas. However, new data on size and sex ratios made available to the BRT by research groups working both at Guadalupe Island and in central California provided some insight into relevant demographic information.

The BRT's goals for analyzing the photo-ID data from the two aggregation sites were to: 1) reexamine both datasets in addition to newly available data, 2) evaluate the potential bias in the NEP white shark population estimates by examining the demographics at each site and the assumption about equal probability of capture, 3) examine trends in abundance at Guadalupe Island and estimate parameters characterizing the population dynamics at this site, and 4) calculate lower bounds for estimates of female abundance and the male-female sex ratios in the NEP.

For a detailed description of the methods of the mark-recapture analyses, see Appendix F of this report. The sections below provide an overview of the methods and the main results and conclusions from the analyses.

4.4.2.2. Methods

The BRT was provided with the raw mark-recapture datasets from research groups working at the central California and Guadalupe Island study areas, including new size and sex data when available. The central California data used by the BRT were the same as used in the analysis by Chapple et al. (2011), but with updated information about the sex of many animals that was not previously known. The Guadalupe Island data used by the BRT included two more years of data

than were used by Sosa-Nishizaki et al. (2012) as well as additional data on the number of days of sampling effort per month (Appendix F, Table 2). This allowed the modeling analysis to incorporate changes in sampling efforts later in the year when females are more likely to be seen. Mark-recapture modeling analysis for data from both sites was conducted using POPAN open population models. POPAN (POPulation ANalysis http://www.cs.umanitoba.ca/~popan/) is a model that estimates population size from mark-recapture data without requiring the assumption that the population was closed during the study period. A closed model is one in which the population is not allowed to change in size whereas in an open model the population is allowed to change in size through emigration, immigration, recruitment or mortality. Chapple et al. (2011) used a closed population model for their mark recapture analysis, but the BRT felt that an open model was more appropriate. The open mark-recapture models provide an estimate of population size, an estimate of apparent survival (φ) of animals in the population (i.e., proportion of animals that return to the site in subsequent year where animals can either be lost to mortality or permanent emigration), and the probabilities of detection (p) and recruitment (p_{ent}).

A total of 3 datasets were modeled 1) the 3-year dataset for central California (2006 - 2008), 2) the complete 11-year dataset for Guadalupe Island (2001-2011), and 3) a 3-year subset (2006-2008) of the Guadalupe Island data. For each dataset, alternative model configurations (combinations of input parameters) were evaluated with the best models selected based on AICc which is a measure of how well a model describes the patterns in the data (i.e., goodness of fit). The different model configurations included various combinations of parameters that either did not vary or varied with time and sex, with and without interaction terms. For the models based on the final 11-year Guadalupe dataset, parameter values were evaluated from the best models with and without covariates for total days of sampling effort per year, and a covariate for the monthly proportion of sampling effort that occurred in November to December.

4.4.2.3. Results and Conclusions

Demographics:
Size and sex data of white sharks at both aggregation sites were examined to characterize the demographics of the sharks "captured" in the photo-ID studies. At both study areas the majority of white sharks were mature and the sex ratio was biased towards males (Figure 4.10). The overall ratio of female to male sharks (adults and subadults) documented in the photo-ID database at Guadalupe Island as of 2012 was 1:1.6 although the sex ratio changed seasonally (Section 4.4.3). At the two sites in central California the female: male sex ratio was 1:3.8, although this could change once the gender for additional white sharks is eventually determined. The percentage of mature white sharks at the central California sites was ~ 85% (Figure 4.10) with 57% of females being mature. At Guadalupe Island the most reliable size data come from SPOT tagging efforts where animals are actually measured. Based on this dataset, 90% of white sharks at Guadalupe Island are estimated to be mature; however, this estimate may be biased by the use of whale blubber as bait which was used in the later years of the study and appeared to attract larger animals. In the first year of the study, fish was used as bait, which likely attracts smaller individuals. Earlier estimates that 60% of the sharks at Guadalupe Island were mature (Domeier and Nasby-Lucas 2007) were based on rough estimates of length and are likely unreliable (N. Nasby-Lucas, MCSI, pers. comm.). The actual portion of the Guadalupe Island white sharks observed that are mature is most likely somewhere between 60% and 90%.

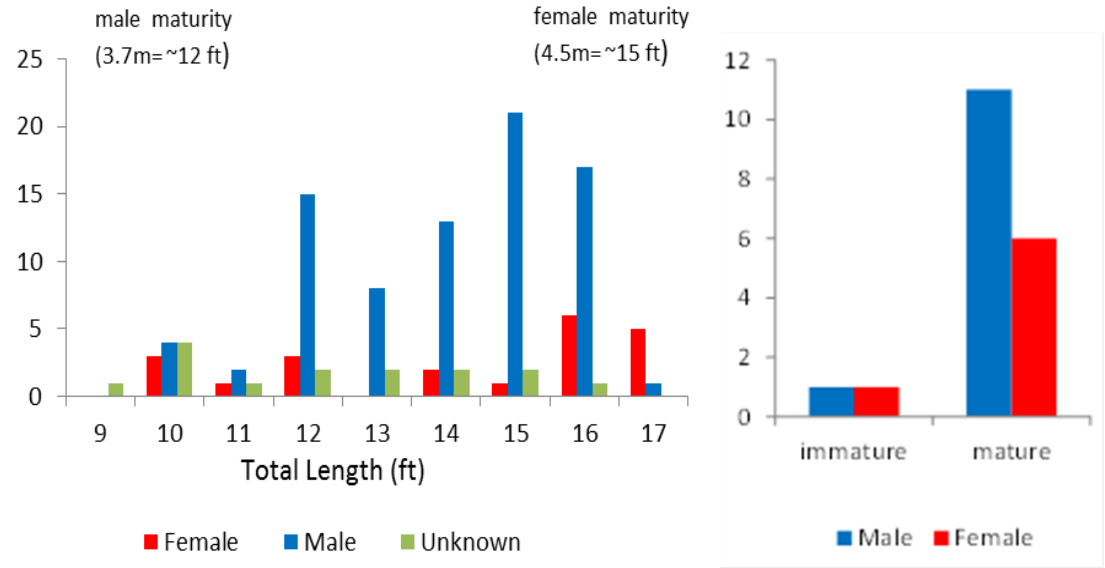

Figure 4.10 Number of mature and immature males and females at the sites in central California (left) and at Guadalupe Island (right). Size at maturity is assumed to be 12 ft. for males and 15 ft. for females. Data from central California and Guadalupe Island courtesy of S. Jorgensen and N. Nasby-Lucas, respectively.

These data suggest that mature males are the dominant component of the population available at both aggregation sites given the sampling designs that have been used. Potential biases that could explain the relatively low number of mature females in the datasets are discussed in Section 4.4.3. There are a number of reasons why fewer subadults than adults may be documented at the two study areas. White sharks are known to exhibit a size-based dominance feeding hierarchy with the presence of larger individuals causing smaller animals to leave the area which would impact the capture probability of subadults (Goldman and Anderson 1999, Sperone et al. 2010, Domeier 2012). Subadults may also be less likely to visit the two aggregation sites if they have not yet started feeding on marine mammals or begun to seek mates. In short, the results of demographic analysis suggest that subadult white sharks do not have an equal probability of capture in the two study areas when studies are conducted and consequently that the number presented in the petition does not include all subadult white sharks in the NEP.

Abundance estimate for central California aggregation sites, 2006-2008:
A total of 131 white sharks (21 female and 81 males) were recorded in the photo-ID studies conducted at the two sites in central California from 2006-2008. Using these data, the BRT used a POPAN model to estimate the "super-population" of NEP white sharks at the two central California sites over the 3 year time period. A super-population represents all the individuals that were observed at the site during the study, including those that have since died or emigrated from the site. Given that the time series includes only three years of data and that mature females visit semi-annually the values calculated by the POPAN model for survival (ω), the probability of detection (p) and entry (p_{ent}) are not reported. The BRT's population model generated a mean 3-year super-population estimate of 166 white sharks at the two central California sites over the 3

year period from 2006 – 2008, including 25 (±9) females, 97 (±28) males, and 44 (±120) individuals of unidentified sex. For comparison, Chapple et al.'s (2011) closed model generated a mean abundance estimate of 219 white sharks at the central California sites, with a "credible" (95% confidence) range in estimates of 130 to 275 white sharks. Thus, the BRT model's mean estimate was within the credible range of the abundance estimates generated by Chapple et al.'s (2011) model. In the supplement, Chapple et al. (2011) report the results for a number of different models including an open population model (Jolly-Seber), similar to the POPAN model. The 3-year super population estimate for this model was 156 sharks, also similar to the results from the POPAN model run by the BRT.

To calculate the number of adult females at the central California sites, the BRT used the POPAN model results, information about sex ratios (estimated to be 26% female) and proportion of females considered to be mature (estimated to be 57%) (Figure 4.10). Of the estimated number of females at the sites, the BRT calculated that 14 were adult females (25 females*0.57 mature = 14). For the estimated number of individual of unknown sex at the sites, the BRT calculated that 6 were adult females (44 individuals*0.26 female*0.57 mature = 6). In total therefore, the BRT estimated there were 20 adult females at the two central California study areas.

Abundance estimates for Guadalupe Island, 2001 - 2011:
During the eleven year study period from 2001 to 2011 at Guadalupe Island, a total of 142 white sharks (54 females and 88 males) were recorded in the Guadalupe Island photo-ID database. The BRT's best super-population estimate for white sharks at Guadalupe Island from 2001 to 2011 was 154 which included 62 (±6) females and 92 (±5) males. Due to the incorporation of two additional years of data and the increasing population trend at Guadalupe Island, this estimate is higher than the super-population estimate of 120 white sharks reported by Sosa-Nishizaki et al. (2012). The BRT also estimated the super-population size from 2006-2008 for Guadalupe Island, to be consistent with the analysis for the central California study. Over this period, the estimated white shark super-population size for Guadalupe Island was 84 white sharks, including 52 males and 32 females. Using the estimated proportion of mature individuals at the site (85%) from Figure 4.10, the BRT calculated that there were 27 mature females at Guadalupe Island (32 females*0.85 mature females). The female:male sex ratio over this three year period was 1:1.7.

Because of the longer time series of data available in the Guadalupe Island study (11 years) compared to the central California study (3 years), the BRT also used the population model to examine other population parameters mentioned above (annual population size, survival, the probability of detection and entry). Trends in the estimated annual population size at Guadalupe Island showed an increasing trend in the annual abundance of males from 2001 to 2011 with the estimated number of males essentially doubling over the 11-year period from about 30 to about 60 individuals (Figure 4.11). Estimated numbers of females also showed an increasing trend for the first several years of the study before stabilizing during the later years of the series. Changes in sampling are likely to have influenced the annual female abundance estimates after 2007 (Figure 4.11) given the late arrival time of females (Figure 2.4) and reduced sampling in November and December (Appendix F, Table 2) later in the time series.

Based on the model, apparent survival varied by year and by sex and was lowest in 2005 and 2009. Females had lower apparent survival values than did males (mean was 0.93 for males and 0.9 for females; median was 0.98 for males and 0.95 for females), indicating that emigration and/or mortality are higher for females than for males Whether emigration or mortality is driving this difference cannot be determined from these models. The probability of detecting a white shark given that it was present (p) also varied by sex, by sampling effort per year, and the proportion of sampling effort during November to December (since males return to Guadalupe Island in July/August and Females return September/October) (see Figure 2.4). The detectability of females was lower (0.5) and more variable than for males (0.6) indicating that temporal patterns in data collection impact the probability of detecting females. The probability of entry/recruitment (p_{ent}) was a constant value of 0.07 for both sexes indicating that new male and female white sharks entered the population at the same rate.

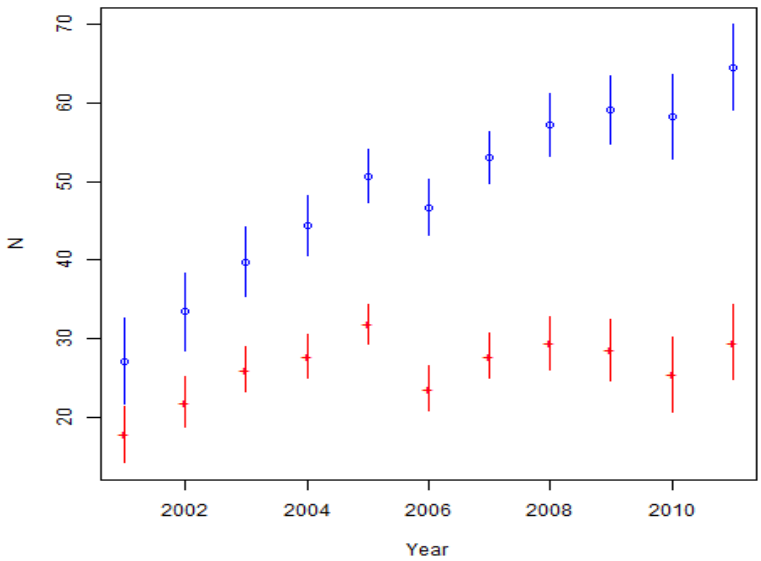

Figure 4.11 Population model estimates of the numbers of white sharks present (N) each year at Guadalupe Island from 2001 to 2011. Error bars represent ± one standard error (SE). Female abundance estimates are indicated in red, males in blue. The models included a sex-specific covariate for monthly sampling effort.

Combined adult female abundance estimate for central California and Guadalupe Island: Combining estimates from the two study areas yields an adult female estimate of 47 for the period 2006 - 2008. To calculate a variance around this estimate for use in subsequent models the BRT used a parametric bootstrap approach. For each bootstrap sample, random abundance values were drawn from the mark-recapture estimates at each study area (which assumed normal distributions with corresponding means and coefficients of variation), along with random values for the proportion of unknown-gender animals from central California that were female and the proportion of females that were mature (using appropriate beta distributions). The resulting combined estimate was 47 ± 4.1.

The BRT's re-analysis of the photo-ID data from central California and Guadalupe Island provided female abundance estimates for the two sites as well as other information needed for the population modeling analysis discussed in Section 4.5. A key question, however, is whether or not the estimates for the two aggregation sites include all subadult and adult white sharks in

the NEP. The BRT's examination of the demographic data for the central California and Guadalupe Island provided important insights into potential sources of bias that relate to this question. As previously discussed, the white shark populations at both sites are dominated by mature males and are underrepresented in terms of females and subadult sharks. The BRT considered this demographic data as well as other information and concluded that the population estimates for the two sites do not represent the actual numbers of subadult and adult white sharks in the NEP. However, the estimates for central California and Guadalupe Island do provide a lower bound estimate of the numbers of white sharks at these sites and both the abundance estimates and observed sex ratios were used by the BRT to characterize bias in sex ratio and the NEP white shark population size as discussed in the following section.

4.4.3. Assessment of bias in sex ratio and abundance estimates

Given the apparent skew in the sex ratios found in central California and at Guadalupe Island based on the demographic data and associated concerns about bias in the studies, the BRT concluded that the direct empirical estimates of adult female abundance from the photo-ID study areas likely underestimated true adult female abundance in the NEP. To address this underestimation and generate estimates of the total number of adult females for use in the population modeling, the BRT used SEDM to characterize the magnitude of potential bias. The first step in this process was to characterize the bias in the observed sex ratios in order to estimate the total number of adult females at the two study sites. The second step was to characterize what proportion of adult females in the NEP population as a whole, do not visit the aggregation sites so that a total NEP population abundance could be estimated. These two SEDM exercises are described in the following sections.

4.4.3.1. Sex Ratio Bias

Background:
At the central California and Guadalupe Island study sites, the photo-ID data are dominated by observations of males. Based on the available data, the observed female to male sex ratio for mature white sharks was 0.6 at Guadalupe Island and 0.2 at the central California sites. There are several possible explanations for the observed skew in the sex ratios including a number of potential biases linked to the assumption that males and females have an equal probability of capture. A more detailed description of these is provided by Sosa-Nishizaki et al. (2012). Possible explanations for the skewed sex ratio are discussed below and were used by the BRT to inform its SEDM assessment of sex ratio bias.

1) The population is truly skewed, which would be possible if female mortality is substantially higher then male mortality (sex ratios of pups are close to parity, indicating that the skew is not due to more males being born than females). This differential mortality would have to occur throughout the juvenile stages, prior to the subadult phase because the observed sex ratio skew is already apparent for the smallest subadults in the central California aggregation (see Figure 4.10). However, there is no evidence of sex-biased mortality on juvenile female white sharks, and the sex ratio of sharks captured in near shore fisheries is close to parity (see Section 4.1.1.3).

2) White sharks, like other lamnids including salmon and mako sharks, have been shown to exhibit sexual segregation. Sexual segregation is reflected in sex ratios that deviate from parity (are not 1:1). In nearly all places around the world where white sharks have been surveyed, the sex ratios of pups both in utero and in the environment are close to parity (see Sections 4.1.1.3), whereas the sex ratios in the observed juvenile, subadult and mature age classes are biased. Globally, female:male sex ratios range from 1:2.4 on the U.S. east coast (Casey and Pratt 1985), 1:2 in New Zealand (C. Duffy pers. comm.), to 1:0.2 in Australia (Bruce 1992). One recent study in South Africa showed a female:male sex ratio of 1:3 with both seasonal and spatial shifts in the sex ratios of juvenile and subadult white sharks over relatively small spatial scales (Robbins 2007). In the NEP, sexual segregation is apparent offshore with females using a larger, more dispersed area and most males occupying a more concentrated region in the offshore focal area (Jorgensen et al. 2010, Domeier and Nasby-Lucas 2012).
3) Some females may not be observed at the studied aggregation sites, because they arrive later in the season after most photo-ID sampling effort has ceased. Due largely to weather patterns, the majority of sampling effort at both Guadalupe Island and at the central California sites has occurred opportunistically over a period of 2-4 months in the late summer and fall and sampling does not cover the entire period for which sharks are present. Based on work at Guadalupe Island the sex ratio shifts from 1:18.8 in August to 1:0.9 in November (Nasby-Lucas and Domeier 2012) indicating that temporal patterns in sampling effort can impact the apparent sex ratio.
4) It is possible that some females at the aggregation sites are simply not available to be sampled for behavioral reasons (a more detailed explanation for which is provided in (Sosa-Nishizaki et al. 2012)). White sharks show strong site fidelity and clear preferences for particular coastal sites (Jorgensen et al. 2010). Females may be less reliant on pinniped prey or have a reduced proclivity to spend time near the surface or approach dive cages.
5) Mature females have a presumed 18 month gestation cycle and many do not return to the coastal aggregation sites each year. The mark recapture models used by the BRT estimated a lower capture probability for females and consequently adjusted for the overall lower capture probability for females (Section 4.4.2). Nevertheless, the relatively short time series at central California could result in poor estimation of the capture probability for females and consequently underestimate abundance.

Quantifying Bias in Sex Ratios using SEDM:

Given the concerns about sex ratio bias in studies at the two aggregation sites, the BRT used SEDM to evaluate the plausibility of a number of different sex ratio alternatives at each site. For both study areas, the highest sex ratio considered was a 1:1 ratio and the lowest sex ratio was that estimated from the 3-year superpopulation estimates (2006-2008) at the two sites (0.6 at Guadalupe Island; 0.2 at the central California sites). For central California, two other intermediate sex ratios were considered (female:male ratios of 0.6 and 0.8). These two intermediate values were used because they were consistent with the ratios considered for Guadalupe Island and to provide intermediate values between the observed and 1:1 ratio. For Guadalupe Island, one intermediate sex ratio was considered (female:male ratio of 0.8) which was intermediate between the observed and 1.1 ratio. Prior to the SEDM voting, the BRT

constructed and considered arguments for each sex ratio value, based on the best available information. These arguments are given in Appendix G.
For the central California sites, the BRT allocated plausibility points across the following 4 sex ratio categories which represent different proportions of females in the population:

- 0.2: The observed proportion of female at the central California sites from 2006 - 2008
- 0.6: The observed proportion of female at Guadalupe Island from 2006 - 2008
- 0.8: The mid-point between 0.6 and 1
- 1.0: Representing the equal sex ratio observed in earlier age classes.

