Endangered Species Act Section 7(a)(2) Biological Opinion

Reinitiation of ESA Consultation on the Continued Operation of the Large Mesh Drift Gillnet Fishery under the U.S. West Coast Fishery Management Plan for Highly Migratory Species

NMFS Consultation Number: 2023-00435

Action Agency: NMFS West Coast Region (WCR) Sustainable Fisheries Division

Affected Species and NMFS' Determinations:

ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species?[1]	Is Action Likely To Jeopardize the Species?	Is Action Likely to Adversely Affect Critical Habitat?[1]	Is Action Likely To Destroy or Adversely Modify Critical Habitat?
Marine Mammals					
Blue whale (*Balaenoptera musculus*)[2]	Endangered	No	N.A.	N.A	N.A.
Fin whale (*Balaenoptera physalus*)	Endangered	Yes	No	N.A.	N.A.
Humpback whale; Mexico Distinct Population Segment (DPS) (*Megaptera novaeangliae*)	Threatened	Yes	No	No	N.A.
Humpback whale; Central America DPS	Endangered	Yes	No	No	N.A.
Sperm whale (*Physeter macrocephalus*)	Endangered	Yes	No	N.A.	N.A.
Sei whale (*Balaenoptera borealis*)[2]	Endangered	No	N.A.	N.A.	N.A.
Gray whale; Western North Pacific population (*Eschrichtius robustus*)[2]	Endangered	No	N.A.	N.A.	N.A.
Killer whale, Southern Resident DPS (*Orcinus orca*)	Endangered	No	N.A.	No	N.A.
North Pacific right whale (*Eubalaena japonica*)[2]	Endangered	No	N.A	N.A.	N.A.
Guadalupe fur seal (*Arctocephalus townsendi*)[2]	Threatened	No	N.A.	N.A.	N.A.
Steller sea lion; Eastern DPS (*Eumetopias jubatus*)	N.A.	N.A.	N.A.	No	N.A.

Sea Turtles					
Leatherback sea turtle (*Dermochelys coriacea*)	Endangered	Yes	No	No	N.A.
Loggerhead sea turtle; North Pacific Ocean DPS (*Caretta caretta*)[2]	Endangered	Yes	No	N.A.	N.A.
Olive ridley sea turtle (*Lepidochelys olivacea*)[2,3]	Endangered	Yes	No	N.A.	N.A.
Green sea turtle; East Pacific DPS (*Chelonia mydas*)[2]	Threatened	Yes	No	N.A.	N.A.
Marine Fish					
green sturgeon; Southern DPS (*Acipenser medirostris*)	Threatened	No	N.A.	No	N.A.
Pacific eulachon, Southern DPS (*Thaleichthys pacificus*)	Threatened	No	N.A.	No	N.A
Scalloped hammerhead shark; Eastern Pacific DPS (*Sphyrna lewini*)[2]	Endangered	No	N.A.	N.A.	N.A.
Gulf Grouper (*Mycteroperca jordani*)[2]	Endangered	No	N.A.	N.A.	N.A.
Giant manta ray (*Manta birostris*)[2]	Threatened	Yes	N.A.	N.A.	N.A.
Oceanic whitetip shark (*Carcharhinus longimanus*)[2]	Threatened	No	N.A.	N.A.	N.A.
Marine Invertebrates					
White abalone (*Haliotis sorenseni*)[2]	Endangered	No	N.A.	N.A.	N.A.
Black abalone (*Haliotis cracherodii*)	Endangered	No	N.A.	No	No
Salmonids					
Sacramento River winter-run Chinook (*Oncorhynchus tshawytscha*)	Endangered	No	N.A.	No	N.A.
Central Valley spring-run Chinook	Threatened	No	N.A.	No	N.A.
California Coastal Chinook	Threatened	No	N.A.	No	N.A.
Snake River fall Chinook	Threatened	No	N.A.	No	N.A.
Snake River spring/summer Chinook	Threatened	No	N.A.	No	N.A.
Lower Columbia River Chinook	Threatened	No	N.A.	No	N.A.
Upper Willamette River Chinook	Threatened	No	N.A.	No	N.A.
Upper Columbia River spring Chinook	Endangered	No	N.A.	No	N.A.
Puget Sound Chinook	Threatened	No	N.A.	No	N.A.
Hood Canal summer run chum (*Oncorhynchus keta*)	Threatened	No	N.A.	No	N.A.
Columbia River chum	Threatened	No	N.A.	No	N.A.

Central California Coast coho (*Oncorhynchus kistuch*)	Endangered	No	N.A.	No	N.A.
S. Oregon/N. California Coast coho	Threatened	No	N.A.	No	N.A.
Oregon Coast coho	Threatened	No	N.A.	No	N.A.
Lower Columbia River coho	Threatened	No	N.A.	No	N.A.
Snake River sockeye (*Oncorhynchus nerka*)	Endangered	No	N.A.	No	N.A.
Lake Ozette sockeye	Threatened	No	N.A.	No	N.A.
Southern California steelhead (*Oncorhynchus mykiss*)	Endangered	No	N.A.	No	N.A.
South-Central California Coast steelhead	Threatened	No	N.A.	No	N.A.
Central California Coast steelhead	Threatened	No	N.A.	No	N.A.
California Central Valley steelhead	Threatened	No	N.A.	No	N.A.
Northern California steelhead	Threatened	No	N.A.	No	N.A.
Upper Columbia River steelhead	Endangered	No	N.A.	No	N.A.
Snake River Basin steelhead	Threatened	No	N.A.	No	N.A.
Lower Columbia River steelhead	Threatened	No	N.A.	No	N.A.
Upper Willamette River steelhead	Threatened	No	N.A.	No	N.A.
Middle Columbia River steelhead	Threatened	No	N.A.	No	N.A.
Puget Sound steelhead	Threatened	No	N.A.	No	N.A.

[1] Please refer to Section 2.12 for the analysis of species or critical habitat that are not likely to be adversely affected
[2] Critical habitat has not been designated for these species along the U.S. West Coast.
[3] The breeding population from the coast of Mexico is listed as endangered; all other populations are globally listed as threatened.

Consultation Conducted By: National Marine Fisheries Service, West Coast Region, Protected Resources Division

Issued By: Chris E Yates

Christopher E. Yates
Assistant Regional Administrator for Protected Resources

Date: July 7, 2023

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List of Acronyms

AFSC	Alaska Fisheries Science Center
APPS	Authorizations and Permits for Protected Species
CA	California
CASWRB	California State Water Resources Control
CDFW	California Department of Fish and Wildlife
CHRT	Critical Habitat Review Team
CI	Confidence Interval
CITES	Convention on International Trade in Endangered Species
CMS	Conservation of Migratory Species of Wild Animals
CPPS	Permanent Commission of the South Pacific
CPUE	Catch-per-unit-effort
CV	Coefficient of Variation
dB	Decibels
DDE	Dichlorodiphenyldichloroethane
DGN	Drift Gillnet Fishery
DIP	Demographically Independent Population
DPS	Distinct Population Segment
DQA	Data Quality Act
DSBG	Deep-Set Buoy Gear
DSLBG	Deep-Set Linked Buoy Gear
DSLL	Deep-Set Longline Gear
DSSL	Deep-Set Shortline
DRIFTNET ACT	Driftnet Modernization and Bycatch Reduction Act
EA	Environmental Assessment
EC	Ecosystem Component
EEZ	Exclusive Economic Zone
EFP	Exempted Fishing Permit
EM	Electronic Monitoring
ENP	Eastern North Pacific
ESA	Endangered Species Act
ESU	Evolutionary Significant Unit
ETP	Eastern Tropical Pacific
FAO	Food and Agriculture Organization
fm	Fathoms
FMP	Fishery Management Plan
Fr	Recovery Factor
GT	Gross Tons
GTC	Grupo Tortuguero de las Californias
HMS	Highly Migratory Species
HMSMT	Highly Migratory Species Management Team
IAC	Inter-American Convention for the Protection and Conservation of Sea Turtles
IATTC	Inter-American Tropical Tuna Commission
IMO	International Maritime Organization

IPCC	Intergovernmental Panel on Climate Change
ITS	Incidental Take Statement
IUCN	International Union for the Conservation of Nature and Natural Resources
IWC	International Whaling Commission
kHz	Kilohertz
km^2	Square Kilometers
LE	Limited Entry
LRB	CDFW License and Revenue Branch
LL	Longline Gear
m	Meter
MMPA	Marine Mammal Protection Act
MSA	Magnuson-Stevens Fishery Conservation and Management Act
mtDNA	Mitochondrial DNA
NMFS	NOAA's National Marine Fisheries Service
nmi^2	Square Nautical Miles
Nmin	Minimum Population Estimate
N_{multi}	Multistrata Model
mt	Metric Tons
Opinion	Biological Opinion
OR	Oregon
OSP	Optimum Sustainable Population
PBR	Potential Biological Removal
PIT	Passive Integrated Transponder
PARS	Port Access Route Study
PBF	Physical or Biological Features
PCB	Polychlorinated Biphenyls
PCE	Primary Constituent Element
PDO	Pacific Decadal Oscillation
PFMC	Pacific Fisheries Management Council
PLCA	Pacific Leatherback Conservation Area
POCTRP	Pacific Offshore Cetacean Take Reduction Plan
POCTRT	Pacific Offshore Cetacean Take Reduction Team
PRD	Protected Resources Division
PSMFC	Pacific States Marine Fisheries Commission
PVA	Population Viability Analysis
PWSA	Port and Waterways Safety Act
RAMP	Risk Assessment Mitigation Program
RCP	Representative Concentration Pathway
RPM	Reasonable and Prudent Measures
SAFE	Stock Assessment and Fisheries Evaluation
SAR	Stock Assessment Reports
SCB	Southern California Bight
SDM	Species Distribution Models
SFD	Sustainable Fisheries Division
SRKW	Southern Resident Killer Whale

SSLL	Shallow-set longline gear
STAJ	Sea Turtle Association of Japan
SWFSC	Southwest Fisheries Science Center
TSS	Traffic Separation Schemes
UME	Unusual Mortality Event
USCG	U.S. Coast Guard
USFWS	U.S. Fish and Wildlife Service
VMS	Vessel Monitoring System
VSR	Vessel Speed Reduction
WA	Washington
WCPFC	Western and Central Pacific Fisheries Commission
WCR	West Coast Region
WNP	Western North Pacific
WWF	World Wildlife Fund
yr	Year

1. INTRODUCTION

This Introduction section provides information relevant to the other sections of this document and is incorporated by reference into Sections 2 and 3, below.

1.1 Background

NOAA Fisheries (NMFS) prepared the biological opinion (opinion) and incidental take statement (ITS) portions of this document in accordance with section 7(b) of the Endangered Species Act (ESA) of 1973 (16 U.S.C. 1531 et seq.), as amended, and implementing regulations at 50 CFR part 402.

Fisheries for highly migratory species (HMS) that occur in the U.S. exclusive economic zone (EEZ) and adjacent waters off the coasts of Washington (WA), Oregon (OR), and California (CA) are authorized by NMFS under the Magnuson-Stevens Fishery Conservation and Management Act (MSA) and must comply with other applicable federal statutes such as the ESA and Marine Mammal Protection Act (MMPA). Specifically, these fisheries are managed under the HMS Fishery Management Plan (FMP; PFMC 2016) and implementing regulations. Consistent with the HMS FMP, any necessary conservation and management measures are adopted on a bi-annual basis through the Pacific Fisheries Management Council (PFMC or Council). These measures are implemented through the federal rulemaking process and sent to the U.S. Secretary of Commerce, through NMFS, for review and approval. Additional measures can be adopted on an emergency basis outside of the Council's bi-annual process.

Information on distribution, life history, stock structure, stock status, and catch of FMP management unit species is summarized in several existing documents including environmental assessment documents (EAs) for Amendments to the HMS FMP the Council's HMS Stock Assessment and Fisheries Evaluation (SAFE) documents[1], and previous ESA biological opinions on HMS FMP fisheries. There were 13 management unit species codified in federal regulations under the original 2004 HMS FMP. NMFS published a final rule implementing Amendment 2 to the HMS FMP on September 13, 2011 (76 FR 56327), reducing the number of management unit species to 11, as follows:

1. Albacore tuna, *Thunnus alalunga*
2. Bigeye tuna, *T. obesus*
3. Skipjack tuna, *Katsuwonus pelamis*
4. Bluefin tuna, *T. orientalis*
5. Yellowfin tuna, *T. albacares*
6. Striped marlin, *Tetrapturus audax*
7. Swordfish, *Xiphias gladius*
8. Blue shark, *Prionace glauca*
9. Common thresher shark, *Alopias vulpinus*
10. Shortfin mako shark, *Isurus oxyrinchus*

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[1] http://www.pcouncil.org/highly-migratory-species/stock-assessment-and-fishery-evaluation-safe-documents/

11. Dolphinfish, *Coryphaena hippurus*

The large mesh drift gillnet fishery (DGN) fishery for swordfish and thresher shark is one of six authorized gear types/fisheries under the HMS FMP. The other fisheries include: the albacore fishery using surface hook-and-line gear by trolling and pole-and-line fishing techniques; the deep-set longline fishery based in California targeting tuna in waters beyond the EEZ; the tropical tuna fisheries using purse seine, including the coastal purse seine fishery (small vessels) that concentrates on small pelagic species but which also harvests northern bluefin and yellowfin tuna when they migrate into the Pacific EEZ; the swordfish harpoon fishery; and the charter boat HMS sport fisheries. On May 8, 2023, NMFS published a final rule to implement Amendment 6 to the HMS FMP, which authorizes deep-set buoy gear (DSBG) as a legal gear type for targeting swordfish and catching other HMS off the U.S. West Coast.

The 2004 HMS biological opinion and supplemental 2016 biological opinion on the deep-set longline fishery remain intact and in force relative to all of these other HMS fisheries besides the DGN fishery, which is the subject of this consultation and biological opinion.

Prior to completion of the last biological opinion on the DGN fishery in 2013, NMFS hosted a workshop in 2011 to address declining swordfish production by the West Coast-based vessels. At that time, attrition (defined as permanent exit from participation) in the DGN fishery over the prior 10-year period was estimated at 60 percent for West Coast ports overall. Between 2001 and 2011, the number of DGN vessels making landings to the West Coast fell from 60 to 21, with an average of 27 from 2009-2011. Participants in the workshop considered a likely future in which attrition would continue without change in the management of the fishery such as to increase access to fishing grounds or to modify fishing gear to create new fishing opportunities, or both (see Rhodes et al. 2012 workshop proceedings). In 2011, research trials began to test DSBG as an additional gear type in the swordfish fishery.

In 2014, the Council solicited applications for exempted fishing permits (EFPs) to test modified DGN gear or alternative gear types. In 2015, the Council recommended NMFS approve several applications to test a range of gear alternative gear types. In 2015, an initial set of EFPs were issued to commercial swordfish fishermen to begin gear trials of alternative methods. This practice of issuing EFPs to commercial fishermen has continued, with an increasing number of DGN permit holders seeking EFPs. The average annual swordfish landings attributed to DGN gear for the period of 1996 to 2014 was 356 round metric tons (mt), and 102 mt for the period of 2015 to 2021. This decline in annual landings may, in part, be a factor of DGN fishermen moving some effort to other gear types. However, other management measures affecting the DGN fishery are also likely to have contributed to this decline in DGN landings of swordfish.

Since the 2013 biological opinion on the DGN fishery was completed, there have been 5 amendments to the HMS FMP that have been adopted by the Council and approved by NMFS. Amendment 3 added a suite of lower trophic level species to the FMP's list of ecosystem component (EC) species and prohibited future development of directed commercial fisheries for the suite of EC species shared between all four FMPs managed by the Council ("Shared EC Species"). Amendment 4 is an administrative amendment designed to clarify the Council's role in the process of making stock status determinations and change the schedule of the Council's three-meeting biennial management cycle for HMS stocks. Amendment 5 establishes a federal limited entry (LE) permit system for the DGN fishery using very similar standards as the existing State of California LE permit program for the DGN fishery that is intended to streamline management and future decision-making for this fishery by assuming federal control over the permit system. Amendment 5 adopted many of the State of California permit requirements as they relate to the DGN fishery, but all State of California permit requirements on the DGN fishery remain intact. Full implementation of these new amendments is still ongoing, but at this time none of these recent amendments are expected to impact fishing effort or change current fishery practices within the DGN fishery. Amendment 7, numbered out of sequence due to the chronology of Council recommendations, described the Standardized Bycatch Reporting Methodology of the HMS FMP.

On May 8, 2023, following the Council's recommendation, NMFS published a final rule (88 FR 29545) implementing Amendment 6 to HMS FMP, which authorizes DSBG as a legal gear type for targeting swordfish and catching other HMS in federal waters off of California and Oregon. The rule establishes a limited entry permit system for fishing DSBG within federal waters of the Southern California Bight (SCB) and an open access permit system for fishing the gear in federal waters outside of the SCB. The rule includes definitions for two configurations of DSBG, standard and linked, and specifies limited entry permitting procedures and requirements for use of the gear. Permitting procedures include requirements for DSBG permit possession, renewal, eligibility, and transferal requirements, and procedures related to ranking of limited entry DSBG applicants and issuance of permits to applicants. Specifications on the gear include standards for the buoy array of both standard and linked DSBG configurations, weights, hook size, and the number of individual pieces of gear used. Management measures include regulations on active tending, gear deployment and retrieval timing, use of multiple gears on a single trip, species retention, and fishery monitoring. Additional regulations include requirements for pre-trip notifications, protected species workshops, and a prohibition on linked DSBG operations shoreward of a line approximating the 400 meter depth contour. The Amendment would allow fishermen to fish with DSBG on the same trips as DGN, and to extend the fishing opportunity for swordfish and other HMS species into summer months. The Council recommended these measures to improve the economic viability of the West Coast swordfish fishery, while minimizing bycatch and protected species interactions.

In addition to amendments to the HMS FMP, new regulatory actions have been taken in the DGN fishery prompted by State of California law as well as recommendations from the Council. Since the 2012-2013 ESA consultation, NMFS has proposed regulations implementing protected species hard caps for the DGN fishery, attempted to withdraw the proposed regulations, promulgated final regulations, and had the final regulations vacated by judicial order. The Council took final action in September 2015 on a proposal that included rolling 2-year hard caps on interactions with fin whales, humpback whales, sperm whales, short-fin pilot whales, bottlenose dolphins, and four sea turtle species (loggerhead, leatherback, olive ridley, and green sea turtles). NMFS drafted and published proposed regulations based on the Council's final preferred alternative; however, NMFS later withdrew the proposed regulations based on an economic analysis indicating that the hard caps regulations violated National Standard 7 in the

MSA, which states that "conservation and management measures shall, where practicable, minimize costs and avoid unnecessary duplication." Subsequent litigation forced NMFS to publish the regulations in February 2020 (85 FR 7246, Feb. 7, 2020). Further litigation resulted in an opinion (*Burke, et al. v. Coggins,* 521 F. Supp. 3d 31 (D.D.C. 2021)) and order from the Federal District Court for the District of Columbia that vacated the regulations on February 18, 2021.

The Council then revisited DGN hard caps as part of its 2022 meeting agendas. In June 2022, the Council's Highly Migratory Species Management Team (HMSMT) was tasked with completing an economic analysis for a new range of alternatives for DGN hard caps. An analysis was available during the November 2022 Council meeting; however, the Council decided at that time to revise its range of alternatives. Following that Council meeting, NMFS staff revised the analysis to consider the new alternatives and evaluate the results in terms of tail-conditional expectations as recommended by the Council's Science and Statistical Committee. Use of tail-conditional expectations as a metric as opposed to the mean (which was used in the November 2022 analysis) to evaluate results of managing rare event bycatch is potentially more appropriate for characterizing the risk of such extreme outcomes. That analysis was made available to the Council for their March 2023 meeting. At its March 2023 meeting, in light of passage of the Driftnet Modernization and Bycatch Reduction Act (Driftnet Act, described below), which amends the MSA to prohibit the use of large mesh DGN gear by December 2027, the Council decided not to proceed with further action to implement hard caps. The Council recognized that hard caps would likely not provide any benefit in terms of reducing high priority protected species mortality/injury over the few seasons they would be in force. In addition, the Council decided its time would be better spent consulting with NMFS on the fishery transition program mandated by the Driftnet Act.

In addition to the federal actions described, in 2018, Senate Bill (SB) 1017 became law in the State of California. Regulations to implement the legislation establish a transition program for the DGN fishery by providing funding to reimburse fishermen who surrender their federal DGN permits and DGN gear. Transition program participants must also surrender their state DGN permit, affirm their net has been destroyed at an accredited facility, and affirm they will not fish under, transfer, or renew their federal DGN permit. Any remaining state permits will be revoked on January 31, 2024. Any federal DGN limited (LE) permit holder that does not participate will have their State of California LE DGN permit revoked in 2024, but would not be prevented by this State legislation from continuing to renew their federal DGN LE permit. Ultimately, this program has driven a significant decline in effort in the DGN fishery. Recent data indicate that as few as seven vessels actively fished DGN during the 2021/2022 fishing season. The deadline to participate in California's program occurred in October 2022. Thus, it is expected that a number of federal LE DGN permits issued for the 2022/2023 fishing year are unlikely to be renewed for the 2023/2024 fishing season. NMFS typically completes these renewals by July of the calendar year. However, in the interim, the California Department of Fish and Wildlife (CDFW) has provided updates to NMFS regarding federal LE DGN permit holders' completion of the state program. For the 2022/2023 fishing season, there were 31 federal LE DGN permit holders, and a 24 of these permit holders renewed their permits for 2023/2024.

In addition, on December 29, 2022, President Joseph Biden signed the Driftnet Act. This Act amends the MSA to add mesh size of 14 inches or greater to the definition of large-scale driftnet fishing at MSA §3(25), which is prohibited (§307(1)(M)), but includes an exception from the prohibition applicable to use of DGN gear within five years of enactment. The Act directs NMFS to "consult with the Pacific Fishery Management Council on a strategy to phase out the use of large mesh driftnets and permit the use of alternative fishing methods to increase the economic viability of the West Coast-based swordfish fishery while minimizing bycatch to the maximum extent possible." During the five years from enactment to the December 2027 prohibition of the use of DGN gear, NMFS must conduct a transition program (§206(i)) to phase out use of the gear and compensate fishery participants for the cost of fishery-related permits, gear forfeiture, and purchase of alternative gear.

Passage of the Driftnet Act substantially changes the context for NMFS' proposed action and should be considered in the analysis of consequences to listed species. The Act sunsets the fishery by December 2027[2], after which there will be no additional sets of large mesh DGN gear in federal waters. Further, the Act changes potential incentives for permit holders within the five-year period before the fishery sunsets, such as to: (1) increase the incentive for DGN permit holders to fish DGN gear before use of the gear is prohibited, or (2) increase the incentive for DGN permit holders to transition their fishing interests to alternative gears before the use of DGN gear is prohibited, or both.

We completed pre-dissemination review of this document using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (DQA) (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). The document will be available within 2 weeks at the NOAA Library Institutional Repository [https://repository.library.noaa.gov/welcome]. A complete record of this consultation is on file at NMFS Long Beach Branch.

1.2 Consultation History

There have been several ESA consultations relevant to operation of the DGN fishery completed over the last two decades that are summarized in brief below.

NMFS issued a biological opinion on September 30, 1997, evaluating the effect of the final regulations to implement the Pacific Offshore Cetacean Take Reduction Plan (POCTRP) for the DGN fishery on listed sea turtle and marine mammal populations (NMFS 1997). This opinion concluded that establishing a minimum extender length requirement of 6 fathoms (fm) (36 feet), conducting skipper workshops, and using pingers on the nets would most likely reduce the incidental catch of listed marine mammals and sea turtles. Based on analyses of the final regulations, we concluded that the continued operation of the DGN fishery under the POCTRP was not likely to jeopardize the continued existence of the humpback and sperm whales, or

[2] While the sunset date of the federal DGN fishery is less than five years from the date of this Opinion, we use five years throughout the opinion based on projections developed in early 2023 for the five seasons of fishing that will occur from this point through the end of 2027.

leatherback and loggerhead sea turtles.

On October 23, 2000, we issued a new biological opinion based upon review of bycatch in the DGN fishery and authorization to take listed marine mammals incidental to commercial fishing operations under section 101(a)(5)(E) of the MMPA. That biological opinion concluded the DGN was not likely to jeopardize the listed marine mammals or green and olive ridley sea turtles. However, that biological opinion concluded that the issuance of the MMPA authorization for the DGN fishery was likely to jeopardize leatherback and loggerhead sea turtles, and subsequent regulations were issued to reduce the likelihood of interactions between these sea turtle species and the DGN fishery.[3]

On February 4, 2004, we issued the biological opinion on adoption of the HMS FMP, which included management of the DGN fishery. That biological opinion concluded that operation of the DGN fishery under the HMS FMP was not likely to jeopardize the continued existence of fin, humpback and sperm whales, or leatherback, loggerhead, green, and olive ridley sea turtles.

On May 2, 2013, we issued a new biological opinion on the continued operation of the DGN fishery under the HMS FMP using updated information on bycatch rates as well as status of ESA-listed species. That biological opinion concluded that operation of the DGN fishery was not likely to jeopardize the continued existence of fin, humpback and sperm whales, or leatherback, loggerhead, green, and olive ridley sea turtles.

On September 8, 2016, NMFS revised the listing of humpback whales from a single global listing to fourteen Distinct Population Segments (DPSs) of humpback whales, each having their own status under the ESA (81 FR 62260). Specifically, the humpback whales found off the contiguous west coast of the United States are divided into three DPSs: the Hawaiian DPS, not warranted for listing under the ESA; the Mexico DPS, listed as threatened under the ESA; and the Central America DPS, listed as endangered under the ESA. In addition, since the 2013 biological opinion on the DGN fishery, reports of entanglements of whales, especially humpback whales, notably increased off the west coast of the United States. Many of these entanglements have been in fishing line and in numerous cases have been identified to fixed gear fisheries like Dungeness crab pot gear. There have also been entanglements in gear identified as gillnet, although none has been positively identified as belonging to the DGN fishery authorized under the HMS FMP.

On November 20, 2017, NMFS WCR Protected Resources Division (PRD) transmitted a memo to WCR Sustainable Fisheries Division (SFD) that concluded reinitiation of the 2013 biological opinion on the continued operation of the DGN fishery was appropriate due to the new species listings of humpback whales as separated into two DPSs, and evaluation of information on the elevated numbers of reported entanglements of humpback whales in recent years (NMFS 2017a). To aid in this consultation, PRD requested certain information from SFD necessary to complete an updated analysis of the impact of the DGN fishery on ESA-listed species and issue a new biological opinion. Beginning in late November, 2017, SFD and PRD regularly exchanged

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[3] Pacific Sea Turtle Conservation Areas (50 CFR 660.713) described in section 1.3.2.

information regarding clarifications of the proposed actions, available information regarding bycatch data and bycatch estimation, and analysis of observer records. On February 21, 2018, staff from SFD and PRD met to discuss plans for analysis of fishing effort and observer coverage in the DGN fishery using up-to-date vessel monitoring system (VMS) data in conjunction with a broader data integration initiative coordinated by Pacific States Marine Fisheries Commission (PSMFC) staff and NMFS. On April 16, 2018, the ESA consultation initiation was withdrawn by NMFS PRD in order to continue working on the integrated VMS analysis.

Throughout the remainder of 2018 and 2019, analytical efforts to evaluate the representative nature of observer coverage in recent years were led by PSMFC, with periodic review and revision that included input from NMFS PRD and SFD staff (see Section 1.3.2 for further detail on recent observer program analyses). In September, 2019, NMFS PRD exchanged requests with SFD for final versions of this analysis, along with updated information about recent DGN effort and developments within the DGN fishery, which would be needed to resume and complete ESA consultation. Work on the integrated VMS analysis continued until the report was presented to the PFMC at the June 2021 meeting (Suter et al. 2021).

In 2021, two humpback whale entanglements in the DGN fishery were reported by fishery observers (NMFS 2022a). In February, 2022, NMFS began to coordinate with the SWFSC on the timeline to generate updated bycatch estimates in the DGN fishery that would include the most recent humpback whale interactions. As described above, in 2022 the Council continued to deliberate on potential actions surrounding implementation of hard caps in the DGN fishery. Starting in June of 2022, NMFS PRD and SFD began to coordinate on getting the ESA consultation reinitiated including all of the most recent information and developments. In October, 2022, updated bycatch estimates for all ESA-listed species bycatch in the DGN fishery were provided (Carretta 2022). As described above, the Driftnet Act was passed in December 2022.

Throughout the beginning of 2023, NMFS PRD and SFD continued to coordinate on generating a final proposed action in support of initiating ESA consultation. Following the transmittal of a final consultation request along with additional information from SFD to PRD on March 30, 2023 (NMFS 2023a), the ESA consultation was reinitiated.

In a final rule effective on September 26, 2019, the U.S. Fish and Wildlife Service and NMFS jointly revised portions of our regulations that implement section 7 of the ESA. On July 5, 2022, the U.S. District Court for the Northern District of California issued an order vacating the 2019 regulations that were revised or added to 50 CFR part 402 in 2019 ("2019 Regulations," see 84 FR 44976, August 27, 2019) without making a finding on the merits. On September 21, 2022, the U.S. Court of Appeals for the Ninth Circuit granted a temporary stay of the district court's July 5 order. On November 14, 2022, the Northern District of California issued an order granting the government's request for voluntary remand without vacating the 2019 regulations. The District Court issued a slightly amended order two days later on November 16, 2022. As a result, the 2019 regulations remain in effect, and we are applying the 2019 regulations here. For purposes of this consultation and in an abundance of caution, we considered whether the substantive analysis and conclusions articulated in the biological opinion and incidental take statement would be any different under the pre-2019 regulations. We have determined that our analysis and conclusions would not be any different.

1.3 Proposed Federal Action

Under the ESA, "action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies (see 50 CFR 402.02).

Management of the Large Mesh Drift Gillnet Fishery under the U.S. West Coast Fishery Management Plan for Highly Migratory Species

California's swordfish fishery began by using harpoon gear to target swordfish basking at the surface, but in the early 1980s, some fishermen began using DGN. Shortly after this time period, landings reached a historical annual high of 3,000 mt of swordfish in 1985 (PFMC 2011). The use of drift gillnet gear to target or capture thresher shark also began to occur during this time. Fishing activity is highly dependent on seasonal oceanographic conditions that create temperature fronts which concentrate prey species for swordfish. Historically, the California DGN fleet operated within U.S. EEZ[4] waters adjacent to the state and as far north as the Columbia River during El Niño years. Starting in 2001 for leatherback turtles, and 2003 for loggerhead turtles, NMFS created areas off the U.S. West Coast that are closed to DGN fishing seasonally or conditionally to protect endangered sea turtles, referred to as Pacific Sea Turtle Conservation Areas, which are described below in section 1.3.2.