For Guadalupe Island the BRT allocated plausibility points across the following 3 sex ratio categories which represent different proportions of females in the population:

- 0.6: The observed proportion of female at Guadalupe Island from 2006 - 2008
- 0.8: The mid-point between 0.6 and 1
- 1.0: Representing the equal sex ratio observed in earlier age classes.

Based on the results of the SEDM assessment, the BRT concluded that the female:male sex ratios at both Guadalupe Island and in central California are most likely higher and that there are more females than suggested by photo-ID data and mark-recapture analyses (Table 4.7). For Guadalupe Island, the BRT felt that the most plausible female:male sex ratio was 0.8 and for central California that the most plausible sex ratio was 0.6. The main factor influencing the BRT's assessment was the timing of the sampling season at both sites relative to the late arrival of the females which would result in under sampling of females. In addition, the BRT noted that skewed sex ratios are common in white sharks around the world and therefore the skewed ratios observed at these sites were not surprising. The BRT, however, did express some concern that the difference in survival (i.e., individuals that return and are not lost due to emigration and mortality) between males and females could be linked to a higher mortality for females given the time they spend in near shore habitat overlapping with fisheries although gear vulnerability for such large sharks is likely lower than for smaller individuals. The observed sex ratios at the two sites also influenced the BRT's assessment as the sex ratios receiving the most plausibility points for the central California sites (0.6) was lower than that for Guadalupe Island (0.8).

Table 4.7 SEDM voting results regarding the plausibility of different sex ratio (Female:Male) values for Guadalupe Island and the central California study sites. The average number of plausibility points and standard deviation across all BRT members are shown for each sex ratio value considered.

	Guadalupe Island sex ratio values			**Central California sex ratio values**			
Sex ratio (F:M)	**0.6 (obs)**	**0.8**	**1.0**	**0.2 (obs)**	**0.6**	**0.8**	**1.0**
Avg	33	48	20	4	43	36	18
Stdev	18	20	10	4	15	14	9

4.4.3.2. Abundance Bias

Background:
In addition to there being a bias in the sex ratio, the BRT felt that there were additional factors which bias the estimate of white shark abundance in the NEP. There are a number of reasons to

believe that there are more adult female white sharks (and a larger overall population in the NEP) than have been estimated at the central California and Guadalupe Island aggregation sites.

1) The central California abundance estimate does not include white sharks at other local sites that are documented to be important white sharks hotspots, including Año Nuevo State Park. There is a long history of white shark activity at Año Nuevo, the site of the largest mainland breeding colony of northern elephant seals. Acoustic tagging studies in central California (Jorgensen et al. 2010) showed that some individuals exhibited site fidelity to particular coastal sites and did not move between sites over the course of the study which would impact their availability to be photographed at Southeast Farallon Island or Tomales Point. Such site fidelity (sometimes over small spatial scales) has been recorded in other areas across the species range including South Africa (Bonfil et al. 2005); Australia (Bruce and Bradford 2012; 2013) and New Zealand (Duffy et al. 2012). Similarly, there are multiple sites around Guadalupe Island where white sharks are observed, but photo-ID studies only occurred at one of these sites.
2) White sharks may occupy unknown or previously unoccupied areas. In the southern California Channel Islands and off central California there appears to be an increase in the occurrence of white sharks (see Section 4.3.3 and 4.3.4). The Channel Islands are home to a population of over 80,000 California sea lions and so the increasing occurrence of white sharks there would not be surprising. Other potential aggregation sites where pinnipeds are known to be common include the Coronados Islands and Cedros Island in Mexico. While not validated by scientists, fishers off Mexico have reported large white sharks at some of these islands (O. Sosa-Nishizaki, CICESE, pers. comm.). White sharks are also observed away from the main aggregation sites off Alaska, British Columbia, Canada, Washington, Oregon, California, Mexico, and into the Gulf of California (Klimley 1985, Martin 2004, Galván-Magaña et al. 2010). Some sharks tagged at the two known aggregation sites have been observed to visit other coastal areas (Jorgensen pers. comm., Domeier and Nasby-Lucas 2013, Nasby-Lucas, MCSI, pers. comm.) but the data are limited and information on the extent of coastal areas used by white sharks tagged at the Guadalupe Island and off central California is still unknown.
3) Recent data using isotopes to characterize white shark diet across life history phases in the NEP suggested that not all white sharks make the transition to preying on marine mammals as adults and consequently those individuals may be less likely to be found near pinniped aggregations (Kim et al. 2012). A limitation of using isotopes for such studies, however, is the challenge of differentiating between signals associated with movement and diet.
4) Based on catch, attack and stranding data, some white sharks do not appear to undergo an annual offshore migration (Ainley et al. 1985, Klimley 1985). It is possible that the sharks that remain near the coast represent a different component of the population than those tagged at the aggregation sites. Very few satellite-tagged white sharks have remained along the coast.
5) The high level of haplotypic diversity in the NEP white shark population suggests that the population is larger than indicated by the photo-based mark-recapture study. The alternate explanation for the high haplotypic diversity, that there was a dramatic (>90%) decline in the population over the last few generations, is not supported by the model results (Section 2.3, 4.1.4 and Appendix B).

Quantifying bias in female abundance estimates using SEDM:
The BRT used SEDM to evaluate the potential bias in abundance estimates for mature female white sharks in the NEP. The BRT allocated plausibility points across four categories of possible bias in the estimates as follows:

- There is no bias in the mark-recapture estimates (sex-ratio corrected abundance estimates should be multiplied by 1). This category indicates that essentially all white sharks in the NEP are available for sampling at the two study sites.
- There are 20% more adult female white sharks in the NEP than the mark-recapture estimates indicate (abundance estimates should be multiplied by 1.2). This category indicates that just a few animals were not sampled at the aggregation sites and the current studies estimate the majority of adult female white sharks.
- There are 2 times more adult female white sharks in the NEP than the mark-recapture estimates indicate (abundance estimates should be multiplied by 2). This indicates that white sharks likely occur at other sites or areas in the NEP in addition to the two aggregation sites and that a substantial portion of the actual population has not been sampled because of their location or the timing and/or design of the sampling at the aggregation sites.
- There are 10 times more adult female white sharks in the NEP than the mark-recapture estimates indicate (abundance estimates should be multiplied by 10). This category indicates that a large number of sharks have been missed by the current sampling studies and is consistent with the high level of population abundance suggested by the high haplotype diversity found in the sampled population (see Section 2.3, 4.1.4 and Appendix B).

Based on the results of the SEDM evaluation, the BRT concluded that abundance of female white sharks in the NEP is most likely at least 2 times higher and possibly much larger (up to 10 times larger) than that estimated by studies at the two aggregation sites. There were several considerations that influenced the BRT's evaluation and conclusion. First, the BRT felt that there are likely areas where white sharks are found, but they have not been sampled (e.g. Año Nuevo Island and possibly others such as the Channel Islands or unsurveyed islands off Mexico). Second, the BRT felt it was plausible that some females never visit the aggregation sites. Finally, the BRT felt that the high level of haplotype diversity indicated the population abundance was much higher than suggested from the estimates at the aggregation sites. A summary of the SEDM results for the assessment of abundance bias are in Table 4.8.

Table 4.8 SEDM voting results regarding the bias in adult female abundance estimates based on the mark-recapture studies. The BRT used SEDM to determine the plausibility of different levels of abundance bias; 1) the actual abundance is likely to be the same as estimated, 2) 20% greater than estimated, 3) two times greater than estimated, or 4) 10 times greater than estimated (values: 1, 1.2, 2, and 10). Shown below are the average number of plausibility points and standard deviation across all BRT members for each value.

	Adult female abundance bias categories			
	1.0	1.2	2.0	10.0
Avg	5	19	50	26
Stdev	9	9	18	11

4.4.3.3. Estimate of weighted adult female white shark abundance scenarios

Both the sex ratio and abundance bias assessments described in the preceding Sections (4.4.3.1 and 4.4.3.2) were used by the BRT to develop abundance estimates for adult female white shark in the NEP and the associated uncertainty for each abundance estimate, as described below. These abundance values were used to formulate abundance scenarios for use in subsequent population modeling designed to assess population risk due to fishery bycatch impacts.

The three SEDM assessments described in Sections 4.4.3.1 and 4.4.3.2 resulted in 48 combinations of bias correction (4 sex ratio adjustments for central California* 3 sex ratio adjustment at Guadalupe Island* 4 adjustments based on the bias in abundance). Adjusting the estimated number of mature females at Guadalupe Island and central California for these bias combinations yielded 48 abundance estimates in addition to the direct empirical estimate of 47 adult females (Section 4.4.2.3). For example, one of the 48 abundance estimates with a sex ratio bias of 0.8 at Guadalupe Island and 0.6 at central California and an abundance bias of 2 has an associated abundance estimate of ((27 females/0.8 sex ratio) + (20 females/0.6 sex ratio))*2=134 adult females. The same approach was taken to compute each of the 48 abundance estimates using the various combinations of bias assessed in the BRTs SEDM. The relative plausibility of each of the 48 possible abundance estimates was calculated by multiplying the plausibility points from each of the three bias estimates. The weights were rescaled so that all 48 weights summed to 1.

For further modeling analyses, the 48 alternative abundance values were binned into the following four categories: 60 to125, 125 to 200, 200 to 400 and 400 to 1600 (1600 was the maximum abundance calculated using the approach above). In addition to these abundance categories, the estimate of 47 adult females from the mark recapture analysis (see Section 4.4.2.3) was used as a lower population bound in the modeling. The sum of rescaled weights for each abundance estimate within each category as described above yielded a total weight for each abundance category (Table 4.9). This summed weight reflected the BRTs initial belief about which abundance category most likely included the actual adult female abundance level for the NEP. Based on this initial analysis, the BRT concluded that the adult female abundance in the NEP was most likely greater than 200 individuals (67% of plausibility points in the 200-400 and >400 adult female categories), although lower populations were possible (34% of plausibility points in the < 200 adult female categories). The BRT re-evaluated abundance bias after initial modeling was completed as discussed below in Section 4.5.2.3.

Table 4.9 SEDM weights associated with the adult female abundance (N_f) categories after the first SEDM point allocation on abundance bias.

Abundance category	Summed SEDM weight
Nf < 125	0.13
125 ≤ Nf < 200	0.21
200 ≤ Nf < 400	0.41
Nf > 400	0.26

Following the initial SEDM point allocation on the bias in sex ratio and abundance, the BRT conducted a PVA analyses that quantified the impact of fisheries mortality (YOY and juveniles) on population growth rates (Section 4.5). These analyses provided information about the feasibility of the 5 abundance categories and associated bias considered in the BRT's initial SEDM assessment. Based on the results of the PVA analyses, fisheries mortality rates for YOYs under abundance scenarios based on the levels of abundance bias (i.e. 1.0 and 1.2) were estimated to be on the order ~70 and 40%, respectively. Because mortality rates of 20% are considered high for gillnet fisheries (see Appendix I), the BRT felt it was highly unlikely that 6 vessels operating in the Sebastián Vizcaíno Bay area (which are responsible for the large majority of documented YOY catches in the NEP, O. Sosa-Nishizaki, CICESE, pers. comm.) could account for this high a level of fisheries mortality. These vessels operate in only a portion of the white shark nursery habitat in the NEP, and catch rates are highest at the same time of year in both Sebastián Vizcaíno Bay and the southern California Bight indicating the YOY are spread throughout this area. In light of these considerations, the BRT conducted a second SEDM to reallocate plausibility points associated with the abundance bias. Based on this second SEDM (Table 4.10) the BRT shifted its assessment of bias to higher population levels with over 90% of the plausibility points in the two highest bias levels (2 and 10). The BRT felt the lower bias level and associated population levels were simply not realistic based on the associated fisheries mortality rates. These new abundance bias estimates were used to calculate new abundance scenario weights as previously described which were then used in subsequent PVA analyses as described in Section 4.5.

Table 4.10 Results from the second SEDM voting exercise regarding the bias in adult female abundance estimates based on the mark-recapture studies. This SEDM point allocation was taken after the initial PVA analysis.

	Adult female abundance bias categories			
	1	1.2	2	10
Avg	1.3	8.1	40.6	50.0
Stdev	3.5	10.0	21.5	26.2

The results of the BRT's second SEDM on abundance bias changed the relative weights of the 48 abundance scenarios and abundance categories (Table 4.11). Based on this reanalysis, the BRT concluded there was an even greater likelihood (83%) that the population of mature adult female white sharks in the NEP was larger the 200 individuals with only 16% supporting the two lower abundance levels (less than 200 adult females). In addition to the sources of bias related to females not being available for the photo-ID studies at the two study sites and the haplotypic diversity in the NEP population, the BRT concluded from the PVA model results that the lower population levels were not plausible or realistic in light of the estimated levels of fishery related mortality. Although there remains some uncertainty in the overall level of fisheries bycatch, the BRT felt that is was unlikely that 6 vessels operating in a small portion of the nursery habitat could remove more than 20% of the YOY white sharks in a given year.

Table 4.11 SEDM weights associated with the adult female abundance (N_f) categories after the second SEDM point allocation on abundance bias.

Abundance category	Summed SEDM weight
Nf < 125	0.05
125 ≤ Nf < 200	0.11
200 ≤ Nf < 400	0.33
Nf > 400	0.50

4.4.4. Estimate of adult female mortality due to fishing

Based on the threats evaluation, the BRT expressed the greatest concern regarding the threat of fisheries mortality (see Section 4.2 and Table 4.5) and consequently attempted to quantify and evaluate the risk posed by this mortality in the modeling efforts. YOY and juvenile fisheries mortality rates were informed using the bycatch data from the U.S. and Mexico (Section 4.1.1). Adult female mortality, for which there are few data, was informed using the SEDM approach. The BRT considered the plausibility of various levels of mortality associated with the potential for adult female white sharks to be entangled in near-shore fishing gear or caught on the high-seas by IUU fishing vessels. Arguments for each mortality level can be found in Appendix H.

Table 4.12 SEDM voting results regarding the potential number of adult female white sharks killed annually. Shown below are the average number of plausibility points and standard deviation across all BRT members for each value.

	Number of adult females killed annually				
	0	**1**	**2**	**5**	**10**
Avg	9	24	34	25	7
Stdev	10	16	13	16	5

The BRT concluded that the adult mortality was likely around 2 adult females annually with 83% of the plausibility points spread between an adult mortality of 1 and 5 females per year (Table 4.12). These values are higher than those indicated by the available catch data (see Section 4.1.1). This reflects the BRT's concern that more adult white sharks could be entangled in gear or caught on the high seas than are seen or reported. Obtaining direct data on fisheries mortality from remote camps (such as in Mexico), on the high seas and/or with limited fisheries observer coverage is challenging. Also, sharks that may become entangled but then swim away to die later are difficult to observe, especially given that white sharks sink when dead.

4.5. Assessment of Population Abundance and Fisheries Risk Assessment:

General Introduction:
The BRT conducted a population assessment to assess the probability that fisheries mortality could lead to future population decline, especially to levels below certain thresholds within relevant time periods. A summary of the analysis is provided below. A more detailed description is provided in section 4.5.1.

Analysis Summary:
Estimates of potential population productivity (intrinsic rate of increase) are fundamental to modeling how threats might impact population growth; populations with higher potential productivity can sustain higher levels of mortality. Annual rates of population growth are represented by λ, where $\lambda > 1$ means a population is increasing and conversely $\lambda < 1$ means a population is decreasing. λ can be calculated using information on age-specific reproduction and survival (vital rates) if the relative proportion of the population in different age classes is stable. If the vital rate estimates come from a population under optimal conditions for survival and reproduction and resources are not limiting then intrinsic or maximum potential growth rate (λ_{max}) can be calculated (Caughley 1977). Smith et al. (1998) referred to this as rebound potential for sharks.

To calculate λ_{max} for white sharks, the BRT needed estimates for annual reproduction (how many female pups are produced) and survival rates. For reproduction, the BRT used the values in Table 2.1. The estimate of annual pup production is 2 female pups per year (8.9 pups/ 2.2 year cycle * 0.5 assuming 1:1 sex ratio at birth). For survival the BRT used a number of age-class-specific survival rates, assuming the survival rate generally increases through life (i.e., survival of YOY < juvenile < subadults and adults). The values for vital rates and the associated error distributions were used as inputs for statistical models to calculate a mean and error distribution for λ_{max}. It is important to note that this distribution of λ_{max} only includes natural mortality and not anthropogenic mortality.

Given that the BRT considered fisheries mortality to be a moderately important threat to white sharks, it was necessary to incorporate fisheries related mortality into the analyses. This was accomplished in a number of different stages.

- YOY and age-1 white shark mortality data were combined with information on reproduction to calculate what percentage of YOY and age-1 white sharks in the population would be caught under each abundance scenario. For example, for YOY the question was, "Of all the pups produced under the different adult female abundance scenarios, what proportion of these would be killed by fisheries given the current levels of bycatch?" One notable result was that recent estimates of annual YOY bycatch are approximately equal to the total annual pup production that would be expected from 47 adult females (the direct empirical estimate of adult female abundance). In addition, under this scenario, bycatch mortality for age-1 white sharks would exceed a 100% mortality rate. This is obviously an unrealistic result and suggests that adult female abundance must be substantially higher than the photo-ID estimates if the available bycatch estimates are accurate. This lowest-abundance scenario was consequently excluded from further analyses.
- The YOY and age-1 mortality rates calculated through the above analyses (number caught/number produced) were used to discount the potential rates of population growth (λ_{max}) to account for the fisheries mortality.
- The population growth rates were similarly discounted for adult female mortality. Because the BRT did not have actual data on adult catch, the SEDM process was used to define plausible alternatives of adult female bycatch; these included 0, 1, 2, 5, and 10 adult females killed per year. The combination of these adult-mortality scenarios and the

four retained scenarios of adult female abundance (the value of 47 was not included) resulted in 20 models that were evaluated.

- The potential population growth rates discounted to account for the bycatch mortality of adult, YOY and age-1 white sharks (termed λ_{max*}) were used to project the population into the future for each of the 20 potential scenarios. Population projections incorporated stochastic variation in growth rates (e.g., due to variable environmental conditions), density-dependence, and uncertainty in all population growth parameters (see details in Section 4.5.1.1).
- These population projections were used to calculate the probability that the NEP white shark population would decline to various population thresholds over certain time periods as defined below; these thresholds corresponded to four categories of extinction risk defined by the BRT: high, medium, low and very low (see Section 4.5.2.3).

4.5.1. Information inputs for population assessment

The assessment was based on estimates of: 1) maximum potential productivity (i.e., the intrinsic population growth rate, estimated using life history parameters from Table 2.1), 2) current adult female abundance (see Section 4.4.3), and 3) current levels of fisheries bycatch mortality for YOY, juveniles and adults (Section 4.1.1and 4.4.4).