The DGN fleet off the U.S. West Coast catches and retains various marketable species caught incidentally, including opah and tuna species (yellowfin and Bluefin tuna). DGN has a long history of catching these marketable species that contributes to drift gillnet revenues (see Pacific Fisheries Information Network [PacFIN] Apex Reports Highly Migratory Species SAFE Reports 003 and 009 at *https://pacfin.psmfc.org/*). These species have become more abundant in U.S. waters in recent years and, therefore, are more likely to be caught during the ongoing prosecution of the DGN fishery.

1.3.1 Vessels and Gear of the DGN Fishery

Drift Gillnet Gear

A drift gillnet is a panel of netting suspended vertically in the water by floats, with weights along the bottom. It is usually used to target swordfish and common thresher shark along the West Coast EEZ. Fish are entangled in the net, which uses relatively large mesh (typically around 18-20 inches measured diagonally knot-to-knot) to target relatively large species of pelagic fish in an effort to minimize the bycatch of smaller unwanted or prohibited species. The number of meshes hanging between the floatline and leadline (bottom of the net) ranges from 100 to 150. The lines that attach the buoys to the floatline that dictate the depth the net is fished are referred

[4] The U.S. Exclusive Economic Zone was established by Presidential Proclamation 5030 on March 10, 1983. It extends from the outer boundary of state waters at three nautical miles (nmi) to a distance of 200 nmi from shore.

to as buoy lines or extenders. DGN gear is anchored to a vessel at one end such that the net and vessel drifts together along with the current. Nets are often set perpendicular to currents, or across temperature, salinity, or turbidity fronts. Nets are typically set in the evening, allowed to soak overnight, and then retrieved in the morning. The average soak time is around 10 hours. See DGN Fishery Regulations section below for more specific gear requirements in this fishery.

Fishing Season

The DGN fishing season runs from May 1 to January 31 of the following year. However, nearly all of the fishing effort occurs from October to January 31 of the following year due to the seasonal migratory pattern of swordfish and other HMS species, in combination with other current seasonal fishing restrictions.

Participation and Permits

The HMS FMP requires a federal HMS permit[5] for all U.S. commercial fishing vessels (including recreational charter vessels) that fish for HMS management unit species within the West Coast EEZ, and for U.S. vessels that pursue HMS management unit species on the high seas (seaward of the EEZ). In order to participate in the DGN fishery specifically, a California-issued DGN permit is also required.[6] This permit is linked to an individual fisherman, not a vessel, and is only transferable under very restrictive conditions.

To keep a California-issued DGN permit active, current permit holders are required to purchase a permit each year, although permit holders are not required to make landings using DGN gear. In addition, a general California resident or non-resident commercial fishing license and a current vessel registration are required to catch and land fish in California caught with DGN gear. Completion and submission of federal logbook records is also required. As mentioned above, Amendment 5 establishes an additional federal DGN permit that mirrors the State of California DGN permit.

About 150 DGN permits were initially issued when the limited entry program was established by the state of California in 1980. The number of permits has declined from a high of 251 in 1986 to a low of 42 issued in 2021. In recent years, far fewer vessels have been active with a low of only 7 vessels actively participating in the DGN fishery in 2021 (Table 1). Additionally, the proportion of vessels active in the fishery relative to permits issued in any given year has declined from a high of 75-100 percent before 1996 to between 70-50 percent between 1996 and 2001. In 2002, the proportion of active vessels relative to permits issued fell to below 50 percent and continued to decline in the following years to below 30 percent by 2011, and under 20 percent by 2021. In recent years, far fewer vessels have been active, with only seven actively fishing in 2021. Surprisingly, some of the vessels that fished in 2021 had not fished in recent fishing seasons. Annual fishing effort (number of sets as reported by NMFS) has also decreased

[5] Federal permits must be renewed every 2 years, but permits are not currently limited and permit applications may be submitted at any time.
[6] States of Oregon and Washington do not permit landing of fish caught with DGN gear.

from a high of 11,243 sets in the 1986 fishing season to a low of 195 in 2021 (Table 1 and 2).

Table 1. Annual drift gillnet permits issued and number of active vessels, 1980–2021.

Year	Active Vessels	Permits Issued	Year	Active Vessels	Permits Issued
1980	100	*	**2001**[1]	61	114
1981	118	*	**2002**	52	106
1982	166	*	**2003**	44	100
1983	193	*	**2004**	36	96
1984	214	226	**2005**	38	90
1985	228	229	**2006**	39	88
1986	204	251	**2007**	40	86
1987	185	218	**2008**	39	85
1988	154	207	**2009**	35	84
1989	144	189	**2010**	26	73
1990	141	183	**2011**	22	76
1991	121	165	**2012**	17	78
1992	120	149	**2013**	18	74
1993	124	117	**2014**	21	74
1994	129	162	**2015**	19	73
1995	118	185	**2016**	21	70
1996	112	167	**2017**	18	67
1997	109	120	**2018**	19	62
1998	99	148	**2019**	14	60
1999	86	136	**2020**	12	58
2000	72	127	**2021**	7	42

Source: CDFW License and Revenue Branch (LRB) 2017, and NMFS Permits & Monitoring Branch 2022.
* actual number of permits issued by LRB not available but the California State Legislature set a cap of 150 in 1982.
[1] Year of implementation of Pacific Leatherback Closure Area regulations.

In the 2013 biological opinion, NMFS estimated annual DGN fishing effort to be a maximum of 1,500 sets per season. Although DGN fishing effort had declined to under 500 sets per year by that time, this estimate was based on the logic that fishing effort could again reach 1,500 sets per year if currently inactive DGN permit holders re-entered the fishery. However, in the decade since this consultation, the opposite trend has prevailed. Since the State of California's gear buyback and transition program went into effect in 2018, the fishery has undergone additional decreases in effort, falling from 618 sets in 2017 to 321 in 2019, and fewer than 200 sets per year in both 2020 and 2021 (see Table 2 for sets per year). This corresponds with decreases in the number of active vessels. That is, the number of sets per year per active vessel has been relatively consistent through time, with an average of 29 sets per year per active vessel (see Table 1 for active vessels per year) for 1990 through 2021, with a standard deviation of 6.8.

In 2022, 26 individual federal LE DGN permits were issued to permit holders that registered vessels to their permits. Of those, CDFW has reported six permit holders to have since completed the state buyback program. Of the remaining 20 permit holders, eight have submitted applications to fish alternative gear types under EFPs. That leaves 12 remaining federal LE DGN permit holders that have registered vessels to their permits and have yet to apply to fish with other gear types in federal waters.

The Driftnet Act makes clear that there will be no future effort in a DGN fishery within five years from the time of this request for consultation. However, changing management and incentives in the DGN fishery make it challenging to predict future effort in the fishery within that five-year period. As a result, NMFS considers a range of possibilities for fishing effort over the next five years, as described in the request for ESA consultation (NMFS 2023a).

At the high end of the range, it is possible that the 12 federal LE permit holders that have registered vessels to their permits and have yet to apply or receive permits to fish with other gear types may be incentivized by the fishery's sunset in the Driftnet Act to fish under their permits while they still can. As indicated in the Background Section, some vessels that have been active in recent fishing seasons had not been active in the fishery for several prior fishing seasons. It is unclear what might have spurred this interest in otherwise latent permits to become active, especially given low swordfish availability in recent years. Because the DGN fishery has always been a mixed-use fishery that harvests other marketable HMS catch, it is possible that increased abundance of Pacific bluefin in areas where the fishery operates has spurred some additional interest in recent years (88 FR 18256).

Thus, it is worthwhile to consider the possibility that the looming deadline on the state buyback and transition program may have played a role, and that it is possible that the imminent sunset on the DGN fishery in the Driftnet Act may spur inactive vessels to become active. If we assume that the 12 federal LE permit holders that have registered vessels to their permits fish 29 sets per year (the average per active vessel from 1990-2021), then it follows that we may assume they will fish 344 sets per year for the next five years, for a total of 1,719 sets in the next five year period. If we assume that the other eight federal LE DGN permit holders that have applied to fish with other gear types would feel more inclined to invest and engage in experimentation of alternative gear with passage of the Driftnet Act, then it follows that we could assume their fishing effort with DGN would decrease over the five-year period. For example, in the first year of the federal transition period, these eight vessels may continue to fish in the drift gillnet fishery at an average rate of 29 sets per year and until they are permitted for and having some success with alternative gear in later years leading to sequentially higher DGN effort reductions over the next four calendar years. If we assume that this results in roughly 25, 50, 75 and 90 percent effort reductions over the four year periods, we expect these 8 vessels to make a total of 596 sets over the next five year period (i.e., 229 sets in year one, 172 in year two, 115 in year three, 57 in year four, and 23 in year five). Thus, at the high end of the range of DGN effort projections, we estimate a total of 2,314 sets would be made over the next five year period (NMFS 2023a).

At the low end of the range, it is possible that future DGN fishing effort could be limited to vessels registered to federal LE DGN permits that have been active in the fishery in recent years or that made a recent vessel transfer on their permit. There are 11 vessels that meet these conditions. However, CDFW has informed NMFS that two of those permit holders recently completed their buyback program. Of the remaining nine active vessels, six have applied for permits to fish with alternative gear types. Using the same calculations as above for the high estimate of future fishing effort, we estimates of sets over next five years based on two vessels making 29 sets per year for five years (up to 290 sets total) and six vessels making 29 sets in the first year, and fewer in the subsequent years (i.e., 172 sets in year one, 123 in year two, 86 in year three, 43 in year four, and 17 in year five; up to 441 sets total). Thus, at the low end of the range of DGN effort projections, we estimate a total of 731 sets would be made over the next five year period. Of course, the more economically viable an alternative gear proves to be, the more likely it is that effort will shift away from DGN sooner.

While it difficult to forecast what will happen over the next five years, we anticipate that effort will be somewhere between the high and low end of the range that has been considered. In any one year, we anticipate that effort could be as much as the highest annual effort that was considered, which is 573 sets. However, over the entire five year period, we anticipate that annual effort will average between the high and low end of the range considered, which is up 305 sets, or a total of up to 1522 sets over the next 5 years.

Table 2. Annual drift gillnet fishing effort and catch of swordfish and thresher shark 1990–2021.

Year	**Number of Sets**	**Swordfish (mt)**	**Thresher Shark (mt)**
1990	4,078	1,133	163
1991	4,778	945	379
1992	4,379	1,407	92
1993	5,442	1,413	210
1994	4,248	762	203
1995	3,673	773	144
1996	3,392	764	168
1997	3,093	704	218
1998	3,353	877	238
1999	2,634	594	90
2000	1,936	548	85
2001	1,665	270	211
2002	1,630	301	110
2003	1,467	217	188
2004	1,084	182	53
2005	1,075	220	125
2006	1,433	444	93
2007	1,241	490	144
2008	1,103	406	98

Source: PFMC 2022.

1.3.2 DGN Fishery Observer Program

NMFS West Coast Region has operated an at-sea observer program in the DGN fishery since July 1990 to the present, while CDFW had operated a DGN observer program from 1980–90. The objectives of the NMFS Observer Program are to record, among other things, information on non-target fish species and protected species interactions that may not be typically nor accurately reported in fishing logbooks, due to focus on target species by fishermen or incentives not to report certain species to avoid increased regulation. These observer data are relied upon to produce estimates of protected species bycatch and forecast potential impacts of future fishing effort on these species. Since 1990, NMFS has sought to obtain a minimum 20 percent observer coverage of the DGN fishery each year, per recommendations from the Southwest Fisheries Science Center (SWFSC; NMFS 1989). Table 3 provides the recent history of observer coverage based on the calendar year fishing effort (although the fishing season typically runs August through January). In response to concern over some relatively low coverage rates from 2008-2010, NMFS SFD worked with the contracted observer provider to address challenges in observer deployment and achieve target coverage rates (NMFS 2013). In addition, in response to concern over sperm whale bycatch in certain portions of the DGN fishery in 2013, NMFS began targeting 30% overall observer coverage and instituted a temporary 100 percent observer coverage zone in waters deeper than 2,000 meters for the DGN fishery. The 100 percent observer coverage zone expired in August 2014. Following the expiration of the temporary observer coverage requirements, NMFS returned to targeting 20-30 percent overall observer coverage of the entire DGN fishery.

Overall, the annual average of observer coverage that occurred from 2000-2021 was 20.5 percent and total observer coverage of all effort during this time was 20.1 percent, which is essentially in line with the overall historical 20 percent coverage goal. During 2013 and 2014, the 30 percent coverage goal for the DGN was also met.

Table 3. Summary of observer coverage by DGN Observer Program from 2000-2021 (calendar fishing year, January to December).

Year	Total sets	Observed sets	% Observer coverage
2000	1936	444	22.9
2001	1665	339	20.4
2002	1630	360	22.1
2003	1467	298	20.3
2004	1084	223	20.6
2005	1075	225	20.9
2006	1433	266	18.6
2007	1241	204	16.4
2008	1103	149	13.5
2009	761	101	13.3
2010	492	59	12.0
2011	435	85	19.5
2012	445	83	18.7
2013	470	176	37.4
2014	379	113	29.8
2015	361	74	20.5
2016	737	134	18.2
2017	618	114	18.4
2018	473	124	26.2
2019	321	86	26.8
2020	147	22	15.0
2021	195	38	19.5

In the 2013 biological opinion, NMFS was required to undertake an evaluation of observer coverage targets and feasibility, along with other measures that may help improve the reliability of bycatch estimates produced from observer data (see *Effects of the Action* Section 2.4.1 for more discussion of bycatch estimates). In 2015, the Council developed alternatives for increased monitoring and recommended that NMFS implement 100 percent monitoring of the DGN fishery, using onboard observers or electronic monitoring (EM). At the March 2018 Council meeting, NMFS submitted a report (March 2018, I.1.a Supp. NMFS Report 3, reposted as June 2018 G.7, Attachment 3) evaluating the potential impacts of increased monitoring alternatives ranging from 50-100 percent coverage, using observers or EM or both. This preliminary analysis indicated that the economic impacts to the fleet (assuming that the fleet would be responsible for funding increased monitoring) may be prohibitive to implementing the alternatives evaluated. In the meantime, NMFS SFD is proposing to maintain a target coverage level of 20-30% percent, primarily due to limitations in current Observer Program funding. Observer coverage relative to the overall distribution of the DGN fishery is illustrated in Figure 5 below.

As a result, of the 2013 biological opinion, NMFS was also required to undertake steps to address questions and concern surrounding unobservable vessels, including implementation of a VMS program to improve monitoring of the DGN fishery. In response, NMFS instituted regulations in 2015 requiring all DGN vessel owners/operators to continuously operate and maintain a VMS unit in good working order 24 hours a day throughout the fishing year (80 FR 10392). The VMS unit must accurately transmit a signal indicating the vessel's position at least once every hour, 24 hours a day throughout the year, unless a valid exemption report has been confirmed by the NMFS Office of Law Enforcement. The final rule implementing this requirement became effective on June 9, 2015 (80 FR 32465).

In 2021, the PSMFC published a report analyzing and comparing the characteristics of observed and unobserved DGN fishing trips (Suter et al. 2021). One purpose of this analysis was to assess the presence of an "observer effect," wherein fishing behavior and associated catch composition are different when an observer is on board versus on unobserved trips. The report found "there were few statistically significant differences in fishing metrics between observed and unobserved trips on periodically observed vessels, or between unobservable and periodically observed vessels." Although an observer effect was not identified in the study, the rare occurrence of bycatch events for most non-target species in the DGN fishery means it is not possible to definitively estimate numbers of discards at sea during unobserved trips. The PSMFC's report can be found on the Council's website (June 2021 Agenda Item F.1.a NMFS Report 2).

We considered, under the ESA, whether or not the proposed action would cause any other activities and determined that it would not.

2. ENDANGERED SPECIES ACT: BIOLOGICAL OPINION AND INCIDENTAL TAKE STATEMENT

The ESA establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat upon which they depend. As required by section 7(a)(2) of the ESA, each Federal agency must ensure that its actions are not likely to jeopardize the continued existence of endangered or threatened species or to adversely modify or destroy their designated critical habitat. Per the requirements of the ESA, Federal action agencies consult with NMFS, and section 7(b)(3) requires that, at the conclusion of consultation, NMFS provide an opinion stating how the agency's actions would affect listed species and their critical habitats. If incidental take is reasonably certain to occur, section 7(b)(4) requires NMFS to provide an ITS that specifies the impact of any incidental taking and includes reasonable and prudent measures (RPMs) and terms and conditions to minimize such impacts.

In this biological opinion, we analyze the likely adverse effects resulting from the proposed action on the following species: fin whale; humpback whale, Mexico DPS and Central America DPS; sperm whale; leatherback sea turtle; North Pacific Ocean DPS loggerhead sea turtle; East Pacific DPS green sea turtle; olive ridley sea turtle; and giant manta ray. We also determined that designated critical habitat for leatherback sea turtles and critical habitat for both the Central America and Mexico DPS of humpback whales may be affected, but is not likely to be adversely affected (see section 2.12 *Not Likely to Adversely Affect Determinations*).

NMFS determined the proposed action is not likely to adversely affect blue whales, sei whales, Western North Pacific gray whales, Southern Resident killer whales, North Pacific right whales, Guadalupe fur seals, Southern DPS green sturgeon, Southern DPS eulachon, Eastern Pacific DPS scalloped hammerhead sharks, gulf grouper, oceanic whitetip sharks, white abalone, black abalone, or any ESA-listed Evolutionary Significant Unit (ESU) or DPS of salmonids from the U.S. west coast, or their critical habitats. Our analysis is documented in the *"Not Likely to Adversely Affect" Determinations* section (2.12).

2.1 Analytical Approach

This biological opinion includes a jeopardy analysis which relies upon the regulatory definition of "to jeopardize the continued existence of" a listed species, which is "to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

The designation(s) of critical habitat for leatherback sea turtles, both the Central America and Mexico DPS of humpback whales, and Southern Resident DPS killer whales use(s) the term primary constituent element (PCE) or essential features. The 2016 final rule (81 FR 7414; February 11, 2016) that revised the critical habitat regulations (50 CFR 424.12) replaced this term with physical or biological features (PBFs). The shift in terminology does not change the approach used in conducting a "destruction or adverse modification" analysis, which is the same regardless of whether the original designation identified PCEs, PBFs, or essential features. In this biological opinion, we use the term PBF to mean PCE or essential feature, as appropriate for the specific critical habitat.

The ESA Section 7 implementing regulations define effects of the action using the term "consequences" (50 CFR 402.02). As explained in the preamble to the final rule revising the definition and adding this term (84 FR 44976, 44977; August 27, 2019), that revision does not change the scope of our analysis, and in this opinion we use the terms "effects" and "consequences" interchangeably.

We use the following approach to determine whether a proposed action is likely to jeopardize listed species or destroy or adversely modify critical habitat:

- Evaluate the rangewide status of the species and critical habitat expected to be adversely affected by the proposed action.
- Evaluate the environmental baseline of the species and critical habitat.
- Evaluate the effects of the proposed action on species and their critical habitat using an exposure–response approach.
- Evaluate cumulative effects.

- In the integration and synthesis, add the effects of the action and cumulative effects to the environmental baseline, and, in light of the status of the species and critical habitat, analyze whether the proposed action is likely to: (1) directly or indirectly reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species; or (2) directly or indirectly result in an alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species.
- If necessary, suggest a reasonable and prudent alternative to the proposed action.

For this proposed action, NMFS relies upon data provided by fisheries observers deployed in the DGN fishery and other available data on interactions with ESA-listed species that may be associated with the DGN fishery to evaluate the potential effects of DGN fishery on ESA-listed species. Estimates of bycatch rates (and mortality or serious injury for some species) for ESA-listed species in the DGN fishery are produced and published by the SWFSC in technical memos (e.g., Carretta 2022a) and in marine mammal stock assessment reports (SARs; e.g., Carretta et al. 2022). Information on the *Status* and relevant *Environmental Baseline* of ESA-listed species that interact with the DGN fishery is gathered from recent SARs, recovery plans, status reviews (e.g., Bettridge et al. 2015), published scientific literature, other publicly available information, and unpublished data available to NMFS.

2.2 Rangewide Status of the Species and Critical Habitat

This opinion examines the status of each species that is likely to be adversely affected by the proposed action. The status is determined by the level of extinction risk that the listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. This informs the description of the species' likelihood of both survival and recovery. The species status section also helps to inform the description of the species' "reproduction, numbers, or distribution" for the jeopardy analysis as described in 50 CFR 402.02. The opinion also examines the condition of critical habitat throughout the designated area, evaluates the conservation value of the various watersheds and coastal and marine environments that make up the designated area, and discusses the function of the PBFs that are essential for the conservation of the species.

One factor affecting the range-wide status of ESA-listed species and aquatic habitat at large is climate change. Climate change has received considerable attention in recent years, with growing concerns about global warming and the recognition of natural climatic oscillations on varying time scales, such as long-term shifts like the Pacific Decadal Oscillation (PDO) or short-term shifts, like El Niño or La Niña. Evidence suggests that the productivity in the North Pacific (Quinn and Niebauer 1995) and other oceans could be affected by changes in the environment. Important ecological functions such as migration, feeding, and breeding locations may be influenced by factors such as ocean currents and water temperature. Any changes in these factors could render currently used habitat areas unsuitable and new use of previously unutilized or previously not existing habitats may be a necessity for displaced individuals. Changes to climate and oceanographic processes may also lead to decreased productivity in different patterns of prey distribution and availability. Such changes could affect individuals that are dependent on those affected prey.

NMFS' policy (NMFS 2016a) is to use climate indicator values projected under the Intergovernmental Panel on Climate Change (IPCC)'s Representative Concentration Pathway (RCP) 8.5 when data are available or best available science that is as consistent as possible with RCP 8.5. RCP 8.5, like the other RCPs, were produced from integrated assessment models and the published literature; RCP is a high pathway for which radiative forcing reaches >8.5 W/m^2 by 2100 (relative to pre-industrial values) and continues to rise for some amount of time. Changes in parameters will not be uniform, and the IPCC projects that areas like the equatorial Pacific will likely experience an increase in annual mean precipitation under scenario 8.5, whereas other mid-latitude and subtropical dry regions will likely experience decreases in mean precipitation. Sea level rise is expected to continue to rise well beyond 2100 and while the magnitude and rate depends upon emissions pathways, low-lying coastal areas, deltas, and small islands will be at greater risk (IPCC 2018).

The potential impacts of climate and oceanographic change on whales and other marine mammals will likely affect habitat availability and food availability. Site selection for migration, feeding, and breeding may be influenced by factors such as ocean currents and water temperature. For example, there is some evidence from Pacific equatorial waters that sperm whale feeding success and, in turn, calf production rates are negatively affected by increases in sea surface temperature (Smith and Whitehead 1993; Whitehead 1997). Any changes in these factors could render currently used habitat areas unsuitable. Changes to climate and oceanographic processes may also lead to decreased prey productivity and different patterns of prey distribution and availability. Research on copepods has shown their distribution may be shifting in the North Atlantic due to climate changes (Hays et al. 2005). Different species of marine mammals will likely react to these changes differently. For example, range size, location, and whether or not specific range areas are used for different life history activities (e.g. feeding, breeding) are likely to affect how each species responds to climate change (Learmouth et al. 2006).

Based upon available information, it is likely that sea turtles are being affected by climate change. Sea turtle species are likely to be affected by rising temperatures that may affect nesting success and skew sex ratios, as some rookeries are already showing a strong female bias as warmer temperatures in the nest chamber leads to more female hatchlings (Kaska et al. 2006; Chan and Liew 1995). Rising sea surface temperatures and sea levels may affect available nesting beach areas as well as ocean productivity. Based on climate change modeling efforts in the eastern tropical Pacific Ocean, for example, Saba et al. (2012) predicted that the Playa Grande (Costa Rica) sea turtle nesting populations would decline 7% per decade over the next 100 years. Changes in beach conditions are expected to be the primary driver of the decline, with hatchling success and emergence rates declining by 50-60% over the next 100 years in that area (Tomillo et al. 2012). Sea turtles are known to travel within specific isotherms and these could be affected by climate change and cause changes in their bioenergetics, thermoregulation, prey availability, and foraging success during the oceanic phase of their migration (Robinson et al. 2008; Saba et al. 2012). While the understanding of how climate change may impact sea turtles is building, there is still uncertainty and limitations surrounding the ability to make precise predictions about or quantify the threat of future effects of climate change on sea turtle populations (Hawkes et al. 2009).

We consider the ongoing implications of climate change as part of the status of ESA-listed species. Where necessary or appropriate, we consider whether impacts to species resulting from the proposed action could potentially influence the resiliency or adaptability of those species to deal with climate change that we believe is likely over the foreseeable future. However, we note that the proposed action only has a five year horizon, after which time the DGN fishery will cease, which limits the potential for the proposed action and its associated effects to also interact with a changing climate.

2.2.1 Fin Whale

Fin whales were listed as endangered worldwide under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained on the list of threatened and endangered species after the passage of the ESA in 1973 (35 FR 8491). Currently there is no designated critical habitat for fin whales. A recovery plan was completed in 2010 (NMFS 2010a). Fin whales feed on planktonic crustaceans, including *Thysanoessa* sp. euphausiids and *Calanus* sp. copepods, and schooling fish, including herring, capelin and mackerel (Aguilar 2009). Association with the continental slope is common, perhaps due to abundance of prey (Schorr et al. 2010). However, fin whales aggregate to areas with large amounts of prey regardless of water depth. For example, fin whales can feed in more shallow waters during the day (less than 330 feet), and feed in deeper waters at night (can be greater than 1,320 feet; EPA 2017).

Fin whales are distributed widely in the world's oceans and occur in both the Northern and Southern Hemispheres. In the northern hemisphere, they migrate from high Arctic feeding areas to low latitude breeding and calving areas. In the Atlantic Ocean, fin whales have an extensive distribution from the Gulf of Mexico and Mediterranean Sea northward to the arctic. The North Pacific population summers from the Chukchi Sea to California, and winters from California southward. Fin whales have also been observed in the waters around Hawai'i. Fin whales can occur year-round off California, Oregon, and Washington (Carretta et al. 2022a). Information suggesting fin whales are present year-round in southern California waters with movements into central California and Baja California returning to the Southern California Bight, as evidenced by individually-identified whales being photographed in all four seasons, and including one satellite tracked individual with movement from south Baja California by February and north to Monterey area by June (Falcone and Schorr 2013). Širovic et al. (2017) propose the possibility of a southern California resident population through acoustic data along with their seasonal movements, although is not yet clear. Additional telemetry studies would be necessary to fully flesh out the population stocks along with genetics and acoustics Martien et al. 2020). The fin whales most likely to be observed within the proposed action area are identified as part of the CA/OR/WA stock as defined under the MMPA.

<u>Population Status and Trends</u>: Although reliable and recent estimates of fin whale abundance are available for large portions of the North Atlantic Ocean, this is not the case for most of the North

Pacific Ocean and Southern Hemisphere. The status of populations in both of these ocean basins in terms of present population size relative to "initial" (pre-whaling, or carrying capacity) level is uncertain. Fin whales in the entire North Pacific are estimated to be less than 38 percent of historic carrying capacity of the region (Mizroch et al. 1984). The best estimate of fin whale abundance in California, Oregon, and Washington waters out to 300 nautical miles is 11,065 (Coefficient of Variation (CV)=0.405) animals from 2018, where species distribution models (SDMs) from 1991-2018 line-transect survey data were used to estimate density of cetaceans in the California Current Ecosystem (Becker et al. 2020a). While this estimate is greater than previously posited by Nadeem et al. (2016) and Moore and Barlow (2011) who applied Bayseian trend analysis, it remains consistent with their conclusions of an increasing population for this stock. SDMs are used for this region as a stable method for estimating densities which incorporate both the changes in species abundance and habitat shifts over time (Becker et al. 2016, 2017, 2020b; Redfern et al. 2017). However, the new abundance estimates are substantially higher than earlier estimates because the new analysis incorporates lower estimates of detection probability (Barlow 2015). There is now evidence of recovery in California coastal waters. Evidence of their increased abundance came from line transect surveys off California and within the California Current (extending from CA, along OR, and WA) between 1991 and 2018 (Moore and Barlow 2011, Nadeem et al. 2016, Becker et al. 2020a), with an estimated mean annual abundance increase of 7.5% from 1991 to 2014 off California, Oregon, and Washington (Nadeem et al. 2016). However, it remains unclear what to attribute the growth to: immigration or their birth and death rates (Carretta et al. 2022a).

<u>Threats</u>: A comprehensive list of general threats to fin whales is detailed in the Recovery Plan (NMFS 2010a). Obvious threats to fin whales besides vessel interactions and fishery entanglements include reduced prey abundance due to overfishing or other factors (including climate change), habitat degradation, and disturbance from low-frequency noise. Because little evidence of entanglement in fishing gear exists, and large whales such as the fin whale may often die later and drift far enough not to strand on land after such incidents, it is difficult to estimate the numbers of fin whales killed and injured by gear entanglements.

Documented ship strike deaths and serious injuries are derived from actual counts of fin whale carcasses and should be considered minimum values, although Rockwood et al. (2017) recently published efforts to generate plausible estimates of ship strike mortality for several whale species, including fin whales, along the U.S. west coast (see Section 2.4 *Environmental Baseline*). Continued research around vessel traffic pattern inconsistencies may inform our understanding of the mitigation of vessel strike risks with Redfern et al. (2019) who suggested consideration of both the reduction of vessel speeds along with expanded areas of avoidance.

The threats to fin whales due to underwater noise, pollutants, marine debris, and habitat degradation, are difficult to quantify, although recent studies are finding concerning levels of persistent organic pollutants (Pinzone et al. 2015) and microplastics (Fossi et al. 2012) in the blubber of fin whales, and there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for fin whales that use low frequency sound to communicate.

Based on a minimum population estimate of 7,970 for the CA/OR/WA stock of fin whales, the 2015-2019 total quantified documented incidental mortality and serious injury (2.2/year (yr)) due to fisheries (0.64/yr) and ship strikes (1.6/yr) is less than the calculated potential biological removal (PBR) of 80 (Carretta et al. 2022a). Total fishery mortality is less than 10% of PBR and, therefore, may be approaching zero mortality and serious injury rate (Carretta et al. 2022a).

Conservation: There are several international agreements in place to protect fin whales. For example, part of the IWC's function is to set catch limits for commercial whaling, which have been set at zero since 1985. Even before then, fin whales have been nominally protected from commercial whaling since 1966 by the IWC. The ban on commercial whaling has likely been a key conservation measure that has allowed fin whales to recover and continue to increase in the North Pacific, although limited harvest of fin whales outside the IWC framework has occurred. Fin whales are currently listed as Appendix I under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), which is aimed at protecting species at risk from unregulated international trade. Appendix I includes species threatened with extinction which may are or may be affected by trade; therefore, trade of Appendix I species is only allowed in exceptional circumstances. The International Union for the Conservation of Nature and Natural Resources (IUCN) Red List identifies and documents those species most in need of conservation if global extinction rates are to be reduced. The last assessment for fin whales was conducted in 2018; fin whales continued to be classified as "vulnerable" (Cooke 2018a).