4.5.1.1. Maximum potential population productivity

Given estimates of demographic vital rates (i.e., survival and reproduction) for a population, it is possible to construct a life table, i.e., the schedule of survival and reproduction through life. Conventional methods, such as solving for the characteristic Euler equation or conducting eigenanalysis of a Leslie or Lefkovitch matrix, may then be used to estimate the corresponding annual rate of population growth (λ) for a population with stable age distribution . If the vital rate estimates come from a population under optimal conditions for survival and reproduction, then one may estimate the maximum potential rate of sustained population growth (λ_{max}), which is the rate that may be achieved as long as resources are not limiting. Smith et al. (1998) referred to this as the rebound potential for sharks (i.e., a measure of the ability to recover following a population decline). These parameters are fundamental to estimating how threats might impact population growth.

Vital rate inputs:
Little information about adult survival rate was available. The BRT's analysis of photo-ID data for adults and subadults at GI provided estimates of mean annual apparent survival for females ($0.90 \pm .11$) and for males (0.95 ± 0.06) (Section 4.4.2). However, these estimates have low precision and do not distinguish between true mortality and permanent emigration (i.e., if there is some permanent emigration from the study area, then true survival is higher than these estimates). Uncertainty in adult female survival (S_{adult}) was therefore expressed using a uniform distribution with minimum and maximum values representing the range of rates considered feasible by the BRT. A minimum value of 0.89 was used because lower adult survival rates were compatible with reasonably positive population growth (e.g., $\lambda_{max} > 1.01$) only when unrealistically high juvenile survival rates (e.g., mean $S_{juv} \approx S_{adult}$) were allowed. Also, the photo ID-based estimate of female apparent survival (0.90) is considered to be an approximate

minimum estimate of true adult survival because it reflects some level of emigration, some human-caused mortality, and possibly some density-dependent reduction in survival from the intrinsic rate. A maximum value of 0.96 was used because it coincides approximately with upper adult survival estimates for other long-lived vertebrates such as albatrosses (Dillingham and Fletcher 2011) and some marine and terrestrial turtles (Heppell 1998). No explicit assumptions about white shark longevity were made in the analysis (i.e., models used by the BRT did not include longevity parameters), but higher mean adult survival rates (e.g., $S_{adult} \geq 0.98$ for some cetaceans) would indicate longevity patterns similar to humans, whose lifespans are likely longer than white sharks' (Anderson et al., 2010). This was the rationale for not considering higher adult mean survival rates.

No information was available about YOY or mean juvenile survival rates. It was assumed that survival rates would generally increase through life so that $S_{yoy} < S_{juv} < S_{subadult} = S_{adult}$, where S_{juv}, $S_{subadult}$ and S_{adult} represent the geometric mean of age-specific values within the age class Therefore, S_{juv} and S_{yoy} were calculated as: $S_{juv} = c_{juv}S_{adult}$ and $S_{yoy} = c_{yoy}S_{juv}$, where c_{juv} and c_{yoy} are constants between 0 and 1. The juvenile stage was assumed to last for 5 years, so that animals become subadults (i.e., achieve adult survival rates) upon reaching their 6th birthday. Uncertainty in these parameters was expressed through use of probability distributions representing the full range of what the BRT considered to be plausible values (Table 4.13).

Table 4.13 Average white shark vital rates and distributions for calculating a distribution for the maximum potential rate of population growth λ_{max}

Vital rate	Estimate or distribution	Information source[a]
Age at first reproduction, α	Uniform(12, 15)	1
Litter size	Normal(8.9, σ = 0.75)	2
Sex ratio at birth	1:1, fixed	3
Inter-birth interval	2.2 years, fixed	4
Adult and subadult survival, $S_{subadult} = S_{adult}$	Uniform(0.89, 0.96)	See text
c_{juv}*	Uniform(0.85, 0.95)	
Juvenile survival	$S_{juv} = S_{adult}c_{juv}$	
c_{yoy}*	Uniform(0.5, 0.99)	
YOY survival	$S_{yoy} = S_{juv}c_{yoy}$	

[a] Information sources: (1) based on von Bertalanffy growth curve from Cailliet et al. (1985) given female maturation size between 4.5 and 5 m; (2) from Mollet et al. (2000): mean = 8.9, range = 4 – 14 from 11 individuals; the specified distribution approximates these data if range assumed to represent a 99% CI; (3) Francis 1996; (4) Nasby-Lucas and Domeier (2012).

* S_{juv} and S_{yoy} were calculated as: $S_{juv} = c_{juv}S_{adult}$ and $S_{yoy} = c_{yoy}S_{juv}$, where c_{juv} and c_{yoy} are constants between 0 and 1. Survival rates represent the geometric mean for each age class.

Conventional approach for estimating λ_{max}:

Using a Monte Carlo randomization approach, a distribution for λ_{max} is constructed by randomly drawing values (thousands of times) from the distributions of vital rates (Table 4.13) and numerically solving λ_{max} for each combination of draws using a version of the Euler equation that does not specify a maximum age (ignores senescence) but assumes a constant annual survival and reproductive rate upon reaching maturity (Skalski et al. 2008):

$$0 = \lambda_{max}^{\alpha - 1}(S_{adult} - \lambda_{max}) + l_{\alpha}F \qquad (1)$$

where l_α is the probability of surviving to reproductive age (α) ($l_\alpha = S_{yoy}S_{juv}^{5}S_{subadult}^{\alpha-6}$; assuming that the juvenile stage lasts for 5 years and animals become subadults at 6 years of age), and F is fecundity in terms of the average number of female pups born per female per year (equal to litter size × 1/inter-birth interval × the proportion of pups that are female).

Integration of life history theory to improve vital rate and λ_{max} estimates:
For bird species across a wide range of taxa, body sizes and life histories, Niel and Lebreton (2005) showed that the product of $\log(\lambda_{max})$ and generation time (T) is a demographic invariant, i.e., $(\lambda_{max})T = a \approx 1$. It followed in their study that λ_{max} could be calculated solely from estimates of adult survival (S_{adult}) and age at reproductive maturity (α) (and the estimate of the allometric constant a), by numerically solving the following equation for λ_{max}:

$$\lambda_{max} = \exp\{ a \cdot [\alpha + S_{adult}/(\lambda_{max} - S_{adult})]^{-1} \}. \qquad (2)$$

Analysis by Dillingham et al. (in prep) suggests that for sharks, the product of $\log(\lambda_{max})T$ may be more variable across species such that for a given species k, a_k may be assumed to come from a normal or lognormal distribution with mean a and variance σ_a^2, where a is the expected $\log(\lambda_{max})T$ and σ_a describes inter-species process variation in the a_k. Based on meta-analysis of shark life history estimates published by Cortes (2002), Dillingham et al. (in prep) have obtained preliminary estimates for sharks of $a \approx 1$ and $\sigma_a = 0.4$ (with CV on $\sigma_a = 0.35$). This information can be used as an informative Bayesian prior to refine the vital rates estimated from the conventional methods described above. Put simply, combinations of vital rates that result in $\log(\lambda_{max})T$ being further from a are considered less plausible than those close to a. Implausible combinations of vital rates are rejected from the set of Monte Carlo samples. This shrinks the distribution of λ_{max} from the estimates obtained from equation 1 toward estimates obtained from equation 2, increasing the precision of the distribution for λ_{max} (Figure 4.12). The amount of shrinkage depends on the precision of estimates from equation 1 (the more data-informed the estimate for species k, the less it will shrink toward a theoretical estimate) and on the value of σ_a (more shrinkage when σ_a is small). Based on the shrinkage estimator, the posterior distribution for λ_{max} for white sharks has a mean of 1.050, SD = 0.017, and 95% credible interval (CRI) of 1.021 – 1.090.

Advantages of using estimates from this integrated matrix-allometric approach are twofold. Conceptually, integrating life history theory into the analysis should result in better informed estimates, particularly in cases where important empirical information from the population of interest is missing. Analytically, this approach reduces the sensitivity of estimates to how uncertainty in the species' life history is specified. An example illustrating this is provided in Appendix J.

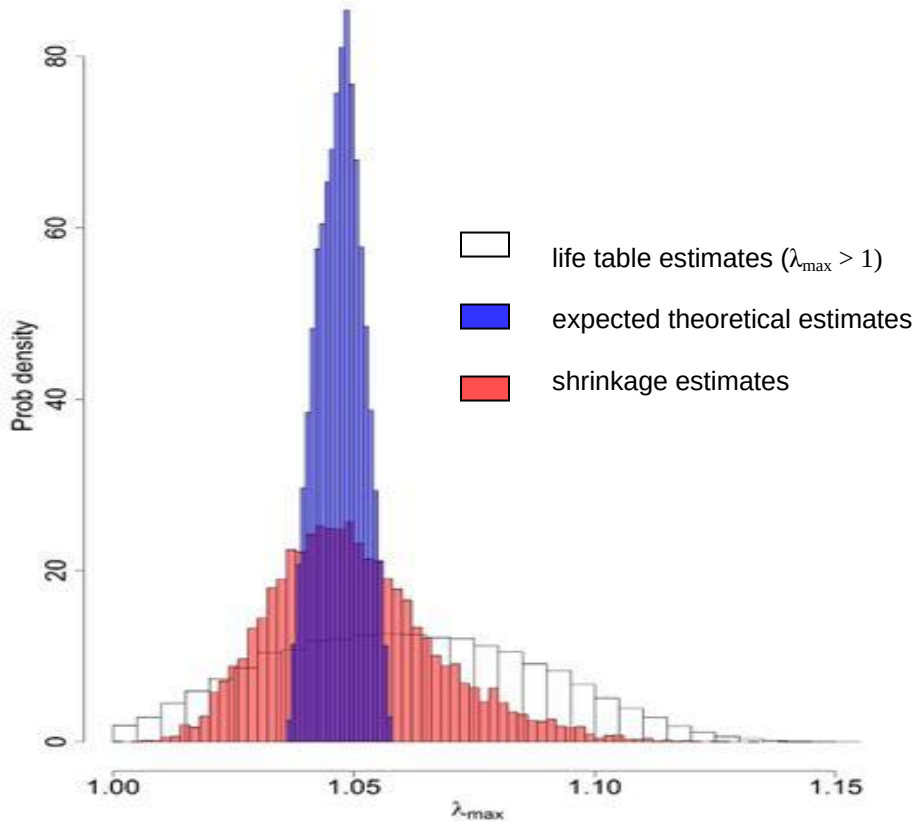

Figure 4.12 Distributions for λ_{max} calculated from vital rate distributions in Table 4.13 using the conventional life table approach (equation 1); expected values using life-history theory (equation 2, with *a* fixed at 1); and Bayesian shrinkage estimates.

4.5.1.2. Female abundance

Adult female abundance:

The BRT conducted the population assessment under five scenarios of plausible adult female abundance levels as discussed in Section 4.4.3.3 of this report. These adult abundance scenarios were: 47 females (lower bound estimate); 60- 125 females; 125 to 200 females; 200 to 400 females; and 400 -1600 females. Associated abundance distributions are shown in
Figure 4.13.

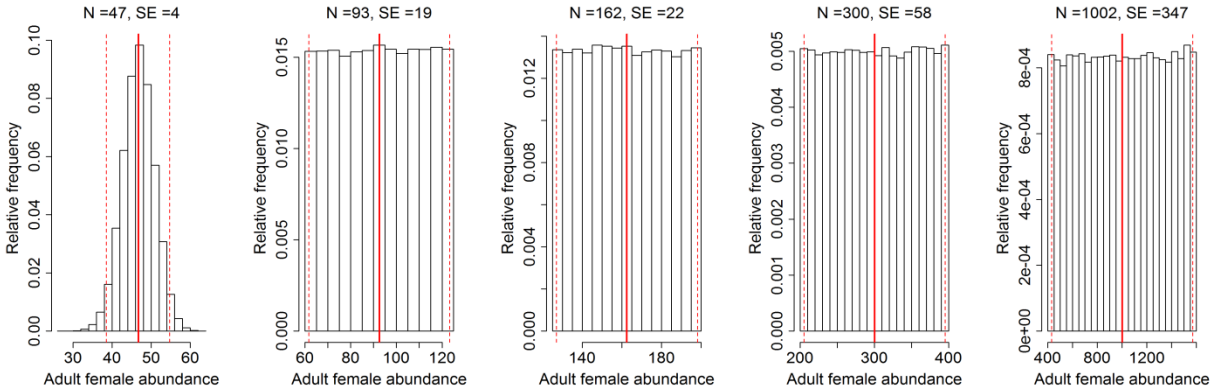

Figure 4.13 Distributions for adult female abundance, under five scenarios. The first (lowest) estimate corresponds to the combined estimates for Guadalupe Island and central California (super-population estimate for 2006 – 2008), based on analysis of photo-ID data and information in those datasets about the proportion of individuals that are mature females (as well as uncertainty in this proportion, described by beta distributions). For the other four scenarios, abundance is uniformly distributed with lower and upper bounds defined as per discussion in Section 4.4.3.3 of this report. Vertical lines represent the distribution mean (solid red lines) and 95% CIs (dotted red lines).

Female abundance across all age classes:

Using vital rate parameters it is possible to estimate the total female abundance (across all life history phases) associated with each of the abundance categories seen in Figure 4.13. Approximations of total female abundance were derived from adult female abundance estimates by assuming the population is stationary (not increasing or decreasing, and with stable age distribution) and constructing basic life tables based on survival estimates used in the Monte Carlo simulations. For example, the number of female YOYs for simulation *i* is estimated from the product of adult abundance, $N_{adult,\,i}$ and the annual reproductive rate (divided by 2, assuming 1:1 sex ratio at birth). The number of 1-year old females, $N_{1,\,i}$ is then $N_{yoy,\,i} * S_{yoy,\,i}$ (where S is the survival rate). The number of 2-year olds is $N_{1,\,i} * S_{juv,\,i}$, and so on. Total abundance (both sexes) is estimated by multiplying the female abundance estimate by 2, assuming 1:1 sex ratio. Because true sex ratio is unknown and because the population is not known to be stationary, this estimation method could be biased, but in the absence of better information it serves as an approximation of the total population size relative to abundance of sampled age classes.

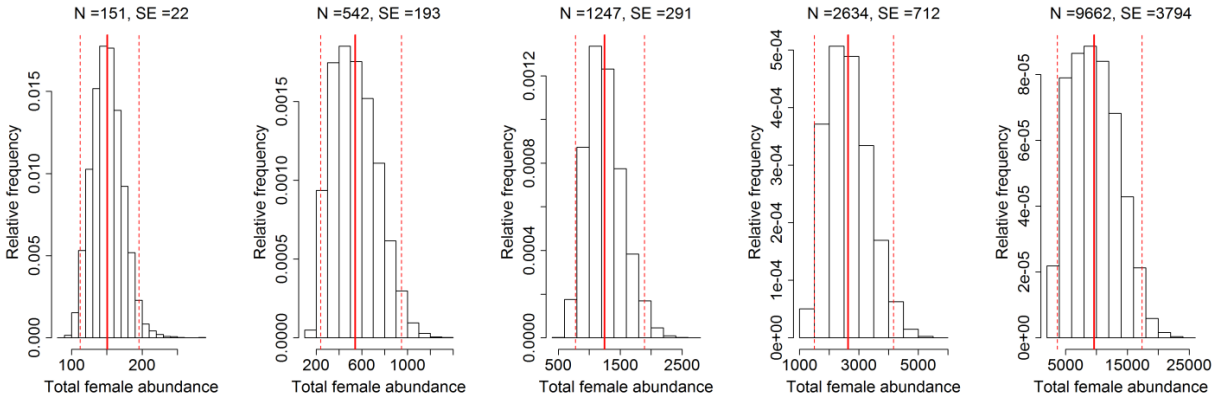

Figure 4.14 Abundance distributions for all females across age classes calculated using the abundance scenarios in Figure 4.13. Vertical lines represent the distribution mean (solid red lines) and 95% CIs (dotted red lines).

Table 4.14 Estimates of the abundance of all white sharks across age classes including only females and both males and females together. Calculations were made using the estimate of 47 mature females and the mean of the 4 abundance scenarios seen in Figure 4.14. The ratio of total females to adult females is also shown for each abundance scenario.

Adult females	Total females	Total:Adult ratio	Total, both sexes (assuming parity)
47	151	3.2	302
93	542	5.8	1084
162	1247	7.7	2494
300	2634	8.8	5268
1002	9662	9.6	19324

The Total:Adult ratio changes because of the impact of bycatch of YOYs on the overall age structure. At low numbers of adult females most YOYs are caught and hence the proportion of immature is low and the Total:Adult ratio is lower than when adult female numbers are higher. For the higher abundance scenarios considered most plausible by the BRT (e.g., > 200 adult females), the total population abundance of NEP white sharks (both sexes) is likely to be > 3000 based on these calculations.

4.5.1.3. Current levels of bycatch mortality

The BRT used the population assessment to examine the influence of fisheries bycatch on the population under each of the scenarios for adult female abundance.

As described in Section 4.1.1 of this report, contemporary estimates of annual bycatch mortality (total individuals killed) for YOY white sharks is 159 and for juveniles between age 1 and 2 is 46. These estimates are based on the following: records of 186 and 6 individuals caught along

the Pacific coast of northern and southern Baja California, respectively (100% mortality rate, 77% of animals smaller than 175 cm or YOY) and about 25 individuals caught in southern California net fisheries (49% mortality rate, 88% smaller than 175 cm) (Section 4.1.1). Fifty percent are assumed to be female based on the bycatch data (Section 4.1.1.3). Note, these are probably minimum bycatch estimates, since not all catches (particularly in Mexico) are likely accounted for in the available reports. For example, catches of juveniles are known to occur in the Gulf of California (Galván-Magaña et al. 2010) and in fishing communities along the Pacific coast of southern Baja California (H. Peckham, Groupo Tortuga, pers. comm.), but annual estimates do not exist for these regions.

The five scenarios of adult female bycatch considered were: 0, 1, 2, 5, or 10 animals killed per year, based on the discussion in Section 4.4.4 and Appendix H.

4.5.2. Assessment of the impact of fisheries catch on the NEP population's extinction risk

The BRT assessed how the different levels of fisheries mortality on YOY, juvenile, and adult female life stages may impact the NEP population, using information on the population's maximum potential growth rates (λ_{max}) and the five different categories of adult female abundance. The impact assessment was conducted in stages, using a stochastic age-structured density-dependent growth model (i.e., a model that factors in a population's size and carrying capacity, along with natural variability in both the population's vital rates and environmental conditions). First, estimates of YOY and age-1 bycatch mortality, combined with information about reproductive rates, were used to calculate anthropogenic YOY and age-1 mortality rates (proportion of animals killed annually) under each scenario of adult female abundance. These YOY/juvenile bycatch mortality rates were used to calculate the maximum potential population growth rate possible given the reduced S_{yoy} and S_1 (survival from age 1 to 2); this rate is termed λ_{max*}. The probability that λ_{max*} would be less than one (corresponding to sustained exponential decline) was also calculated. Second, estimates of adult female bycatch and corresponding adult mortality rates were included and λ_{max*} metrics were re-calculated. Finally, the distributions for λ_{max*} combined with plausible specifications of stochastic variation in λ and uncertainty in density-dependent population growth parameters were used to simulate populations projections. These were used to calculate the probability that the NEP white shark population would decline to various population thresholds based on risk categories decided on by the BRT (Section 4.5.2.3).

4.5.2.1. Impact of bycatch and mortality rates for YOYs and 1-year olds

The bycatch mortality rate for YOYs ($M_{yoy,b}$) may be calculated as $M_{yoy,b} = C_{yoy}/N_{yoy}$, where C_{yoy} is the number of lethal catches and N_{yoy} is the number pups born in a year, which is simply the product of the adult female abundance and fecundity. Similarly, the bycatch mortality rate for 1-year olds is $M_{1,b} = C_1/N_1$, where $N_1 = N_{yoy}S_{yoy}(1 - M_{yoy,b})$.