The International Maritime Organization (IMO) is the United Nations specialized agency with responsibility for the safety and security of shipping and the prevention of marine pollution by ships. One way that the IMO facilitates efficient and safe passage for ships is by establishing Traffic Separation Schemes (TSSs), which are voluntary routing measures aimed at separating opposing traffic streams and encouraging the flow of vessels in and out of port within designated traffic lanes. The IMO also designates regions as "Particularly Sensitive Sea Areas" and "Areas to be Avoided" for various ecological, economic or scientific reasons. The IMO was approached for the first time regarding conservation of an endangered whale species in 1998 for the North Atlantic right whale (*Eubalaena glacialis*), and since then they have been approached with nations' proposals to establish or amend routing measures in various locations to reduce the threat of vessel collision with endangered whales, including humpbacks. The U.S. Coast Guard (USCG) is responsible for making recommendations to the IMO, and for establishing and modifying shipping lanes within U.S. waters. When considering changes, the USCG will initiate a Port Access Route Study (PARS), and recommendations for shifting or modifying shipping lanes are presented to the IMO for approval. IMO-endorsed modifications to TSSs have been established in areas off San Francisco/Oakland, off Santa Barbara and Los Angeles/Long Beach. Most recently, in January 2023, the IMO adopted a U.S. proposal to increased protections for blue, fin and humpback whales off southern California, which will take effect in the summer of 2023. The modifications will include a 13 nautical mile extension of the existing TSS in the Santa Barbara Channel, resulting in vessels lining up in deeper waters where there are lower concentrations of whales which should help reduce the risk from ship strikes. In addition, an area to be avoided by vessels will be expanded by more than 2,000 nm^2 and will cover approximately 4,476 nm^2 of important foraging habitat off Point Conception and Point Arguello in Santa Barbara County, CA.

2.2.2 Humpback Whale

Humpback whales are found in all oceans of the world and migrate from high latitude feeding grounds to low latitude calving areas. Humpbacks primarily occur near the edge of the continental slope and deep submarine canyons, where upwelling concentrates zooplankton near the surface for feeding. Humpback whales feed on euphausiids and various schooling fishes, including herring, capelin, sand lance, and mackerel (Clapham 2009).

Humpback whales were listed as endangered under the Endangered Species Conservation Act in June 1970 (35 FR 18319) and remained on the list of threatened and endangered species after the passage of the ESA in 1973 (35 FR 8491). On September 8, 2016, NMFS published a final rule to divide the globally listed endangered humpback whale into 14 DPSs and place four DPSs as endangered and one as threatened (81 FR 62259). NMFS has identified three DPSs of humpback whales that may be found off the coasts of Washington, Oregon and California. These are the Hawaiian DPS (found predominately off Washington and southern British Columbia [SBC]) which is not listed under the ESA; the Mexico DPS (found all along the U.S. West Coast) which is listed as threatened under the ESA; and the Central America DPS (found predominately off the coasts of Oregon and California) which is listed as endangered under the ESA. A recovery plan for humpbacks (globally listed as endangered at the time) was issued in November 1991 (NMFS 1991). Given the change in status of humpback whales throughout the world, NMFS is currently working on an updated DPS-specific recovery plan for three ESA-listed DPSs found in U.S. waters of the Pacific Ocean: Central America, Mexico, and the Western North Pacific.

Critical habitat for the endangered Central America DPS and the threatened Mexico DPS of humpback whales within waters off the U.S. West Coast was designated on April 21, 2021 (86 FR 21082). Essential features for both DPSs were identified as prey species, including euphausiids and small pelagic schooling fishes such as Pacific sardine (*Sardinops sagax*), northern anchovy (*Engraulis mordax*) and Pacific herring (*Clupea pallasii*). Critical habitat for the Central America DPS of humpback whales contains approximately 48,521 square nautical miles (nmi^2) of marine habitat in the North Pacific Ocean within the portions of the California Current off the coasts of Washington, Oregon, and California. Specific areas designated as critical habitat for the Mexico DPS of humpback whales contain approximately 116,098 nm^2 of marine habitat in the North Pacific Ocean, including areas within portions of the eastern Bering Sea, Gulf of Alaska, and California Current Ecosystem.

The most recent final SAR (2021) for humpback whales on the west coast of the United States (Carretta et al. 2022a) noted that NMFS was currently evaluating the stock structure of humpback whales under the MMPA in response to the new ESA listings; thus we will refer in part to the status of the populations that are found in the action area using the existing SAR. We note and acknowledge, however, that the draft 2022 SAR (Carretta et al. 2023a, b) reviewed by the Pacific Scientific Review Group and published for public comment (88 FR 4162) proposes to define two new humpback whale stocks off the U.S. west coast: the Central America/Southern Mexico-CA/OR/WA stock and the Mainland Mexico-CA/OR/WA stock. While NMFS will continue to evaluate the relationship between the humpback whale DPSs and recognized "demographically independent populations"[7] (DIPs; Martien et al. 2019), we will consider and rely heavily upon the draft 2022 SAR and most recent publicly available information in assessing the status (including abundance and trends) of the two listed DPSs, and in considering the proportional risk of anthropogenic activities on humpbacks found within the action area.

Humpback whales found along the U.S. West Coast spend the winter primarily in coastal waters of Mexico and Central America, and the summer along the U.S. West Coast from California to British Columbia. As a result, both the endangered Central America DPS and the threatened Mexico DPS at times travel and feed off the U.S. West Coast and may be exposed to the DGN fishery. In July 2021, NMFS WCR updated a memo outlining evaluation of the distribution and relative abundance of ESA-listed DPSs that occur in the waters off the U.S. West Coast using the best available scientific information available, which included genetic analyses (Lizewski et al. 2021; Martien et al. 2020; 2021), photo-identification analyses, (Calambokidis and Barlow 2020; Wade 2017; 2021) and species distribution models (Becker et al. 2020b; NMFS 2021a). NMFS (2021) recommended that for ESA Section 7 analyses, that the WCR should apply a proportional approach based on the most recent abundance information on the CA/OR/WA stock from Calambokidis and Barlow (2020) and the proportions of the various DPSs feeding off CA/OR (and WA/Southern British Columbia in Wade (2021). NMFS recommended considering that for actions occurring off the coast of California of Oregon, 42 percent of the humpback whales that could be affected by a proposed action would be members of the endangered Central America DPS and 58 percent would be members of the threatened Mexico DPS. For actions off the coast of Washington, NMFS recommended considering that six percent of humpback whales would be from the Central America DPS, 25 percent would be from the Mexico DPS, and 69 percent would be from the non-listed Hawai'i DPS. In addition, the most recent draft SARs for humpback whales uses the same underlying information for apportioning human impacts among the newly proposed humpback stock delineations (Carretta et al. 2023a, b). We will use these recommended proportions in our analysis of the relative threats to each of the two listed DPSs, both with respect to the proposed action and other impacts that occur in the action area.

Based on the best available information, all of the whales from the Central America DPS appear to migrate to feed only off the west coast of the United States. Conversely, whales from the Mexico DPS migrate in varying proportions to the U.S. West Coast, British Columbia, and various areas off Alaska. Based on the management objectives of the MMPA to delineate marine mammal stocks[8], NMFS has determined that stocks should represent "DIPs" (Martien et al. 2019). Within the ESA-listed DPSs in the Pacific Ocean, multiple lines of evidence indicate that the animals migrating to and along the U.S. West Coast from waters off mainland Mexico

[7] NMFS' Guidelines for Preparing Stock Assessment Reports Pursuant to the 1994 Amendments to the MMPA specify that a stock under the MMPA should comprise a demographically independent population (DIP), where "demographic independence" is to mean that "…the population dynamics of the affected group is more a consequence of births and deaths within the group (internal dynamics) rather than immigration or emigration (external dynamics). Thus, the exchange of individuals between population stocks is not great enough to prevent the depletion of one of the populations as a result of increased mortality or lower birth rates (NMFS 2016b).

[8] The term stock, as defined by statute under the MMPA means a group of marine mammals of the same species or smaller taxa in a common spatial arrangement that interbreed when mature.

comprise a single DIP, and could possibly be considered a stand-alone stock under the MMPA (Martien et al. 2021). In the waters off California and Oregon, this DIP co-occurs with a newly described DIP of animals originating from waters off of Central America and southern Mexico (Taylor et al. 2021). In waters off of Washington and southern British Columbia (outside of the Action Area), animals from the Hawai'i DPS also occur; however their status as a separate DIP within the Hawai'i DPS has not been established.

Given the identification of two DIPs (also termed "migratory herds" that share both wintering and feeding areas by Martien et al. 2020) within the endangered Central America and the threatened Mexico DPSs, Curtis et al. (2022) recently estimated the abundance of the Central America/Southern Mexico DIP. Although the Central America DPS is comprised of those whales that winter along the Pacific coast of Central America from Panama to Guatemala (Bettridge et al. 2015), this DIP's wintering area is understood to extend north into southern Mexico to at least the state of Guerrero, with animals sighted as far north as Michoacán and Colima (Taylor et al. 2021). Given that this DIP forages exclusively off the U.S. West Coast, and given the most recent abundance estimate for the CA/OR/WA stock of humpbacks, we can use a proportional approach to estimate the likelihood of a whale occurring in the action area and affected by the DGN fishery originating from Central America/Southern Mexico and/or mainland Mexico. This will be discussed in more detail in Population Status and Trends sections below.

<u>Population Status and Trends:</u> NMFS-WCR reviewed the best available scientific information on the distribution and abundance of the two DPSs foraging off the U.S. West Coast. There are two primary lines of evidence for the origin of humpback whales found off the U.S. West Coast: photo identification catalogues and genetic identification of sampled individuals.

Wade et al. (2016) estimated abundance within all sampled winter breeding and summer feeding areas in the North Pacific and estimated migration rates between these areas using a comprehensive photo-identification study of humpback whales in 2004-2006 during the "SPLASH" (Structure of Populations, Levels of Abundance and Status of Humpbacks) project. Subsequently, Wade (2017) reanalyzed the Wade et al. (2016) data because "the multistrata model analyses were not necessarily converging to the correct answer," as stated in Wade (2017). Further revisions and refinements were made in Wade (2021) as part of the ongoing comprehensive assessment of humpback whales by the International Whaling Commission (IWC). The revised results led to different estimates of abundance for both the breeding (winter) and feeding (summer) grounds and different estimates of the proportional representation of animals from the different breeding grounds foraging off areas of the U.S. West Coast. We note that the SPLASH surveys were conducted around 15 years ago, which indicates that those abundance estimates are outdated; specifically, they are greater than 8 years old, which is not considered a reliable estimate of current abundance, as summarized in NMFS' Guidelines for Preparing Stock Assessment Reports (NMFS 2016b; NMFS 2023b). For the 2004-2006 humpback populations, the Wade (2021) revised abundance estimate for the Central America DPS is 755 (CV=0.242) animals, and the revised abundance estimate for the Mexico DPS is 2,913 (CV=0.066) animals, using the Multistrata model (N_{multi}) (which uses both winter and summer data; Table 4 in Wade 2021).

Recent analyses by Calambokidis and Barlow (2020) updated humpback whale abundance estimate for the previous CA/OR/WA stock of humpbacks, which included 2018 survey data. Capture-recapture models for humpback whales off CA/OR showed a dramatic increase in recent years, with a trend for the population starting in 1989 (~500 animals) through 2018 increasing an average 7.5% per year, with a higher rate of increase in the late 2000s[9]. While multiple abundance estimates for humpbacks along the U.S. West Coast were reported, the most recent (i.e., 2018) estimate of 4,973 whales (with a standard error of 239 and lower and upper 20th percentile values of 4,776 to 5,178 whales) was produced for CA/OR based on the Chao model using rolling 4-year periods for the last four most recent available years (2015-2018; Table 3 in Calambokidis and Barlow 2020). While the estimates of humpback whale abundance for WA/SBC were also presented (1,593 animals, standard error of 108) and showed increases, particularly in recent years and extending into the Salish Sea[10], the abundance estimate for the U.S. West Coast only included CA/OR. There are two main reasons why the authors did not add the two estimates from both foraging areas. First, the WA/SBC estimate included a fairly large number of animals that would be outside U.S. waters, since some of the major areas of concentration were just north of the U.S. border. Animals outside of U.S. waters are generally not included in abundance estimates generated for SARs under the MMPA[11]. Secondly, there is some interchange between the CA/OR and the WA/SBC areas, which would mean that each individual estimate is to some degree including a portion of animals from the other area (J. Calambokidis, Cascadia Research Collective, personal communication, September 2020).

The final 2021 SAR (Carretta et al. 2022a) relied on this abundance estimate for the CA/OR/WA stock of humpback whales: 4,973 (CV=0.048) whales, with a minimum abundance estimate of 4,776 whales. The current population trend is approximately 8.2 percent annually since the late 1980s, which is consistent with observed increases for humpback whale across the entire North Pacific, from 1,200 whales in 1966 to 18,000 to 20,000 whales during 2004 to 2006 (Calambokidis et al. 2008). Calambokidis and Barlow (2020) note that the apparent increase in abundance from 2014 to 2018 is likely too great to represent real population growth, and may reflect negatively-biased estimates during 2009 to 2014 due to less representative sampling compared with 2018. Based on the minimum abundance estimate for the CA/OR/WA stock (4,776) as well as the estimated population growth rate for this stock (8.2% x ½) and a recovery factor (Fr) of 0.3, the PBR for this stock was 58.7 whales. However, because this stock spends

[9] These estimates will further be evaluated and tested in the future with results from planned Bayesian models and an analysis of the 2018 sighting survey conducted by NMFS. A “SPLASH 2” program is also underway, with increased field efforts off southern Mexico and Central America, as well as workshops to update the databases and analyses.

[10] Photographs of humpback whales in the inland waters of Washington (Strait of Juan de Fuca, Haro Strait and Puget Sound) are currently being analyzed by Cascadia Research Collective to match individuals to the breeding ground photo identification catalogues. Until that analysis is complete, we will use the same proportions in inland waters as for the outer coast of Washington.

[11] Note that they may be included when estimates are based on mark-recapture for transboundary stocks such as humpback whales. The 2019 SARs included the abundance estimate for WA/SBC feeding group of humpback whales (n=526) and then this estimate was prorated for the time spent outside of U.S. waters (where data are available) to calculate the potential biological removal level (J. Carretta, NMFS-SWFSC, personal communication, December 2020).

approximately half its time outside of the U.S EEZ, PBR for this stock was 29.4 whales per year (final 2021 SAR; Carretta et al. 2022a). While generally informative about the status of humpback whales off the U.S. West Coast, including ESA-listed humpback whale DPSs, the SARs for humpback whales off the U.S. West Coast is being replaced using the new stock delineations proposed by Carretta et al. (2023a, b). Based on new information from Curtis et al. (2022) on abundance estimates for the Central America-Southern Mexico-CA/OR/WA stock, new PBRs relevant for both the Central America-CA/OR/WA and Mexico-CA/OR/WA stocks are summarized below along with their relevance to the Central America and Mexico DPS status.

Threats: A comprehensive list of general threats to humpback whales is detailed in the Recovery Plan (NMFS 1991) and the 2015 status review (Bettridge et al. 2015). Similar to other large whales, humpbacks throughout the North Pacific Ocean are potentially affected by loss of habitat, loss of prey (for a variety of reasons including climate variability), underwater noise, and pollutants. Substantial coastal development is occurring in many regions throughout the range of the two listed DPSs considered in this biological opinion, and noise associated with construction (e.g., pile driving, blasting or explosives) has the potential to affect humpbacks by generating sound levels that may disturb humpback whales or adversely affect their hearing. Port construction may result in a higher volume of ship traffic. Contaminants, including heavy metals, persistent organic pollutants, effluent, airborne contaminants, plastics and other marine debris can affect humpback whales through the accumulation of lipophilic compounds in their blubber (e.g., pesticides such as DDT) and may have detrimental effects, including disease susceptibility, neurotoxicity, and reproductive and immune impairment. Humpbacks may also ingest or become entangled in marine debris. Whale watching and scientific research may also disturb or harm humpback whales through disruption of essential biological functions, harassment or injury from inadvertent close approaches or application of tags, etc. In general, anthropogenic sound has increased in all oceans over the last 50 years, and is thought to have doubled each decade in some areas of the ocean over the last 30 years or so. Low-frequency sound comprises a large proportion of this increase, stemming from a variety of sources, including shipping, oil and gas exploration, and military activities. Detrimental effects associated with anthropogenic sound include hearing loss, masking, and temporary threshold shifts in hearing, so social communication could be impacted, as well as fluctuations in stress hormones, change in behavior such as departure from prime foraging areas or alteration in migratory routes or timing. Given the sensitivities of humpbacks to low and mid-frequency sounds, researchers may be able to detect changes and adverse effects to individuals; however, population-level impacts on cetaceans in general has not been confirmed and is difficult to detect on a large scale, including long time-frames.

Little is known of the anthropogenic threats to the two listed humpback whale DPSs while they are outside of U.S. waters and on their breeding grounds. When we do have reports of injured or dead whales reported off Canada and Mexico entangled in gear originating from the United States, these reports are included in the SAR. Given their long migrations between feeding and breeding grounds, we assume both DPSs are subject to the anthropogenic threats summarized above, but any reports (particularly published reports) are rare to nonexistent.

Entanglement in fishing gear poses a significant threat to individual humpback whales throughout the Pacific Ocean. For fisheries interactions/entanglements off the U.S. West Coast, pot and trap fishery entanglements are the most prevalent source of serious injury and mortality, and reported entanglements increased considerably in 2014. Between 1982 and 2013, NMFS' confirmed stranding/sighting records of entangled whales (all species) averaged around 9 whales/year. Entanglement reports spiked between 2014 and 2017 with an average of 41 confirmed entanglements/year (Saez et al. 2021). The estimated impact of fisheries on the Central America and Mexico DPSs is likely underestimated, since the mortality or serious injury of large whales due to entanglement in gear may go unobserved because whales swim away with a portion of the net, line, buoys, or pots. Non-commercial fisheries may include tribal and recreational fisheries as well as marine debris (including research buoys) but are likely responsible for a small fraction of all entanglements. Most of the details of the interactions are summarized in the *Environmental Baseline* section 2.4.1.1, since the action area includes most of the U.S. West Coast.

Humpback whales, especially calves and juveniles, are highly vulnerable to ship strikes (Stevick 1999) and other interactions with non-fishing vessels. Off the U.S. West Coast, humpback whale distribution overlaps significantly with the transit routes of large commercial vessels, including cruise ships, large tug and barge transport vessels, and oil tankers in the proposed action area. Whale watching boats and research activities directed toward whales may have direct or indirect impacts on humpback whales as harassment may occur, preferred habitats may be abandoned, and fitness and survivability may be compromised if disturbance levels are too high. Over the past 30 years, our known (and considered minimum) estimate of vessel strikes of large whales is considered low. More details of interactions between vessels and humpback whales is provided in the *Environmental Baseline* section 2.4.1.2.

<u>Conservation</u>: There are several international agreements in place to protect humpback whales. For example, part of the IWC's function is to set catch limits for commercial whaling, which have been set at zero since 1985. Even before then, the North Pacific humpback whales have been nominally protected from commercial whaling since 1966 by the IWC. Illegal catches continued in the region for several years after protection but the last substantial catches occurred in 1968 (IUCN 2018). Since that time, the IWC's Scientific Committee has developed a stock assessment and catch limit methodology ("revised management procedure) with the goal of providing information on catch limits consistent with maintaining sustainable populations. Catch limits for humpback whales have been allowed for some aboriginal whaling, but not within the North Pacific Ocean. The ban on commercial whaling has likely been a key conservation measure that has allowed humpbacks to recover and continue to increase in most areas of the North Pacific. Humpback whales are currently listed as Appendix I under the CITES, which is aimed at protecting species at risk from unregulated international trade. Appendix I includes species threatened with extinction which may are or may be affected by trade; therefore, trade of Appendix I species is only allowed in exceptional circumstances. The IUCN Red List identifies and documents those species most in need of conservation if global extinction rates are to be reduced. The last assessment for humpback whales was conducted in 2018; humpbacks continued to be classified as "least concern" (Cooke 2018b).

As described above, the IMO recently adopted a U.S. proposal to increased protections for blue, fin and humpback whales off southern California, which will take effect in the summer of 2023. The modifications will include a 13 nautical mile extension of the existing TSS in the Santa Barbara Channel, resulting in vessels lining up in deeper waters where there are lower concentrations of whales, which should help reduce the risk from ship strikes. In addition, an area to be avoided by vessels will be expanded by more than 2,000 nm^2 and will cover approximately 4,476 nm^2 of important foraging habitat off Point Conception and Point Arguello in Santa Barbara County, CA.

Domestically, all marine mammals (including humpbacks) are protected by the MMPA. Any "take" of a marine mammal requires approval of NMFS, with some exceptions. ESA-listed marine mammals receive additional protections under the MMPA from activities such as commercial fishing. The Mexico DPS also forages in Alaskan waters, and under the authority of the ESA and the MMPA, NMFS issued a final rule effective in 2001 making it unlawful for anyone (i.e., vessels) to approach humpback whales within 100 yards or disrupt their normal behavior in Alaska (66 FR 29502). Off the U.S. West Coast, there are five national marine sanctuaries that provide additional protection for both humpback whale DPSs: the Olympic Coast (3,188 mi^2), the Greater Farallones (3,295 mi^2), Cordell Bank (1,286 mi^2), Monterey Bay (6,094 mi^2), and the Channel Islands (1,470 mi^2). In general, the sanctuaries provide additional protection within their waters that may restrict acoustic impacts, discharge of pollutants, cruise ships, fishing (through marine protected areas, for example), offshore wind energy development, oil and gas development, vessel traffic, etc., all of which benefit humpback whales when foraging within sanctuary waters.

In 2015, a working group was formed to address large whale (and humpback whales in particular) and leatherback entanglements in the California Dungeness crab fishery. Members of the working group included non-governmental organizations, industry, biologists, and state and federal representatives. Together, they developed a Risk Assessment Mitigation Program (RAMP) to reduce the risk of entanglements of blue whales, humpback whales, and leatherback turtles. The CDFW finalized RAMP regulations on November 1, 2020 (14 CFR Sec. § 132.8) which requires the agency Director to evaluate the risk of entanglement and the need for management action at least monthly during the fishing season (November 1-June 30). The regulations include triggers for management actions, (e.g., fishery closures, advisories, depth constraints, gear requirements, alternative gear) if entanglements of the three key species are confirmed or concentrations of these species is observed.

The states of California, Oregon, and Washington are currently applying for a permit to allow the states' Dungeness crab fishery to incidentally take humpback whales (and blue whales and leatherbacks) through section 10 of the ESA. Section 10 requires the states to develop a conservation plan that would describe the anticipated impact of the requested take levels and how the state agencies will minimize those impacts (e.g. establishing an entanglement detection network, implementing a lost/abandoned gear retrieval program, implementing electronic vessel position monitoring, gear marking, testing innovative gear).

2.2.2.1 Mexico DPS Humpback Whale

The Mexico DPS consists of whales that breed along the Pacific coast of Mexico, the Baja California Peninsula and the Revillagigedos Islands. This DPS feeds across a broad geographic range from California to the Aleutian Islands, with concentrations in California-Oregon, northern Washington-southern British Columbia, northern and western Gulf of Alaska and Bering Sea feeding grounds. The Mexico DPS was determined to be discrete based on significant genetic differentiation as well as evidence for low rates of movements among breeding areas in the North Pacific given sightings information. The DPS was determined to be significant due to the gap in breeding grounds that would occur if this DPS were to go extinct and the marked degree of genetic divergence to other populations (Bettridge et al. 2015).

Population Status and Trends: The Mexico DPS of humpback whales, which occurs throughout U.S. west coast and the DGN action area, was estimated to be 6,000 to 7,000 animals from the SPLASH project (Calambokidis et al. 2008) and in the most recent status review (Bettridge et al. 2015). More recently, Wade et al. (2021) estimated the abundance of the Mexico DPS to be 2,913 (CV=0.242) based on revised analysis of the available data from 2004-2006. Because the abundance estimates for the Mexico DPS were derived from the ~15 year old SPLASH project, NMFS (2021a) generated a more current estimate of the abundance using the most recent data from Calambokidis and Barlow 2020, which was included in the 2021 SAR (Carretta et al. 2022a). Based on an assumed 6 percent annual growth rate from the Wade et al. (2021) abundance from 2006, NMFS (2021a) estimated the minimum abundance estimate for the total Mexico DPS to be around 6,981 animals, but could be a higher abundance of approximately 9,000 animals based on recent growth rate estimates used in the SARs.

As described above, Martien et al. (2021) delineated the Mainland Mexico-CA/OR/WA DIP based on two strong lines of evidence indicating demographic independence: genetics and movement data. The draft 2022 SAR (Carretta et al. 2023a, b) designated the DIP as a "stock" because available data make it feasible to manage it as a stock and because there are conservation and management benefits to doing so. Given the Curtis et al. (2022) abundance estimate for whales wintering in southern Mexico and Central America (1,496) and the most recent estimate of humpback whales foraging off the U.S. West Coast (4,973; Calambokidis and Barlow 2020), the estimated abundance for the Mainland Mexico-CA/OR/WA stock is 3,477 animals (CV=0.101), with a minimum estimate of 3,185 whales, taken as the lower 20th percentile of the difference. The stock trend for this particular stock could not be estimated since two stocks of humpbacks utilize the area (Carretta et al. 2023a, b). However, given the most recent annual growth rate estimated by Calambokidis and Barlow (2020) of 8.2 %/year, we can assume that if the Central America/Southern Mexico - CA/OR/WA stock is only growing at around 1.6%/year (Curtis et al. 2022), that the Mainland Mexico – CA/OR/WA stock is likely increasing at a level near or greater than 5-6%, which is similar to trend estimates in earlier SARs. The current net productivity rate for this stock is unknown. However, as stated in the draft 2022 SAR (Carretta et al. 2023a, b), the theoretical maximum net productivity rate can be taken to be at least as high as the maximum observed for the combined stocks, or 8.2% annually (Calambokidis and Barlow 2020), though it could be higher if one of the stocks is growing faster than another.

Given the information described above, the calculated PBR for this stock is 65 animals/year (Carretta et al. 2023a, b). Ryan et al. (2019) notes that humpbacks are present in central California waters at least 8 months annually, with the beginning half of December and the ending half of April representing "transition months," where whales are moving in/out of the region (summarizing sighting and acoustic data). Assuming 8 months of residency time is U.S. West Coast waters, or 2/3 of the year, this yields a PBR in U.S. waters of 43 whales per year for this stock (Carretta et al. 2023a, b)

At this time, the current total abundance of the entire Mexico DPS is unknown, beyond the estimates of 6-7,000 made using data from fifteen years ago. Likely, given the growth rates that have been observed for the portion of this DPS that occurs off the U.S. West Coast since that time, the population of the DPS has likely increased significantly as well. The threats for the Mexico DPS have been generally summarized above, and more information from activities that affect this population in the action area will be described further in section 2.4 *Environmental Baseline*.

2.2.2.2 Central America DPS Humpback Whale

The Central America DPS is composed of humpback whales that breed along the Pacific coast of Costa Rica, Panama, Guatemala, El Salvador, Honduras and Nicaragua (Bettridge et al. 2015), although new evidence has shown that humpback whales found off southern Mexico belong to the "Central American population unit," which migrates north to the feeding areas of the U.S. West Coast using a migratory corridor along mainland Mexico to the mouth of the Gulf of California along the Baja California Peninsula (Martínez-Loustalot et al. 2022). Martien et al. (2020) also introduced that DIPs of humpbacks are delineated as "migratory herds" that share both wintering (breeding) and feeding grounds. The Central America DIP's wintering ground is understood to extend into southern Mexico, and therefore is termed the Central America/Southern Mexico-CA/OR/WA DIP for its wintering area and its feeding area off CA/OR/WA (Taylor et al. 2021), which will likely describe the "stock," as defined under the MMPA. Therefore, while the Central America DPS is defined by Bettridge et al. (2015) and the final rule identifying the 14 DPSs of humpbacks (81 FR 62260; September 16, 2016) under the ESA, we consider the inclusion of southern Mexico humpbacks and the abundance estimate recently published by Curtis et al. (2022), using photo-identification data collected in their wintering area from 2019 to 2021. However, NMFS will continue to evaluate the relationship between the humpback whale DPSs and recognized DIPs moving forward.

<u>Population Status and Trends</u>: The Central America DPS of humpback whales occurs throughout the U.S. west coast and DGN action area, although individuals are more likely to be found off the coast of California and Oregon. Earlier estimates of abundance for the Central America DPS ranged from approximately 400 to 600 individuals (Bettridge et al. 2015; Wade et al. 2016), although Wade (2021) reanalyzed the SPLASH data to estimate the abundance estimate for the Central America DPS at 755 (CV=0.242). We note that, similarly as with the Mexico DPS, the abundance estimate was derived from the ~15 year old SPLASH project data.

Since the 2021 SAR was finalized (Carretta et al. 2022a), Curtis et al. (2022) published new information regarding the abundance estimate of the Central America/Southern Mexico DPS, which has resulted in significant changes to the currently draft 2022 SAR (Carretta et al. 2023a, b). First, the "Central America/Southern Mexico – California/Oregon/Washington DIP" has been identified as a "stock," as defined under the MMPA. This was based on the two strong lines of evidence indicating demographic independence: genetics and movement data (Taylor et al. 2021). Second, the Curtis et al. (2022) abundance estimate for this stock was incorporated into the draft 2022 SAR. Using spatial capture-recapture methods based on photographic data collected between 2019 and 2021, researchers estimated the abundance of this stock to be 1,496 (CV=0.171) whales, which represents the best estimate of the Central America/Southern Mexico-CA/OR/WA stock of humpback whales. The minimum population estimate was taken as the lower 20th percentile of the capture-recapture estimate from Curtis et al. (2022), or 1,284 whales. Given the inclusion of whales from southern Mexico in the current estimate, Curtis et al. (2022) derived a population growth rate based on the differences between the 2004-2006 estimate (755 animals, CV=0.242) (Wade, 2021)) and the current estimate by excluding whales in southern Mexico waters in the spatial recapture model. This yielded an annual growth rate for this stock of 1.6% (SD=2.0%) for the Central America/Southern Mexico-CA/OR/WA stock; however, this estimate has high uncertainty. As described in the draft 2022 SAR (Carretta et al. 2023a, b), the maximum net productivity level can be taken to be at least as high as the maximum observed the two stocks summering off California and Oregon (for which most of the mark-recapture estimates were based on), or 8.2%, although it could be higher if one of the stocks is growing faster than the other.