The distribution for λ_{max*} can be calculated from the distributions for λ_{max}, $M_{yoy,b}$ and $M_{1,b}$. For each Monte Carlo combination of vital rates that was retained (not rejected) by the algorithm for estimating λ_{max} (see section above on estimating maximum potential population productivity), l_α is replaced with $l_\alpha(1-M_{yoy,b})(1 -$

$M_{1,b}$) and the Euler method (equation 1) is used to calculate λ_{max*}. Estimates of YOY catch rate (proportion of YOYs captured annually, including retained and released animals), $M_{yoy,b}$ and $M_{1,b}$ under the different scenarios of adult female abundance are shown in

Figure 4.13. One notable result is that if the total adult female abundance were equal to the photo-ID based estimates (i.e., 47), then present-day estimates of annual YOY bycatch are approximately equal to the total annual pup production (Figure 4.15 upper leftmost panel). Moreover, under this adult female abundance scenario, bycatch mortality for 1-year olds would be 100% (Figure 4.15, bottom left-most panel). This is an unrealistic result and suggests that adult female abundance must be substantially higher than the photo-ID estimates in order to support the level of pup production indicated by catch data. This adult female abundance scenario is therefore not considered further in the BRT's analyses and is not presented in subsequent result summaries. Under the remaining four adult female abundance scenarios, the mean estimated YOY bycatch rates ranged from 5 to 48% (Figure 4.15, top panel) and the mean estimated YOY mortality rates ranged from 5 to 45% (Figure 4.15, middle panel). The mean mortality rates for 1-year olds ranged from 2 to 41% (Figure 4.15, bottom panel).

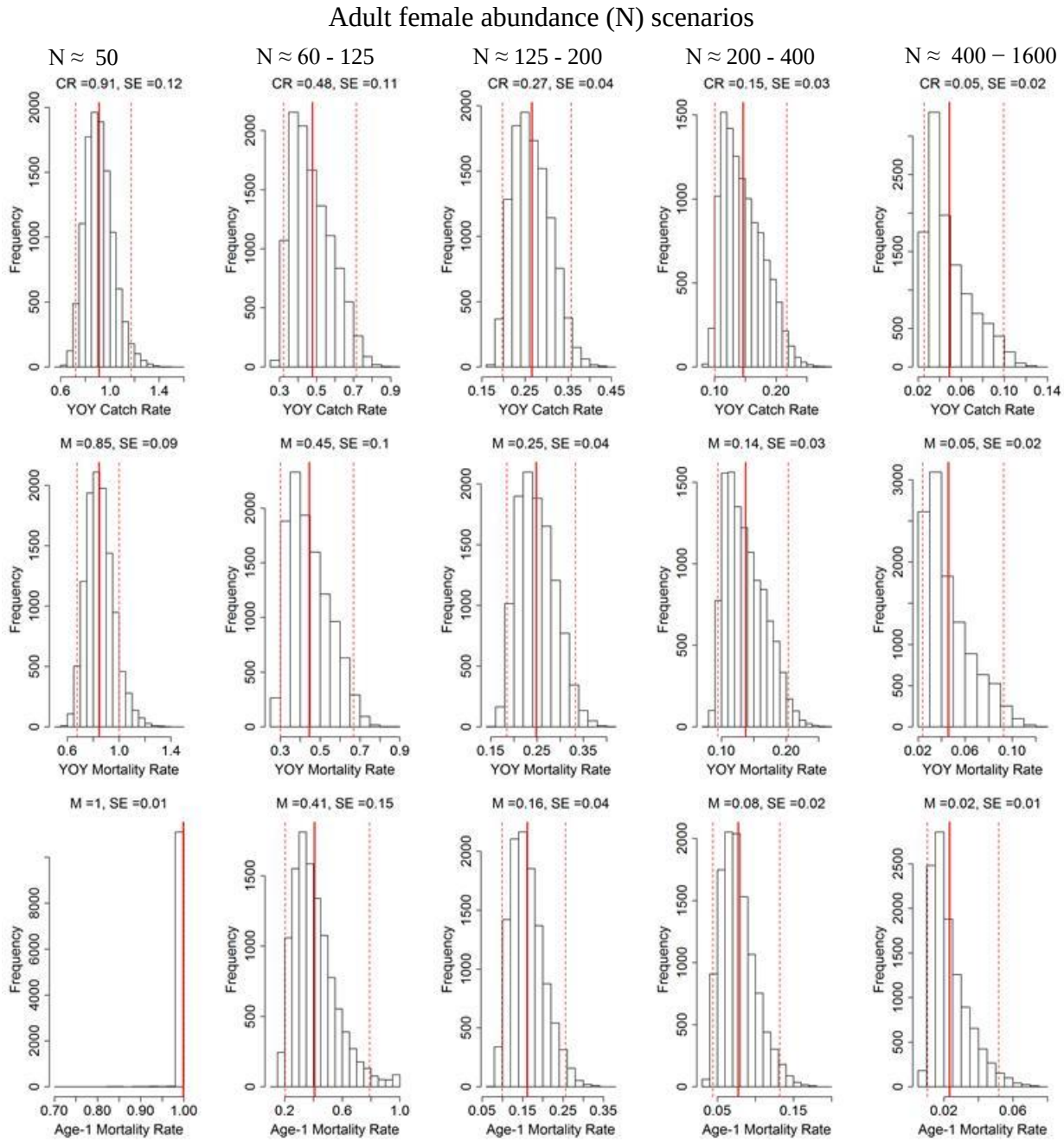

Figure 4.15 Distributions for estimated YOY catch rates (the proportion of the YOY population captured each year; top panel), the corresponding YOY mortality rates (the proportion of the YOY population that are captured and killed each year; middle panel), and the mortality of 1-year olds (the proportion of the juvenile (1-year old) population that is captured and killed each year; bottom panel) under 5 scenarios for the number of adult females (N) in the population (noted across the top). The mean catch rates (CR) and mortality rates (M) and standard error (SE) for each distribution are noted at the top of each plot. Vertical red lines represent the distribution mean (solid lines) and 95% CIs (dotted lines).

The BRT assessed the impact of YOY and juvenile bycatch estimates on the potential population growth rate (Figure 4.16). For the lowest plausible abundance scenario (N ≈ 60 – 125 adult females), the mean estimate for λ_{max*} is 1.00 and the probability of negative population growth

(P) is 0.52. For the three higher abundance scenarios, the probability of YOY and juvenile bycatch mortality forcing sustained deterministic population decline is small (P = 0.043, if N ≈ 125 – 200 adult females) or negligible (P ≈ 0 for scenarios of N > 200 adult females).

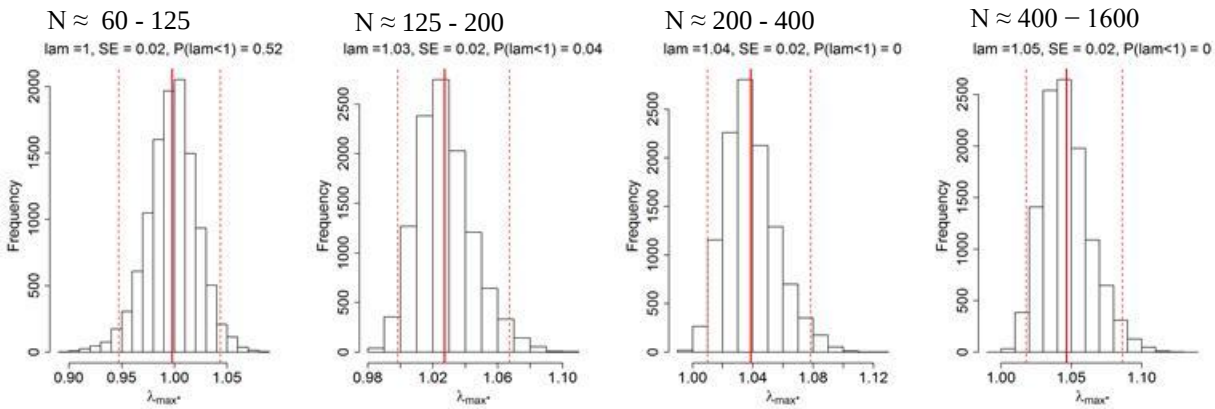

Figure 4.16 Distributions for λ_{max*}, the maximum population growth rate that could be achieved, under four adult female abundance scenarios (excluding the lowest adult female abundance scenario), given reduced survival of YOYs and 1-year olds due to current bycatch levels in southern California and Mexico. Mean estimates of λ_{max*}, the standard error (SE), and the probability of negative population growth (P) are given above each plot. Vertical red lines represent the distribution mean (solid lines) and 95% CIs (dotted lines).

4.5.2.2. Combined impacts of YOY, juvenile and adult bycatch mortality rates

The bycatch mortality rate for adult females, $M_{adult,b}$, may be calculated as C_{adult}/N_{adult}, where C_{adult} is the number of lethal catches of adult females and N_{adult} is the number of adult females under the different abundance scenarios. Following methods of the previous section, l_α for each retained Monte Carlo sample is replaced with $l_\alpha(1-M_{yoy,b})(1 - M_{1,b})$, S_{adult} is replaced by $S_{adult}(1 - M_{adult,b})$, and the Euler method (equation 1) is used to calculate the new λ_{max*}.
The BRT then analyzed the additional impact (in addition to the estimated YOY and juvenile mortality rates discussed above) of capturing and killing 1, 2, 5, or 10 adult females per year on the population's maximum growth rate (λ_{max*}) (Figure 4.17). For the lowest plausible abundance scenario (N ≈ 60 – 125 adult females), all considered levels of adult female mortality resulted in a high probability of causing population decline (P ranged from 0.61 to 0.95; Figure 4.17, top row). For the low-intermediate abundance scenario (N ≈ 125 – 200 adult females; Figure 4.17, second row), catching one or two adult females annually resulted in a small but non-trivial probability of population decline (P = 0.06 or 0.08, respectively), but catching 5 or 10 adult females per year results in a substantial probability of population decline (P = 0.19 or 0.43, respectively). For the two highest abundance scenarios (N ≈ 200 – 400 adult females, or 400 – 1600 adult females; Figure 4.17, bottom two rows), the probability of population decline was generally low for all considered levels of adult female mortality (P ≈ 0 to P = 0.05).

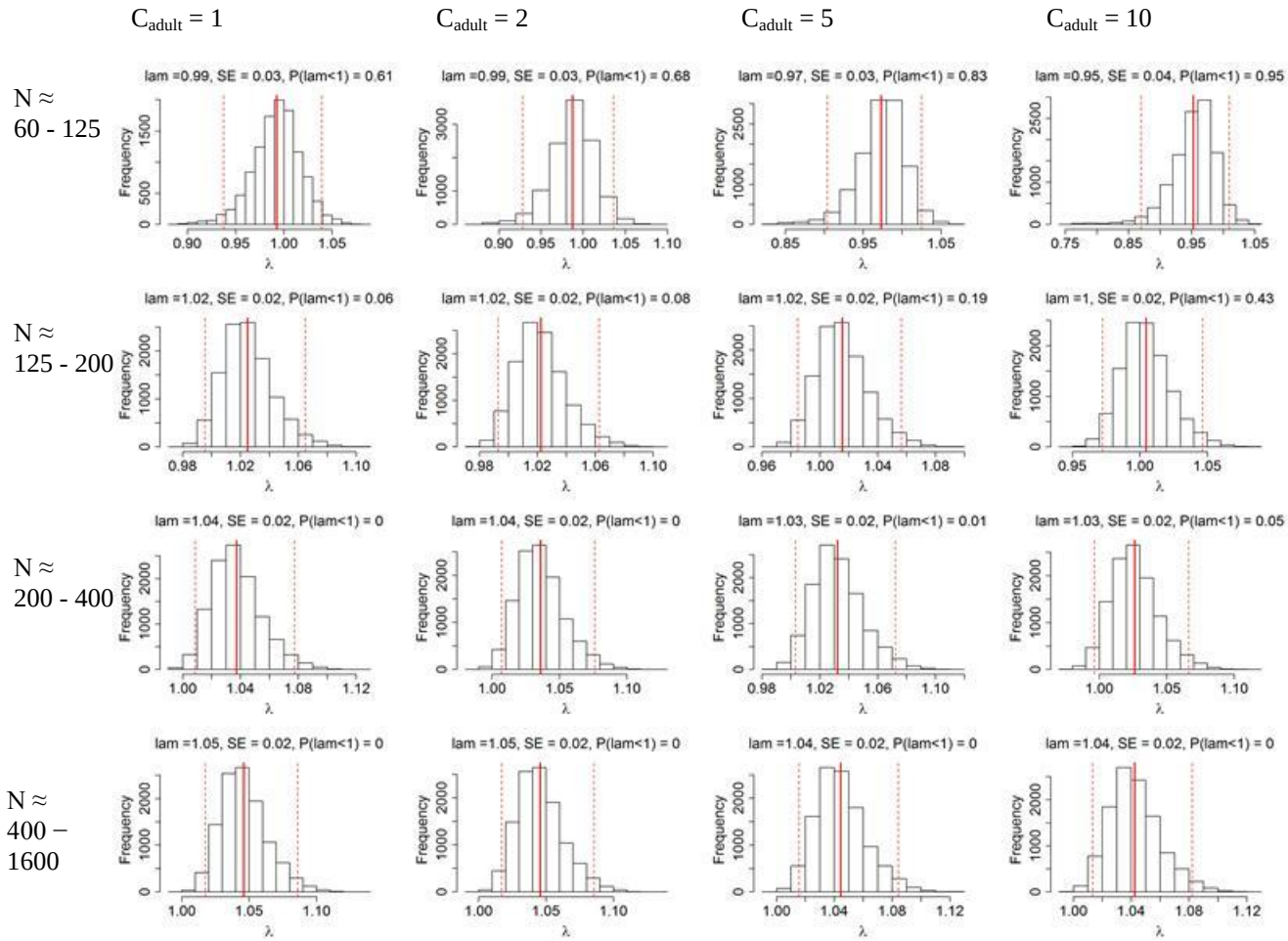

Figure 4.17 Distributions for λ_{max*}, the maximum population growth rate that could be achieved, under four adult female abundance (N) scenarios (noted along the left margin) and four scenarios of adult female catch and mortality ($C_{adult.}$ noted along the top margin) of 1, 2, 5, or 10 adult females per year. Adult female catch and mortality was considered in addition to the estimated YOY and juvenile mortality rates (see Figure 4.15) Mean estimates of λ_{max*}, the standard error (SE), and the probability of negative population growth (P) are given above each plot. Vertical red lines represent the distribution mean(solid lines) and 95% CIs (dotted lines).

4.5.2.3. Probability of declining below defined population thresholds

For each Monte Carlo sample *i* of a particular abundance and bycatch scenario, the NEP white shark population's trajectory was projected over specified time frames. The model projected population trajectories under different scenarios of present-day adult female abundance and bycatch mortality levels, as described in the sections above. The BRT then considered these modeled population projections to evaluate the threat of fisheries bycatch and its contribution to the population's risk of extinction.

Population trajectories were projected according to the following stochastic density-dependent model:

$$N_{t+1} = N_t\lambda_t$$

$$\lambda_t = \{\lambda_{max} \exp[-\log(\lambda_{max})(N_t/K)^{\theta}] - \Delta\lambda_{byc}\} \exp(\varepsilon_t)$$

where N_t is the adult female abundance in year t, K (carrying capacity) and θ are theta-logistic growth model parameters, $\Delta\lambda_{byc}$ is the difference between λ_{max} and λ_{max*}, and $\varepsilon_t \sim$ Normal(0, σ = 0.10) describes annual

stochastic variation in λ_t. The choice for σ was based on several empirical examples (see next paragraph). Parameter K is unknown and was therefore calculated for each population trajectory *i* as N_0/κ, where N_0 is the abundance estimate at time zero (from distributions in
Figure 4.13) and κ ~ Uniform (0.1, 0.9), implying that present-day abundance could be anywhere between 10% and 90% of what the environment can currently support. Parameter θ determines the shape of the density-dependent response and is also unknown. When θ = 1, the model describes simple logistic growth. For large values of θ, the population is capable of high growth rates (i.e., close to λ_{max}) until abundance gets fairly close to its carrying capacity. For long-lived late-maturing species, it is generally believed that θ > 1 (e.g., Fowler 1988). Uncertainty was therefore expressed as θ ~ Uniform (1, 10). The upper bound corresponds to net population production being maximized at around N = 0.8K (rather than at 0.5K for the simple logistic model).

The choice for σ was based on data from Guadalupe Island white sharks and empirical examples from other species. Based on the photo-ID data at Guadalupe Island, variation in $\hat{N}_{t+1} / \hat{N}_t$ for adult female white sharks from 2001 – 2011 (see Section 4.4.2.3) corresponded to σ ≈ 0.15, but this should overestimate the true σ because variance in the annual abundance estimates reflects sampling error and an unknown level of annual emigration and immigration from the study area. Estimates of variation in $\hat{\lambda}_t$ for several vertebrate populations with similar generation times (e.g., 15 – 30 years) are also around 0.1 – 0.2 (Regan et al. 2009), but these also probably overestimate variation for the adult class due to sampling error in the underlying abundance estimates and because those $\hat{\lambda}_t$ are based on population abundance estimates across all age classes (e.g., variation in abundance for gray whales age ≥ 1) or just the youngest age classes (e.g., variation in pinniped pup counts, sea turtle nest counts). Variation in these groups is relatively volatile (due to highly variable reproductive or juvenile survival rates) compared to what would be expected for the adult age class.

Defining risk categories:
The BRT chose four categories for overall levels of risk of extinction: high, medium, low and very low. The specific criteria were based on current estimated abundance, population trajectories over specific time horizons and the probability that the population is declining. For population trajectories, the BRT used different time horizons that were based on the white shark generation time (~20 years), 3 and 5 generation times were used for different levels of risk and 2 generation times (40 years) used to delineate "foreseeable future". For abundance levels in the risk categories, the BRT used two numbers, "near extinction" and "dangerously small". These reference points are defined below.

The two higher risk categories have elements that correspond to risks faced by a declining population and risks faced by small populations. We chose to have both these elements in our higher risk definitions. This is similar to a memorandum developed by the FWS in response to a request by the Court to provide additional explanation for the legal basis of the FWS's determination that polar bears be listed as threatened (TH) rather than endangered (EN). The memorandum reviews previous decisions and lists 4 categories that cover species listed as endangered. The two that relate to this white shark petition are:

- Species formerly more widespread that have been reduced to such critically low numbers or restricted ranges that they are at a high risk of extinction due to threats that would not otherwise imperil the species.
- Species with still relatively widespread distribution that have nevertheless suffered ongoing major reductions in its numbers, range, or both, as a result of factors that have not been abated.

We considered what timeframes and abundance levels were appropriate to the white shark case below.

"Foreseeable future" is considered to be the timeframe over which predictions about the future in making determinations about the future conservation status of the species can be reasonably relied upon (M-37021 2009 DOI memorandum). In quantifying the foreseeable future the BRT should consider the life history, habitat characteristics, availability of data, kinds of threats, ability to predict threats, and the reliability of models used to forecast threats. There are several factors to consider for white sharks when seeking timeframes meaningful for risk assessment: 1) the only threat considered by the Team to have moderate risk is the threat category "Overutilization: commercial, recreational or education", 2) the primary age category killed in fisheries are young of the year, and 3) white sharks are long-lived. These factors suggest that 'foreseeable' future should be based on generation time since effects on yearlings will manifest into population risk on that timeframe. The lifespan of white sharks (30 years) was also used to consider risk since the 'species' would persist for at least one lifespan even if there were no recruitment. There is considerable uncertainty about how variable population growth is for white sharks given the lack of information on basic life history parameters.

The BRT did not find that consideration of the threat of overutilization led to adopting any particular number of generations. Instead the Team considered other precedents for looking at risk timeframes. The IUCN uses 3 generations for its category of Endangered and 5 generations for its category of Vulnerable. These timeframes seemed reasonable points to assess risk for white sharks. It also seemed reasonable to consider a period of 2 generations as the timeframe for foreseeable future given that the threat of overutilization is best considered on a generational timescale and using more than one generation seemed prudent given the uncertainties in population size and variability.

"Near-extinction" is defined by Regan et al. (2009) as *a population that has declined to a size at which the probability of extinction in the near future (50 years or the lifespan of the species whichever is the longer) is extremely high.* The IUCN (2001) criteria use 50 mature individuals as a criterion (D1) for their category of Critically Endangered. Following discussion of the potential for depensatory processes at various population sizes in the BRT felt that the value of 50 mature individuals used by IUCN for a Critically Endangered species seemed appropriate for the near-extinction threshold.

"Dangerously small" population size *is a population that is sufficiently small that density depensation may occur and that variability in population size resulting from fluctuations in the environment could result in reaching near-extinction.* The IUCN criterion D1 uses 250 mature individuals for their category of Endangered. A population with less than 250 mature individuals

would also be considered at high risk of extinction and this level seems to fit numerous cases also listed under the ESA. Therefore we use 250 mature adults as our threshold for a dangerously small population.

Given that population models for white sharks were constructed in terms of females only, threshold were defined in terms of mature females rather than mature individuals, assuming a sex ratio of approximately 50:50. Further details on the term 'near extinction' can be found in Regan et al. (2009) together with more rationale for the use of 50 mature individuals.

Definitions of risk categories

To determine high risk we considered two categories:

1) the declining population category (where the population declines to near-extinction within a certain timeframe),
2) the small population category (where the population is dangerously small currently).

High Risk--a population is at high risk if it meets either condition:

A. the population has a 5% chance of falling below 50 mature individuals (25 mature females) in 60 years (~3 generations)
B. the current population is less than 250 mature individuals (125 mature females).

We related Medium risk to our High risk categories by adding 40 years (~2 generations) as our definition of ‘foreseeable future’. The medium risk category is based on the idea of becoming a high risk population within ~2 generations.

Medium Risk--a population is at medium risk if it meets either condition:

A. the population has a 5% chance of falling below 50 mature individuals (25 mature females) in 100 years (~5 generations)
B. the population has a 5% chance of falling below 250 mature individuals (125 mature females) in 40 years (~2 generation times).

Low and very low risk categories do not have the two categories like the high or medium risk categories.

Low risk-- The population does not meet the criteria for medium or high risk but the population has at least a 10% probability of showing a decline within 100 years (~5 generations).

Very low risk-- The population does not meet any of the above criteria for high, medium, or low risk. The population has a high probability of being stable or increasing.

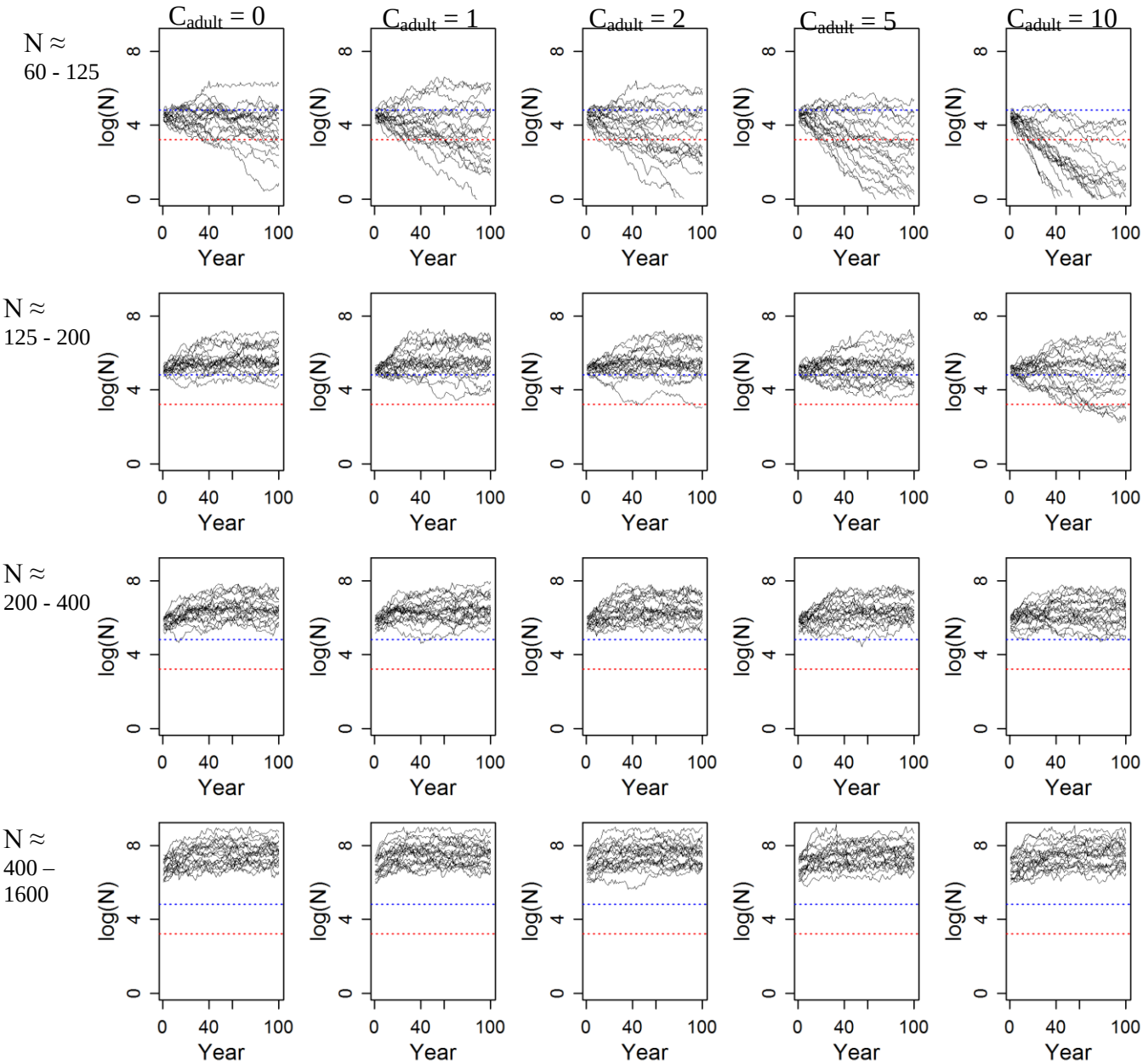

Figure 4.18 Stochastic projection example (20 trajectories per panel) of adult female abundance (on log scale) out to 60 and 100 years from present, under four scenarios of present-day adult female abundance (indicated in left margin) and five scenarios of adult female bycatch mortality (C_{adult};top margin). Horizontal dotted lines indicate the thresholds of: 25 ("near-extinction", red) or 125 ("dangerously small", blue) adult females.

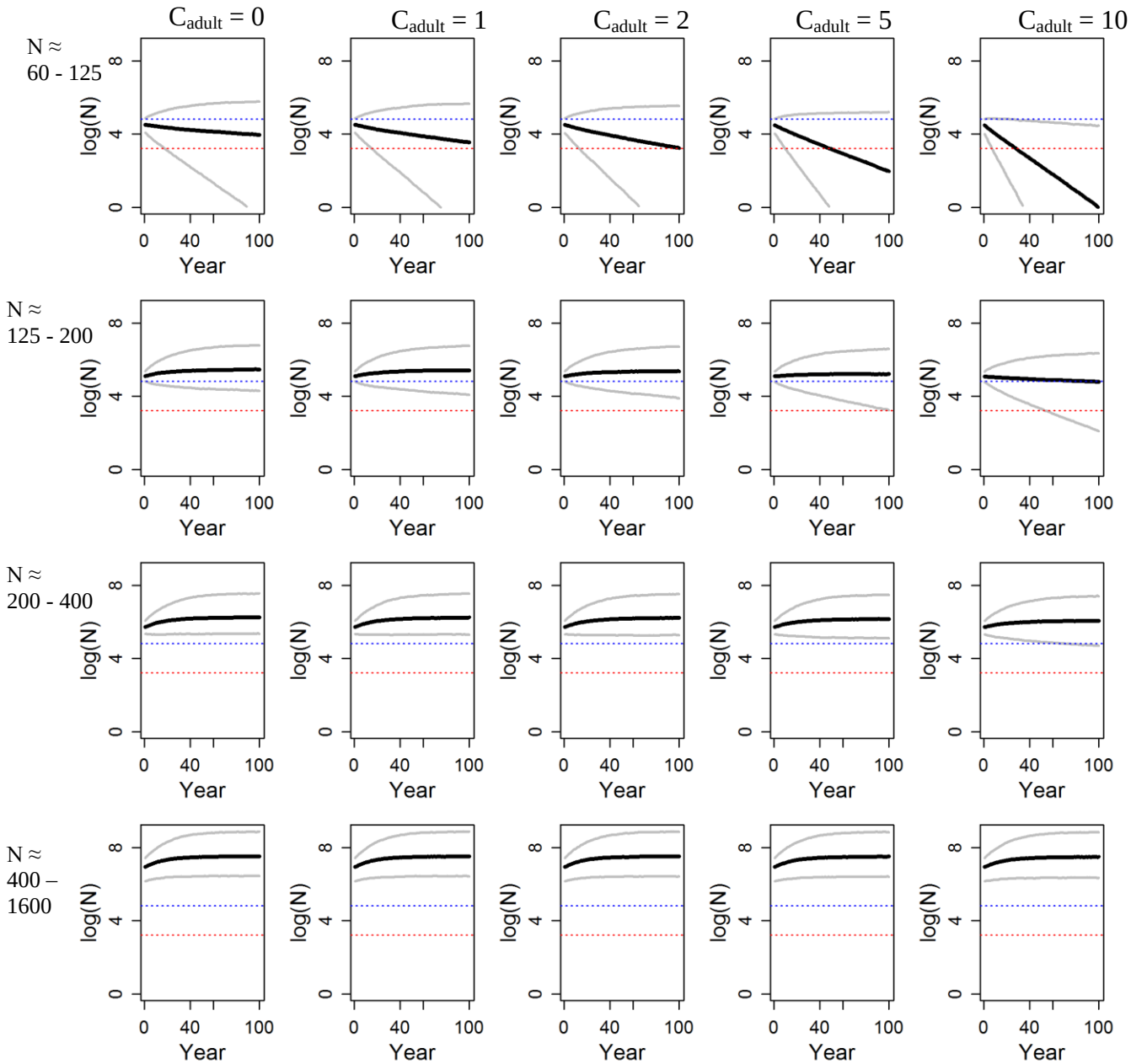

Figure 4.19 Stochastic projection summaries of adult female abundance (on log scale) out to 60 and 100 years from present, under four scenarios of present-day adult abundance (indicated in left margin) and five scenarios of adult female bycatch mortality (C_{adult}; top margin). Horizontal dotted lines indicate the thresholds of: 25 ("near-extinction", red) or 125 ("dangerously small", blue) adult females. The solid black line represents the median projected abundance through time. Gray lines depict the 5th and 95th percentiles of the abundance projections corresponding to the 90% confidence intervals.

Table 4.15 Probability (P) of adult female abundance declining to "near-extinction" (≤ 25 adult females) within 60 years (N_{60}; high risk) or 100 years (N_{100}; medium risk), under 4 scenarios of present-day adult female abundance (N_0) and 5 scenarios of adult female bycatch mortality (C_{adult}). Probabilities are calculated as the proportion of projections out of ≈11,000 that decline to ≤ 25 adult females within 60 or 100 years. Colors correspond to high (red), medium (orange), low (yellow), or very low (green) risk levels (see footnotes and text for threshold criteria).

	$C_{adult} = 0$	$C_{adult} = 1$	$C_{adult} = 2$	$C_{adult} = 5$	$C_{adult} = 10$
Time frame: 60 years					
$N_0 \approx 60 – 125$	0.263	0.332	0.393	0.556	0.760
$N_0 \approx 125 – 200$	0.002	0.003	0.005	0.017	0.067
$N_0 \approx 200 – 400$	0.000	0.000	0.000	0.000	0.001
$N_0 \approx 400 – 1600$	0.000	0.000	0.000	0.000	0.000
Time frame: 100 years					
$N_0 \approx 60 – 125$	0.360	0.432	0.508	0.677	0.854
$N_0 \approx 125 – 200$	0.006	0.013	0.017	0.046	0.152
$N_0 \approx 200 – 400$	0.000	0.000	0.000	0.001	0.004
$N_0 \approx 400 – 1600$	0.000	0.000	0.000	0.000	0.000

Red (high risk): $P(N_{60} < 25) > 0.05$
Orange (medium risk): $P(N_{100} < 25 \text{ or } N_{40} < 125) > 0.05$
Yellow (low risk): Above criteria not met but $P(N_{100} < N_0) > 0.1$
Green (very low risk): Above criteria not met

Table 4.16 Probability of adult female abundance declining to <125 adult females within 40 years (medium risk), under 4 scenarios of present-day adult female abundance (N_{40}) and 5 scenarios of adult female bycatch mortality (C_{adult}). Probabilities are calculated as the proportion of projections out of ≈11,000 that decline to < 125 adult females within 40 years. Colors correspond to high (red), medium (orange), low (yellow), or very low (green) risk levels (see footnotes and text for threshold criteria).

	$C_{adult} = 0$	$C_{adult} = 1$	$C_{adult} = 2$	$C_{adult} = 5$	$C_{adult} = 10$
Time frame: 40 years					
$N_0 \approx 60 – 125$	0.762	0.806	0.832	0.901	0.959
$N_0 \approx 125 – 200$	0.141	0.163	0.19	0.272	0.436
$N_0 \approx 200 – 400$	0.007	0.01	0.01	0.016	0.035
$N_0 \approx 400 – 1600$	0.000	0.000	0.000	0.000	0.000

Red (high risk): $P(N_{60} < 25) > 0.05$
Orange(medium risk): $P(N_{100} < 25 \text{ or } N_{40} < 125) > 0.05$
Yellow (low risk): Above criteria not met but $P(N_{100} < N_0) > 0.1$
Green (very low risk): Above criteria not met

Table 4.17 Probability of adult female abundance declining ($N_{t=100} < N_{t=0}$) over the course of 100 years (Low risk), under 4 scenarios of present-day adult female abundance (N_0) and 5 scenarios of adult female bycatch mortality (C_{adult}). Probabilities are calculated as the proportion of projections out of ≈11,000 that decline over 100 years. Colors correspond to high (red), medium (orange), low (yellow), or very low (green) risk levels (see footnotes and text for threshold criteria).

	$C_{adult} = 0$	$C_{adult} = 1$	$C_{adult} = 2$	$C_{adult} = 5$	$C_{adult} = 10$
Time frame: 100 years					
$N_0 \approx 60 – 125$	0.675	0.737	0.782	0.887	0.964
$N_0 \approx 125 – 200$	0.244	0.287	0.316	0.423	0.616
$N_0 \approx 200 – 400$	0.132	0.139	0.145	0.189	0.264
$N_0 \approx 400 – 1600$	0.078	0.079	0.080	0.088	0.103

Red (high risk): $P(N_{60} < 25) > 0.05$
Orange (medium risk): $P(N_{100} < 25 \text{ or } N_{40} < 125) > 0.05$
Yellow (low risk): Above criteria not met but $P(N_{100} < N_0) > 0.1$
Green (very low risk): Above criteria not met

The proportions of population trajectories declining below 25 ("near-extinction") and 125 ("dangerously small") adult females after 60 and 100 years were calculated (Table 4., Table 4.,

Table 4.17) for all adult female abundance and mortality scenarios. Individual stochastic population projections, in relation to these thresholds, are shown in Figure 4.18 and summarized in Figure 4.19.

For the scenario in which present-day adult female abundance is estimated to be between 60 – 125, the population would be considered at high risk under all adult female bycatch scenarios, both by definition (because the current abundance is below 125 adult females) and because the probability that the population will decline to less than 25 adult females within 60 years is greater than 0.05 (Table 4.).

For the scenario in which present-day adult female abundance is estimated to be between 125 – 200, the population would be considered at medium risk under the four lowest adult bycatch scenarios, because the probability of declining to less than 125 adult females within 40 years exceeds 0.05 (Table 4.). However, the probability of declining to less than 25 adult females within 100 years did not exceed 0.05 (Table 4.). The population would be considered at high risk under the highest adult female bycatch mortality (C_{adult} = 10), because the probability of declining to less than 25 adult females within 60 years exceeds 0.05 (Table 4.).

For the scenario in which present-day adult female abundance is estimated to be between 200 – 400, the population would be considered at low risk under all of the adult female bycatch scenarios considered (Table 4., Table 4.,

Table 4.17). The probability of declining to less than 25 adult females within 60 or 100 years, or to less than 125 adult females within 40 years, would be very low (less than 0.05). However there is some non-trivial probability (> 0.10) that the population would experience some level of decline within the next 100 years.

Finally, for the highest abundance scenario in which present-day adult female abundance is estimated to be greater than 400 adult females, the population would be considered at low or very low risk under all of the adult female bycatch scenarios (Table 4., Table 4.,

Table 4.17). Only for the highest level of adult female bycatch considered (C_{adult} = 10) would the population be expected to decline at all with any substantial probability (P > 0.1) (

Table 4.17). The probability of declining to less than 25 adult females within 60 or 100 years, or to less than 125 adult females within 40 years, would be essentially zero. Overall the PVA results indicated that at population levels of 200 and greater, there is low to very low risk associated with fisheries mortality on adult females and YOY and juvenile white sharks.

In addition to assessing extinction risk under different scenarios of female abundance and bycatch, the model approach estimated catch and mortality rates of YOY and juveniles. Catch and mortality rates informed the plausibility of the different abundances. For the lowest two abundance scenarios considered analytically plausible (N ≈ 60 – 125 and N ≈ 125 – 200), the corresponding annual catch rates for YOYs (means = 0.49 and 0.27, respectively) are extremely and perhaps unrealistically high. Population removal rates for sharks, even for selective gears (such as pelagic longlining), are probably less than 0.20 (Worm et al. 2013). For populations of marine mammals and sea turtles known or suspected to be declining because of high bycatch mortality, the mortality rate on affected age classes from gill net bycatch is typically less than 0.10 (see Appendix I, Table I.1). Vaquita porpoises – which are Critically Endangered (IUCN Red List), believed to be precipitously declining, and subjected to extremely intensive gill net exposure throughout their range– have an estimated bycatch mortality rate near 0.20 based on the model used in Jaramillo-Legorreta et al. (2007). It would be astonishing if gill nets had the efficiency to capture > 0.20 of white shark YOYs each year, especially given that the available bycatch estimates come from small fishing fleets operating throughout a limited portion of known or suspected YOY nursery habitat (i.e., most of the recorded bycatch comes from just 6 boats operating in Sebastián Vizcaíno Bay, Mexico, O. Sosa-Nishizaki, CICESE, pers. comm.). If even 0.20 of YOYs are captured annually, then existing YOY catch estimates (> 150) suggest YOY abundance on the order of at least 1000. Given suspected underestimates of bycatch and the likelihood that < 0.20 of YOYs are captured each year, the true number of YOYs may be much greater (low thousands). This information challenges the plausibility of the lower abundance-level scenarios (< 200 adult females) considered in this assessment. The BRT concluded (Section 4.4.3.3) that a female white shark abundance greater than 200 was most plausible.