In the draft 2022 SAR (Carretta et al. 2023b), the PBR for this stock was calculated to be 5.2 animals. Assuming 8 months of residency time as described above, the total PBR for this stock (5.2) is prorated by 2/3, to yield a PBR in U.S. waters of 3.5 whales per year (Carretta et al. 2023a, b).

The threats for the Central America DPS have been generally summarized above, and more information from activities that affect this population in the action area will be described further in section 2.4 *Environmental Baseline*.

2.2.3 Sperm Whale

Sperm whales were listed as endangered worldwide under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained on the list of threatened and endangered species after the passage of the ESA in 1973 (35 FR 8491). NMFS has not designated critical habitat for sperm whales. Sperm whales are one of the most widely distributed of marine mammals worldwide, an can be found between 60°N and 70°S latitude (Leatherwood and Reeves 1983). Sperm whales are found throughout the North Pacific and are distributed broadly from tropical and temperate waters to the Bering Sea as far north as Cape Navarin, Russia. They are often concentrated around oceanic islands in areas of upwelling, and along the outer continental shelf and mid-ocean waters. Known as deep diving whales, their diet consists of many larger organisms that also occupy deep waters of the ocean. Their principle prey is large squid but they will also eat large demersal and mesopelagic sharks, skates, and fishes. Within the

Pacific US EEZ there are three distinct, are not adjacent, populations of sperm whales described in the Marine Mammal Protection Act stock assessment reports that include: 1. California, Oregon, and Washington; 2. Hawaiian, and 3. Alaska waters. Mesnick et al. (2011) used multiple data streams including genetic analysis of single-nucleotide polymorphisms (SNPs), microsatellites, and mitochondrial DNA (mtDNA) confirming sperm whales in the California Current are demographically independent from Hawai'i and the Eastern Tropical Pacific.

Population Status and Trends: Whitehead (2002) estimated current sperm whale abundance to be approximately 300,000–450,000 worldwide, growing at about 1 percent per year. Abundance in the Pacific is approximately 152,000–226,000 using Whitehead's 2002 methods. There are large populations of sperm whales in waters that are within several thousand miles west and south of California, Oregon, and Washington, although there is no evidence of sperm whale movements into this region from either the west or south. Sperm whales are found year-round in California waters (Dohl et al. 1983; Forney et al. 1995). They reach peak abundance from April through mid-June and from the end of August through mid-November (Rice 1974). They have been seen in every season except winter (Dec-Feb.) in Washington and Oregon (Green et al. 1992). It is the CA/OR/WA stock of sperm whales that is likely to be exposed to the DGN fishery.

Previous estimates of sperm whale abundance for the CA/OR/WA stock of sperm whales from 2005 (3,140, CV=0.40, Forney 2007) and 2008 (300, CV=0.51, Barlow 2010) show a ten-fold difference that cannot be attributed to human-caused or natural population declines. The most recent estimates of sperm whale abundance in California, Oregon, and Washington waters out to 300 nmi are available from a trend-model analysis of line-transect data collected from seven surveys conducted from 1991 to 2016 (Moore and Barlow 2017), using methods similar to previous abundance trend analyses for fin whales (Moore and Barlow 2011) and beaked whales (Moore and Barlow 2013). Sperm whale abundance estimates based on the trend-model ranged between 2,000 and 3,000 animals for the 1991-2014 time series (Moore and Barlow 2014). The best estimate of sperm whale abundance in the California Current is the trend-based estimate corresponding to the most recent survey (2014), or 1,997 animals (CV=0.57). Assessment of current population trends is inconclusive given the low precision of estimates (Carretta et al. 2022a)

Threats: A comprehensive list of general threats to sperm whales is detailed in the Recovery Plan (NMFS 2010b). Entanglement in fishing gear poses a threat to individual sperm whales and overall to the CA/OR/WA sperm whale stock. The vulnerability of sperm whales to incidental entanglement in fishing gear especially gillnets set in deep water for pelagic fish (*e.g.*, sharks, billfish, and tuna) is well-documented (Di Natale and Notarbartolo di Sciara 1994; Felix et al. 1997). Sperm whales may become entangled in fishing gear while attempting to take fish off the gear such as demersal long-line gear (Angliss and Outlaw 2008). Sperm whales are also vulnerable to ship strikes as they raft on the surface after long dives. Similar to other whale species, threats to sperm whales besides vessel interactions and fishery entanglements include reduced prey abundance due to overfishing or other factors (including climate change), habitat degradation, and disturbance from low-frequency noise.

For the CA/OR/WA stock of sperm whales, the total 2013-2017 quantified documented incidental mortality and serious injury from fisheries is greater than or equal to 0.64 (CV=1.4) and without any ship strikes reported, due to a low probability of carcasses washing ashore this is less than the calculated PBR of 2.5 (Carretta et al. 2022a). Total human-caused mortality is greater than 10% of the calculated PBR, and due to ship strikes, entanglement, and unknown total amount of deaths or serious injury from anthropogenic causes resulting in underestimates, and therefore, cannot be considered to be approaching a zero mortality and serious injury rate. Although acoustic pingers are known to reduce the entanglement of cetaceans in the DGN fishery (Carretta and Barlow 2011), it is unknown whether pingers have any effect on sperm whale entanglement in this fishery due to low sample sizes.

Conservation: There are several international agreements in place to protect sperm whales. For example, part of the IWC's function is to set catch limits for commercial whaling, which have been set at zero since 1985. Even before then, fin whales have been nominally protected from commercial whaling since 1966 by the IWC. The ban on commercial whaling has likely been a key conservation measure that has allowed sperm whales to recover and continue to increase in the North Pacific. Sperm whales are currently listed as Appendix I under CITES, which is aimed at protecting species at risk from unregulated international trade. Appendix I includes species threatened with extinction which may are or may be affected by trade; therefore, trade of Appendix I species is only allowed in exceptional circumstances. The IUCN Red List identifies and documents those species most in need of conservation if global extinction rates are to be reduced. The last assessment for sperm whales was conducted in 2008 and amended in 2019; sperm whales continued to be classified as "vulnerable" (Taylor et al. 2019).

2.2.4 North Pacific DPS Loggerhead Sea Turtle

Loggerheads are circumglobal, inhabiting continental shelves, bays, estuaries, and lagoons in temperate, subtropical, and tropical waters. Major nesting grounds are generally located in temperate and subtropical regions, with scattered nesting in the tropics. Until 2011, loggerheads were listed globally as a threatened species under the ESA. A recovery plan for the then threatened U.S. Pacific loggerhead populations was completed over 20 years ago (NMFS and U.S. Fish and Wildlife Service (USFWS) 1998a). The most recent status review for the North Pacific DPS of loggerheads was completed in 2020, which reaffirmed the endangered status of this species (NMFS and USFWS 2020a)

In 2011, a final rule was published describing ESA-listings for nine DPSs of loggerhead sea turtles worldwide (76 FR 58868). The North Pacific Ocean DPS, is the only species found in the *Action Area* of the proposed action listed as endangered under the ESA. Since the loggerhead listing was revised in 2011, a recovery plan for the North Pacific loggerhead DPS has not been completed. However, through a U.S. initiative, three countries (United States, Japan, and Mexico) have been developing a tri-national recovery plan although there have been complications in collaborations with Japan and Mexico. Further resolution and clarification needs to take place between the three countries to determine if the recovery plan will move forward through a "formal" process (B. Schroeder, NMFS-headquarters, personal communication, 2022).

North Pacific loggerhead sea turtles occur north of the equator in the Pacific Ocean. Like other sea turtle species, the North Pacific loggerhead exhibits a complex life cycle: egg, hatchling, juvenile, subadult, and adult. Juvenile and subadult life stages are also frequently distinguished according to whether they occur in neritic or pelagic waters.

North Pacific loggerheads nest exclusively in Japan, in three regions (or management units): mainland Japan, Yakushima, and Okinawa. After the turtles emerge as hatchlings on their natal beaches in Japan, they spend their developmental years foraging in the North Pacific, moving with the predominant ocean gyres for many years before returning to their neritic foraging habitats. Satellite tracking of juvenile loggerheads indicates the Kuroshio Extension Bifurcation Region in the central Pacific to be an important pelagic foraging area for juvenile loggerheads (Polovina et al. 2006; Howell et al. 2008). Researchers have identified other important juvenile turtle foraging areas off the coast of Baja California Sur, Mexico (Peckham et al. 2007; Conant et al. 2009). Resident times of juvenile North Pacific loggerheads foraging at a known hotspot off Baja California were recently estimated at over 20 years, with turtles ranging in age from 3 to 24 years old (Tomaszewicz et al. 2015). South of Point Eugenia on the Pacific coast of Baja California, pelagic red crabs (*Pleuroncodes planipes*) have been found in great numbers, attracting top predators such as tunas, whales and sea turtles, particularly loggerheads (Pitman 1990; Wingfield et al. 2011; Seminoff et al. 2014). After spending years foraging in the central and eastern Pacific, mature loggerheads migrate to forage in oceanic or neritic waters closer to Japan in between breeding seasons (Hatase et al. 2002; Hatase et al. 2010), with adult females returning to nest, on average, every 3.3 years (mean "remigration interval") and laying a mean of 4.6 nests per season ("clutch frequency") (see Hatase et al. 2013). Thus, adult loggerheads remain in the western Pacific for the remainder of their life cycle (Iwamoto et al. 1985; Kamezaki et al. 1997; Conant et al. 2009; Hatase et al. 2002).

Loggerheads documented off the U.S. west coast in the action area are primarily found south of Point Conception, California, in the Southern California Bight.

<u>Population Status and Trends</u>: The North Pacific loggerhead DPS nests primarily in Japan (Kamezaki et al. 2003), although low level nesting may occur outside of Japan in areas surrounding the South China Sea (Chan et al. 2007; Conant et al. 2009). Along the Japanese coast, nine major nesting beaches (greater than 100 nests per season) and six "submajor" beaches (10–100 nests per season) exist, including Yakushima Island where over 50% percent of nesting occurs (Kamezaki et al. 2003; Jones et al. 2018). Census data from 12 of these 15 beaches provide composite information on longer-term trends in the Japanese nesting assemblage. From this data, Kamezaki et al. (2003) concluded a substantial decline (50–90%) in the size of the annual loggerhead nesting population in Japan had occurred since the 1950s. As discussed in the 2011 final ESA listing determination, current nesting in Japan represents a fraction of historical nesting levels (Conant et al. 2009; 76 FR 58868). Nesting declined steeply from an initial peak of approximately 6,638 nests in 1990–1991, to a low of 2,064 nests in 1997. Since that time, nesting has been variable, increasing and decreasing over time as is typical of sea turtle nesting trends. Overall, since 2003/2004, an increasing trend of approximately 9 percent annual growth in the number of nests has been documented for the entire nesting assemblage, through 2015

(i.e., all nesting beaches combined) (Y. Matsuzawa, Sea Turtle Association of Japan, personal communication, 2017).

In terms of abundance, of North Pacific loggerheads, Van Houtan (2011) estimated the total number of adult nesting females in the population was 7,138 for the period 2008-2010. An abundance assessment using data available through 2013 was conducted by Casale and Matsuzawa (2015) as part of an IUCN Red List assessment that estimated 8,100 nesting females in the population. Jones et al. (2018) used a model estimate of 3,632 females nesting at Yakushima, assumed to represent 52% of all nesting females in the population, to estimate the total number of North Pacific loggerhead nesting females at 6,984 (NMFS 2019a).

Most recently, Martin et al. (2020a, 2020b) used a Bayesian state-space population growth model to estimate the range of intrinsic population growth rates, or *r*. They used data from three index beaches in Yakushima: Maehama, Inakahama, and Yotsuehama, from 1985 to 2015. The nest count data was converted to nester count data by dividing the number of nests each year by the mean clutch frequency (4.6 nests per female; Hatase et al. 2013). As with all sea turtles, these trends will not necessarily represent the true growth of the population because annual nester counts, which represent the bulk of data on sea turtles, only represent a portion of the population, specifically adult females. However, as this is the only estimate of population growth rate available for this species, we consider it the best available scientific data to describe trends. Drawing from the resulting distribution of *r* values as well as the distributions of the nesting female population size at the end of the time series and a process error term, the researchers projected population trends 100 years into the future, conducting 10,000 simulations to capture the variability in projects. In the future projections, Martin et al. (2020a, 2020b) computed the proportion of simulations for which the projected number of annual nesters fell below (and remained below) 50%, 25% and 12.5% of the estimated abundance of nesters within specific time intervals. For the set of runs ending below a threshold, they calculated the mean, median, and 95% credible interval for the number of years until the population fell below the threshold. They also calculated the probability of the projected total reproductive females falling below each threshold at 5, 10, 25, 50, and 100 years in the future.

Results of the PVA model suggest that the adult female portion of the North Pacific loggerhead sea turtle population is increasing at a rate of 2.3%/year (95% confidence interval (CI): −1.1% to 15.6%), and the probability of the population as indicated by the index nesting beaches falling to less than half of its current abundance within 100 years is 33% (Martin et al. 2020a). For those simulations in the PVA that indicated a decline, 50% of current population size was reached in a mean of 25.2 years (95% CI: 5 to 82 years). PVA modeled estimates suggest that the modeled population presently consists of a minimum of 4,541 adult female loggerheads (95% CI: 4,074-5063; total nesters for the three index beaches in Japan). It is estimated that there are approximately 328,744 juvenile (year 1-25) North Pacific loggerhead sea turtles (T. Jones, NMFS, personal communication, 2019). Using the estimate of 4,541 females nesting in Yakushima, representing 52% of nesting females, the total number of North Pacific loggerhead nesting females is 8,733 (4,541*100/52). Using a sex ration of 65% female (Martin et al. 2020) suggests that the abundance of the North Pacific loggerhead DPS is approximately 13,435 (8,733*100/65) adults, or a total population size of 342,179 (328,744 juveniles + 13,435 adults).

As noted above, North Pacific loggerheads have been documented in high numbers off the central Pacific coast of Baja California, Mexico. Aerial surveys conducted from 2005 through 2007 in the Gulf of Ulloa, a known “hot spot,” provided an estimated foraging population of over 43,000 juvenile loggerheads (Seminoff et al. 2014). NMFS conducted aerial surveys of the SCB in 2015 (a year when the sea surface temperatures were anomalously warm, and an El Niño was occurring) and estimated more than 70,000 loggerheads throughout the area (Eguchi et al. 2018), likely feeding on pelagic red crabs and pyrosomes which are the turtle’s preferred prey. Recent analysis of loggerhead sea turtle presence in the SCB suggests that loggerhead presence offshore of Southern California is tied not just to warm temperatures, but to persistently warm temperatures over a period of months such as what occurred during the recent large marine heatwave experienced by the Eastern North Pacific Ocean (Welch et al. 2019).

Recent efforts have examined potential relationships between significant climate/environmental variables and influences on turtle populations. Van Houtan and Halley (2011) identified correlations between loggerhead juvenile recruitment and breeding remigrations and two strong environmental influences: sea surface temperature and the Pacific Decadal Oscillation index of ocean circulation. The mechanisms that could influence loggerhead survival at important stages may be relevant to understanding past nesting beach trends, and this is a promising avenue of research. However, there are many more anthropogenic and natural factors that may influence sea turtle populations and future trends, and a consideration of the differences in ocean basins, nesting assemblages, demographics, and habitat, among other variables, needs to be included in any characterization of status and trend of a particular population or DPS such as North Pacific loggerheads. Relating environmental variance into population dynamics is an important component in our attempts to understand the fate of long-lived and highly migratory marine species such as sea turtles. However, we cannot currently reliably predict the magnitude of future climate change and its impacts on North Pacific loggerheads. In addition, as noted by Arendt et al. (2013), the Van Houtan and Halley (2011) paper proposed an alternative to a long-held paradigm that the survivorship of large juveniles and adult sea turtles is more predictive of population change than juvenile recruitment. Van Houtan and Halley (2011) suggested that cohort effects stemming from survival in the first year of life had a greater effect on population growth. Analyses conducted by Arendt et al. (2013) on climate forcing on annual nesting variability of loggerheads in the Northwest Atlantic Ocean showed that trends in annual nest counts are more influenced by remigrants rather than neophytes, which contradicts in part the Van Houtan and Halley (2011) study. As summarized above, the North Pacific loggerhead nesting population has been generally increasing, considering the most recent trend analyses (using data from three index beaches from 1985 to 2015 (Martin et el. 2020a)) not included in the Van Houtan and Halley (2011) analysis, which may be explained by conservation efforts on the nesting beaches, at the foraging grounds (e.g., Gulf of Ulloa, in Baja California, Mexico), and potentially realized reduction of threats from large-scale fisheries such as longlining.

At this time, uncertainty remains related to the North Pacific loggerhead nesting beach trend forecasts and correlations with climate indices related to the PDO, for example. The mechanisms that could influence loggerhead survival at important stages are logical, and this is a promising avenue of research. Relating environmental variance into population dynamics will be an important step in trying to understand the fate of marine species such as sea turtles. However, it is not possible to reliably predict the magnitude of future climate change and the impacts on loggerhead sea turtles. The existing data and current scientific methods and analysis are not able to predict the future effects of climate change on this species or allow us to predict or quantify this threat to the species (Hawkes et al. 2009).

<u>Threats</u>: A detailed account of natural and anthropogenic threats of loggerhead sea turtles around the world is provided in recent status reviews (NMFS and USFWS 2007a; Conant et al. 2009; NMFS and USFWS 2020a). Loggerhead nesting beaches are threatened by hurricanes and tropical storms as well as storm surges, sand accretion, and rainfall associated with hurricanes. Hatchlings are killed by predators such as herons, gulls, dogfish, and sharks. Juvenile and adult loggerheads are also killed by sharks and other large marine predators. Loggerheads are also killed by cold stunning and exposure to biotoxins.

The most significant threats facing loggerheads in the North Pacific include coastal development and bycatch in commercial fisheries. Destruction and alteration of loggerhead nesting habitats are occurring throughout the species' range, especially coastal development (including breakwaters that alter patterns of erosion and accretion on nesting beaches), beach armoring, beachfront lighting, and vehicular/ pedestrian traffic. As the size of the human population in coastal areas increases, that population brings with them secondary threats such as exotic fire ants, feral pigs, dogs and growth of populations that tolerate human presence (e.g., raccoons, armadillos and opossums) which feed on turtle eggs. Overall, the NMFS and USFWS have concluded that coastal development and coastal armoring on nesting beaches in Japan are significant threats to the persistence of this DPS (Conant et al. 2009; 76 FR 58868; NMFS and USFWS 2020a).

For both juvenile and adult individuals in the ocean, bycatch in commercial fisheries, both coastal and pelagic fisheries (including longline, drift gillnet, set-net, trawling, dredge, and pound net) throughout the species' range is a major threat (Conant et al. 2009). Specifically in the Pacific, bycatch continues to be reported in gillnet and longline fisheries operating in 'hotspot" areas where loggerheads are known to congregate (Peckham et al. 2007). Interactions and mortality with coastal and artisanal fisheries in Mexico and the Asian region likely represent the most serious threats to North Pacific loggerheads (Peckham et al. 2007; Ishihara 2009; Conant et al. 2009). In Mexico, loggerhead mortality has been significantly reduced, particularly in a previously identified hotspot, where thousands of loggerheads may forage for many years until reaching maturity. In 2013, Mexico was notified that, unless it established a regulatory program comparable in effectiveness to that of the United States, Mexico would receive a "negative certification" under section 403(a) of the Magnuson-Stevens Act. This notification was made as a result of documented evidence of hundreds of loggerheads found stranded or bycaught in coastal artisanal fisheries in the Gulf of Ulloa, off the Pacific coast of Baja California. As a result, in 2016, Mexico published new regulations, which established a reserve located in the loggerhead hotspot area. Within this reserve, the 2016 regulation sets a loggerhead turtle mortality limit for commercial fishing vessels of 90 turtles. If that 90 mortality threshold is met, Mexico would suspend gillnet fishing from May through August to protect loggerhead sea turtles. Restrictions on mesh size and soak time were also included to reduce mortalities. After reviewing the regulations, the United States was able to positively certify Mexico in September 2016 (Department of Commerce 2016). This restriction likely reduces loggerhead bycatch by an order of magnitude and addresses one of the primary threats identified in Conant et al. (2009). In spite of the measures Mexico has taken, Mexican Wildlife Law Enforcement reported significant strandings of dead North Pacific loggerhead sea turtles on the shores of the Gulf of Ulloa in each of the past three years: 459 in 2018, 331 in 2019, and 351 from January to June 2020. In communication with NMFS, Mexico noted that the Gulf of Ulloa measures are still in place, including the refuge area. Based on the recent strandings data, NMFS was concerned that the measures were not being fully or effectively implemented. Therefore, in 2021, NMFS re-identified Mexico for not having management measures to end or reduce the bycatch of loggerhead turtles in the Gulf of Ulloa fisheries that are not comparable in effectiveness to U.S. regulations (NMFS 2021b). NMFS is currently in discussions with Mexico to understand more fully the cause of the strandings, as well as the effectiveness of management measures, including enforcement, use of observers (including cameras), analysis of observer data, bycatch reduction measures, and effectiveness of the reserve.

There are interactions between North Pacific loggerheads and domestic longline fishing for tuna and swordfish based out of Hawai'i. The Hawai'i-based longline fisheries were estimated to have captured and killed several hundred loggerheads before they were closed in 2001. Under requirements established in 2004 to minimize sea turtle bycatch (69 FR 17329), vessel operators in the Hawai'i-based shallow-set swordfish fishery must use large (sized 18/0 or larger) circle hooks with a maximum of 10 degrees offset and mackerel-type bait. Between 2004 and 2018, the shallow-set longline fishery was estimated to have captured about 177 North Pacific loggerhead sea turtles, killing about 28 of these turtles (NMFS 2019a). From 2012-2017, the incidental take statement for the Hawai'i-based shallow-set fishery was 34 loggerhead turtles per year, which served as the "hard cap" for the fishery that requires closure of the entire fishery during any year if reach. Recently, the hard cap for loggerhead sea turtle bycatch was removed, with the expectations that up to 36 may be caught and 6 may be killed each year, and that vessels would be restricted to no more than 5 loggerheads taken during any one trip (NMFS 2019a).

In the deep-set longline tuna fishery based out of Hawai'i from 2004-2021, there were 16 loggerheads observed taken (estimated 79 total, based on observer coverage). Based on historical capture events, NMFS anticipated that over 10 years, up to 86 North Pacific loggerheads will be captured, and of those, 48 will be killed. NMFS exempted the take (includes interactions, injuries or mortalities) of up to 43 North Pacific loggerheads over any given 5-year period (NMFS 2023c).

In the Western Central Pacific Ocean U.S. purse seine fishery, NMFS authorized the incidental take of up to 36 North Pacific loggerhead turtles annually, with an anticipation that 6 of those turtles would die (NMFS 2021c). Additionally, NMFS anticipated the take of 30 loggerhead interactions to occur in the U.S. Eastern Tropical Pacific (ETP) purse seine fishery, with one mortality anticipated every seven years (NMFS 1999, 2004a). In the ETP non-U.S. purse seine fisheries rarely interact with loggerhead sea turtles. For example, from 1993 through 2021, nearly 26 loggerheads were estimated to have been killed, with no deaths estimated since 2015. With 100 percent observer coverage in the U.S. purse seine fleet operating in the ETP, there have been zero loggerheads observed killed in this fishery (IATTC, personal communication, 2022). Because effort in this area may take place south of the equator, some of these turtles may be from the South Pacific loggerhead DPS, but without genetic information or tags to verify their origin, we will assume they are from the North Pacific DPS.

Estimating the total number of sea turtle interactions in other Pacific fisheries that interact with the same sea turtle populations as U.S. fisheries is difficult because of low observer coverage and inconsistent reporting from international fleets. Lewison et al. (2004) estimated 2,600 – 6,000 loggerhead mortalities from pelagic longlining in the Pacific in 2000. Beverly and Chapman (2007) more recently estimated loggerhead and leatherback longline bycatch in the Pacific to be approximately 20 percent of that estimated by Lewison et al. (2004), which would equate to between 520 and 1,200 loggerhead mortalities during the year assessed. More recently, Peatman et al. (2018) estimated that a median estimate of 29,405 loggerheads were captured in longline fisheries operating in the North Pacific from 2003-2017. These various estimates cover different time intervals, were produced by a variety of assumptions, and rely on data collected from fisheries with limited observer coverage (generally <1%, particularly for the fleets with the highest expended effort, except for Hawai'i-based longline fisheries, which range from ~20-100%), so their differences are not surprising. Nevertheless, they capture the approximate scale of the number of sea turtles that have been captured by fisheries outside of the action area.

In 2015 a workshop was convened to analyze the effectiveness of sea turtle mitigation measures in the tuna RFMOs and 16 countries provided data on observed sea turtle interactions and gear configurations in the Western Central Pacific Ocean. Based on the information gathered there, 549 loggerhead sea turtles were reported leading to a total estimate of 10,980 loggerheads caught in the region from 1989-2015 in these countries (NMFS unpublished data). Finally, bycatch estimates of sea turtles were summarized from annual reports by the Western and Central Pacific Fisheries Commission (WCPFC) (2021), which included the Hawai'i deep-set long line fishery, which represented around 5-6% of the total hooks set by Western Central Pacific Ocean longline fisheries. From 2013-2020, an average of 2,387 loggerheads (95% CI: 1,318 – 3,457) were captured per year, with an average of 390 loggerheads (95% CI: 327-452) killed per year (WCPFC 2021).

Between recent developments to reduce sea turtle bycatch in domestic fisheries that have been working their way into some international fisheries, and the incomplete data sets and reporting that exists, the exact level of current sea turtle bycatch internationally is not clear. However, given the information that is available, we believe that international bycatch of sea turtles in fisheries throughout the Pacific Ocean, continues to occur at significant rates several orders of magnitude greater than what is being documented or anticipated in U.S. domestic fisheries.

Conservation: Considerable effort has been made since the 1980s to document and reduce loggerhead bycatch in Pacific Ocean fisheries, as this is the highest conservation priority for the species. NMFS has formalized conservation actions to protect foraging loggerheads in the North Pacific Ocean which were implemented to reduce loggerhead bycatch in United States fisheries. Observer programs have been implemented in federally managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-hooking mortality. These include developing gear solutions to prevent or reduce capture (e.g., circle hooks) or to allow the turtle to escape without harm (e.g., turtle exclusion devices), implementing seasonal time-area closures to prevent fishing when turtles are congregated, modifying existing gear, and developing and promoting "Sea Turtle Handling Guidelines" (NMFS and USFWS 2007a). For example, switching to large circle hooks and mackerel bait in 2004 reduced the interaction rate by approximately 90% in the Hawai'i shallow-set longline fishery (Gilman et al. 2007a, WPFMC 2009b) and more recent analyses showed a reduction of 95% in this fishery (Swimmer et al. 2017). NMFS has also developed a mapping product known as TurtleWatch that provides a near real time product that recommends areas where the deployment of pelagic longline shallow sets should be avoided to help reduce interactions between Hawai'i-based pelagic longline fishing vessels and loggerhead sea turtles (Howell et al. 2008, 2015).

Since loggerhead interactions and mortalities with coastal fisheries in Mexico and Japan are of concern and are considered a major threat to North Pacific loggerhead recovery, NMFS and United States non-governmental organizations have worked with international entities to: (1) assess bycatch mortality through systematic stranding surveys in Baja California Sur, Mexico; (2) reduce interactions and mortalities in bottom-set fisheries in Mexico; (3) conduct gear mitigation trials to reduce bycatch in Japanese pound nets; and (4) convey information to fishers and other stakeholders through participatory activities, events and outreach. In 2003, Grupo Tortuguero's ProCaguama (Operation Loggerhead) was initiated to partner directly with fishermen to assess and mitigate their bycatch while maintaining fisheries sustainability in Baja California, Mexico. ProCaguama's fisher-scientist team discovered the highest turtle bycatch rates documented worldwide and has made considerable progress in mitigating anthropogenic mortality in Mexican waters (Peckham et al. 2007, 2008). As a result of the 2006 and 2007 tri-national fishermen's exchanges run by ProCaguama, Sea Turtle Association of Japan (STAJ), and the Western Pacific Fisheries Management Council, in 2007 a prominent Baja California Sur fleet retired its bottom-set longlines (Peckham et al. 2008; Peckham and Maldonado-Diaz 2012). Prior to this closure, the longline fleet interacted with an estimated 1,160-2,174 loggerheads annually, with nearly all (89%) of the takes resulting in mortalities (Peckham et. al. 2008). Because this fleet no longer interacts with loggerheads, conservation efforts have resulted in the continued protection of approximately 1,160-2,174 juvenile loggerheads annually (final loggerhead listing rule: 76 FR 58868; September 22, 2011). Additionally, stranding data collected since 2003 at Playa San Lazaro indicates a 60% reduction in standings' during 2010 compared to previous 2003-2009 averages (Peckham 2010). To date, 90% of the gillnet fleet has retired their gear (a total of 140 gillnets), 18 crews have converted to hook and line fishing (a more sustainable practice in the 'hotspot' area), and local government enforcement has increased to ensure compliance with local laws (Peckham pers.comm.). In Japan, due to concerns of high adult loggerhead mortality in mid-water pound nets, researchers with the STAJ, ProCaguama, and NMFS have begun collaborations, together with local fishermen throughout several Japanese prefectures, to investigate and test pound net mitigation options to reduce the impact and mortality of sea turtle bycatch. This work is ongoing as of 2011 and has received high media attention both within Japan and internationally that has helped to raise public awareness and maintain momentum (Ishihara et al. 2014).

Led by the Mexican Wildlife Service, a federal loggerhead bycatch reduction task force, comprised of federal and state agencies and non-governmental organizations, was organized in 2008 to ensure loggerheads receive the protection they are afforded by Mexican law. In 2009, while testing a variety of potential solutions, ProCaguama's fisher-scientist team demonstrated the commercial viability of substituting bycatch-free hook fishing for gillnet fishing. ProCaguama, in coordination with the task force, is working to develop a market-based bycatch solution consisting of hook substitution, training to augment ex-vessel fish value, development of fisheries infrastructure, linkage of local fleets with regional and international markets, and concurrent strengthening of local fisheries management (Conant et al. 2009).