One model consideration that should be discussed is the uncertainty in bycatch estimates. Consideration must be given to how the assessment would be affected by different catch estimates. If true bycatch is higher than the estimates, this would, as noted above, make all but the highest abundance (low-risk) scenario seem dubious. On the other hand, true juvenile bycatch could be lower than what was used for the assessment. The large numbers of catches reported from Sebastián Vizcaíno Bay have not been scientifically verified, and although they are believed by Mexican researchers to be valid (Section 4.1.1.4), there is the possibility that species were misidentified. If true catches are lower than the available estimates, this would decrease the estimates of YOY mortality rates and increase the plausibility of results for the lower abundance (higher-risk) scenarios. Results are conditional on existing bycatch estimates, which likely contain an unquantifiable degree of bias and should be interpreted with caution.

4.6. Risk assessment conclusions and overall assessment of extinction risk for the NEP white shark population

4.6.1. Risk Assessment Conclusions

To evaluate the overall extinction risk of the NEP white shark population, the BRT conducted a final SEDM assessment that considered all available information from the status review including the threats assessment (Section 4.2), direct or indirect indicators of population trends (Section 4.3), information on population abundance including updated mark-recapture analysis, haplotypic diversity and evaluation of factors biasing available population abundance estimates (Section 4.4) and extensive population modeling to assess risks associated with fisheries bycatch

impacts (Section 4.5). Based on this information and uncertainty about the future, the BRT allocated plausibility points among 4 risk categories as defined in section 4.5.2.

High Risk - The population has at least a 5% chance of falling below 50 mature individuals (25 mature females) within 60 years (~3 generations) or the current population has less than 250 adults (125 mature females).

Medium Risk - The population has at least a 5% chance of falling below 50 mature individuals (25 mature females) within 100 years (~5 generations) or below 250 mature individuals (125 mature females) within 40 years (~2 generations).

Low Risk- The population is neither at high or medium risk, but has at least a 10% chance of decline within the next 100 years (~5 generations).

Very low Risk - The population is not at high, medium or low risk. The population has a high probability of being stable or increasing

Based on the results of the SEDM (Table 4.17) the BRT concluded that the NEP white shark population is likely to be at a very low to low risk of extinction (86% of plausibility points) not likely to become endangered in the foreseeable future (for complete voting record see Appendix K).

Table 4.18 Summed SEDM weights for each risk category for the final SEDM point allocation assessing extinction risk.

Risk Category	Average SEDM weight (SD)
Very low	50 (19.3)
Low	36 (12.8)
Medium	9 (8.3)
High	5 (5.6)

The following text summarizes the BRT's conclusions with regards to current and future threats, uncertainty, current abundance estimates and population trends that support the voting results.

The level of extinction risk facing a population depends on information about population abundance, trends in abundance or other population indicators, potential threats to the population over time and uncertainty about the future. Fisheries-related mortality was the only anthropogenic threat considered by the BRT to be a potentially substantial risk factor for the NEP white shark population. Other threats such as physiological effects of contaminants or ocean acidification from climate change were acknowledged as factors that could adversely affect the population, but the BRT thought these factors would have relatively minor population-level consequences within the foreseeable future compared to direct fisheries-related mortality. Depletion of white shark prey (pinnipeds and fish stocks) from human activities may have had historical consequences for white shark population levels, but pinniped populations have increased over the last several decades and many fish stocks have similarly recovered or have

been recovering and so prey depletion was not considered to be a concern for the white shark population now or in the foreseeable future.

A number of factors influenced the BRT's assessment of population abundance and extinction risk. One of the most important pieces of evidence for a low to very low risk of extinction comes from the available bycatch data. The level of YOY and juvenile bycatch mortality estimated for U.S. fisheries and reported for Mexican fisheries is not consistent with a "very small" population. If adult female abundance is presently less than 200 individuals, the estimated YOY bycatch would correspond to removing on the order of 20% to 70% of estimated annual pup production. The BRT considered such removal levels to be highly implausible for gillnets which are not selective, given that removal rate estimates for other shark populations (Worm et al. 2013) and gillnet mortality estimates for other coastal protected species (e.g., marine mammals, sea turtles) are generally less than 10% and virtually always less than 20%. Removing more than 20% of the population seems especially unlikely given that the available white shark bycatch estimates come from fisheries operating in only a relatively small portion of YOY nursery habitat. Assuming, therefore, that gillnets are most likely removing less than 20% of annual pup production, the level of YOY and juvenile bycatch currently documented to be occurring suggests that the NEP population consists of at least a few hundred adult females.

The genetic haplotypic diversity for the NEP white shark population is another factor that the BRT considered in assessing adult female abundance. The available haplotype diversity for this population is consistent with a population consisting very recently (within the last few generations) of at least several hundred to several thousand adult females. The question is whether the abundance remains at this level presently or whether the population was recently severely depleted. The historical projection models suggest that while some level of population decline may have occurred in recent decades, the level of historical bycatch that would have been required to deplete a population of 1,000 adult females in 1971 to fewer than 100 or 200 individuals now is inconsistent with available information about historical bycatch levels (Sections 2.3, 4.1.4 and Appendix B). This suggests that the haplotypic diversity reflects a current adult female population size of at least a few hundred to a couple thousand adult females.

If current adult female abundance exceeds 200 adult females as the BRT has concluded is most likely, then this implies that the empirical estimates based on the photo-ID data from Guadalupe Island and central California do not represent an accurate assessment of the abundance of the entire population. This bias could be explained by several factors considered highly plausible by the BRT, including:

1) Under sampling of females at the coastal aggregation sites due to a temporal mismatch in sampling with respect to the timing of arrival by females to those sites;
2) Under sampling of females relative to males at the coastal aggregation sites due to spatio-behavioral factors (e.g., differences in territoriality patterns, time spent at depth, likelihood of approaching bait);
3) Non-availability of both females and males at the fixed sampling locations at the coastal aggregation sites, due to area -restricted movements of sharks around the pinniped rookery areas; or

4) Non-availability of both females and males at the coastal aggregation sites because they do not use the surveyed aggregation areas at all (e.g., use other pinniped rookery areas or do not feed substantially on marine mammal prey).

The strongly skewed sex ratio of observed white sharks at the coastal aggregation sites is more consistent with differential availability of males and females to sampling, as seen in other location, than with a true skew in the population's sex ratio. This is based on the lack of evidence for the sex-biased mortality of juveniles that would be required to explain the observed sex ratio skew in the subadult and young adult class as a true population phenomenon. In addition, differential habitat use between the sexes is common among sharks and in lamnids specifically.

Although population trend information is somewhat inconclusive, the BRT concluded that the available information is consistent with a stable or increasing population and not obviously indicative of a declining population (and hence not consistent with most high-risk population projection scenarios). White shark CPUE has increased since the mid-1990s in the U.S. west coast set net fishery, which would be expected for an increasing population. This increase coincides with changes in fishery regulations (e.g., the high seas drift gillnet ban, time-area closures within U.S. waters) and declining fishery effort in U.S. west coast net fisheries. Increasing abundance at Guadalupe Island and increased incidence of white shark attacks on marine mammals at the California Channel Islands may also suggest that the population is increasing, but could simply indicate increased *use* of certain areas, and not necessarily increasing population-wide abundance. However, the possibility that increased use at Guadalupe Island and the California Channel Islands could be decoupled from abundance trends for the whole population implies the existence of a larger source population apart from the aggregation areas (unless all external recruitment to Guadalupe Island is from the central California aggregation), again indicating an adult population size larger than what has been estimated based on photo-ID data for Guadalupe Island and central California.

4.7. Suggestions for Future Research

There are a number of gaps in our understanding of the biology, ecology, population dynamics and human interaction that lead to a level of uncertainty in the status review. Additional research is needed. The BRT recommends continuation of photo-ID studies at Guadalupe Island, Mexico and central California including: 1) comparing the two databases; and 2) expanding temporal and spatial scope to encompass more of the range of white sharks. We also recommend expanding the genetic studies to: 1) include nuclear markers; 2) compare samples from both aggregation sites and across the range; and 3) analyze samples of YOYs and juveniles with a focus on identifying parents. Additional tagging studies should: 1) focus efforts outside of the two current aggregation sites; 2) deploy more SPOT tags on mature females; 3) tags more juveniles in the U.S; 4) and tag juveniles and YOY in Mexico. We recommend the continuation of fisheries monitoring in Mexico, possibly including sampling at markets and using genetics to validate species identification. The MBA biological sampling program has provided valuable data on catch in U.S. waters and should be continued. Additional efforts should examine the potential impacts of both pollutants and ocean acidification on sharks in general. A number of important

studies are currently underway and we expect that in the next five year some of these data gaps will be filled.

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APPENDICIES:

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Appendix A: Summary of all the SEDM votes taken by the BRT

A number of decisions made by the BRT were made by a structured expert decision making (SEDM) voting system. A summary of all factors informed through SEDM voting are provided in Table A.1.

Table A.1 Summary of all SEDM voting made by the BRT.

Decision	Description	Categories
Discreteness	NEP white sharks discrete from other white sharks found in the Atlantic and Pacific Oceans based on behavioral factors.	Yes, No
Significance – genetics	NEP markedly separated based on genetic factors	Yes, No
Significant gap in range	Loss of NEP white sharks would result in a significant gap in the range of the taxon.	Yes, No
SPOR	Identify whether there are any specific portions of the white sharks geographic range in the NEP that are significant in terms of the population's overall viability, and if lost it would significantly increase the population's risk of extinction.	Yes, No
Risk	Evaluate the level of historic and current risks	low, medium, high
Trend	Evaluate the information suggesting population trends	Increasing, stable, decreasing, uncertain
Sex ratio CC	The sex ratio for adult white sharks at central California study sites	0.2, 0.6, .08, 1
Sex ratio GI	The sex ratio for adult white sharks at GI	0.6, 0.8, 1
Abundance bias	By what factor is the abundance in the NEP underestimated.	none, 1.2, 2, 10
Female mortality	Potential number of adult females killed each year	0, 1, 2, 5, 10
Post model abundance bias	Modeling exercises indicated that the known number of YOY's killed each year is not possible from the observed number of females. A second SEDM point allocation was taken on levels of bias for abundance estimates.	1, 1.2, 2, 10
Final risk assessment	Examine all factors that inform risk, abundance and trends to determine the level to which the BRT felt that NEP white sharks are at risk of extinction.	High, medium, low, very low

Appendix B: Examination of the potential for a dramatic population decline over the last two generations

The NEP white shark population's genetic haplotype diversity is moderate in comparison to other species and haplotypic diversity can provide insight into population abundance (see Section 2.3). There are two possible scenarios that explain the moderate level of haplotypic diversity. 1) The number of mature females is at least several hundred to a few 1000 or 2) it was at that level within the last few generations but suffered a recent and dramatic population decline (>90%) (see Section 2.3). The two possible explanations have very different implications regarding the level of extinction risk facing the population, with the first explanation indicating the present-day population is abundant, and the second explanation indicating that the present-day population has undergone a severe decline in the recent past. The BRT conducted the following Monte Carlo exercise to attempt to distinguish between these two possible scenarios. The objective was to impose relatively high fisheries related mortality to determine whether it was feasible to induce a 90% population decline over ~ 2 generations. The goal was not to model recent population trends.

Methods

For each sample *i* out of 1000 Monte Carlo simulations, an initial population of $n_{adult,0,i}$ = 500 or 1000 adult females was projected forward 41 years (mimicking the period 1971 – 2011) using an age-structured density-dependent stochastic population model. The initial population was assumed to be at its carrying capacity in 1971 (K_i). For simplicity, density-dependence was assumed to act on the population by affecting juvenile survival rate, $S_{juv,i}$, linearly as a function of $N_{t,i}/K_i$:

$$S_{juv,t,i} = S_{juv,K,i} + (1 - N_{t,i}/K_i)(S_{juv,max,i} - S_{juv,K,i}),$$

where $S_{juv,K,i}$ is the juvenile survival rate for a population at equilibrium or carrying capacity (K_i) such that $\lambda = 1$, $N_{t,i}$ is the sum of female abundance across ages in year t, and $S_{juv,max,i}$ is the juvenile survival rate corresponding to the estimate of $\lambda_{max,i}$ (see Section 4.5.2.1). $S_{juv,K,i}$ was found by setting $\lambda = 1$ and solving for $S_{juv,i}$ using the Euler equation (holding other vital rates for Monte Carlo sample *i* at their values corresponding to the estimate for $\lambda_{max,i}$). K_i was set equal to $N_{0,i}$ (i.e., $\Sigma n_{j,0,i}$), which was estimated by assuming that the population was initially in a stable age distribution (according to the Leslie matrix for $\lambda = 1$, $\mathbf{M}_{K,i}$) so that female abundance for each age j at time zero was equal to $n_{j,0,i} = n_{adult,0,i} * u_{K,j,i}/u_{K,adult,i}$, where $\mathbf{u}_{K,i}$ is the stable age eigenvector for the Leslie matrix $\mathbf{M}_{K,i}$.

The population of adult females was projected through time using an approximating model:

$$\mathbf{n}_{t+1,i} = (\mathbf{M}_{byc,t,i}\, \mathbf{n}_{t,i}) \exp(\varepsilon_{t,i})$$
$$\varepsilon_{t,i} \sim \text{Normal}(0, \sigma = 0.1)$$

where $\mathbf{n}_{t+1,i}$ is the population size in successive time steps, $\mathbf{M}_{byc,t,i}$ is the Leslie matrix that varies through time with changing $S_{juv,t,i}$, and $\varepsilon_{t,i}$ describes annual stochastic variation. $\mathbf{M}_{byc,t,i}$ also

reflected intense bycatch mortality. For years 1971 to 1990, fecundity and age-1 survival parameters were each multiplied in the matrix by 0.8 to indicate 20% annual mortality on YOYs and 1-year olds, and the subadult and adult survival parameters were multiplied by 0.98 to indicate 2% annual mortality on these age classes. For years 1991 to 2011, these rates were reduced to 0.15 for YOYs and 1-year olds and 0.01 for subadults and adults, reflecting reduced mortality associated with changing fishery regulations in the U.S. and on the high seas. These rates correspond to the white shark bycatch estimates indicated in Table B.1, which are all substantially higher than what has been documented in the available information about historical bycatch. The environmental stochastic parameter ε_t is a scalar error term applied to abundance in all age classes, thus mimicking extreme covariance in the vital rates (i.e., conditions in a given year are good or bad for all ages). A more realistic model would apply stochastic error to the individual vital rates each year (with variance on the rates selected to produce the desired variance for $n_{adult,t}$) rather than directly to the age-specific abundances. The current method was used merely for simplicity, with the goal of approximating realistic dynamics in adult abundance, not necessarily in the age-specific dynamics from year to year.

Table B.1 Average white shark bycatch (across 1000 Monte Carlo simulations) estimated for 1971 (start of time series) and 1995 (after cessation of high seas drift net fishing and protective regulations in the U.S.), given initial populations of 500 or 1000 adult females in 1971, and given bycatch mortality rates in population transition matrix $M_{byc,t,i}$. Values in the table are for females (and for both sexes in parentheses assuming sex ratio parity).

	$n_{0,adult}$ = 500 females		$n_{0,adult}$ = 1000 females	
	1971	1995	1971	1995
YOY and juvenile bycatch	325 (650)	160 (320)	650 (1300)	325 (650)
subadult and adult bycatch	20 (40)	7 (14)	40 (80)	13 (25)

Results

The assumed levels of bycatch mortality in this exercise were sufficient to cause strong population declines over the course of 40 years (Figure B.1), although the probability that by 2011, the population would decline to 10% of the population levels in 1971 (100 or 50 mature females) was essentially 0. With respect to the different risk scenarios, the probability that they would reduce the population from 500 or 1000 adult females to levels that would correspond to high risk scenarios (< 125; see Section 4.5.2.3) was low. If the initial adult female abundance in 1971 was 500, then there was a 20% chance of decline to < 200 adult females (corresponding to medium risk abundance scenarios) and a 5% chance of decline to < 125 adult females (corresponding to high risk scenarios) by 2011, given the assumed levels of bycatch mortality. If initial adult female abundance in 1971 was 1000, then depletion of the population to levels that would correspond presently to medium (<200 adult females) or high (<125 adult females) risk would be highly unlikely.

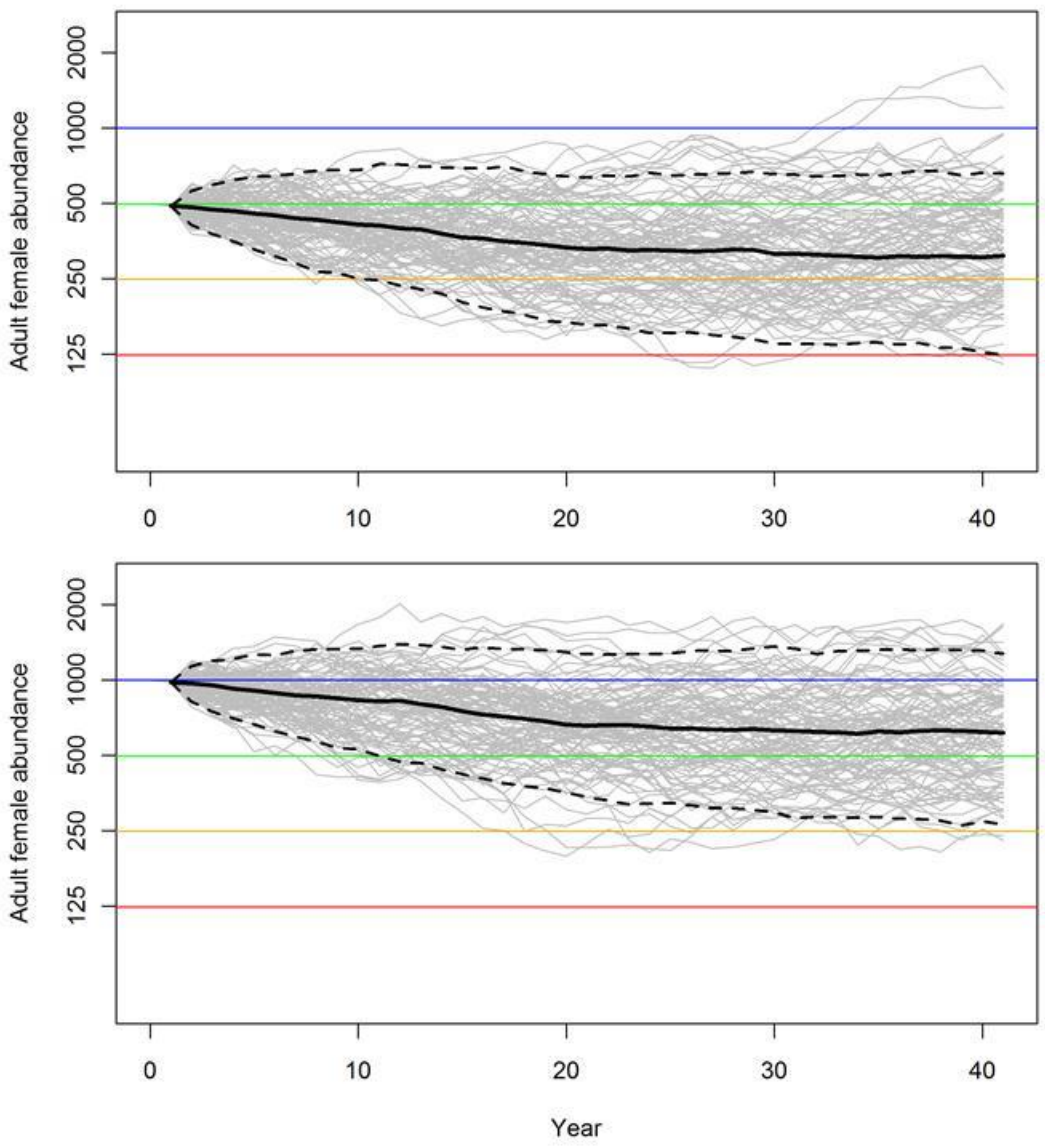

Figure B.1 Projections of adult female abundance from the stochastic density-dependent age-structured model. Initial adult female population size at year 0 (1971) was either 500 (top) or 1000 (bottom). The solid black line represents the median projection and the dotted lines represent the 90% CIs.

Discussion

This model is conservative and precautionary in several respects. Initial female abundance was assumed to be only 500 or 1000, whereas genetic haplotype diversity suggests abundance may presently be or may have been considerably higher (e.g., in the high hundreds to low thousands). The bycatch mortality rates used in the model are high and correspond to numbers of white sharks caught and killed in the early-1970s and mid-1990s that far exceed what available information suggest were likely to have occurred in the NEP (see Section 4.1.1). The BRT acknowledged, however, that elasmobranch discards in high-seas large-mesh drift gill net

fisheries during the 1970s and 1980s were poorly documented and could have been substantial. In addition, carrying capacity (K) was fixed so that population growth was limited throughout the time series by what the environment supported in the early 1970s. Since that time, prey populations for white sharks have increased in abundance, so higher population growth might have been achievable in more recent years.

In summary, the scenarios modeled here are intended to represent a pessimistic case, in which the historical population abundance included only 500 to 1000 adult females, bycatch mortality rates were high (in comparison to rates indicated by the available bycatch data), and the carrying capacity of the population remained fixed at historical, potentially reduced levels (due to reduced prey abundance in the 1970s).

Based on results of these analyses, it seems unlikely that the NEP population of adult female white sharks would have declined over a short time period (e.g., last 40 years) from levels consistent with the genetic haplotype data (adult females numbering in the high hundreds or thousands) to levels represented by the survey estimates from central CA and Guadalupe Island (~ 50 adult females). Some declines could have occurred since the 1970s, but model results suggest that the number of mature females still likely numbers at least in the low hundreds today.

Appendix C: DPS determination

DPS - Determination of Discreteness

The BRT considered whether the NEP white shark population is discrete and significant and constitutes a DPS, as defined by the NMFS/FWS DPS policy (61 FR 4722, February 7, 1996). For each issue, detailed arguments for and against each option--and the resulting distribution of plausibility points--are provided below to fully document the BRT's decisions outlined in the text of this status review.

Evaluation of DPS Discreteness Criteria- Are NEP white sharks markedly separated from other populations of the same taxon as a consequence of behavioral factors?

Arguments <u>in favor of</u>* marked separation *based on behavioral uniqueness

The distribution of white sharks in the NEP is markedly separated from the South and western North Pacific and other global populations based on

- Tagging data
  - Satellite tagging data show no movement outside the NEP.YOYs are known to remain in coastal shelf waters off California, primarily south of Point Conception to Sebastián Vizcaíno Bay. Adults migrate from at least the two primary aggregation sites to offshore areas stretching as far west as the Northwest Hawaiian Islands. With over 100 sharks tagged with satellite tags in the NEP, there have been no observed movements across the equator or west of the Hawaiian Islands. Furthermore, white sharks satellite-tagged in the Southwest Pacific near Australia and New Zealand have similarly not migrated into the NEP (Bruce et al. 2006, Duffy et al. 2012).

- Photo-ID data
  - Photo-ID shows strong site fidelity of adults to coastal aggregation sites. Males are observed to return predominantly on a yearly basis with some individuals returning to the same sites for up to 22 years (Anderson et al. 2010). Females are observed to return on either an annual or semiannual basis. This pattern is thought to be linked to reproduction and parturition.

- Genetic data (mitochondrial DNA)
  - Genetic data from the central California coast show a group of closely related haplotypes that are not found in the nearest other white shark populations (Japan, Australia/New Zealand or South Africa). Although the sample sizes are relatively small from all the regions, the general pattern of finding only related haplotypes within most regions (monophyletic) strengthens the inference that little movement is occurring between these regions. Although mitochondrial data represent female inheritance, the genetic data coupled with the movement data from tagging (that shows that males stay within the region and have more limited movements than females) suggests separation of the NEP from other regions.

Arguments <u>against</u> marked separation based on behavioral uniqueness

There are inadequate data from white sharks that potentially never are available for sampling at the coastal adult aggregation areas. Arguments against behavioral separation are:

- Tagging Data
    - Tagging bias likely underestimates the extent of potential mixing with other white shark populations. All individuals have been tagged in only a few locations, e.g., YOYs in southern California and subadults and adults at the central California or Guadalupe Island aggregation sites. Although both tagging data and photo-ID data indicate site philopatry, there may be a portion of NEP white sharks with significantly different behavior, especially those that may feed on fish rather than pinnipeds. The tagging and photo-ID data may bias our understanding of the magnitude of separation of NEP white sharks from other white shark populations.

- Genetic data
    - The sample distribution is inadequate to characterize genetic separation, because the samples included in the genetic analysis were collected only from the central California aggregation sites. Site fidelity combined with restricted sampling likely results in non-representative sampling of the overall NEP white shark population. In addition, relatively few samples are available from other areas (South Africa, Australia/New Zealand and Japan). The high number of haplotypes represented by only a single individual strongly suggests that haplotypic diversity is as yet poorly quantified. Consequently, the conclusions that there are no shared haplotypes between white sharks in the NEP and other regions are weakened.

NEP white sharks also show no behaviorally unique characteristics such as dietary specialization, life history differences or migratory behavior in comparison to white sharks observed in other regions.

DPS – Determination of Significance

Evaluation of DPS Significance Criteria- Is there evidence that NEP white sharks differ markedly from other populations of the species in their genetic characteristics?

Arguments <u>in favor of</u> marked genetic differentiation

NEP white sharks differ markedly from other populations of the species in their genetic characteristics. Evidence supporting marked differences is:

- NEP white shark mtDNA haplotypes form a weakly supported monophyletic clade.
    - NEP white sharks cluster together in relation to haplotypes from Japan, Australia, New Zealand, and South Africa and no haplotypes are shared with these other regions.

    - The magnitude of mtDNA differentiation is large enough to infer that time has been sufficient and female-mediated gene flow low enough to potentially allow adaptation to the NEP habitat and that the NEP would not be readily repopulated by neighboring shark populations 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 of the mtDNA of the NEP white sharks to the mtDNA of white sharks in all other geographic strata indicate less than one migrant per generation between the NEP and these other regions. The genetically most-closely related population is Australia/New Zealand and the time estimated since divergence is estimated to be ~150kya.

- Although there are no published nuclear data for the NEP, tagging data show no movement of males outside the NEP and these tagged males have demonstrated site-fidelity through both tracking data and photo-ID data.
    - Many of the individuals in the genetic study that included NEP samples were known adult males with photo-ID histories. All of these males had haplotypes indicating a NEP origin. These observations are consistent with a closed population without male-mediated gene flow between the NEP and other regions.

Arguments <u>against</u> marked genetic differentiation

NEP white sharks might not differ markedly from other populations of the species in their genetic characteristics. Arguments against marked differentiation are:

- The sample size and distribution is inadequate to characterize genetic separation
    - The proportion of the NEP population sampled is likely to be relatively small given the relatively high number of haplotypes represented by only a single individual. Further, the published data do not include any samples from Guadalupe Island which represents a substantial proportion of the estimated adult population. There are also no samples from other areas within the range of the NEP population (e.g., near Hawaii), which may exclude animals that don't regularly visit the two studied aggregation sites.

- NEP white sharks are potentially not significantly different from adjacent white shark populations because of possible male-mediated gene flow. The lack of published analysis of nuclear markers in the NEP does not allow inference about how marked genetic differentiation might be.
    - Another plausible scenario that may reduce the genetic differentiation of NEP white sharks involves male mediated gene flow. Pardini et al. (2001) showed that nuclear markers indicated male mediated gene flow in white sharks whereas mitochondrial markers showed limited female mediated gene flow. The low level of gene flow needed to overwhelm local adaptation (roughly 1 migrant per generation) would be difficult to detect using photo-ID and tagging studies, which are short in duration (the longest time series is about10 years, which is roughly half of one shark generation). Even if breeding occurs at the aggregation sites (which remains uncertain), detecting low numbers of males from outside the NEP would be difficult. If breeding occurs offshore, then detection of males from outside the NEP would not be possible.

Evaluation of DPS Significance Criteria- Would loss of NEP white sharks result in a significant gap in the range of the taxon?

Arguments <u>in favor</u> of loss of the NEP population resulting in a significant gap in the range of the taxon

White sharks are found in temperate waters in the North Pacific, Southwestern Pacific, North Atlantic and South Atlantic. Evidence in support of the loss of the NEP white sharks resulting in a significant gap in the range of the taxon:

- The NEP represents a large portion of the range of the species (over 20%).
- The strong phylogeographic patterns of mtDNA are consistent with rare colonization events with little to no immigration events thereafter.

Arguments <u>against</u> loss of NEP resulting in a significant gap in the range of the taxon

White sharks have similar life histories and prey types globally. Loss of the NEP population would not diminish the ability of white sharks to persist as a species because:

- The similar ecological function of white sharks globally reduces the importance of losing any single population to the evolutionary potential of white sharks.
- The presence of white sharks in adjacent waters off Japan and Australia/New Zealand suggests that recolonization of the NEP could occur on an evolutionary timescale.

DPS – Plausibility point allocation by BRT team members

The BRT used a structured expert decision making (SEDM) approach to evaluate whether the NEP white shark population met the DPS criteria of discreteness and significance. BRT members independently allocated plausibility points on the discreteness and significance criteria using the likelihood point method, whereby each member distributed 100 points between the arguments for and against each factor. Point allocation in this manner allowed individual BRT members to express their level of certainty on each of the factors, such that placement of all 100 points either for or against a particular factor would indicate certainty in the arguments or evidence presented. BRT members agreed to view resulting plausibility points with the members' names associated to facilitate discussion and assure that linguistic uncertainty was not responsible for any disparate point allocations. The BRT discussed the point allocations and in some cases adjusted points when prior articulation of the arguments had been unclear. Individual final BRT member point allocations, renumbered on each ballot to retain anonymity, and the overall proportion of points for arguments for or against each factor are shown in the tables below.

Table C.1 Summary of BRT point allocations on A) the discreteness, B) and significance based on genetics and C) significance based on gap in the range for NEP white sharks.

A. Factor 1. Are NEP white sharks markedly separated based on behavioral factors			
Member	Yes	No	Total
1	95	5	100
2	90	10	100
3	95	5	100
4	90	10	100
5	90	10	100
6	95	5	100
7	95	5	100
8	90	10	100
Avg	**0.92**	**0.08**	**1**

B. Factor 1. Are NEP white sharks significant based on genetic factors			
Member	Yes	No	Total
1	85	15	100
2	85	15	100
3	55	45	100
4	90	10	100
5	80	20	100
6	80	20	100
7	85	15	100
8	85	15	100
Avg	**0.81**	**0.19**	**1**

C. Factor 2. Loss of NEP white sharks would result in a significant gap in range			
Member	Yes	No	Total
1	100	0	100
2	100	0	100
3	90	10	100
4	95	5	100
5	95	5	100
6	100	0	100
7	90	10	100
8	100	0	100
Avg	**0.96**	**0.04**	**1**

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Appendix D: MBA sampling program characterization

In 2002 the Monterey Bay Aquarium initiated a scientific collection program for both live and dead sharks, asking fishermen to notify them of any white shark bycatch. Additional data for the period prior to the program were obtained from one vessel that caught 3 sharks over 5 years. From 2002-2005 the program included 4 vessels. In 2005 they started a more extensive outreach campaign and by 2011, 15 vessels were involved. A monetary award was given to fishermen who brought in sharks to compensate them for their time given that for live fish in particular it could take hours to transfer them to aquarium staff and collaborators. The amount of the award and the low incidence of catch are inconsistent with fishermen intentionally shifting fishing effort to target white sharks as a part of this program. On average fishermen working with MBA caught one shark a year. It is believed, however, that during the period of the MBA program reporting rates increased especially after 2005. Over the course of the program a total of 53 live and 52 dead sharks were recorded. The majority of dead sharks (19) came from one fisherman who has since left the fishery. The program has provided invaluable data on movements, behaviors, habitat use, pollutant loads, fisheries interactions and genetics and is the source of most of the data available for YOY and juvenile white sharks in the NEP.

Appendix E: Summary of SEDM voting on population trends

The results from the BRT SEDM voting process on the plausibility that the population of NEP white sharks is increasing, stable or decreasing. Also included is a level for uncertainty. See Section 4.3.

Table E.1 Summary of BRT point allocations on population trends.

BRT Member	increase	stable	decreasing	uncertain
1	45	30	0	25
2	50	25	25	0
3	40	30	10	20
4	50	40	0	10
5	30	10	5	55
6	40	30	5	25
7	40	40	0	20
8	40	30	0	30
Avg	**41.9**	**29.4**	**5.6**	**23.1**
Stdev	**6.5**	**9.4**	**8.6**	**16.0**

Appendix F. Mark recapture analysis methods

Mark-recapture analyses were conducted using the Photo-ID data from both Guadalupe Island and the two sites in central California. Analyses conducted by the BRT included the data used in publications (Chapple et al. 2011, Sosa-Nishizaki et al. 2012), as well as new data on white shark size and sex at both study areas and two additional years of sightings data and monthly sampling effort (Table F.2) at Guadalupe Island. Analyses were conducted to reexamine the white shark abundance estimate cited in the petition of 339 subadults and adults (based on the estimates from Chapple et al. 2011, Sosa-Nishizaki et al. 2012), and to examine the potential for trends at Guadalupe Island where there is a longer time series.

For the analyses, the BRT used R package RMark version 2.1.3 (Laake and Rexstad 2011) as an interface to the program MARK version 6.1 (Cooch and White 2011) to apply mark-recapture models to the data from each of the study areas. The open-population POPAN model was applied to both study areas.

POPAN estimates 4 parameters directly (N, φ, p, and p_{ent}). N is the superpopulation size or all animals at the site over the study period. The parameter φ is sometimes called "apparent survival", and quantifies the individuals that have remained at the site and were not lost to emigration or mortality. The p parameter is the probability of detecting an individual on a given sampling occasion, given that it was actually present at the site. The p_{ent} parameter is the probability of recruitment of new individuals into the group at the site on each new sampling occasion. The N parameter, or "superpopulation", represents all the individuals that were ever at the site during the study, including those that have since died or emigrated away.

Mark-recapture models can be configured to estimate each of the parameters as either a single constant, or as multiple values that can vary with time or by sex. A variety of additional derived parameters, such as the numbers of individuals present at a site in each year, can be subsequently calculated from the model using the estimated parameters.

For central California, a single data configuration consisting of all the observations over the three year period (2006-2008) was supplied to the open-population POPAN model. For model runs, configurations using the data categorized as males, females, and unknowns, and alternative configurations without sex as a covariate were compared using AICc.

For Guadalupe Island, the longer time series (2001 – 2011) allowed the data to be configured in multiple ways. In addition to the combined model representing all 11 years the BRT calculated annual population estimates as well as a three year super population estimate covering the years from 2006 - 2008.

For each assemblage from the two sites, at least 500 alternative model configurations (combinations of input parameters) were evaluated. From each set of alternative model configurations the best model describing that dataset was selected based on AICc. The POPAN models selected based on AICc for both sites are shown in Table F.1.

Table F.1 Models selected by AICc for central California and Guadalupe Island.

Site	Configuration selected by AICc
Central California	ω(~sex + time)p(~time) p_{ent} (~time)N(~1)
Guadalupe Island	ω (~sex + time)p(~effort * sex) p_{ent} (~sex)N(~1)

The different configurations included various combinations of non-varying, and time- and sex-varying parameters, with and without interaction terms. For the models based on the final 11-year Guadalupe dataset, parameter values from the best models with and without covariates for total days of sampling effort per year, and a covariate for the monthly proportion of sampling effort that occurred in November to December (totaling 650 alternative model configurations based on this final dataset) were evaluated. Model-averaged parameter estimates for the best 5 models of the 650 total models for the final 11-year dataset were compared to the estimates from the best model.

Two measures of sampling effort, the total number of effort-days per year, and the proportion of these effort-days occurring in November-December were included as covariates in the population assessment modeling.

Table F.2 The number of days of sampling effort in each year at Guadalupe Island from 2001-2011.

	July	Aug	Sept	Oct	Nov	Dec	Total
2001	0	0	1	1	1	0	3
2002	2	2	6	6	6	1	23
2003	0	9	5	6	15	2	37
2004	0	3	10	14	19	5	51
2005	0	6	23	12	13	3	57
2006	0	11	17	13	11	0	52
2007	0	6	18	21	6	2	53
2008	0	14	17	14	2	6	53
2009	2	10	20	21	12	0	65
2010	0	16	21	10	1	0	48
2011	0	9	13	16	6	0	44

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Appendix G. Pre-model assessment of sex ratio and abundance biases

Prior to conducting the population analyses the BRT used SEDM to determine the plausibility of different sex ratios and abundance biases using the observed data from the two studies at central California and Guadalupe Island as a starting point (see Section 4.4.3). The arguments for each value considered are provided below.

Questions 1 & 2--Sex ratio at central California sites and at Guadalupe Island: The BRT evaluated the plausibility of different sex ratio (adult female:adult male) estimates for central California (Question 1) and Guadalupe Island (Question 2). The low (minimum) estimates of sex ratio considered by the BRT were based on observations of the numbers of subadults and adults at the different study areas and the sex ratios at the different areas. The data on sex were not included in the model run by Chapple et al. (2011) but were provided to the BRT for the purpose of the status review. The high (maximum) estimates assume a 1:1 ratio based on the parity observed at early life history stages. One or two intermediate values between the minimum and maximum estimates were also considered for each study area. Arguments for the different sex ratio values considered are given below.

<u>Minimum values:</u> These values assume that the photo-ID data accurately reflect the sex ratios at the two study areas. The lower number of adult females relative to adult males could be accounted for by the potential increased risk of mortality to adult females that must come into coastal waters to pup. These adult females could become fatally entangled in gear and swim away with that gear without being detected. A low entanglement rate would be sufficient to account for the apparent difference between survival and emigration rates of females (0.9) compared to males (0.93) (see Section 4.4.2).

Question 1: Minimum sex ratio (R_{min}) for Guadalupe Island =0.6
Question 2: R_{min} for central California = 0.2

<u>Maximum values (1:1=females:males):</u> Roughly equal number of males and female YOY and juveniles are caught suggesting an equal sex ratio for males and females in their first years. Other long-lived species with low reproductive rates tend to have equal sex ratios throughout life.

Questions 1 and 2: Maximum sex ratio (R_{max}) = 1

<u>Intermediate values:</u>
Sex ratio could be lower than parity reflecting a higher risk for females but not as low as the observed values, reflecting the underestimation resulting from the later arrival of females to the aggregation sites.