Conservation efforts have also focused on protecting nesting beaches, nests, and hatchlings. Much of Japan's coastline is "armored" using concrete structures to prevent and minimize impacts to coastal communities from natural disasters. These structures have resulted in a number of nesting beaches losing sand suitable for sea turtle nesting, and nests often need relocating to protect them from erosion and inundation. In recent years, a portion of the concrete structures at a beach in Toyohashi City, Aichi Prefecture, was experimentally removed to create better nesting habitat (76 FR 58868; September 22, 2011). The STAJ along with various other organizations in Japan, are carrying out discussions with local and federal Government agencies to develop further solutions to the beach erosion issue and to maintain viable nesting sites. Recently, the Ministry of Environment has supported the local NGO conducting turtle surveys and conservation on Yakushima in establishing guidelines for tourism to minimize impacts by humans on nesting beaches (Y. Matsuzawa, STAJ, personal communication; Conant et al. 2009). Yet, beach erosion and armament still remain one of the most significant threats to nesting beaches in Japan (Conant et al. 2009). Since 2003, the Council has been contracting with STAJ to protect loggerhead nests and increase hatchling survivorship at several nesting beaches in southern Japan, including at the two primary beaches on Yakushima Island. Beach management activities include conducting nightly patrols during the summer nesting season to relocate nests from erosion prone areas, protecting nests from predators and people with mesh and fences, and cooling nests with water and shading to prevent overheating during incubation. STAJ has developed techniques for nest relocation that now result in an average of 60% hatchling success rates (compared to nearly zero survival of the same nests laid in erosion prone areas). Nest relocation in 2004-08 resulted in an estimated 160,000 hatchlings being released that otherwise may have been lost (76 FR 58868; September 22, 2011).

The conservation and recovery of loggerhead turtles is facilitated by a number of regulatory mechanisms at international, regional, national, and local levels, such as the Food and Agriculture Organization's (FAO) Technical Consultation on Sea Turtle-Fishery Interactions, the Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC), CITES, and others. In 2008 the WCPFC adopted CMM 2008-03 to mitigate the impacts on turtles from longline swordfish fisheries in the western central Pacific Ocean. The measure includes the adoption of FAO guidelines to reduce sea turtle mortality through safe handling practices and to reduce bycatch by implementing one of three methods by January 2010. The three methods to choose from are: 1) use only large circle hooks, or 2) use whole finfish bait, or 3) use any other mitigation plan or activity that has been approved by the Commission. As a result of these designations and agreements, many of the intentional impacts on sea turtles have been reduced:

harvest of eggs and adults have been slowed at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to slow the take of turtles in foraging areas. Moreover, as shown by the above examples from Hawai'i, Japan, and Baja Mexico, international efforts are growing to reduce sea turtle interactions and mortality in artisanal and industrial fishing practices (Gilman et al. 2007b; Peckham et al. 2007; NMFS and USFWS 2007a; Ishihara et. al. 2014).

2.2.5 Leatherback Sea Turtle

A recovery plan for the U.S. Pacific populations of leatherbacks was completed over 20 years ago (NMFS and USFWS 1998b), and leatherbacks remain listed globally as an endangered species under the ESA. In 2012, NMFS revised critical habitat for leatherbacks to include additional areas within the Pacific Ocean (77 FR 4170). The revised designation includes approximately 17,000 square miles stretching along the California coast from Point Arena to Point Arguello east of the 3,000 meter depth contour and approximately 25,000 miles stretching from Cape Flattery, Washington, to Cape Blanco, Oregon east of the 2,000 meter depth contour. The principal biological feature identified as essential to leatherback conservation was prey, primarily scyphomedusae. The proposed action occurs within Pacific leatherback critical habitat, and we analyze potential effects to designated leatherback critical habitat in section 2.12 of this Opinion.

Leatherback sea turtles have been observed at sea between about 71° N to 47° S (Eckert et al. 2012). Globally, seven populations are currently recognized under the ESA: (1) Northwest Atlantic; (2) Southeast Atlantic; (3) Southwest Atlantic; (4) Northeast Indian; (5) Southwest Indian; (6) West Pacific; and (7) East Pacific (NMFS and USFWS 2020b).

Leatherback turtles lead a completely pelagic existence, foraging widely in temperate and tropical waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Leatherbacks are highly migratory, exploiting convergence zones and upwelling areas for foraging in the open ocean, along continental margins, and in archipelagic waters (Morreale et al. 1994; Eckert 1999; Benson et al. 2007a, 2011). Leatherback sea turtles feed from near the surface to depths exceeding 1,000 m, including nocturnal feeding on tunicate colonies within the deep scattering layer (Spotila 2004). Although leatherback sea turtles can dive deeper than any other reptile, most dives are less than 80 m (Shillinger et al. 2011). Migrating leatherback sea turtles spend a majority of their time submerged and display a pattern of continual diving. They appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting continual foraging along the entire depth profile (Eckert et al. 1988). Stable isotope analysis can complement satellite data of leatherback sea turtle movements and identify important foraging areas that reflect regional food webs (Seminoff et al. 2012).

In the Pacific, leatherback nesting aggregations are found in the eastern and western Pacific. Aerial surveys conducted between 2004 and 2007 identified Indonesia, Papua New Guinea, and Solomon Islands as the core nesting areas for the population (Benson et al. 2011; Benson et al. 2012). The majority of nesting occurs along the north coast of the Bird's Head Peninsula, Papua Barat, Indonesia at Jamursba-Medi and Wermon beaches (Dutton et al. 2007). A recent discovery of a previously undocumented nesting area on Buru Island, Indonesia and relatively new sites in the Solomon Islands suggests that additional undocumented nesting habitats may exist on other remote or infrequently surveyed islands of the western Pacific Ocean (NMFS and USFWS 2020b). Low levels of nesting are also reported in Vanuatu (Petro et al. 2007; Wan Smolbag 2010).

The population exhibits genetic population structure. While mtDNA analyses of 106 samples from Indonesia, Papua New Guinea, and Solomon Islands did not detect genetic differentiation among nesting aggregations (Dutton et al. 2007), microsatellite DNA analyses indicate fine-scale genetic structure (Dutton 2019; NMFS SWFSC unpublished data). Hence, we treat these nesting aggregations as subpopulations. Two life history strategies are documented in the West Pacific Ocean population: winter boreal nesters (December to March) and summer boreal nesters (June to September). Migration and foraging strategies vary based on these life history strategies, likely due to prevailing offshore currents and seasonal monsoon-related effects experienced as hatchlings (Benson et al. 2011; Gaspar et al. 2012). Summer nesting females forage in Northern Hemisphere habitats in Asia and the North Pacific Ocean, while winter nesting females migrate to tropical waters in the South Pacific Ocean (Benson et al. 2011; Harrison et al. 2018; Figure 1). The lack of crossover among seasonal nesting populations suggests that leatherback turtles develop fidelity for specific foraging regions likely based on juvenile dispersal patterns (Benson et al. 2011; Gaspar et al. 2012; Gaspar and Lalire 2017). Stable isotopes, linked to particular foraging regions, confirm nesting season fidelity to specific foraging regions (Seminoff et al. 2012). Adult West Pacific leatherback sea turtles interacting with the DGN fishery are most likely summer nesters using the North Pacific transition zone (or Kuroshio extension), equatorial eastern Pacific, or the California Current Extension (Figure 1).

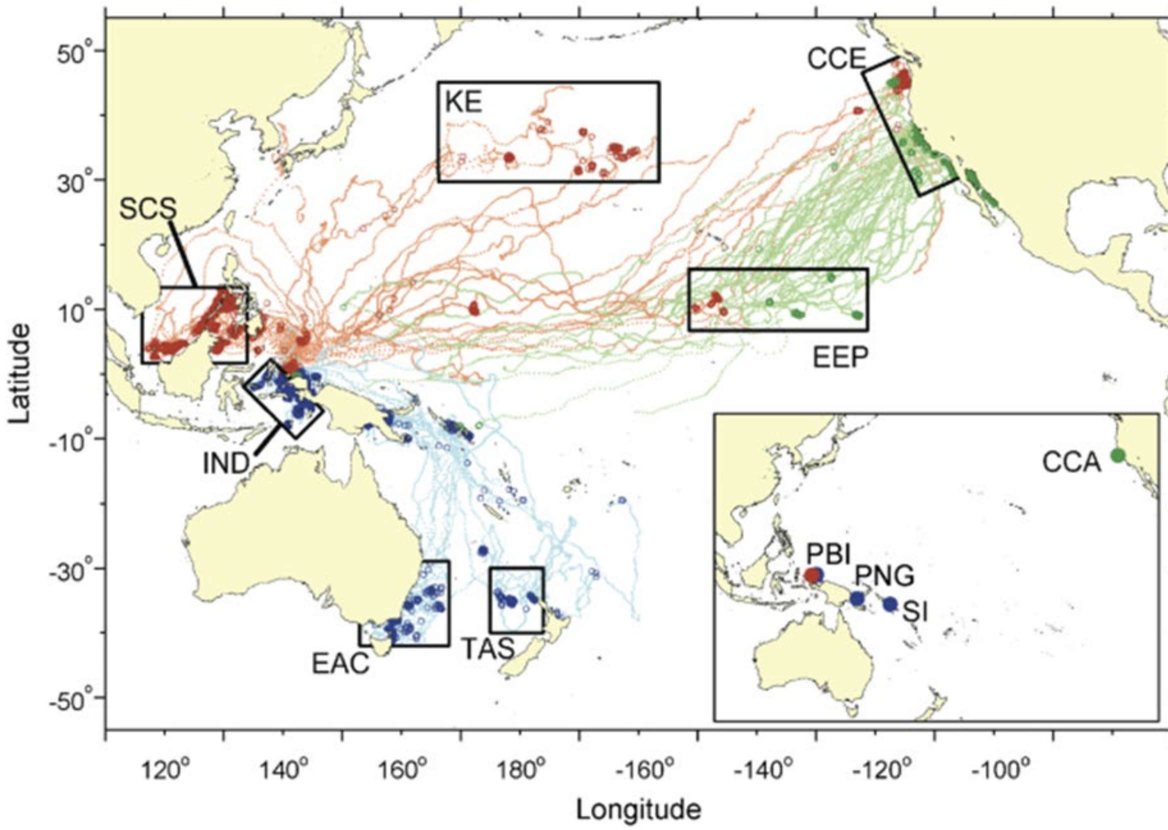

Figure 1. Satellite tracks from 126 West Pacific leatherback sea turtles. Color of track indicates deployment season: red = summer nesters, blue = winter nesters, green = deployments at central California foraging grounds. Inset shows deployment locations; PBI = Papua Barat, Indonesia, PNG = Papua New Guinea, SI = Solomon Islands, CCA = central California. Black boxes represent ecoregions for which habitat associations were quantitatively examined: SCS = South China, Sulu and Sulawesi Seas, IND = Indonesian Seas, EAC = East Australia Current Extension, TAS = Tasman Front, KE = Kuroshio Extension, EEP = equatorial eastern Pacific, and CCE = California Current Ecosystem (from Benson et al. 2011).

The most recent status review (NMFS and USFWS 2020b) defines the East Pacific subpopulation as leatherback turtles originating from the East Pacific Ocean, north of 47° S, south of 32.531° N, east of 117.124° W, and west of the Americas. The subpopulation generally occupies a distribution distinct from the West Pacific population and is considered to be located outside of the action area for the proposed action. However, based on interactions with the Hawai'i-based deep-set longline fishery, there are some areas where East and West Pacific populations can overlap, such as south of Hawai'i. Based on the genetic analyses of leatherbacks found off the U.S. West Coast, we consider the probability of the East Pacific leatherback sea turtles occurring in the action area, to be extremely low. No leatherbacks taken and sampled in the DGN fishery or captured off the U.S. West Coast for research have ever been genetically assigned to the East Pacific nesting beach subpopulation.

Leatherbacks nesting in the Eastern Pacific (primarily in Mexico and Costa Rica, and to a lesser extent, Nicaragua) migrate thousands of miles into tropical and temperate waters of the South Pacific (Eckert and Sarti 1997; Shillinger et al. 2008). Tagging studies have shown that eastern Pacific post-nesting females migrate southward to the south Pacific after nesting in Costa Rica (Shillinger et al. 2008, 2011; Figure 2). The adult turtles commonly forage offshore in the South Pacific Gyre in upwelling areas of cooler, deeper water and high productivity (Shillinger et al. 2011). During the nesting season, they stay within the shallow, highly productive, continental shelf waters (Shillinger et al. 2010). There are also data on at-sea distribution that were collected via observers and fishers onboard fishing vessels in the eastern Pacific. The primary data available were developed by the Inter-American Tropical Tuna Commission (IATTC) and shows a wide distribution of leatherback sea turtles throughout the eastern Pacific, ranging from the Gulf of California, Mexico to Peru (IATTC 2012). However, genetic analyses of juvenile and adult leatherback sea turtles caught in fisheries off Peru and Chile indicate that a proportion (approximately 16% of sampled turtles) are from West Pacific rookeries (Donoso and Dutton 2010; NMFS and USFWS 2013).

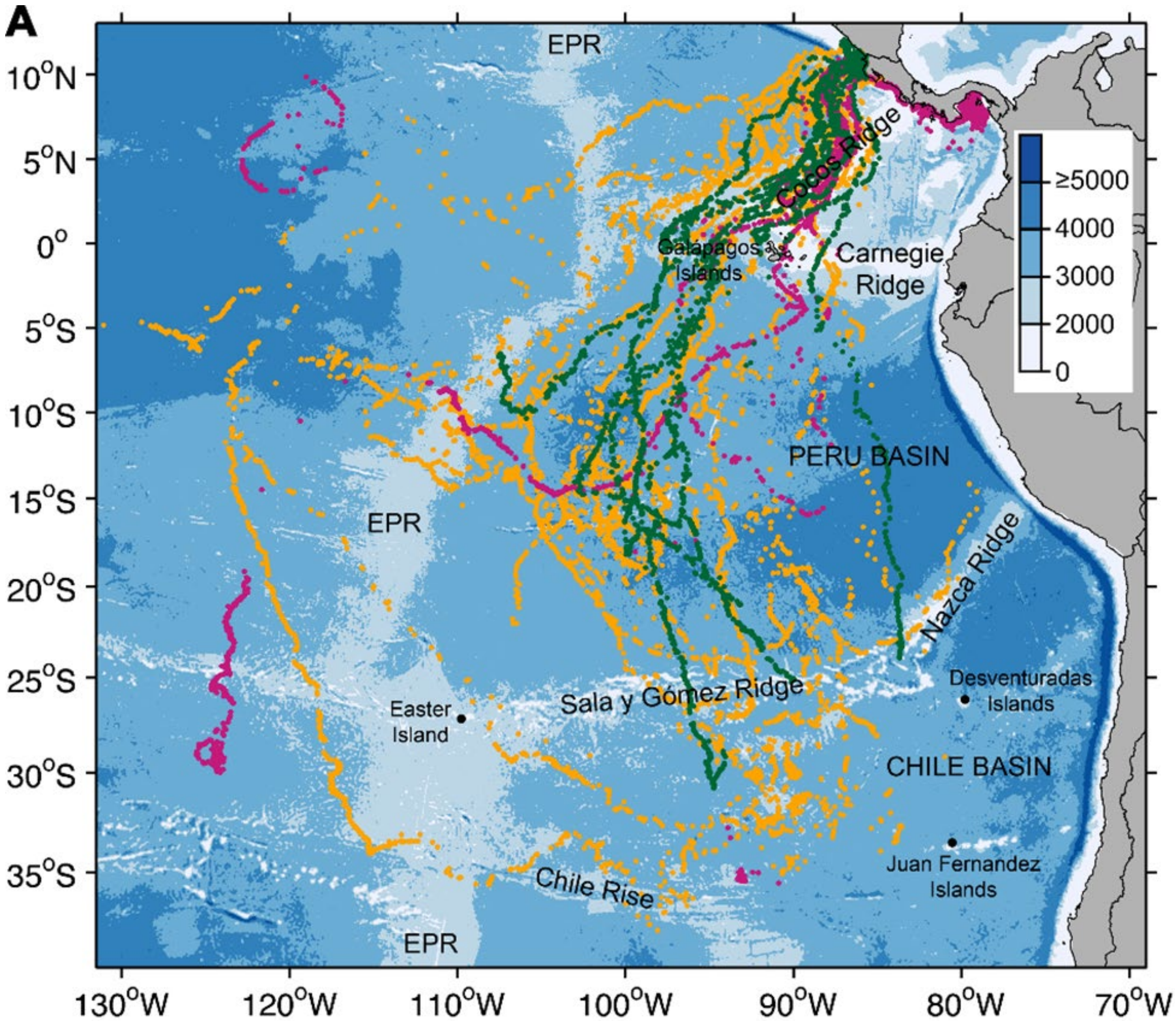

Figure 2. Satellite tracks for 46 post-nesting female leatherback sea turtles from the East Pacific population and nesting at Playa Grande, Costa Rica. Colors indicate the year of deployment: 2004 (*n* = 27, orange), 2005 (*n* = 8, purple), and 2007 (*n* = 11, green; From Shillinger et al. 2008).

The IUCN Red List conducted its most recent assessment of the West Pacific Ocean subpopulation in 2013 and listed it as "Critically Endangered" due in part to its continual decline in nesting, the continued threat due to fishing, and the low number of estimated nesting females. Genetic samples from leatherback sea turtles interacting with the CA DGN fishery indicate that all of these individuals are from the West Pacific population (P. Dutton, personal communication, SWFSC, unpublished data).

<u>Population Status and Trends</u>: Leatherbacks occur throughout the world and populations and trends vary in different regions and nesting beaches. In 1980, the leatherback population was approximately 115,000 (adult females) globally (Pritchard 1982). By 1995, one estimate claimed

this global population of adult females had declined to 34,500 (Spotila et al. 1996). In 2020, NMFS and USFWS published a global status review for leatherback sea turtles. Abundance and trend estimates of nesting females for five of the DPSs not located in the Pacific Ocean indicated that all were at risk of extinction. The Northwest Atlantic DPS has a total index of nesting female abundance of 20,659 females, with a moderate level of confidence. This DPS exhibits a decreasing nest trend at nesting beaches with the greatest known nesting female abundance. For the Southwest Atlantic DPS, NMFS and USFWS estimated only 27 females, with most nesting occurring in Brazil and exhibiting an increasing, although variable nest trend. The Southeast Atlantic DPS was estimated to have 9,198 nesting females, with most nesting in Gabon where a declining nest trend has been observed at this largest nesting aggregation. The Southwest Indian Ocean DPS was estimated to have 149 nesting females with an overall nesting trend to be slightly decreasing. Lastly, the Northeast Indian DPS total index of nesting female abundance was estimated to be 109 females with a declining trend, particularly with the extirpation of its largest nesting aggregation in Malaysia (NMFS and USWFS 2020b).

In the Pacific, leatherback populations are declining at all major Pacific basin nesting beaches, particularly in the last three decades (Spotila et al. 1996; Spotila et al. 2000; NMFS and USFWS 2007b; NMFS and USFWS 2020b).

East Pacific leatherbacks

Using the best available data for the East Pacific population, NMFS and USFWS (2020b) estimate that there are approximately 755 adult females in the East Pacific population with 76% of nesting occurring on beaches in Mexico (572 females), 22% (165 females) in Costa Rica and 2% (18 females) in Nicaragua. This estimate, 755 adult females, is based on index beaches that comprise approximately 75% of the total nesting for the population (NMFS and USFWS 2020b); therefore, we estimate a total of 1,007 adult females. Assuming a sex ratio of 79% female (Santidrian Tomillo et al. 2014) suggests a total of 1,274 adults in 2020 inclusive of both males and females. We do not have data to assess the total population size; however, based on data in Table 2 of Jones et al. (2012), we expect that adults comprise a mean of 2.1% (CI: 1.3% to 3.7%) of the total population size, which would suggest a total population size of 60,611 (CI: 34,050 to 95,462) individuals in 2020.

This population is declining, with a 97.4 % decline since the 1980s or 1990s (Wallace et al. 2013a). The declines have generally not been reversed despite intense conservation efforts (NMFS and USFWS 2020b). Where there were enough data to estimate trends (at least 9 years of data), NMFS and USFWS (2020b) estimated mean trends with 95% CI as specified in 4. Historically, the majority of nesting in Costa Rica has occurred at Las Baulas; however, nesting at this beach has been depleted and to date has not shown any signs of recovery, with nesting from 2010 to 2015 ranging from 22 to 38 nesting females per year (NMFS and USFWS 2020b). For some beaches, trends from 2011 through 2016 (the end of the time series) suggest an increase; however, there are not enough data to determine if this reflects interannual variation or a true change in trends. Overall, the current and potential future trend for the population is uncertain, and additional years of data are needed to ascertain if recovery is occurring in Mexico. Given that the majority of nesting for the population is currently occurring in Mexico, we consider the declining trend of -4.3% per year at Cahuitan to be the worst-case scenario because it is the lowest population growth rate (i.e., highest rate of decline) in Mexico. The highest measured rate of decline for the East Pacific population is -15.5% per year at what was historically the primary nesting beach, Las Baulas Costa Rica; however, given the current low levels of nesting at this beach, it is not clear that this rate of decline is the most representative of the population.

Table 4. Trends in nesting females for nesting beaches in the East Pacific leatherback population with at least 9 years of data (from NMFS and USFWS 2020b).

Beach	Years	Low/High Nest Numbers	Mean Annual Trend	95% Confidence Interval
		Mexico		
Tierra Colorada	1996-2017	12/503	0.6%	-17.1 to 18.9%
Barra de la Cruz/ Grande	1996-2016	5/365	+9.5%	-6.5 to 25.8%
Cahuitan	1997-2016	4/75	-4.3%	-22.1 to 17.6%
		Costa Rica		
Las Baulas	1988-2015	22/1,504	-15.5%	-23.1 to -7.8%

Western Pacific leatherbacks

The Western Pacific leatherback metapopulation that nests in Indonesia, Papua New Guinea, Solomon Islands and Vanuatu harbors the last remaining nesting aggregation of significant size in the Pacific.

The leatherback status review (NMFS and USFWS 2020b) conservatively estimated adult female abundance at 1,277 individuals in 2017. This value is based only on nesting at Jamursba-Medi and Wermon beaches in Papua Barat, Indonesia, as these are the only beaches with long-term monitoring. Despite a slight uptrend in the most recent data, NMFS and USFWS (2020b) estimated the long-term trend in annual nest counts for Jamursba Medi (data collected from 2001 to 2017) at -5.7 percent annually. These two beaches likely represent between 50% and 75% of all nesting for this population (NMFS and USFWS 2020b). To assist with analysis in the Hawai'i shallow-set longline fishery biological opinion (NMFS 2019a), NMFS conducted a population viability analysis (PVA) on West Pacific leatherback sea turtles (Martin et al. 2020a, 2020b). They used the same data as the status review (NMFS and USFWS 2020b) from the Jamursba-Medi and Wermon index beaches, and used Bayesian models to impute missing data and to estimate the range of intrinsic population growth rates (r). Drawing from the resulting distribution of r values as well as the distributions of the nesting female population size at the end of the time series and a process error term, Martin et al. (2020a, 2020b) projected population trends 100 years into the future, conducting 10,000 simulations to capture the variability in projections. In the future projections, Martin et al. (2020a, 2020b) computed the proportion of simulations for which the projected number of annual nesters fell below (and remained below)

50%, 25%, and 12.5% of the estimated abundance of nesters. For the set of runs ending below a threshold, they calculated the mean, median, and 95% credible interval for the number of years until the population fell below the threshold. They also calculated the probability of the projected total reproductive females falling below each threshold at 5, 10, 25, 50, and 100 years in the future.

Results of the PVA model suggest that the adult female portion of the West Pacific leatherback sea turtle population is declining at a long-term rate of 6% per year (95% CI: -23.8% to 12.2%), and the population as indicated by the index beaches is at risk of falling to less than half of its current abundance in as few as five years (range 5-26 years, mean 12.7 years; Martin et al. 2020a). PVA modeled estimates suggest the population in 2017 from these two beaches consisted of about 790 adult female leatherback sea turtles (95% CI: 666-942) using the median values for nest counts. As trends at these beaches between 2017 and 2022 appear to be stable (Figure 3), we consider the 2017 abundance estimate to be the best estimate of current (2022) adult females for the index beaches.

The index of total nesting females in Jamursba Medi and Wermon (1,277 females) provided in the status review of the species (NMFS and USFWS 2020b) was based on a simple calculation that does not provide confidence or credible intervals. While NMFS and USFWS (2020b) determined that this index was a suitable representation of total nesting female abundance for their purposes (i.e., evaluating extinction risk), they acknowledged that the degree to which the index represents the actual abundance of nesting females is unknown. We consider the values from Martin et al. (2020a) using the median values for nest counts to be the best available estimates for abundance for two reasons. First, Martin et al. (2020a) imputed missing data for months during which data were not collected, providing a more accurate estimate of total nesting. Second, their model evaluated variation due to natural causes (i.e., changes in nesting over time due to environmental or demographic factors) and observational error (i.e., imperfect data collection; Martin et al. 2020a).

To estimate the total number of nesting females from all nesting beaches in the West Pacific, we need to consider nesting at unmonitored or irregularly monitored beaches. Approximately 50% to 75% of West Pacific leatherback nesting occurs at Jamursba Medi and Wermon beaches (Dutton et al. 2007; NMFS and USFWS 2020b). Applying the conservative estimate of 75% to the Martin et al. (2020a) estimate of 790 females in the West Pacific population would be 1,053 females with an overall 95% CI of 888 to 1,256 females.

Preliminary data from the Jamursba Medi and Wermon index beaches indicate that nest numbers were relatively stable from 2017 to 2021 (Lontoh et al. in prep) but the data are not yet available in sufficient detail to update the model of Martin et al. (2020a). Hence, we acknowledge that there is a great deal of uncertainty associated with the current status of West Pacific leatherback sea turtles, as represented by the two index beaches.

Additional but lower levels of nesting have been documented elsewhere in Indonesia, including a new monitoring program established in 2017 on Buru Island (World Wildlife Fund (WWF) 2022), plus locations in Papua New Guinea, Solomon Islands, Vanuatu and the Philippines.

Monitoring at most of these additional sites has not been going on long enough to establish trends or abundance; therefore, data from these nesting beaches cannot be used to reliably calculate those metrics at this time. An exception to this is the WWF program at Buru Island in Indonesia, where data have been consistently collected since 2017 (WWF 2022). While there is only 6 years of data available, this period does span almost two remigration intervals. These data indicate an increasing trend of 10.1% per year (CI: -26.1% to 46.3%) based on an exponential growth curve. To encompass full boreal winter and summer nesting, the nesting data are censused from October to September – thus data from 2018 represents data from October 2017 to September 2018. We note that the collection of data started in January of 2017 and the nest number for 2017 is missing data from October to December 2016 and therefore does not represent a full year of data. Using the same method to calculate total adult females as Martin et al. (2020a; remigration interval multiplied by the average of the last 4 years of nesters; see Equation 13 in Martin et al. 2020a), there are approximately 103 adult females nesting at Buru Island, which would constitute an addition to the modeled estimate of 790 annual nesting females at Jamursba Medi and Wermon in 2017 (Martin et al. 2020a). Assuming a 73% female sex ratio (Benson et al. 2011) and based on NMFS' PVA results for median nest counts, the total number of adult leatherback sea turtles in the West Pacific Ocean population would be 1,443 ([790/0.73]/0.75; 95% CI: 1,216-1,720) if the index beaches represent 75% of the population.

Based on the estimates presented in Jones et al. (2012) for all Pacific populations, NMFS inferred an estimated West Pacific leatherback total population size (i.e., juveniles and adults) of 250,000 (95 CI: 97,000-535,000) in 2004. Based on the relative change in the estimates derived from Jones et al. (2012) and the more recent Martin et al. (2020a), NMFS estimates the juvenile and adult population size of the West Pacific leatherback population is around 100,000 sea turtles (95 percent CI: 47,000-195,000). As nesting numbers have been stable since 2017, we assume these abundance estimates are representative of 2022 abundance estimates as well.

The Western Pacific population has been exhibiting low hatchling success and decreasing nesting population trends due to past and current threats (NMFS and USFWS 2020b). The low estimated nesting female abundance of the West Pacific population places it at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback and catastrophes. These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb impacts to habitat or losses in individuals. Low site fidelity, which is characteristic of the species, results in the dispersal of nests among various beaches. This may help to reduce population level impacts from threats which may disproportionately affect one area over another, but may also place nests in locations that are likely unmonitored and not protected from human poaching or predation, thereby increasing threats to the population. Due to its small size, this population has restricted capacity to buffer such losses (NMFS and USFWS 2020b).

Tapilatu et al. (2013) found a 78% decline in nesting from 1984 to 2011 at Jamursba Medi and a 62.8% decline in nesting in Wermon from 2002 to 2011. Overall they estimated a 5.9% per year decline in nesting abundance for both nesting beaches over this time period. The median trend in annual nest counts estimated for Jamursba Medi nesting beaches from data collected from 2001-2017 was -5.7 percent annually (95% CI: -16.2% to 5.3%; NMFS and USFWS 2020b). The median trend in annual nest counts estimated for Wermon nesting beaches from data collected from 2006-2017 (excluding 2013-2015 due to low or insufficient effort) was -2.3 percent annually (95% CI: -19.8 to 14.9%; NMFS and USFWS 2020b). As previously described, Martin et al. (2020a) estimated the combined trends for Jamursba Medi and Wermon to be a mean of -6.0% annually (95% CI: -24.1 to 12.2%). We note that the nesting data in Figure 3 from 2018 to 2021 are preliminary and only provided to NMFS from the authors (Lontoh et al. in prep) as the figure shown in Figure 3. Until we receive the detailed raw monthly data from the nesting beaches, the growth trend analysis of Martin et al. (2020a, 2020b) cannot be updated. Therefore, since we do not have any updated modeled estimates of future growth rates based on this new information, we rely on the estimates of Martin et al. (2020a, 2020b) for current population growth rates. In addition, given the substantial declines in the population from 1984 to 2012, the data from 2012 to 2021 are likely not of long enough duration to definitely state that the population is now at least stable, but this may be reevaluated when raw data are available for analysis. NMFS (2023c) notes that New Zealand shallow-set longline fishery has shown a marked increase in leatherback interactions, from a low of one in 2008 to a high of 50 in 2022, which indicates a significant positive trend of 19.9% per year (CI: 8.4% to 31.3%). While fishery captures can be influenced by numerous environmental factors that can disconnect them from population trends, NMFS postulates that the strength of the trend suggests the potential for more leatherback turtles in the water in recent years.

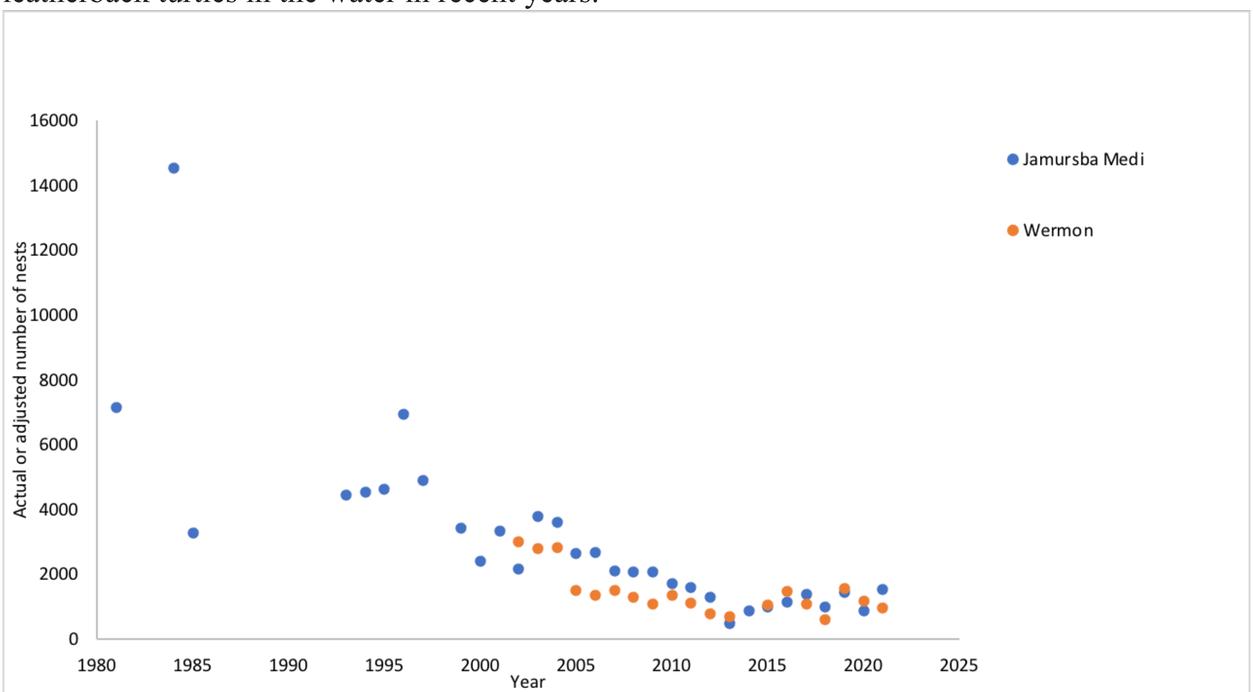

Figure 3. Actual and adjusted number of leatherback nests between 1981 and 2021 at Jamursba Medi and Wermon. Each year represents nests laid from April of one year to March of the following year (Lontoh et al. in prep).

Although human interactions are a major source of mortality for this declining population, there are indications that natural fluctuations in environmental and oceanic conditions could be significant influences on survival rates across various life stages or on reproductive rates (Van Houtan 2011; Tomillo et al. 2012).

Satellite tracking of post-nesting females and foraging males and females, as well as genetic analyses of leatherback turtles caught in U.S. Pacific fisheries or stranded on the west coast of the U.S., along with stable isotope analysis, all indicate that all of the leatherbacks found off the U.S. West Coast are from the western Pacific nesting populations, specifically boreal summer nesters. Approximately 38-57 percent of summer-nesting females from Papua Barat migrate to distant foraging grounds off the U.S. West Coast, including the neritic waters off central California. Researchers recently assessed the abundance and trend of leatherbacks foraging off central California using 28 years of aerial survey data from coast-wide and adaptive fine-scale surveys (Benson et al 2020). Results indicate that leatherback abundance has declined at an annual rate of -5.6% (95% credible interval of -9.8% to -1.5%) to less than 200 individuals.

Martin et al. (2020a) estimated the mean and median time until the West Pacific population declines to 50 percent, 25 percent, and 12.5 percent of its 2017 estimated abundance, and Siders et al. (2023) updated these results to 2021, assuming the population declined at a rate of 6% per year from the 2017 estimates. Results of this updated modeling effort indicate that the adult female portion of West Pacific leatherbacks nesting at Jamursba Medi and Wermon beaches are predicted to decline to 50 percent of their 2017 abundance in a mean of about 9 years beginning in 2021 (or by about 2030; CI from 1 to 22 years) and to 25 percent of their 2017 abundance in a mean of about 20 years (or by about 2041; 95% CI from 8 to 37 years). Again, these estimates assume a mean decline of 6% per year since 2017, which may not be accurate given some of the recent nesting data.

Threats: Leatherback sea turtles are probably already beginning to be affected by impacts associated with climate change given low hatch success due to lethal beach temperatures and beach erosion (Tapilatu and Tiwari 2007; Bellagio Steering Committee 2008; PLAWG 2012; NMFS and USFWS 2013b). Over the long-term, climate change-related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003).

Natural factors, including the 2004 tsunami in the Indian Ocean (see detailed report by Hamann et al. 2006) and the tsunami that affected Japan in 2011, may have impacted leatherback nesting beach habitat through encroachment and erosion (2004 tsunami) or may have resulted in increased debris into leatherback marine habitat (e.g., impacting migratory routes and foraging hotspots). Shifting mudflats in the Guianas have also made nesting habitat unsuitable (Crossland 2003; Goverse and Hilterman 2003).

Predation on sea turtle hatchlings by birds and fish (see Vose and Shank 2003) has been commonly reported. Reported predation of leatherback hatchlings includes tarpons (Nellis and Henke 2000), gray snappers (Vose and Shank 2003), ghost crabs, great blue and yellow-crowned herons, and crested caracaras (Santidrian Tomillo et al. 2010). Adult leatherbacks are preyed upon by large predators, such as jaguars, tigers, killer whales, sharks, and crocodiles (reviewed by Eckert et al. 2012).

Major anthropogenic threats to the species, are fisheries bycatch, direct harvest, alteration of nesting habitat, and predation, which are briefly described below (NMFS and USFWS 2020b). In addition, habitat changes attributed to changing environmental conditions (i.e., sand temperatures that result in mortality or changes in sex ratios, erosion), pollution and marine debris are also threats to this species (Tiwari et al. 2013).

The drivers of these species decline - both anthropogenic (e.g., bycatch, egg harvest, exploitation of females) as well as environmental (e.g., lethal sand temperatures, predation, erosion) - have been described in detail (Eckert 1993; Bellagio Steering Committee 2008; Tapilatu and Tiwari 2007; Tapilatu et al. 2013). Egg harvest and exploitation of females have been minimized at the two most significant nesting beaches of Papua Barat, Indonesia, and the impact of environmental factors is being addressed through a science-based management and conservation programme. Fisheries bycatch is still considered the major obstacle to this population's recovery (Benson et al. 2011; Bailey et al. 2012; Tapilatu et al. 2013; Wallace et al. 2013b).

In Indonesia, the poaching of turtles and eggs continues, though egg harvest and exploitation of females has been minimized at Jamursba Medi and Wermon beaches due to the presence of monitoring programs and educational outreach (NMFS and USFWS 2020b). Before the monitoring programs, approximately 4 to 5 boats per week (from May to August) collected 10,000 to 15,000 eggs per boat at Jamursba Medi between 1980 and 1993 (Tapilatu et al. 2013). While the commercial egg harvest has been effectively eliminated since beach monitoring was established at Jamusrba Medi in 1993 (Hitipeuw et al. 2007), recent survey efforts suggest that most, if not all, sea turtle eggs (including leatherback turtles) are poached at other Bird's Head Peninsula beaches and sold in local markets (Tapilatu et al. 2017). Between 2016 and 2017, eight females were poached at Buru Island, Indonesia, (WWF 2018), and it is likely that three to five nesting females have been killed annually over the past two decades (J. Wang, NMFS, pers. comm., 2018). In 2017, 114 of 203 leatherback nests were harvested at Buru Island (WWF 2018). In 2018, due to education provided by the newly established WWF program on Buru Island, local community-based efforts in four villages now prohibit adult female and egg harvest. Indonesian laws prohibit the harvest of sea turtles and eggs; however there is very little enforcement of these laws in areas where monitoring programs to not exist (NMFS and USFWS 2020b).

In the Western Pacific, leatherbacks are also subjected to traditional harvest, which was well documented in the 1980s and continues today. Traditional hunters from the Kei Islands continue to kill leatherbacks for consumption and ceremony. Recent surveys indicate that harvest continues with estimates of 431 mortalities over the past 8 years (53.9/yr), and 104 leatherbacks harvested in 2017 (WWF 2018 as cited in NMFS and USFWS 2020). Since 2017, the harvest has declined significantly from the high of over 100 leatherbacks in 2017, to less than 25 in 2019-2021, including only 9 turtles in 2021 (92 percent reduction; J. Wang, NMFS-PIFSC, personal communication, 2022).

Leatherbacks are vulnerable to bycatch in a variety of fisheries, including longline, drift gillnet, set gillnet, bottom trawling, dredge, and pot/trap fisheries that are operated on the high seas or in coastal areas throughout the two species' range. Off the U.S. west coast, a large time/area closure was implemented in 2001 to protect Pacific leatherbacks by restricting the DGN fishery, which significantly (at least 80%) reduced bycatch of leatherbacks in that fishery. On the high seas, bycatch in longline fisheries is considered a major threat to leatherbacks (Lewison et al. 2004). In addition to anthropogenic factors, natural threats to nesting beaches and marine habitats such as coastal erosion, seasonal storms, predators, temperature variations, and phenomena such as El Niño also affect the survival and recovery of leatherback populations (Eckert et al. 2012).

There are interactions between leatherbacks and domestic longline fishing for tuna and swordfish based out of Hawai'i. Under requirements established in 2004 to minimize sea turtle bycatch (69 FR 17329), vessel operators in the Hawai'i-based shallow-set swordfish fishery must use large (sized 18/0 or larger) circle hooks with a maximum of 10 degrees offset and mackerel-type bait. In addition, NMFS requires 100% observer coverage in this fishery, so every interaction is observed. From 2012-2017, the incidental take statement for the Hawai'i-based shallow-set fishery was 26 leatherback sea turtles per year, which served as the "hard cap" for the fishery that requires closure of the entire fishery during any year if reached. Recently, the hard cap for leatherback sea turtle bycatch was reset to 16 per year, with the expectations that up to 16 may be caught and 3 may be killed each year, and that vessels would be restricted to no more than 2 leatherbacks taken during any one trip (NMFS 2019a). Between 2004 and 2022, there were a total of 121 leatherback sea turtles captured in the Hawai'i-based shallow-set fishery, with zero leatherback sea turtles observed killed as a result, but an estimated 21% of those killed given post-interaction mortality estimates (NMFS 2019a; updated in NMFS 2023c). From 2004-2018, NMFS estimated that the Hawai'i-based shallow set fishery annually interacted with around 21 leatherbacks/year, with an estimated 3 dead per year (given also post-interaction mortality) (NMFS 2019a).

From 2004-2022, the Hawai'i-based deep-set longline fishery (~20% observer coverage) was observed to interact with 46 leatherbacks, with an estimated 246 taken (around 13/year). On average, a mean of 17 (95th percentile: 43) were anticipated to be captured. When at-vessel and post-release mortality rates are combined, the effective mortality rate in this fishery is 35%, resulting in a mortality rate of 6 turtles to die each year. The anticipated take level (incidental take statement) over a 5-year period (running sum) was 92 leatherbacks (interactions, injuries and/or mortalities) (NMFS 2023c). In the American Samoa longline fishery, NMFS anticipated a mean of 10 leatherbacks to be taken each year, given observer data from 2010 to 2019. With a mortality rate of 65%, approximately 7 leatherbacks would be killed per year (NMFS 2023d).

Estimating the total number of sea turtle interactions in other Pacific fisheries that interact with the same sea turtle populations as U.S. fisheries is difficult because of low observer coverage and inconsistent reporting from international fleets. Lewison et al. (2004) estimated 1,000 – 3,200 leatherback mortalities from pelagic longlining in the Pacific in 2000. Beverly and Chapman (2007) more recently estimated loggerhead and leatherback longline bycatch in the Pacific to be approximately 20 percent of that estimated by Lewison et al. (2004), which would equate to 200 – 640 leatherbacks during that time period. Chan and Pan (2012) estimated that there were approximately 1,866 total sea turtle interactions of all species in 2009 in the central and North Pacific by comparing swordfish production and turtle bycatch rates from fleets fishing in the central and North Pacific area. In 2015, a workshop was convened to analyze the effectiveness of sea turtle mitigation measures in the tuna RFMOs and 16 countries (including the United States, which reported 27% of the interactions) provided data on observed sea turtle interactions and gear configurations in the Western Central Pacific Ocean. Based on the information gathered there, 331 leatherback sea turtles reported with a total estimate of 6,620 leatherbacks caught in the region from 1989-2015 in these countries (mortality rates were not reported (Common Oceans (ABNJ) Tuna Project 2017). Most recently, Peatman et al. (2018) estimated that 8,362 leatherbacks (annually 557 (95% CI: 439-676) were captured in longline fisheries operating in the North Pacific from 2003-2017. Finally, bycatch estimates of sea turtles were summarized from annual reports by the WCPFC (2021). Sea turtle data included U.S. fishery data, with the Hawai'i-based deep-set longline fishery representing 5 to 6% of the total hooks set by Western Central Pacific Ocean longline fisheries. From 2013 to 2020, an average of 722 leatherbacks (CI: 468 – 976) were caught annually, with an estimated 76 leatherbacks (CI: 16 – 136) killed per year. With low observer coverage in these international fleets (~3%), confidence in these estimates are low. Nonetheless, we have more confidence in understanding the effects of our domestic longline fisheries, given 100% observer coverage in the Hawai'i-based shallow-set fishery and approximately 20% observer coverage in the Hawai'i-based deep-set fishery and variable coverage in the American Samoa longline fishery.

Given that recent developments to reduce sea turtle bycatch in fisheries have been working their way into some international fisheries, and the incomplete data sets and reporting that exist, the exact level of current sea turtle bycatch internationally is not clear. However, given the information that is available, we believe that international bycatch of sea turtles in fisheries throughout the Pacific Ocean continues to occur at significant rates several orders of magnitude greater than what NMFS documents or anticipates in domestic U.S. Pacific Ocean fisheries.

In an attempt to develop a tool for managers to use locally (e.g. in an EEZ) to reduce threats in a particular area of interest, Curtis et al. (2015) developed biological "limit reference points" for western Pacific leatherback turtles in the U.S. west coast EEZ, similar to a PBR approach calculated for marine mammal stocks. Depending on the model used and the various objectives sought (e.g. achievement of maximum net productivity, or no more than a 10% delay in the time for the population to rebuild) and incorporation of conservative assumptions accounting for broad uncertainty in abundance and productivity estimates, the limit reference point estimate for human-caused removals in the U.S. west coast EEZ ranged from 0.8 to 7.7 leatherbacks over 5 years. Although these results are useful for consideration, NMFS is not currently using this approach to managing threats to sea turtles foraging within the U.S. EEZ pending further discussion of how this approach or other approaches relate to the standards of the ESA. We anticipate that the management tool presented by Curtis et al. (2015) and other approaches to managing threats to sea turtles will be continue to be subject to future discussion by scientific and policy experts.

<u>Conservation</u>: Considerable effort has been made since the 1980s to document and address leatherback sea turtle bycatch in fisheries around the world. In the United States, observer programs have been implemented in most U.S. federally managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-interaction mortality. These include developing gear solutions to prevent or reduce capture (e.g., circle

hooks in combination with fin-fish bait for longline fisheries) or to allow turtles to escape without harm (e.g., turtle exclusion devices in trawl fisheries), implementing seasonal time-area closures to prevent fishing when turtles are congregated, modifying existing gear (e.g., reducing mesh size of gillnets), and developing and promoting Sea Turtle Handling Guidelines. For example, switching to large circle hooks and mackerel-type bait in 2004 with complimentary fishery-based outreach and education resulted in an 84% reduction in the leatherback sea turtle interaction rate in the Hawai'i SSLL fishery (Swimmer at al. 2017). In addition, in 2020, NMFS issued a final rule for the SSLL that reduced the annual interaction limit from 26 to 16 for leatherbacks, and included trip (not more than 2 leatherbacks per vessel trip) and vessel (vessels that reach the trip limit twice in a calendar year are prohibited from the fishery for the remainder of the year) limits (85 FR 57988).

NMFS developed a 5-year action plan (2016-2020), identifying the top five recovery actions to support this "Species in the Spotlight" (species listed under the ESA for which immediate, targeted efforts are vital for stabilizing their populations and preventing their extinction) over the next five years: (1) reduce fishery interactions; (2) improve nesting beach protection and increase reproductive output; (3) international cooperation; (4) monitoring and research; and (5) public engagement (NMFS 2016c). This initiative was recently renewed in 2021 for 2021-2025 (NMFS 2021d).

Community-based conservation projects in Wermon and Jamursba-Medi in Papua, Barat, Papua New Guinea, Solomon Islands, and Vanuatu in the West Pacific population and in Mexico, Costa Rica and Nicaragua in the East Pacific Population have been developed that monitor nesting and protect nests from harvest and predation, increasing the production of hatchlings from these nesting areas.

Specifically, with the continuing conservation efforts at Jamursba Medi and Wemon, hatchling production from 2017 to 2019 between April and September alone (i.e., exclusive of the Wermon boreal winter season) increased to 32,000-50,000 hatchlings per year in contrast with a mean hatchling production of 21,966 from 2005 to 2013 (Tapilatu 2014; Pakiding et al. 2020). This is due in part to increased effort to protect nests from predation, tidal inundation, erosion and high sand temperatures. Nest success rates increased from about 35% prior to 2017 to over 50% since 2017, including the stable numbers of nesting females ranging from 87 to 279 annually at Jamursba Medi and 109 to 285 annually at Wermon. At Buru Island, a multi-year action plan, developed with the involvement of local government agencies, local village elders, and community members continues to be implemented. When the plan was first implemented in 2017, over 60% of nests were being poached or predated, with nesting females also taken. By 2022, less than 1% of nests were being poached, with no nesting females taken, with benefits continuing into the present (ESA Bienniel Report to Congress, 2023).

In partnership with NOAA's PIFSC and PIRO, WWF-Indonesia actively works to monitor and reduce the poaching of leatherback turtles in the Kei islands, Indonesia. As mentioned above, over 100 leatherback turtles were harvested annually during certain years, with numbers varying over the years. Over the past 4 seasons (2017-2022), the project has documented a reduction of leatherback takes by 86% (ESA Bienniel Report to Congress, 2023).

The conservation and recovery of leatherback sea turtles is facilitated by a number of regulatory mechanisms at international, regional, national and local levels, such as the FAO Technical Consultation on Sea Turtle-Fishery Interactions, the IAC, CITES, and others. In 2008, the WCPFC adopted [CMM 2008-03] to mitigate the impacts on turtles from longline swordfish fisheries in the Western Central Pacific Ocean. In 2018, the WCPFC adopted CMM 2018-04 to mitigate the impacts of the purse seine fisheries on sea turtles.

As a result of these designations and agreements, many intentional impacts on sea turtles have been reduced: harvest of eggs and adults have been reduced at several nesting areas through nesting beach conservation efforts (although significant more effort is needed to reduce harvest pressure), and a number of community-based initiatives have helped reduce the harvest of turtles in foraging areas.

2.2.6 Olive Ridley sea turtle

Two populations of olive ridleys were listed under the ESA in 1978 (43 FR 32800; July 28, 1978): the breeding colony populations on the Pacific coast of Mexico was listed as endangered, and all other olive ridleys found other than on the Pacific coast of Mexico were listed as a threatened species. Since olive ridleys found off the U.S. West Coast are likely to originate from Pacific Mexican nesting beaches, we assume that any olive ridleys affected by the proposed action are endangered. A recovery plan for the U.S. Pacific populations of olive ridleys was completed nearly 20 years ago (NMFS and USFWS 1998c). A 5-year status review of olive ridley sea turtles was completed in 2014 (NMFS and USFWS 2014).

Olive ridley sea turtles occur throughout the world, primarily in tropical and sub-tropical waters. Nesting aggregations in the Pacific Ocean are found in the Marianas Islands, Australia, Indonesia, Malaysia, and Japan (western Pacific), and Mexico, Costa Rica, Guatemala, and South America (eastern Pacific). Like leatherback turtles, most olive ridley sea turtles lead a primarily pelagic existence (Plotkin et al. 1993), migrating throughout the Pacific, from their nesting grounds in Mexico and Central America to the deep waters of the Pacific that are used as foraging areas (Plotkin et al*.* 1994). While olive ridleys generally have a tropical to subtropical range, with a distribution from Baja California, Mexico to Chile (Silva-Batiz et al. 1996), individuals do occasionally venture north, some as far as the Gulf of Alaska (Hodge and Wing 2000). Olive ridleys live within two distinct oceanic regions including the subtropical gyre and oceanic currents in the Pacific. The gyre contains warm surface waters and a deep thermocline preferred by olive ridleys. The currents bordering the subtropical gyre, the Kuroshio Extension Current, North Equatorial Current and the Equatorial Counter Current, all provide for advantages in movement with zonal currents and location of prey species (Polovina et al. 2004). In the eastern Pacific, the post-reproductive migrations of olive ridleys are unique and complex. Their migratory pathways vary annually, there are no apparent migratory corridors, and there is no spatial and temporal overlap in migratory pathways among groups or cohorts of turtles (NMFS and USFWS 2014). Unlike other sea turtles that show site fidelity from a breeding ground to a single feeding area, where they reside until the next breeding season, olive ridleys are nomadic migrants that swim thousands of miles over vast oceanic areas. This nomadic behavior may be unique to olive ridleys in the eastern Pacific Ocean, as studies in other ocean basins indicate these species occupy neritic waters, not making extensive migrations observed in the eastern Pacific.

Individual olive ridleys experience three different reproductive strategies or behaviors: mass or *arribada* nesting, dispersed or solitary nesting, and a mixed strategy of both.

<u>Population Status and Trends</u>: It is estimated that there are over 1 million female olive ridley sea turtles nesting annually at one of the major beaches (arribada) in Mexico (La Escobilla) (NMFS and USFWS 2014). Unlike other sea turtle species, most female olive ridleys nest annually. According to the Marine Turtle Specialist Group of the IUCN, there has been a 50 percent decline in olive ridleys worldwide since the 1960s, although there have recently been substantial increases at some nesting sites (NMFS and USFWS 2007c). A major nesting population exists in the eastern Pacific on the west coast of Mexico and Central America. Both of these populations use the north Pacific as foraging grounds (Polovina et al. 2004). As described above, because the proposed action is most likely to occur closer to eastern Pacific nesting and foraging sites, we assume that this population would be more likely (i.e., than the western Pacific population) to be affected by the proposed action, and that any affected turtles may have originated from the endangered Mexican breeding population. The eastern Pacific population is thought to be increasing, while there is inadequate information to suggest trends for other populations. Eastern Pacific olive ridleys nest primarily in large *arribadas* on the west coasts of Mexico and Costa Rica. Since reduction or cessation of egg and turtle harvest in both countries in the early 1990s, annual nest totals have increased substantially.

Based on the current number of olive ridleys nesting in Mexico, three *arribada* beaches appear to be stable (Mismaloya, Tlacoyunque, and Moro Ayuta), two are increasing (Ixtapilla, La Escobilla) and one is decreasing (Chacahua), but none of these populations have recovered to their pre-1960s abundance. At the major *arribada* nesting beach, La Escobilla, olive ridleys rebounded from approximately 50,000 nests in 1988 to over 700,000 nests in 1994, and more than a million nests by 2000. From 2001-2005, Abreu-Grobois and Plotkin (2008) estimated a mean annual estimate of over one million females nesting annually at Escobilla. Minor *arribada* nesting beaches in Mexico range from around 2,000 nests (Chacahua) to 10,000-100,000 nests (Moro Ayuta) (NMFS and USFWS 2014).

Regarding non-*arribada* beaches, population trends for most indicate they are stable or increasing. Stable beaches include El Verde, Maruata-Colola, Puerto Arista, and Moro Ayuta. Increasing trends are reported for Platanitos and Cuyutlán (Abreu-Grobois and Plotkin 2008). These increases observed on the nesting beaches are supported by at-sea estimates of density and abundance. Eguchi et al. (2007) analyzed sightings of olive ridleys at sea, leading to an estimate of 1,150,000 – 1,620,000 turtles in the eastern tropical Pacific in 1998-2006. In contrast, there are no known *arribadas* of any size in the western Pacific, and apparently only a few hundred nests scattered across Indonesia, Thailand, and Australia (Limpus and Miller 2008).

<u>Threats</u>: Threats to olive ridleys are described in the most recent five year status review (NMFS and USFWS 2014). Direct harvest and fishery bycatch are considered the two biggest threats. In

the 1950s through the 1970s, it is estimated that millions of olive ridleys were killed for meat and leather and millions of eggs were collected at nesting beaches in Mexico, Costa Rica, and other locations in Central and South America. Harvest has been reduced in the 1980's and 1990's, although eggs are still harvested in parts of Costa Rica and there is an illegal harvest of eggs in parts of Central America and India (NMFS and UWFWS 2014).

Olive ridleys have been observed caught in a variety of fishing gear including longline, drift gillnet, set gillnet, bottom trawl, dredge and trap net. Fisheries operating in coastal waters near *arribadas* can kill tens of thousands of adults. This is evident on the east coast of India where thousands of carcasses wash ashore after drowning in coastal trawl and drift gillnets fishing near the huge arribada (NMFS and USFWS 2007c). Based upon available information, it is likely that olive ridley sea turtles are being affected by climate change through sea-level rise and rising sea surface temperatures as well as related changes in ice cover, salinity, oxygen levels and circulation. Impacts from climate change could include shifts in ranges and changes in algal, plankton and fish abundance, which could affect olive ridley prey distribution and abundance. However, olive ridleys are wide ranging and could shift from an unproductive habitat to more biologically productive waters. Sea level rise and other environmental and oceanographic changes such as the frequency and timing of storms may accelerate the loss of suitable nesting habitats could increase beach loss via erosion or inundation of nests (NMFS and USFWS 2014).

<u>Conservation</u>: The conservation and protection of olive ridleys is enhanced by a number of regional and local community conservation programs. Efforts to decrease or eliminate poaching of nesting females and eggs and protect their habitat have been implemented in many areas of Mexico. In 1986, Mexico established 17 reserve areas to protect sea turtles. In 1990, Mexico banned the harvest and trade of sea turtles. Mexico requires the use of turtle excluder devices in their shrimp fishery to reduce sea turtle bycatch. Local community efforts are numerous. For example, the nongovernmental organization, Grupo Tortuguero, established 30 community sites for monitoring beaches and in-water surveys along the Baja Peninsula and Gulf of California (Esliman et al. 2012). In the state of Nayarit, Mexico, there are seven centers for Sea Turtle Protection and Conservation and two Sea Turtle Protection Camps covering nearly 80 km of nesting beaches (Maldonado-Gasca and Hart 2012).

The U.S. implemented several fisheries regulations that remain in effect to reduce sea turtle bycatch including olive ridleys. For example, all commercial fishermen in the U.S. who incidentally take a sea turtle during fishing operations must handle the animals with due care to prevent injury to live sea turtles, resuscitate (if necessary), and return safely to the water. No sea turtles may be consumed, sold, landed, kept below deck, etc. The U.S. Hawai'i-based longline fishery operating in the central Pacific also incidentally takes olive ridleys from the endangered populations (NMFS 2008). Olive ridley interaction and mortality rates have been reduced by requiring specific gear configurations and operational requirements that include use of circle hooks and non-squid bait; fishery closures based on maximum annual turtle interaction limits; area restrictions; proper handling of hooked and entangled turtles; use of disentangling and dehooking equipment such as dip nets, line cutters, and de-hookers; and reporting sea turtle interactions. Vessel owners and operators are also required to participate in protected species workshops to raise awareness of sea turtle ecology and ensure compliance with sea turtle protective regulations.

As a result of these international, national, and local efforts, many of the anthropogenic threats have been lessened. The ban on direct harvest resulted in stable or increasing nesting Endangered breeding colony populations on the Pacific coast of Mexico, although the Chacahua *arribada* beach continues to decline. Conservation measures to reduce incidental bycatch have benefited the endangered populations; however, fisheries remain a concern.

2.2.7 East Pacific DPS Green Sea Turtle

In 2016, NMFS finalized new listings for 11 green sea turtle DPSs, including listing the East Pacific DPS as threatened (81 FR 20057). The East Pacific DPS includes turtles that nest on the coast of Mexico which were historically listed under the ESA as endangered. All of the green turtles DPSs were listed as threatened, with the exception of the Central South Pacific DPS, Central West Pacific DPS, and the Mediterranean DPS which were listed as endangered Seminoff et al. 2015).[12] Recently the IUCN assessed the East Pacific "regional management unit" of green sea turtles as "vulnerable," which was downlisted from a previous "endangered" status (IUCN 2021). Currently, NMFS and USWFS are considering designating critical habitat for the East Pacific green sea turtle DPS as well as several other (five) DPSs within U.S. jurisdiction. Based on a settlement agreement with several non-profit organizations, the agencies shall propose a determination concerning the designation of critical habitat on or before June 30, 2023, with the proposed rule anticipated to be published in the Federal Register around mid-July, 2023.

Throughout the Pacific Ocean, nesting assemblages group into two distinct regional areas: 1) western Pacific and South Pacific islands; and 2) eastern Pacific and central Pacific, including the rookery at French Frigate Shoals, Hawai'i. In the eastern Pacific, green sea turtles forage coastally from the U.S. West Coast (42°N) in the north, offshore in waters up to 1,000 miles from the coast, and south to central Chile (40°S). The boundaries of this DPS extend from the aforementioned locations in the U.S. and Chile, out to 143°W and 96°W, respectively (Seminoff et al. 2015). Green turtles found in the Gulf of California originate primarily from the Michoacán nesting stock. Green turtles foraging in southern California and along the Pacific coast of Baja California originate primarily from rookeries of the Islas Revillagigedos (Dutton 2003) and within the state of Michoacán (Dutton et al. 2019).

Green sea turtles in the eastern Pacific are migratory as adults, conducting reproductive migrations every three years on average between their natal nesting sites and foraging areas. Individuals show fidelity to foraging areas, often returning to the same areas after successive nesting seasons. In neritic foraging areas, green turtles in the eastern Pacific are omnivorous, consuming marine algae, seagrass, mangrove parts and invertebrates. Green turtles in the wild

[12] The 2015 biological status report that was used to support the recent listing activities (Seminoff et al. 2015) can be found at: http://www.nmfs.noaa.gov/pr/species/Status%20Reviews/green_turtle_sr_2015.pdf

are estimated to attain maturity at 15-50 years (Avens and Snover 2013), with East Pacific green turtles averaging 30 years to maturity.