For Guadalupe Island the intermediate value (R_{avg}) was calculated as the average between the R_{min} and R_{max}.
Question 1: R_{avg} for Guadalupe Island = 0.8

For central California, two intermediate values were used given the large spread between R_{min} and R_{max}. These values were the R_{min} for Guadalupe Island and R_{avg} for Guadalupe Island.

Question 2: R_{avg} for central California = 0.6 and 0.8.

Question 3--Aggregation site sampling bias arguments (accounting for unsampled individuals)

A SEDM point allocation was conducted to assign plausibility points to the level to which mature female white sharks in the NEP were underestimated based on data collected at only two study areas (two central California sites and one Guadalupe Island site)..

B1: No bias--All or nearly all adult female white sharks are sampled at the sampling sites.
B1.2. 20% more adult female white sharks occur in the NEP than are sampled at central California and Guadalupe Island sites.
B2: Double the estimated value--About half of the adult female white sharks are sampled at the sampling sites.
B10: There are an order of magnitude more adult female white sharks in the NEP than are sampled at the sampling sites.

The full voting results regarding the sex ratio bias and bias in adult female abundance estimates are shown in Table G.1 and G.2.

Table G.1 Results from SEDM voting on the plausibility of different sex ratio bias estimates at Guadalupe Island and central California.

BRT Member	Guadalupe Island sex ratios considered			Central California sex ratios considered			
	0.6	**0.8**	**1.0**	**0.2**	**0.6**	**0.8**	**1.0**
1	70	15	15	10	70	10	10
2	20	50	30	0	40	40	20
3	10	85	5	0	60	30	10
4	30	60	10	0	30	60	10
5	40	40	20	10	40	40	10
6	30	40	30	5	30	35	30
7	30	40	30	5	35	35	25
8	30	50	20	0	35	35	30
Avg	**33**	**48**	**20**	**4**	**43**	**36**	**18**
Stdev	**18**	**20**	**10**	**4**	**15**	**14**	**9**

Table G.2 Results from SEDM voting on the plausibility of different abundance biases for the NEP adult female white shark population.

BRT Member	none	1.2	2.0	10.0
1	25	25	25	25
2	0	30	50	20
3	0	10	75	15
4	0	5	70	25
5	10	20	40	30
6	0	20	60	20
7	5	15	30	50
8	0	30	50	20
Avg	**5**	**19**	**50**	**26**
Stdev	**9**	**9**	**18**	**11**

References Cited

Chapple, T. K., S. J. Jorgensen, S. D. Anderson, P. E. Kanive, A. P. Klimley, L. W. Botsford, and B. A. Block. 2011. A first estimate of white shark, *Carcharodon carcharias*, abundance off Central California. Biol Lett 7:581-583.

Appendix H. SEDM arguments for levels of adult female mortality

Using the photo-ID data at Guadalupe Island the BRT estimated apparent survival; the portion of the male and female white sharks that returned to the aggregation site over the course of the study and were not lost to emigration or mortality. The value for apparent survival was less for females (0.9) than males (0.93) at Guadalupe Island by approximately 3% (median values for apparent survival are higher for both males (0.98) and females (0.95) than the mean.) While the relative contribution of emigration and mortality cannot be determined, this difference raises the possibility that the mortality rate for females is higher than that for males. Based on the SPOT data for sharks at Guadalupe Island, adult females are more likely to visit near-shore habitats, presumably to pup, and may be vulnerable to entanglement in near shore fishing gear. Large females also spend more time on the high seas then males, returning to coastal aggregation sites only every other year (Domeier and Nasby-Lucas 2013), and thus may be more vulnerable to being caught on the high seas by IUU fishing vessels (Bonfil 1994). To address the potential for higher human-induced mortality for females, the BRT considered the plausibility of various levels of fisheries mortality on adult females.

Large white sharks are documented in the catch data in the U.S. (Klimley 1985), Mexico (Galván-Magaña et al. 2010) and on the high seas (Bonfil 1994) indicating the potential for entanglement in fishing gear. Klimley (1985) documented the catch of 27 subadult and adult white sharks along the west coast of the US and Canada from 1941 to 1984. Galván-Magaña et al. (2010) documented incidental catch of 16 subadult and adult white sharks in the Gulf of California from 1964 to 2010, with white shark captures occurring mainly in totoaba gill-nets or shrimp trawls. There are reports of five additional subadult and adult mortalities in the Gulf of California between 2004 and 2012 (Castro 2012, O. Sosa-Nishizaki, CICESE, pers. comm., E. Oñate-González, CISESE, pers. comm., and M. Domeier, MCSI, pers. comm.), and a SPOT tagged adult female that is presumed to have died in 2009, potentially due to fisheries catch (Domeier and Nasby-Lucas 2013). In summary, in the last 10 years, 9 mortalities of subadult and adult white sharks have been documented in the Gulf of California Of these 5 were over ~450 cm TL and 2 were documented to be females, 2 as males and 5 unknown. These numbers do not include the adult female SPOT-tagged shark.

Obtaining direct data on subadult and adult mortality from both natural and human causes is difficult. Adults spend most of their time in offshore waters and white sharks will sink when dead and thus are not likely to wash up on a beach. Therefore, deaths due to natural causes are largely undocumented. Mortality associated with fisheries interactions is likely underrepresented by the available data. Given their protected status in both the U.S. and Mexico, catch in fisheries may not always be documented. There may also be sharks that become entangled in gear but go unobserved because at their size they could swim away with part or all of the fishing gear and later die. An estimated carcass detection probability of 17% was made for North Atlantic right whales killed by either entanglement in fishing gear or ship strikes. Most whales float when dead, so the carcass detection rate is likely to be lower in white sharks that swim away with fishing gear.

SEDM on the level of annual adult female human-caused mortalities

The BRT used SEDM to express its uncertainty in the number of adult female white sharks that may die annually resulting from human activities. Plausibility points were distributed among the following options: 0, 1, 2, 5 and 10 adult female white sharks killed per year. Arguments for each of these levels are given below.

1) Argument for 0 human-caused adult female deaths per year
There are only 2 adult female white sharks known to have been killed from fishing-related activities over the past 10 years. This low level would be rounded to zero deaths annually in most years. Conversations with fishers and experts working in Mexico indicate that catching large white sharks is very rare and that most are documented through research or the popular press. Also, no females have been reported at Guadalupe Island trailing fishing gear. The male-female difference in estimated apparent survival would then be attributed to differences in emigration which would be consistent with the strong sexual segregation observed in white sharks, or to an increased rate of natural mortality for females (e.g., perhaps associated with the costs of reproduction).

2) Argument for 1 human-caused adult female death per year
If reported deaths where the sex is marked as unknown are counted as females and the abrupt disappearance of the SPOT-tagged female is included as a fatality, then there have been 8 adult female white sharks killed due to fishing activities over the past 10 years. This rounds to one adult female white shark killed per year.

3) Argument for 2 human-caused adult female deaths per year
Although the number of adult female white sharks killed in fisheries may be close to one per year (see argument #2 above), this is likely an underestimate. Some white sharks that are captured may not be reported and/or entangled animals may swim away with gear and subsequently die at sea undetected. To account for these undocumented mortalities, the BRT considered the plausibility of 2 adult female white sharks killed per year due fisheries interactions. This assumes that only half of the actual mortalities per year are documented.

4) Argument for 5 human-caused adult female deaths per year
Zero, one or two deaths per year are all likely underestimates of actual adult female mortality due to human activity. Since white shark carcasses sink, it is difficult to estimate the amount of mortality that goes unreported, but hypothetically, if white shark carcasses had the same detection rate as right whales (17%), then one observed death would translate to about 6 deaths. If the number of adult females is around 200 (see Table 4.11), then the resulting decrease in survival rate would be about 2.5%, which is similar to the estimated difference between male (0.93) and female apparent survival, based on the analysis of photo-ID data at Guadalupe Island. This would also indicate that there is no difference in rates of emigration between males and females in the NEP population.

5) Argument for 10 human-caused adult female deaths per year
Although based on the available information, the estimated number of adult female white shark deaths may be close to one per year, it is possible that this may under-represent mortality by an

order of magnitude given the difficulty of detecting dead sharks. The BRT included 10 as a potential upper limit to the number of females that might be killed in a given year.

Table H.1 Summary of SEDM voting regarding levels of annual adult female mortality due to fisheries interactions.

	Annual number of adult females killed by fisheries interactions				
BRT member	0	1	2	5	10
1	0	25	40	25	10
2	10	25	35	20	10
3	25	35	20	15	5
4	0	5	45	35	15
5	0	10	30	50	10
6	10	30	40	20	0
7	5	15	40	35	5
8	25	50	25	0	0
Avg	**9.4**	**24.4**	**34.4**	**25.0**	**6.9**
Stdev	**10.5**	**14.5**	**8.6**	**15.1**	**5.3**

The BRT SEDM voting results (Table H.1) indicated that the BRT felt that more mortalities were occurring than accounted for in the data and that some animals are either unreported or undetected. Plausibility points were spread between 1 and 5 adult female mortalities per year. The BRT felt that there is a bit of uncertainty in the actual amount of annual adult female mortality for NEP white shark and it is captured in this distribution.

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Appendix I: Post-model assessment of adult female abundance bias

Results of the modeling exercises contributed further insight as to the plausibility that the observed number of mature females at the photo-ID study sites represented the entirety of the NEP adult female population and informed estimates of female abundance bias. For adult female abundance associated with bias scenarios of 1 or 1.2 (i.e., NEP adult female abundance = 1x or 1.2x the sex-ratio-bias-corrected photo-ID estimates) fisheries mortality rates for YOYs are estimated to be ~70% and ~ 40% respectively (see Section 4.5.2.1). (Catch and mortality rates are similar given the 100% mortality assumed for the majority of the catch). These rates seem highly implausible for a number of reasons. Population removal rates for sharks, even for selective gears (such as pelagic longlining), are probably less than 0.20 (Worm et al. 2013). For populations of marine mammals and sea turtles known or suspected to be declining because of high bycatch mortality, the mortality rate on affected age classes from gill net bycatch is typically less than 0.10 (Table I.1). Vaquita porpoises – which are Critically Endangered (IUCN Red List), believed to be precipitously declining, and subjected to extremely intensive gill net exposure throughout their range– have an estimated bycatch mortality rate ~0.20 based on the model used in Jaramillo-Legorreta et al. (2007). It would be exceptional if gill nets had the efficiency to capture >> 0.20 of white shark YOYs each year, especially given that the majority of available bycatch estimates come from small fishing fleets operating throughout a limited portion of known or suspected YOY nursery habitat. Catch rates are highest at the same time of year in both Sebastián Vizcaíno Bay and the Southern California Bight, indicating a geographic spread in the YOY population.

Table I.1 Estimated fisheries mortality rates for a number of long-lived marine mammals and turtles.

Species	Estimated fisheries mortality	Reference
Vaquita	20%	Jaramillo-Legorreta et al. (2007)
Gulf of Maine harbor porpoise	5.4%	Moore and Read (2008)
Fransicana dolphins	3%-5%	5%, Kinas 2002; 3%, Negri et al. 2012
Pacific leatherback seaturtles	11%	Tomillo et al. 2007
Australia sea lions	3.9%	Goldsworthy et al. 2009

Given the model results, it was apparent that the low levels of abundance bias (1 and 1.2) previously considered by the BRT may not be plausible. Therefore, the BRT assigned plausibility points a second time to determine the level to which mature female white sharks in the NEP were underestimated based on the available photo-ID data. Below are the arguments for and against each level of bias considered by the BRT, incorporating information from the modeling exercises.

<u>B1 = No bias in abundance estimates</u> --All or nearly all adult white sharks come to the sampling sites. (for bias=1 N_f = 48 to 157 depending on the sex ratio bias as the two sites)

Arguments for: Assuming no bias comes closest to the estimated number of females at the sampling sites and is therefore supported with direct data. Although the single report of YOY mortality from Mexico is considered to be the most accurate available estimate, it has not been verified and there is no way to verify that this level of catch has been and will be constant through time.

Arguments against: The level of YOY mortality associated with abundance estimates linked to bias=1 have unrealistically high mortality (~70%) rates. Such a high level of YOY catch is unlikely for reasons discussed above. Haplotypic diversity is also not consistent with an adult female population size below 200 (see Section 2.3, 4.3 and Appendix B).

B1.2= There are a small number of adult female white sharks (20% of the population) that do not visit the sampling sites (for bias=1.2 N_f = 57 to 188 depending on the sex ratio bias at the two sites).

Arguments for: Estimates of mature female abundance resulting from this small level of bias primarily match the estimated number of adult females at the sampling sites based on the photo-ID data, allowing for some missed animals. Some small number of adult females likely visit other sites or no sites at all and are thus, not counted in photo-ID studies.

Arguments against: If these estimates were accurate, then the known levels of YOY mortality would constitute ~ 40% of the annual pup production possible from this many adult females. Such a high level of YOY mortality is unlikely for the reasons discussed above. As with the bias of 1, the genetic data are not consistent with an adult female abundance <200.

B2= About half of the adult female white sharks in the NEP population come to the sampling sites. The population has twice as many adult females than estimated based on the photo-ID data (for bias=1 N_f = 96-314 depending on the sex ratio bias as the two sites).

Arguments for: These values allow for a reasonable number of missed animals that are either at the sites and not observed in sampling efforts, or never come to the two study areas. These higher estimates of adult female abundance are more compatible with the observed level of YOY bycatch and the observed level of genetic diversity than the lower estimates (under B1 and B1.2). Known levels of YOY mortality would constitute an estimated 25% of the annual pup production possible from this many adult females. Such a mortality rate is more plausible than those associated with bias values of 1 and 1.2.

Arguments against: A 25% YOY mortality rate is still higher than the 20% level and unlikely for YOY white sharks in the NEP for reasons discussed above. Genetic data are also not consistent with these adult female abundance estimates, because they would still be considered low numbers based on the observed level of genetic diversity in the population.

<u>B10=</u> At least 10 times the number of adult female white sharks exist in the NEP than are sampled at the sampling sites (N_f = 481 to 1570 depending on the sex ratio bias as the two sites).

Arguments for: Known levels of YOY catch would constitute an estimated 7% of the annual pup production possible from this number of adult females. This mortality rate is more consistent with levels of fisheries mortality for other species. These adult female abundance values are also more consistent with the observed genetic diversity levels.

Arguments against: These adult female abundance levels are not supported by the photo-ID data. Also, given the lack of data there are no measures of the NEP population's historical population size that would allow for the determination of the level of depletion experienced in the recent past. Despite the modeling results in Appendix B, it is possible that the population experienced a recent depletion, which would explain the high observed level of genetic diversity despite a small estimated population size.

SEDM Results:

After taking into account the modeling results, the BRT's considered the higher levels of bias (B2 and B10) to be even more plausible than previously (compare Table I.1 to Table G.2). The BRT assigned the majority of plausibility points (>90%) to bias categories B2 and B10, indicating that the team felt that there are likely more than 2 times the number of adult females in the NEP population than were indicated by the photo-ID studies at the sites. The shift in plausibility points to higher levels of bias were influenced by the model results. The catch rates and mortality rates for the lower levels of bias were considered to be implausible and inconsistent with evidence that the population of NEP white sharks is increasing.

Table I.2 Summary of SEDM voting results regarding the levels of bias in adult female abundance estimates associated with sampling only at the two sites in central California and one location at Guadalupe Island.

BRT Member	**Adult female abundance bias categories considered**			
	B1	B1.2	B2	B10
1	0	0	40	60
2	10	25	45	20
3	0	10	40	50
4	0	0	10	90
5	0	0	20	80
6	0	20	30	50
7	0	10	70	20
8	0	0	70	30
Avg	**1.3**	**8.1**	**40.6**	**50.0**
Stdev	**3.5**	**10.0**	**21.5**	**26.2**

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Appendix J: Alternative analyses for estimating the distribution of maximum population growth rate

The integrated matrix-allometric approach reduces the sensitivity of estimates to how uncertainty in the species' life history is specified as illustrated in the following analysis. Table J.1 shows an alternative set of prior vital rate distributions to those in Table 4.13 that provided a similar result. Contrasting features in this example are that there are only three age classes (no subadult class), age to maturity is fixed at 15 years, the survival rate distributions are slightly different, and a longevity constraint is applied so that no individuals survive to their 31st birthday. Eigenanalysis of Leslie matrices for the randomly drawn vital rates is used to calculate the λ_{max} distribution. Figure J.1A shows the difference in distributions for λ_{max} using conventional approaches and vital rates from Table J.1 vs. Table 4.13. The associated shrinkage estimates from the two sets of vital rate inputs are both more precise and more similarly distributed (Figure J.1).

Table J.1 Alternative set of white shark vital rate estimates or distributions for calculating a distribution of λ_{max}

Vital rate	Estimate or distribution
Age at first reproduction, α	15 fixed
Litter size	Uniform (8, 10)
Sex ratio at birth	1:1 fixed
Inter-birth interval	2.2 years fixed
Adult survival, S_{adult}	Uniform(0.90, 0.97)
Juvenile survival, S_{juv}	Uniform(0.88, S_a)
YOY survival, S_{yoy}	Uniform(0.45, 0.75)
Maximum age	30 fixed

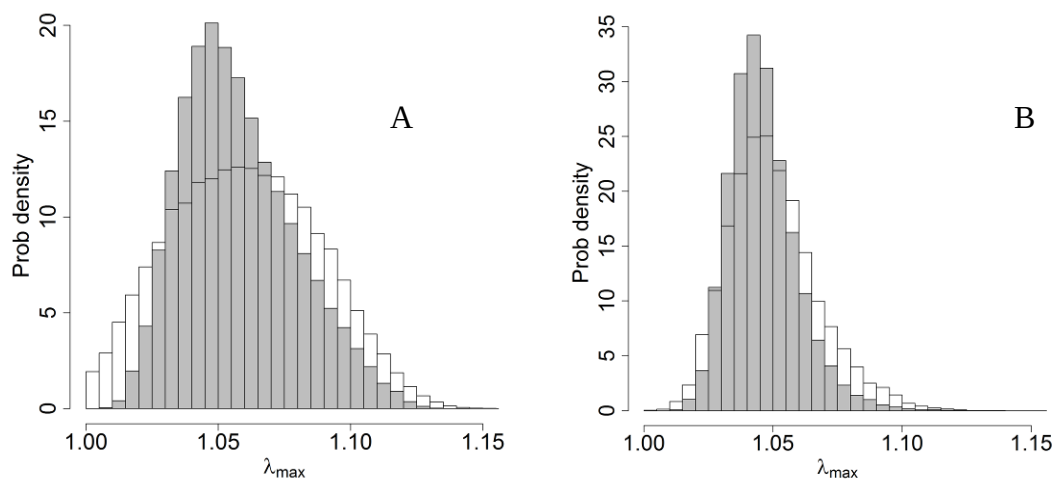

Figure J.1 Distributions for λ_{max} calculated from vital rate distributions in Table 4.13 (white bars) and Table J.1 (gray bars). In A, the estimates are derived from a conventional life table or Leslie matrix approach, respectively. In B, the shrinkage estimates reflect the influence of the allometric model.

Appendix K. Summary of BRT point allocation on overall extinction risk

Table K.1 Summary of all BRT plausibility points from SEDM assessing extinction risk, taking all factors into account.

	Extinction risk categories			
BRT Member	**High**	**Medium**	**Low**	**Very Low**
1	0	3	43	54
2	15	25	30	30
3	11	17	22	50
4	5	11	37	47
5	1	1	22	76
6	1	2	25	72
7	0	5	50	45
8	5	10	55	30
Avg	**4.75**	**9.25**	**35.5**	**50.5**
Stdev	**5.6**	**8.4**	**12.8**	**16.9**