The effects of climate change include, among other things, increases in sea surface temperature, the alteration of thermal sand characteristics of beaches (from warming temperatures), which could result in the reduction or cessation of male hatchling production (Hawkes et al. 2009) and a significant rise in sea level, which could significantly restrict green sea turtle nesting habitat. While sea turtles have survived past eras that have included significant temperature fluctuations, future climate change is expected to happen at unprecedented rates, and if sea turtles cannot adapt quickly, they may face local to widespread extirpations (Hawkes et al. 2009). Impacts from global climate change are likely to become more apparent in future years (IPCC 2018; 2021). However, in some areas like the primary nesting beach in Michoacán, Mexico (Colola), the beach slope aspect is very steep and the dune surface at which the vast majority of nests are laid is well elevated. This site is likely buffered against short-term sea level rise as a result of climate change. In addition, many nesting sites are along protected beach faces, out of tidal surge pathways. For example, multiple nesting sites in Costa Rica and in the Galapagos Islands are on beaches that are protected from major swells.

<u>Population Status and Trends</u>: A complete review of the most current information on green sea turtles is available in the 2015 Status Review (Seminoff et al. 2015). Based on genetic data, this DPS consists of at least five populations: two in Mexico, one in Costa Rica, one in the eastern Pacific and one in the Galapagos Islands. Those populations are represented by at least 39 nesting sites, with most of these sites concentrated in Mexico, Ecuador, and Costa Rica (Seminoff et al. 2015).

Although trend information is lacking for the majority of nesting beaches, based on a 25-year trend for the nesting aggregation at Colola, Mexico, the abundance of East Pacific green turtles appears to have increased since the population's low point in the mid-1980s. (which is the most important green turtle nesting area in the eastern Pacific). Based on nesting beach data, the current adult female nester population for Colola, Michoacan is 11,588 females, which makes this the largest nesting aggregation in the East Pacific green turtles, comprising nearly 58% of the total adult female population. The total for the entire East Pacific green turtle is estimated at 20,062 nesting females (Seminoff et al. 2015). This observed increase may have resulted from the onset of nesting beach protection in 1979, as is suggested by the similarity in timing between the onset of beach conservation and the age-to-maturity for green turtles in Pacific Mexico. Similarly, data from the Galapagos Archipelago suggest that the abundance of nesting females in that population may be increasing. Given the likely increasing trend in this population, NMFS recently estimated a total mean population size of 3,580,207 animals in the East Pacific DPS (NMFS 2023c).

Most green turtles found off the U.S. West Coast and in the action area likely originate from the Revillagigedos Archepelago, a secondary nesting site, and the coast of Michoacán, Mexico (Dutton et al. 2019). The most recent survey (2008) from Revillagigedos estimated that as many as 500 nests were laid over a 4-week period, which the most recent status review (Seminoff et al. 2015) used to estimate nester abundance at 500 females. Two foraging populations of green turtles are found in U.S. waters adjacent to the proposed action area. South San Diego Bay serves as important habitat for a resident population of up to about 60 juvenile and adult green turtles in this area (Eguchi et al. 2010). There is also an aggregation of green sea turtles that is persistent in the San Gabriel River and surrounding coastal areas in the vicinity of Long Beach and Seal Beach, California (Lawson et al. 2011; Crear et al.2016; Crear et al. 2017). Seasonal shifts in movement and distribution of green turtles in the Long Beach/Seal Beach area show that green turtles in the San Gabriel River use warm effluent from two power plants as a thermal refuge, although the river sustains juveniles and adults year-round (Crear et al. 2016).

<u>Threats</u>: A thorough discussion of threats to green turtles worldwide can be found in the most recent status review (Seminoff et al. 2015). Major threats include: coastal development (including heavy armament and subsequent erosion) and loss of nesting and foraging habitat; incidental capture by fisheries; and the harvest of eggs, sub-adults and adults. Climate change is also emerging as a critical issue. Destruction, alteration, and/or degradation of nesting and near shore foraging habitat is occurring throughout the range of green turtles. These problems are particularly acute in areas with substantial or growing coastal development, beach armoring, beachfront lighting, and recreational use of beaches. In addition to damage to the nesting beaches, pollution and impacts to foraging habitat are a concern. Pollution run-off can degrade sea grass beds that are the primary forage of green turtles. The majority of turtles in coastal areas spend their time at depths less than 5 m below the surface (Schofield et al. 2007; Hazel et al. 2009) and hence collisions with boats are known to cause significant numbers of mortality every year (NMFS and USFWS 2007d; Seminoff et al. 2015). Marine debris is also a source of concern for green sea turtles especially given their presence in nearshore coastal and estuarine habitats.

The bycatch of green sea turtles, especially in coastal fisheries, is a serious problem because in the Pacific, many of the small-scale artisanal gillnet, setnet, and longline coastal fisheries are not well regulated. These are the fisheries that are active in areas with the highest densities of green turtles (NMFS and USFWS 2007d). In the northern portions of the East Pacific DPS, bycatch in fisheries has been less well-documented. However, along the Baja California Peninsula, Mexico, green turtles were reported stranded (suspected bycatch) in the hundreds in Bahia Magdalena (Koch et al. 2006). In Baja California Sur, Mexico, from 2006-2009, small-scale gillnet fisheries caused massive green sea turtle mortality at Laguna San Ignacio, where an estimated 1,000 turtle were killed each year in a fishery targeting guitar fish (Mancini et al. 2012). Bycatch of green turtles has also been reported in Peru and Chile, and while the problem persists, innovated bycatch reduction techniques and monitoring approaches have likely reduced bycatch of all sea turtle species.

In Peru, where the fishing industry is the second largest economic activity in the country, there is evidence of sea turtle bycatch in a range of industrial fisheries. Large impacts may also result from similar interactions with small-scale fisheries, largely due to their diffuse effort and large number of vessels in operation. From 2000 to 2007, shore-based and onboard observer programs from three ports in Peru were used to assess the impacts of marine turtles of small-scale longline bottom set nets and driftnet fisheries. During this time, a total of 807 turtles were captured, of which nearly 92% were released alive. Researchers estimated that 2,400 green turtles were captured annually, and estimated that, given the low observer coverage, that the number of turtles (all species) captured per year is likely to be in the tens of thousands (Alfaro-Shigueto et al. 2011).

In the Western Central Pacific, from 1989-2015, the reported number of green turtles reported was 325 turtles, resulting in an "estimated" 6,500 turtles taken (expanded to account for 5% observer coverage used by the WCPFC, although no mortality details were available (Common Oceans (ABNJ) Tuna Project 2017). Of these green turtles, there was no further information supplied that may have apportioned the capture to the six DPSs that may be present in the area of effort. In the Hawai'i-based deep-set longline fishery, between 2004 and 2022, 25 green turtles were observed caught, adjusted to an estimated 128 green turtles taken. Over 10 years, NMFS estimated that a cumulative mean of 154 captures of East Pacific green turtles, of which 148 would be expected to die as a result of their interactions (NMFS 2023c).

The meat and eggs of green turtles has long been favored throughout much of the world that has interacted with this species. As late as the mid-1970s, upwards of 80,000 eggs were harvested every night during nesting season in Michoacán (Clifton et al. 1982). Even though Mexico has implemented bans on the harvest of all turtle species in its waters and on the beaches, poaching of eggs, females on the beach, and animals in coastal water continues to happen. In some places throughout Mexico and the whole of the eastern Pacific, consumption of green sea turtles remain a part of the cultural fabric and tradition (NMFS and USFWS 2007d; IUCN 2021).

Like other sea turtle species, increasing temperatures have the potential to skew sex ratios of hatchling and many rookeries are already showing a strong female bias as warmer temperatures in the nest chamber leads to more female hatchlings (Kaska et al. 2006; Chan and Liew 1995). Increased temperatures also lead to higher levels of embryonic mortality (Matsuzawa et al. 2002). An increase in typhoon frequency and severity, a predicted consequence of climate change (Webster el al. 2005), can cause erosion which leads to high nest failure (Van Houtan and Bass 2007). Rising sea levels can cause repeated inundation of nests and abrupt disruption of ocean currents used for natural dispersion during the green turtle life cycle. Green sea turtles feeding may also be affected by climate change. Seagrasses are a major food source for green sea turtles and may be affected by changing water temperature and salinity (Short and Neckles 1999; Duarte 2002).

<u>Conservation</u>: There have been important conservation initiatives and advances that have benefited East Pacific green turtles. There are indications that wildlife enforcement branches of local and national governments are stepping up their efforts to enforce existing laws, although successes in stemming sea turtle exploitation through legal channels are infrequent. In addition, there are a multitude of non-profit organizations and conservation networks whose efforts are raising awareness about sea turtle conservation. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels or improve. Among the notable regional and/or multinational conservation groups and initiatives are the Central American Regional Network for the Conservation of Sea Turtles, Grupo Tortuguero de las Californias (GTC), Permanent Commission of the South Pacific (CPPS), and IAC. The Central American Regional Network resulted in the creation of a national sea turtle network in each country of the Central American region, as well as the development of firsthand tools, such

as a regional diagnosis, a 10-year strategic plan, a manual of best practices, and regional training and information workshops for people in the region . The GTC is a regional network in Mexico that brings together scientists, conservation practitioners, fishers, and local peoples to address sea turtle conservation issues. Perhaps the greatest achievement of this group was the large decrease in green turtle hunting and local consumption throughout Northwestern Mexico. CPPS is a regional body that includes Panama, Colombia, Ecuador, Peru and Chile, that has conducted many regional workshops on sea turtle conservation, but most importantly has developed a regional management plan for sea turtles. The IAC is the world's only binding international treaty for sea turtle conservation. Signatory nations in the Eastern Pacific include Chile, Peru, Ecuador, Panama, Costa Rica, Honduras, Guatemala, Mexico, and the United States. This treaty endeavors to reduce fisheries bycatch and habitat destruction through a series of binding conservation agreements across these nations. All three of these initiatives work under the principle that benefits and achievements from working in alliance are much higher than those from working alone.

Specific details regarding individual country's (i.e., Chile, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Honduras, Nicaragua, Panama, Peru, and the U.S.) protective legislation with respect to the East Pacific DPS can be found in Seminoff et al. (2015) and IUCN (2021).

In Southern California, NMFS has increased its outreach and education efforts to improve public awareness of the presence of green turtles and to reduce threats to foraging populations, particularly in San Diego Bay, the San Gabriel River and adjacent watershed, as well as estuaries such as Agua Hedionda and Mission Bay. Local threats to green turtles primarily include recreational fishing and vessel strikes, and NMFS has worked with partners to develop educational materials and signs to specifically address those threats.

2.2.8 Giant Manta Ray

The giant manta ray was listed as a threatened species under the ESA on January 22, 2018 (83 FR 2916). NMFS conducted a status review for both the giant manta ray and the reef manta ray (*Manta alfredi*) in 2017 (Miller and Klimovich 2017) in response to a 2015 petition from Defenders of Wildlife to list both species (and the Caribbean manta ray (*M. c.f. birostris*), for which NMFS determined was not a taxonomically valid species or subspecies for listing).

Because manta rays are migratory and considered ecologically flexible (e.g., low habitat specificity), they may be less vulnerable to the effects of climate change compared to other elasmobranchs. However, manta rays frequently rely on coral reef habitat for important life history functions such as feeding and cleaning, and depend on planktonic food resources for nourishment and growth, both of which are highly sensitive to environmental changes. Therefore, climate change is likely to have an impact on distribution and behavior of giant manta rays (Miller and Klimovich 2017).

The giant manta ray occurs across the globe in tropical and warm temperate bodies of water from 36°S to 40°N (Mourier 2012; Figure 4). The documented range for this species within the northern hemisphere includes: Mutsu Bay, Aomori, Japan; the Sinai Peninsula and Arabian Sea, Egypt; the Azores Islands, Portugal; and as far north as southern California (west coast) and New Jersey (east coast), United States (Kashiwagi et al. 2010; Moore 2012; CITES 2013). In the southern hemisphere, the giant manta has been documented as far south as Peru, Uruguay, South Africa, French Polynesia, New Zealand, and most recently, photographed in eastern Australia off Montague Island and Tasmania at 40° S (Mourier 2012; CITES 2013; Corturier et al. 2015). In addition, the giant manta ray has been observed in a predictable seasonal pattern in estuarine waters of Florida, Uruguay, and Brazil, suggesting that they may use estuaries as nursery areas during summer months (Adams and Amesbury 1998; Milessi and Oddone 2003; Medeiros et al. 2015).

The environmental variables that drive giant manta ray habitat use in the ocean are largely unknown, although temperature is a clear correlate (Jaine et al. 2014). Giant manta rays are found offshore in oceanic waters near productive coastlines, continental shelves, offshore pinnacles, seamounts and oceanic islands. In a satellite tracking study off the Yucatán Peninsula, Mexico, Graham et al. (2012) found that 95% of giant manta ray locations occurred in waters warmer than 21.6° C, and that most locations were correlated with high surface chlorophyll concentrations and in waters shallower than 50 meters, representing thermally dynamic and productive waters.

Stewart et al. (2016a) also reported that giant manta rays off the Revillagigedo Archipelago, Mexico tend to occur near the upper limit of the pelagic thermocline where zooplankton aggregate but also shift their activity from surface waters to 100-150 meters, likely targeting surface-associated zooplankton to vertical migrators. Burgess (2017) suggested that giant manta ray specifically feed on mesopelagic plankton, which would place them at depths as deep as 1,000 meters (also see Marshall et al. 2018). Giant manta rays are also observed at cleaning sites at offshore reefs where they are cleaned of parasites by smaller organisms.

The population structure of giant manta rays is largely unknown. At a minimum, the evidence suggests that giant manta rays in the Atlantic and giant manta rays in the Indo-Pacific represent separate populations because this species does not appear to migrate to the Pacific through the Drake Passage (or vice versa) and they do not appear to migrate around the Cape of Good Hope to the Indian Ocean (Lawson et al. 2017; Marshall et al. 2018). Several authors have reported that giant manta ray likely occur in small regional subpopulations (Lewis et al. 2015; Stewart et al. 2016b Marshall et al. 2018; Beale et al. 2019) and may have distinct home ranges (Stewart et al. 2016b). The degree to which subpopulations are connected by migration is unclear, but is assumed to be low (Stewart et al. 2016b; Marshall et al. 2018), and regional or local populations are not likely to be connected through immigration and emigration (Marshall et al. 2018), making them effectively demographically independent. To date there have been limited genetics studies on giant manta ray; however, Stewart et al. (2016b) found genetic discreteness between giant manta ray populations in Mexico suggesting isolated subpopulations with distinct home ranges within 500 km of each other. While NMFS concluded that the species is likely to become endangered within the foreseeable future throughout a significant portion of its range (the Indo Pacific and eastern Pacific), NMFS did not find the species met the criteria to list as a DPS (final rule; January 22, 2018; 83 FR 2916).

A vulnerability analysis conducted by Dulvy et al. (2014) indicates that mobulid populations can only tolerate very low levels of fishing mortality and have a limited capacity to recover once their numbers have been depleted (Couturier et al. 2012; Lewis et al. 2015). Furthermore, Lewis et al. (2015) suggests local populations in multiple areas in Indonesia have been extirpated due to fishing pressure, noting that *M. birostri*s was the most common species previously caught in these areas. Additionally, White et al. (2015) documented an 89% decline in the observed *M. birostris* population in Cocos Island National Park (Costa Rica) over a 20 year period. This decline is believed to be from overfishing outside of the park. Note that these declines are from directed fishing and not bycatch.

Figure 4. Distribution map for the giant manta ray. Extent of occurrence is depicted by light blue and the area of occupancy is noted in darker blue (Figure 3 from Lawson et al. 2017).

<u>Population Status and Trends</u>: As mentioned above, NMFS listed giant manta rays globally as threatened in 2018. The IUCN lists them as vulnerable (the category that immediately precedes endangered in the IUCN classification system), with a decreasing population trend. Although the number of regional subpopulations is unknown, the sizes of those identified as regional subpopulations tends to be small, ranging from 600 (Mozambique) to 1,875 (Raja Ampat, Indonesia) (CITES 2013; Marshall et al. 2018; Beale et al. 2019). CITES (2013) highlights two giant manta ray subpopulations that have been studied and population estimates provided, and counts for more than ten aggregations, where individuals have been recorded (Table 4 in Miller and Klimovich (2017)). The number of individually identified giant manta ray for each studied aggregation ranges from less than 50 in regions with low survey effort or infrequent sightings to more than 1,000 in some regions with targeted, long-term studies. However, ongoing research including mark-recapture analyses suggests that typical subpopulation abundances are more likely in the low thousands (e.g., Beale et al. 2019) and in rare cases may exceed 22,000 in areas with extremely high productivity, such as in coastal Ecuador (Harty et al. 2022).

Thus, while some subpopulations may have been reduced to very small population sizes due to fisheries (direct harvest or bycatch), in general, stable giant manta ray subpopulations are likely to be larger, potentially greater than 1,000 individuals, which would be in keeping with the literature that suggests subpopulations are isolated with limited movement. More importantly, the size of some of these subpopulations has declined significantly in regions subject to fishing (Marshall et al. 2018). Fisheries catch and bycatch have caused giant manta rays to decline by at least 30% globally and by up to 80% in significant portions of its range (i.e., Indonesia, Philippines, Sri Lanka, Thailand, Madagascar; Marshall et al. 2018). Lewis et al. (2015) collected data on daily landings of Manta and Mobula species from 2002 to 2014 for eight locations in Indonesia, and found landings of *Manta* species declined by 71% to 95% three locations with the most complete data. Reports from fishermen suggest that these data are representative of declines in abundance rather than shifts in effort. Tremblay-Boyer and Brouwer (2016) present CPUE data for giant manta ray observed incidentally captured in the WCPO longline and purse seine fisheries, and concluded giant manta rays are observed less frequently in recent years compared to 2000-2005, suggesting a decline in abundance (Tremblay-Boyer and Brouwer 2016).

In most areas of the world, there are mainly recorded individuals, but few subpopulation estimates have been made within subregions. Within the eastern Pacific, where giant manta rays found off the U.S. west coast may originate from, the following locations indicate both individuals recorded as well as subpopulation estimates, if known: Isla de la Plata (Ecuador): 2,804 individuals (which provided an estimate of 22,316 (a "super-population") (Harty et al. 2022); Revillagigedos (Mexico): 916 individuals, no estimate on the subpopulation abundance estimates; Costa Rica (not clear if Atlantic or Pacific Ocean): 52 individuals recorded, no estimate on subpopulation abundance estimates (NMFS 2022b).

<u>Threats</u>: The 2017 status review report provides extensive details of the known threats facing giant manta rays (Miller and Klimovich 2017). The most significant threat to the giant manta ray is overutilization for commercial purposes. They are taken as bycatch in a number of global fisheries throughout their range and are most susceptible to industrial purse-seine (particularly the Indian Ocean and the Eastern Pacific Ocean) and artisanal gillnets. They are also targeted for their parts (primarily meat) given the expansion of the international mobulid gill raker market (thought to have healing properties in Asian medicine) and increasing demand for manta ray products, particularly in many portions of the Indo-Pacific. As mentioned above, declines of sightings have been reported in several areas. Efforts to address overutilization of the species through regulations appear to be inadequate.

In the eastern Pacific Ocean, tuna fisheries, giant manta rays are frequently reported as bycatch in the large-scale purse seine fisheries; however, most manta and devil ray captures are pooled together as identification to species level is difficult. Hall and Roman (2013) reported catch and bycatch (defined as individuals retained for utilization and individuals discarded dead, respectively) prior to 2005 as below 20 tons (data from 1998-2004). By 2005, estimated catch and bycatch was around 30 tons, and increased to around 150 tons in 2006.

While AIDCP observers have been tracking purse seine ray interactions for decades, but up until 2016 observers only recorded rays killed in fishing operations, and not live releases. Based on reported giant manta ray catch to the IATTC, including available national observer program data, an average of 135 giant manta rays were estimated caught per year from 1993-2015 in the eastern Pacific purse seine fishery (Table 8 in Miller and Klimovich 2017). During this same time period, estimates of "unidentified manta/devil ray" were 1,795 animals captured per year. From 2016-2021, a total of 14 giant manta rays were discarded dead in the international fleet. This includes information from the U.S. fleet, although no giant manta rays were discarded dead during that time frame in the domestic fishery. As summarized in the final listing rule (and final status review), a preliminary productivity and susceptibility analysis that indicated that the giant manta ray is one of the most vulnerable species to overfishing in the purse seine fishery by IATTC vessels.

As mentioned above, changes in climate and oceanographic conditions (e.g., acidification) may affect zooplankton size, composition and diversity as well as distribution. Therefore, migration and distribution of giant manta rays may be affected, particularly those that exhibit site-fidelity to particular areas. Climate change is expected to cause shifts in the productivity of the Humboldt Current System, and increased ocean temperatures, deepening stratification, and changes in wind patterns may lead to variable effects on primary production and upwelling strength.

Because giant manta rays are filter feeders, plastics ingestion is likely, as is entanglement in marine debris, potentially contributing to increased mortality rates. In just the year 2010, Jambeck et al. (2015) estimated that the United States produced 0.25 to 1 metric tons of plastic waste available to enter the ocean, while Mexico and Central America were estimated to produce 0.01 to 0.25 metric tons.

Tourist attractions at manta ray "hot spots" may also pose a threat to giant manta rays through behavioral disruption or potential inadvertent habitat destruction by scuba divers. In addition, giant manta rays are subjected to boat strikes, particularly in areas of high maritime traffic. Mooring and boat anchor lines may also wound manta rays or cause drowning. Conservation: Giant manta rays were listed in Appendices I and II of the Convention of the Convention on the Conservation of Migratory Species of Wild Animals (CMS) in 2011. As a result, harvesting of giant manta rays is no longer permitted internationally. Exceptions include traditional subsistence users. The species was also listed under Appendix II of CITES in September, 2014, Inclusion in Appendix II restricts trade of the species between counties. International trade of Appendix-II species may be authorized by the granting of an export permit or re-export certificate. Permits should only be granted if above all that trade will not be detrimental to the survival of the species in the wild.

Conservation: Despite some national and regional (e.g., regional fishery management organizations) protections for giant manta rays, the lack of enforcement and illegal fishing have generally rendered existing regulatory protections inadequate particularly for protecting the species from fishing mortality. However, there have been successes. In Indonesia, giant manta rays were fully protected in the nation's waters with the creation of the world's largest manta ray sanctuary at around 6 million km^2. Targeted fishing for giant manta rays as well as trade in manta ray parts are banned; however, illegal fishing and trade has been documented. Similarly, the Philippines introduced legal protection for manta rays in 1998 although existing regulatory mechanisms are lacking to curb illegal fishing.

In the eastern Pacific portion of the giant manta ray's range, the IATTC implemented a prohibition on the retention, transshipment, storage, landing and sale of all devil and manta rays taken in large-scale fisheries in 2016 (Resolution C-15-04). The success of this resolution of course depends on the post-release mortality rate, particularly when released in purse seine nets, and that rate is currently unknown. Developing countries were granted an exception for small-scale and artisanal fisheries that catch these species for domestic consumption. Prohibitions on fishing and sale of giant manta rays was implemented in Peru in 2016, and Ecuador implemented similar regulations in 2010. Given that the largest population of giant manta rays is found in waters between Peru and Ecuador, with the Isla de la Plata population estimated at around 1,500 individuals, these prohibitions should provide some protection to the species. However, as with other national protections in place, illegal fishing still occurs in these waters.

2.3 Action Area

"Action area" means all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR 402.02). The action area for this proposed action is the U.S. EEZ and adjacent high seas waters off the coast of California, Oregon, and Washington.[13] To a large degree, the action area for the proposed action is further reduced by the combination of state and federal regulations that have influenced where this fishery has occurred in the past, and would be expected to occur in the foreseeable future. For the purposes of this proposed action, the range and extent of the DGN fishery that has occurred in this area since 2001 represent the current state and expected extent of the DGN fishery over the next five years (Figure 5). The descriptions of regulations that govern the DGN are provided below to help define the actual action area for the fishery based on when and where the fishery is expected to occur.

2.3.1 DGN Fishery Regulations

In 2001, the Pacific Leatherback Conservation Area (PLCA) went into effect to protect leatherback sea turtles. This significantly altered the availability of fishing grounds for the DGN fleet and ultimately the distribution of DGN fishing effort off the coast of California. While Oregon and Washington state laws do not allow landings caught with drift gillnet gear, vessels may fish in a portion of federal waters offshore of Oregon and land their catch in California. The large-mesh DGN fishery (14" minimum stretched mesh size) is managed through both federal and state regulations to conserve target and non-target stocks including protected species

[13] 50 CFR § 660.701 defines the action area for the HMS FMP, but does not define "adjacent high seas waters." For the purposes of the DGN fishery, adjacent high seas waters could include a small amount DGN fishing effort occurring in the high seas waters outside the EEZ in relatively close proximity to the EEZ, although fishing effort has not been observed in high seas waters since 2001 (Figure 4).

that are incidentally captured. These regulations are described in Appendices B and C to the original HMS FMP Final Environmental Impact Statement (PFMC 2003), the latter being the California code for fishing swordfish and shark with minimum stretched mesh of 14 inches required. The regulations for ≥14" stretched mesh drift gillnets are summarized as follows:

Federal Regulations

Pacific Offshore Cetacean Take Reduction Team (POCTRT) measures to protect marine

- Acoustic deterrent devices (pingers) are required on drift gillnets to deter entanglement of marine mammals. Pingers, when immersed in water, must broadcast a 10 kilohertz (kHz) (± 2kHz) sound at 132 decibels (dB) (±4 dB) re 1 micropascal at 1 meter lasting 300 milliseconds (± 15 milliseconds) and repeating every 4 seconds (±0.2 seconds). They must also remain operational to a water depth of at least 100 fm. Pingers must be attached in a staggered configuration no more than 300 ft (91.44m) apart along the floatline and leadline (Figure 6).
- All drift gillnets must be fished at minimum depth of 6 fm (10.9 m).
- Attendance at skipper workshops is required after notification from NMFS.
- Vessels must provide accommodations for observers when assigned.[15]

[14] Initially implemented in 1997 (62 FR 51805; October 3, 1997); amended in 1998 (63 FR 27860; May 21, 1998); amended again in 1999 (64 FR 3431; January 22, 1999).
[15] This is a regulatory requirement under 50 CFR 660.719.

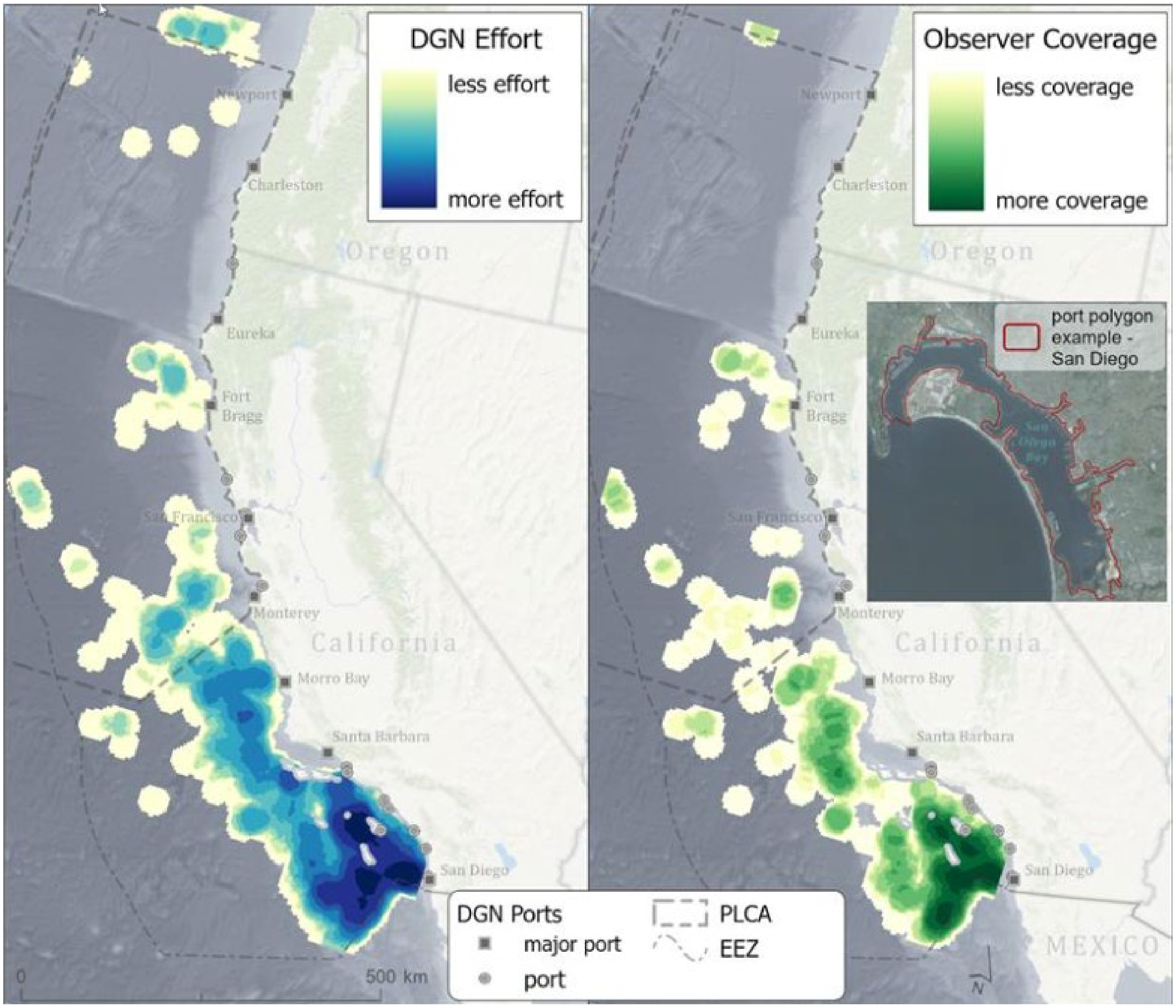

Figure 5. Spatial extent of DGN fishing effort and observer coverage from September 2013 through January 2019. Although, the fishing season runs a full year (May 1-January 31), no reported effort occurred during this time period outside of the August 15-January 31 timeframe.

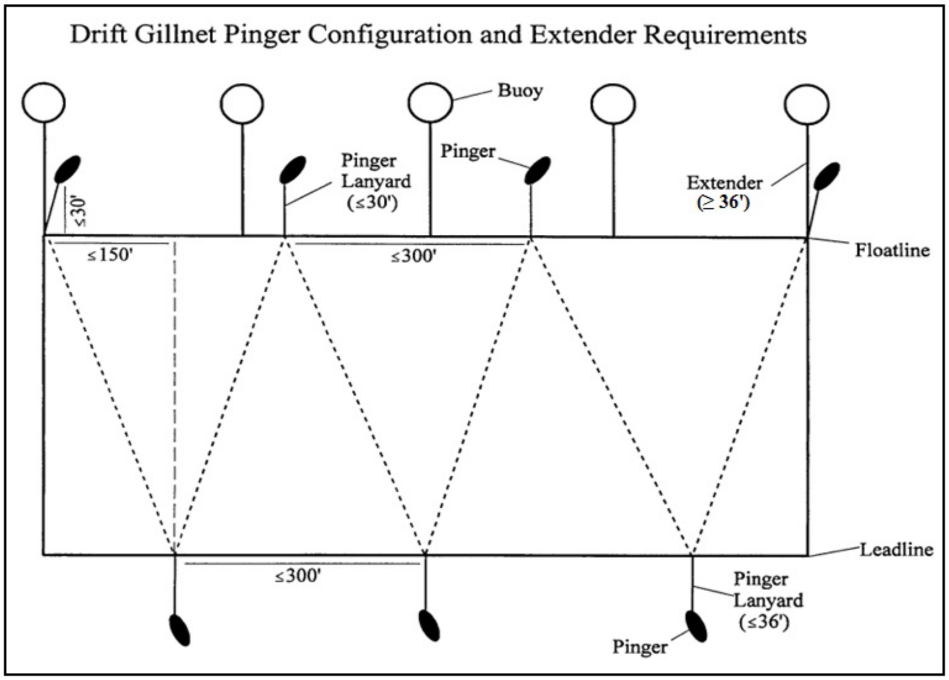

Figure 6. Diagram of required pinger placement on a drift gillnet (50 CFR 229.31).

Pacific Sea Turtle Conservation Areas (50 CFR 660.713)

Drift gillnet fishing may not be conducted:

- From August 15 to November 15 in the portion of the EEZ bounded by the coordinates 36° 18.5' N latitude (Point Sur), to 34°27' N latitude, 123° 35' W longitude; then to 129°W longitude; then north to 45° N latitude; then east to the point where 45° N latitude meets land - Pacific Leatherback Conservation Area.
- From the months of June, July, and/or August during a forecast or declared El Niño, as announced by NMFS in the Federal Register, in the portion of the EEZ south of Point Conception, California (34°27' N latitude) and west to 120° W longitude - Pacific Loggerhead Conservation Area.

The Pacific Sea Turtle Conservation Areas are based on a NMFS October 23, 2000, biological opinion on the DGN fishery and subsequent recommendations made by the POCTRT in 2001 (POCTRT 2001). To minimize the economic impact of the time and area closures, NMFS modified the POCTRT recommendations in the 2001 final rule creating the PLCA for leatherback sea turtles (66 FR 44549; August 24, 2001). The POCTRT recommended a line heading due west from shore at 36°15' N latitude as the southern boundary of the PLCA. NMFS moved the southern boundary's intersection with shore to Point Sur because it is a more recognizable landmark and only three miles north of 36° 15' N latitude. NMFS also modified the POCTRT recommendation (a line heading due west) to a diagonal line from Point Sur to 34° 27' N latitude, 123° 35' W longitude based on satellite tracking data of two leatherback turtles tagged in Monterey Bay in September 2000. This precaution was intended to protect a potential migratory corridor of leatherbacks departing Monterey Bay for western Pacific nesting beaches.

The original trigger language identified by the POCTRT to extend the area closure in a southerly direction to Point Conception if a leatherback was observed taken was also removed because NMFS did not consider this extra precaution to be necessary based on the distribution of the turtles that had historically been taken incidentally in the fishery. In addition, the final PLCA for leatherback sea turtles did not include lowering the top of drift gillnets to at least 60 feet deep as recommended by the POCTRT, because observer data (1990-2000) did not suggest that this would result in a definite decrease in leatherback interactions. Modifications provided access to the productive fishing grounds north of Point Conception, which is consistent with the intent of the POCTRT proposal, while still providing at least an equal, if not greater, level of protection for leatherback and loggerhead sea turtles. The Pacific Sea Turtle Conservation Areas, as well as other seasonal time/area closures for the DGN fishery and designated leatherback critical habitat, are shown below in Figure 7. Specific information on the current status of the Pacific Loggerhead Conservation Area and the underlying supporting information at any time can be found here.

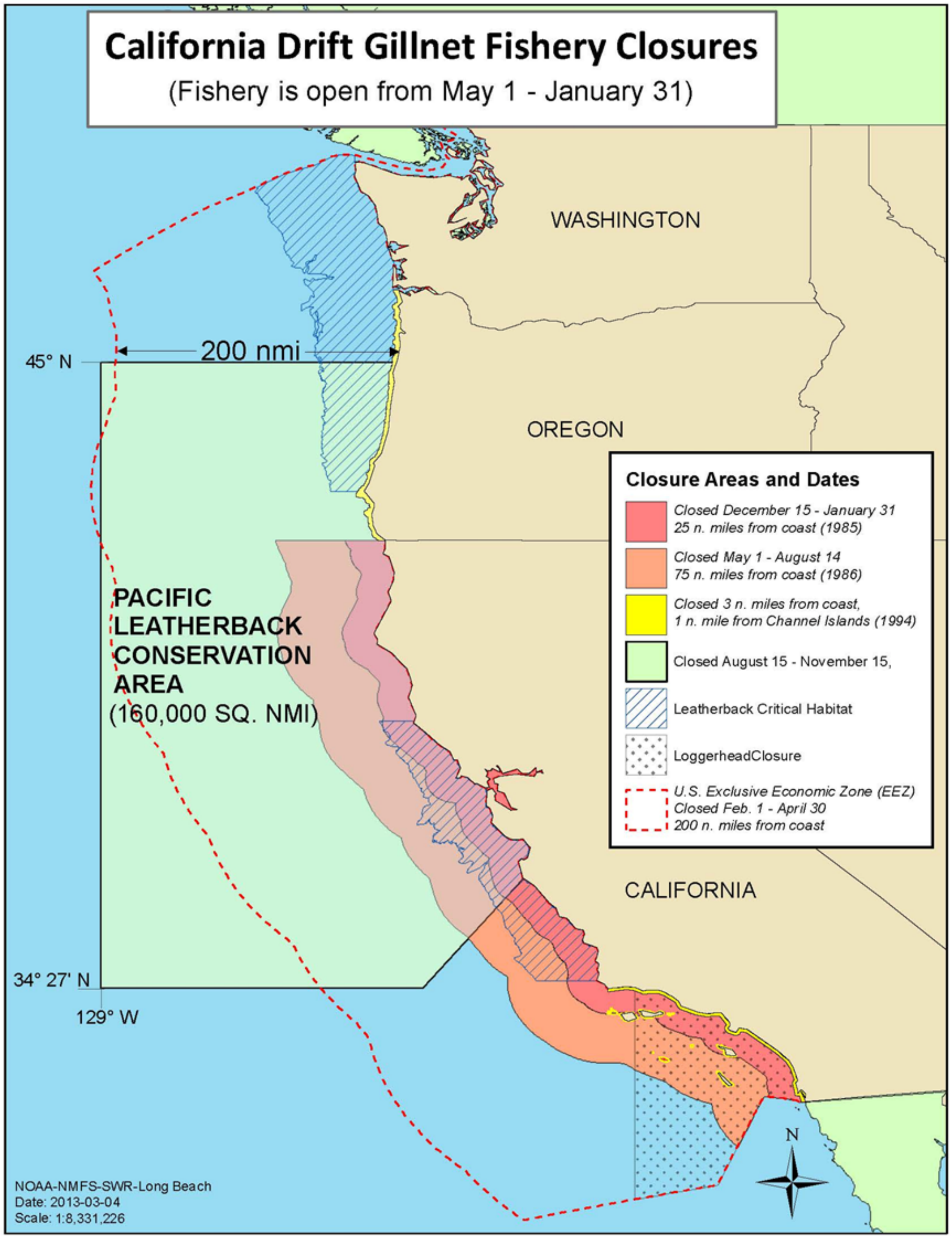

Figure 7. Pacific Sea Turtle Conservation Areas and other time/area closures for the DGN, and designated leatherback critical habitat. Source NOAA-NMFS-SWR-Long Beach March 4, 2013.

Marine Mammal Protection Act Requirements for Category II Fisheries

Section 118 of the MMPA requires NMFS to place all U.S. commercial fisheries into one of three categories based on the level of incidental mortality and serious injury of marine mammals occurring in each fishery (16 U.S.C. 1387(c)(1)). The classification of a fishery determines whether participants in that fishery may be required to comply with certain provisions of the MMPA, such as registration, observer coverage, and take reduction plan requirements. NMFS must reexamine the classifications annually, considering new information in the Marine Mammal Stock Assessment Reports and other relevant sources, and publish in the Federal Register any necessary changes to the LOF after notice and opportunity for public comment (16 U.S.C. 1387 (c)(1)(C)). In 2023, the DGN fishery was categorized as a Category II fishery under this system due to annual mortality and serious injury to sperm whales, short-finned pilot whales, and minke whales (88 FR 16899).

<u>State Restrictions</u> (applicable to vessels operating from the state's ports)

Participation restrictions:

- California has a limited entry program, which was adopted into the HMS FMP, for the swordfish/thresher shark DGN fishery. No new permits will be issued and current permits (issued to vessel operator) can only be transferred under certain conditions (i.e., health concern or death of permit holder) to another fisherman currently holding or eligible for a general gillnet/trammel net permit by the State of California. A federal limited entry DGN permit is required.

Gear restrictions (California):

- The maximum cumulative length of a shark or swordfish gill net(s) on the net reel of a vessel, on the deck of the vessel, and/or in the water at any time shall not exceed 6,000 ft in float line length, except that up to 250 fm of spare net (in separate panels not to exceed 100 fm each) may be on board the vessel stowed in lockers, wells, or other storage.
- The use of quick disconnect devices to attach net panels is prohibited.
- DGN gear must be at least 14 inch stretched mesh.
- The unattached portion of a net must be marked by a pole with a radar reflector.

Mainland area restrictions/closures where DGN gear cannot be used:

- In the EEZ off California from February 1 to April 30.
- In the portion of the EEZ off California within 75 nm of the coastline from May 1 to August 14.
- In the portion of the EEZ off California within 25 nm of the coastline from Dec. 15 through Jan. 31.
- In the portion of the EEZ bounded by a direct line connecting Dana Point; Church Rock on Catalina Island; and Point La Jolla, San Diego County; and the inner boundary of the EEZ from August 15 through September 30 each year.
- In the portion of the EEZ within 12 nm from the nearest point on the mainland shore north to the Oregon border from a line extending due west from Point Arguello.
- East of a line running from Point Reyes to Noonday Rock to the westernmost point of southeast Farallon Island to Pillar Point.
- In the portion of the EEZ within 75 nm of the Oregon shoreline from May 1 through August 14, and within 1000 fm the remainder of the year.
- In State waters off the Washington coast (Washington does not authorize the use of this HMS gear).

Channel Islands (California) closures where DGN gear cannot be used:

- In the portion of the EEZ within six nm westerly, northerly, and easterly of the shoreline of San Miguel Island between a line extending six nm west magnetically from Point Bennett and a line extending six nm east magnetically from Cardwell Point and within six nm westerly, northerly, and easterly of the shoreline of Santa Rosa Island between a line extending six nm west magnetically from Sandy Point and a line extending six nm east magnetically from Skunk Point, from May 1 through July 31 each year.
- In the portion of the EEZ within 10 nm westerly, southerly, and easterly of the shoreline of San Miguel Island between a line extending 10 nm west magnetically from Point Bennett and a line extending 10 nm east magnetically from Cardwell Point and within 10 nm westerly, southerly, and easterly of the shoreline of Santa Rosa Island between a line extending 10 nm west magnetically from Sandy Point and a line extending 10 nm east magnetically from Skunk Point from May 1 through July 31 each year.
- In the portion of the EEZ within a radius of 10 nm of the west end of San Nicolas Island from May 1 through July 31 each year.
- In the portion of the EEZ within six of the coastline on the northerly and easterly side of San Clemente Island, lying between a line extending six nm west magnetically from the extreme northerly end of San Clemente Island to a line extending six nm east magnetically from Pyramid Head from August 15 through September 30 each year.

2.4 Environmental Baseline

The "environmental baseline" refers to the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultations, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 CFR 402.02).

As described above in the *Status of the Species* sections, the ESA-listed species that may be adversely affected by the proposed action that belong to the same species group (marine mammal, sea turtle, or manta ray) are generally exposed to many similar threats throughout their range. Although the action area for this proposed action (large portion of the U.S. west coast EEZ, featuring the waters offshore the Southern California Bight and Central California as allowed by regulation) represents only a portion of relatively large ranges for all of these species that are highly mobile and migrate great distances, many of these same threats are present for animals when they do occur in U.S. west coast waters where the DGN fishery occurs. In this section, we review the available information regarding impacts to ESA-listed species by species group, with reference to individual species as necessary or appropriate based on the available information. Information provided in this section comes from a review of the NMFS marine mammal and sea turtle stranding databases, biological opinions, current scientific research permits, current SARs, and other material as cited below. There is little to no information on human-related interactions within the action area for giant manta rays, other than risk from gillnet and purse seine fisheries including entanglement and forced submergence.

Globally averaged annual surface air temperatures have increased by about 1.0°C over the last 115 years (1901 to 2016). The earth's climate is now the warmest in the history of modern civilization. All of the relevant evidence points to human activities, particularly emissions of greenhouse gases since the mid-20th century, as the probable cause of this warming pattern (Wuebbles et al. 2017). Without major reductions in emissions, the increase in annual average global temperature relative to preindustrial times could reach 5°C or more by the end of this century. With significant reductions in emissions, the increase in annual average global temperature could be limited to 2°C or less. There is broad consensus that the further and faster the earth warms, the greater the risk of potentially large and irreversible negative impacts (Wuebbles et al. 2017).

Increases in atmospheric carbon and changes in air and sea surface temperatures can affect marine ecosystems in several ways including changes in ocean acidity, altered precipitation patterns, sea level rise, and changes in ocean currents. Global average sea level has risen by about seven to eight inches since 1900, with almost half of that rise occurring since 1993. It is very probable that human-caused climate change has made a substantial contribution to sea level rise, contributing to a rate of rise that is greater than during any preceding century in at least 2,800 years. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100 (Wuebbles et al. 2017). Climate change can influence ocean circulation for major basin-wide currents including intensity and position of western boundary currents. These changes have potential for impact to the rest of the biological ecosystem in terms of nutrient availability as well as phytoplankton and zooplankton distribution (Gennip et al. 2017).

Effects of climate change on marine species include alterations in reproductive seasons and locations, shifts in migration patterns, reduced distribution and abundance of prey, and changes in the abundance of competitors or predators. Variations in sea surface temperature can affect an ecological community's composition and structure, alter migration and breeding patterns of fauna and flora and change the frequency and intensity of extreme weather events. For species that undergo long migrations (e.g., sea turtles, humpback whales), individual movements are usually associated with prey availability or habitat suitability. If either is disrupted, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott 2009). Over the long term, increases in sea surface temperature can also reduce the amount of nutrients supplied to surface waters from the deep sea leading to declines in fish populations, and therefore, declines in those species whose diets are dominated by fish.

The ranges of elasmobranch species, such as the giant manta ray, are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012; Harty et al. 2022). Climate-related shifts in range and distribution have already been observed in some marine mammal populations. Specialized diets, restricted ranges, or reliance on specific foraging sites may make many may make many marine mammal populations particularly vulnerable to climate change (Silber et al. 2017). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by climate change, 47% would be negatively affected, and 21% would be put at risk of extinction. Hazen et al. (2012) examined top predator distribution and diversity of top marine predators in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. The researchers predicted up to a 35% change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback sea turtles were predicted to gain core habitat, whereas loggerhead sea turtles (and blue whales) were predicted to lose core habitat area. Such range shifts could affect marine mammal and sea turtle foraging success as well as sea turtle reproductive periodicity (Kaschner et al. 2011).

Significant impacts to marine mammals and sea turtles from ocean acidification may be indirectly tied to foraging opportunities resulting from ecosystem changes. Nearshore waters off California have already shown a persistent drop in pH from the global ocean mean pH of 8.1 to as low as 7.43 (Chan et al. 2017). The distribution, abundance and migration of baleen whales reflects the distribution, abundance and movements of dense prey patches (e.g., copepods, euphausiids or krill, amphipods, and shrimp), which have in turn been linked to oceanographic features affected by climate change (Learmonth et al. 2006).

Sea turtles have temperature-dependent sex determination, and many populations already produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Jensen et al 2018). Female-biased green sea turtle sex ratios have been reported for East Pacific green turtles at foraging locations in San Diego Bay, California (Allen et al. 2015). A fundamental shift in the demographics of species such as sea turtles may lead to increased instability of populations that are already at risk from several other threats. In addition to altering sex ratios, increased temperatures in sea turtle nests can result in reduced incubation times, reduced clutch size, and reduced nesting success due to exceeded thermal tolerances (Fuentes et al. 2011).

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

2.4.1 Marine Mammals

As described above in the status section, fin, humpback (both Mexico and Central America DPSs), and sperm whales are all exposed to threats associated with fisheries bycatch, including U.S. fisheries that occur in the action area. Other impacts to ESA-listed marine mammals that may occur while present along the U.S. west coast include vessel collisions, scientific research, and exposure to environmental changes or hazards. Although the whales considered in this biological opinion are not listed as species that match up directly with stock definitions under the MMPA, we use information provided in the SARs for affected stocks of whales to understand what impacts are occurring the ESA-listed species of whales in the action area.

2.4.1.1 Fisheries Interactions

Most data on human-caused mortality and serious injury for this population is based on opportunistic stranding and at-sea sighting data that represents a minimum count of total impacts. However more recent SARs have included records of entangled *unidentified whales* prorated to humpback whale based on location, depth, and time of the year, which at least provides a more conservative estimate of entanglements specific to humpback whales (and other large whale species).

From 1982 to 2017, there were 521 whale entanglements reported and 434 confirmed entanglements reported off the coast of California, Oregon, and Washington (Saez et al. 2021). Whale entanglement reports were confirmed using criteria that include reviewing photos/videos or through direct observation by NMFS staff. Between 1982 and 2013, there were an average of 9 confirmed entangled large whales, while from 2014 to 2017, there were an average of 41 confirmed entangled large whales. There were multiple factors which may contribute to this increase in the number of reports, including an increase in public awareness and reporting, changes in the spatial distribution and abundance of whales, fishing effort, and ocean conditions. Humpback whales were the second-most frequently reported species, with 167 confirmed entanglements between 1982 and 2017. Overall, the majority of confirmed entanglements (all species) were reported from California (85%), with 7% from Washington, 6% from Oregon, and 1% from Mexico and Canada. The highest number of entanglements were reported in March and April, corresponding with the northern migration of gray whales as well as the early presence of foraging humpbacks. Many of the reports, and as reflected in recent SARs, included “unidentified gear” and could not be assigned to a particular fishery. For known fisheries, “netting” accounted for 34% of the reported gear and pot/traps accounted for 22% of the gear (Saez et al. 2021). More recent information specific to each ESA-listed species is described below.

<u>Fin Whales</u>: Information provided in the 2021 SARs indicated that three fin whales sighted at-sea along the U.S. west coast were determined to be seriously injured as a result of interactions with unknown fishing gear during 2015-2019 (Carretta et al. 2022a). A review of entanglement records through 2022, including records that have not been evaluated in the most recent SARs, indicate that were two fin whale entanglements reported to NMFS in 2015, one reported in 2018, and one reported in 2022, in gear that could not be identified (NMFS Entanglement Response

Program data, NMFS 2023e). The SARs also considers unidentified whale entanglements as potentially contributing to the fin whale fishery interactions totals. Carretta et al. (2022a) reported 16 additional unidentified whale entanglements from 2015-2019. Applying Carretta's (2018) method prorating 0.28 of these entanglements to fin whales, and prorating the unidentified gear entanglements to 0.75 serious injuries per entanglement case (0.28 x 0.75), results in approximately 0.21 additional fin whale serious injuries from these 16 unidentified entangled whale cases. The total mean annual fishery-related serious injury and mortality including both observed and prorated cases is 0.64 fin whales (Carretta et al. 2022a).

Humpback Whales: Off the U.S. West Coast, humpbacks have been documented injured and killed from numerous, fisheries, vessel strikes, and entanglement in marine debris. Pot and trap fisheries are the most commonly documented source of mortality and serious injury of humpback whales in U.S. west coast waters (Carretta et al. 2023a, b). From an overall perspective on threats to humpback whales off the U.S. West Coast, Carretta et al. (2022b) provided a summary of humpback whale human-related injury and mortality sources, number of cases, and total mortality and serious injury for 2016 through 2020. The number of cases include non-serious injuries as well. We provide these as a snapshot but provide more information on the annual average estimates from the draft 2022 SAR (Carretta et al. 2023a, b), specifically for known threats within the action area (section 2.3).

Table 5. Humpback whale human-related injury and mortality sources, number of cases, and total mortality and serious injury, 2016-2020, across California, Oregon, and Washington.

Source	Number of Cases	Mortality/Serious Injury Total (and Annual Average), 2016-2020
Unidentified Fishery Interaction	58	43.75 (8.75)
Dungeness Crab Pot Fishery (CA)	34	23.75 (4.75)
Vessel Strike	14	13.20 (2.64)
Unidentified Pot/Trap Fishery Entanglement	13	9.50 (1.9)
Dungeness Crab Pot Fishery (WA)	7	5.50 (1.1)
Gillnet Fishery	6	2.00 (0.4)
CA spot prawn trap fishery	5	3.25 (0.65)
Gillnet fishery, tribal	3	2.50 (0.5)
Dungeness crab pot fishery (commercial)	2	2.00 (0.4)

Dungeness crab pot fishery (OR)	2	1.75 (0.44)
Dungeness crab pot fishery (recreational)	2	1.00 (0.2)
WA/OR/CA sablefish pot fishery	2	1.50 (0.3)
Hook and line fishery	1	0.75 (0.15)
Marine debris	1	1.00 (0.2)
Pot fishery, tribal	1	1.00 (0.2)
Spot prawn trap/pot fishery (recreational)	1	0.00
WA/OR/CA sablefish pot fishery and CA coonstripe shrimp pot fishery	1	0.00
Total	153	112.45 (22.5/year)

As shown in Table 5 and summarized more generally in the *Status of the Species* (section 2.2), of the documented 153 cases of human-related interactions with humpback whales over the most recent 5 year period (2016-2020), the majority of those are attributed to fisheries interactions, with most of them attributed to unidentified fisheries, followed by interactions with the California Dungeness crab pot fishery. Other pot/trap fisheries also contribute to the majority of fishery interactions with humpback whales and this reflects much of the historic entanglements dating back to the early 1980s. However, recent analyses indicates that since 2000, the proportion of whales (all species) entangled in pot/trap gear has increased, whereas net entanglements have decreased in prevalence (through 2017; Saez et al. 2021).

Table 6 provides a summary of estimated mortality and serious injury associated with different sources of interactions attributed to different stocks of humpback whales in U.S. West Coast commercial fisheries for the period 2016-2020, unless otherwise noted (Carretta 2022; Carretta et al. 2023a, b; Jannot et al. 2021). Records also include entanglements detected outside of U.S. waters confirmed to involve U.S. West Coast commercial fisheries. Most cases are derived from opportunistic strandings and at-sea sightings of entangled whales. Also included are records of entangled *unidentified whales* prorated to humpback whale based on location, depth, and time of year (Carretta 2018). Sources derived from systematic observer programs with statistical estimates of bycatch and uncertainty are shown with CVs. Totals in the first two numerical columns include whales from two stocks: the Central America / Southern Mexico – CA-OR-WA stock, and the Mainland Mexico – CA-OR-WA stock. As described earlier in the *Status of the Species* section 2.2.2, the totals are prorated to the Central America/Southern Mexico-CA/OR-WA stock (proration factor=0.42) and the Mainland Mexico-CA/OR/WA stock (proration factor=0.58), unless the interaction is known to have come from Washington waters where totals are prorated to the Central America/Southern Mexico-CA/OR-WA stock (proration factor=0.06) and the Mainland Mexico-CA/OR/WA stock (proration factor=0.25), and Hawai'i stock (proration factor=0.69; not represented in the table).

Table 6. Sources of serious injury and mortality of humpback whale stocks in U.S. West Coast commercial fisheries for the period 2016-2020, unless noted otherwise (Jannot et al. 2021; Carretta 2022; Carretta et al. 2023a, b).

Fishery Source	Observed Interactions (% observer coverage if applicable)	∑ Total CA-OR-WA Mortality and Serious Injury (CV if applicable)	**Mean Annual M/SI** **Central America / Southern Mexico – CA-OR-WA stock prorated totals**	**Mean Annual M/SI** **Mainland Mexico-CA/OR/WA stock prorated totals**
Unidentified fishery	58	43.75	**3.52**	**4.89**
Dungeness crab pot (California)	34	23.75	**2.01**	**2.74**
Unidentified pot/trap fishery	13	9.50	**0.62**	**0.94**
Dungeness crab pot (Washington)	7	5.50	**0.07**	**0.28**
Unidentified fishery interactions involving *unidentified whales* prorated to humpback whale	7	5.25	**0.44**	**0.61**
Unidentified gillnet fishery	6	2.00	**0.17**	**0.23**
California spot prawn fishery	5	3.25	**0.28**	**0.38**

Dungeness crab pot fishery (Oregon)	2	1.75	**0.15**	**0.20**
WA/OR/CA sablefish pot fishery (observer program) †*	1 (31% - 72%)	7.82	**0.66 (CV>0.8)**	**0.90 (CV>0.8)**
Dungeness crab pot fishery (commercial, state unknown)	2	2.00	**0.17**	**0.23**
CA swordfish and thresher shark drift gillnet fishery (observer program)**	0 (21%)	0.10	**0.01 (CV>4.7)**	**0.01 (CV>4.7)**
Totals CA-OR-WA waters	136	104.7	**8.1**	**11.4**

† At-sea sightings of entangled whales in the WA/OR/CA sablefish pot fisheries that were not recorded in observer programs during 2016-2020 (2) are included in mean annual mortality and serious injury totals because observer data are used to estimate total entanglements for two separate sablefish pot fisheries in this category (Jannot et al. 2021). These two records are not included in 'Observed Interactions.'

* Jannot et al. (2021) report one humpback entanglement in the limited entry sector in 2014, over an observation period spanning 2002 – 2019 where 13% - 72% of landings were observed. Jannot et al. (2021) report one humpback entanglement in the open access sector in 2016, over an observation period spanning 2002 – 2019 where 2% - 12% of landings were observed. This estimate is based on 2015-2019 data.

** There were no observed entanglements during 2016-2020[16], however the model-based estimate of bycatch is based on pooling 1990-2000 data, resulting in a small positive estimate (Carretta 2022).

Non-commercial sources of anthropogenic mortality and serious injury, including tribal fisheries, recreational fisheries, and marine debris (including research buoys) are responsible for a small fraction of all reported cases annually (Carretta et al. 2023a, b). As described earlier in the *Status of the Species* section 2.2.2, the totals are prorated to the Central America/Southern Mexico-CA/OR-WA stock (proration factor=0.42) and the Mainland Mexico-CA/OR/WA stock (proration factor=0.58), unless the interaction is known to have come from Washington waters where totals are prorated to the Central America/Southern Mexico-CA/OR-WA stock (proration factor=0.06) and the Mainland Mexico-CA/OR/WA stock (proration factor=0.25), and Hawai'i stock (proration factor=0.69; not represented in the table.

[16] Carretta 2022 also includes estimates of mortality and serious injury in the DGN fishery through 2021, which are used to support the analysis in section 2.5 *Effects of the Action*.

Table 7. Non-commercial fishery sources of anthropogenic mortality and serious injury observed and reported during 2016-2020 in U.S. West Coast waters (Carretta et al. 2023 a, b).

Source	Observed Interactions	Total Mortality and Serious Injury	**Mean Annual M/SI** **Central America / Southern Mexico – CA-OR-WA stock prorated totals**	**Mean Annual M/SI** **Mainland Mexico – CA/OR/WA stock prorated totals**
Gillnet fishery (tribal)	3	2.5	**0.03**	**0.13**
Dungeness crab pot fishery (recreational)	2	1	**0.09**	**0.12**
Hook and line fishery	1	0.75	**0.06**	**0.09**
Marine debris	1	1	**0.09**	**0.12**
Pot fishery (tribal)			**0.09**	**0.12**
Totals CA-OR-WA waters	**9**	**6.25**	**0.35**	**0.56**

Some limited humpback whale bycatch is expected and has been analyzed in other U.S. West Coast fisheries. Associated with the Pacific Coast Groundfish FMP, and specifically the sablefish pot fishery, NMFS estimated a maximum of 2.05 Mexico DPS humpback whales per year (1.44 maximum five-year running average) would be entangled by the fishery, while a maximum of 1.28 Central America DPS humpbacks per year (0.9 maximum five-year running average) would be entangled by the fishery. NMFS concluded that the Pacific Coast Groundfish fishery would not jeopardize ESA-listed DPSs of humpback whales (NMFS 2020a). As shown in Table 6 from the draft 2023 SAR (covering 2016-2020; Carretta et al. 2023a, b), one humpback interaction was reported in the WA/OR/CA Limited Entry sablefish pot fishery sector (31% to 72% observer coverage). In addition, one humpback interaction was reported in the WA/OR/CA Open Access sablefish pot fishery sector (7% to 12% observer coverage). Given the mean annual mortality/serious injury attributed to these two fisheries using the most recent estimates available generated using observer data from 2015-2019 (7.8/5=1.6), and applying the proration factors yields estimates for the Central America/Southern Mexico-CA/OR/WA stock (and effectively the Central America DPS) of 0.66 humpbacks, and estimates for the Mainland Mexico-CA/OR/WA stock (effectively the Mexico DPS) of 0.90 humpbacks, seriously injured or killed each year (Table 6).

Preliminary information on humpback whale interactions which have been shared publicly, but have not undergone scientific review and serious injury determinations, include data from 2021 and 2022. In 2021, there were 17 confirmed humpback whale interactions, 11 from California, one from Oregon, two in Washington, and three from Mexico. Fisheries that were identified as attributing to the interactions include: five commercial Dungeness crab, two California large mesh drift gillnet, one California experimental box crab, one unidentified gillnet, one California commercial spiny lobster, one California commercial spot prawn, one recreational hook and line, and five attributed to unknown fishery interactions (NMFS 2022a). In 2022, there were 18 confirmed humpback whale interactions, with 16 reported in California and two in Mexico. Fisheries that were identified as attributing to the interactions include: seven commercial Dungeness crab, two unidentified gillnet, and nine unknown (NMFS 2023e).

Sperm Whales: In the 2021 SARs, the most recent information on fisheries interactions with sperm whales includes time periods up to 2017. In 2010, two sperm whales (one death and one serious injury in the same set) were entangled by the DGN fishery. A review of entanglement records through 2022, including records that have not been evaluated in the most recent SARs, indicate that there has been one sperm whale entanglement reported to NMFS (in 2020) since 2015 (NMFS Entanglement Response Program data). Other fisheries may injure or kill sperm whales through entanglement or ingestion of marine debris. Three separate sperm whale strandings in 2008 showed evidence of fishery interactions; 2 whales died from gastric impaction as a result of ingesting multiple types of floating polyethylene netting, and a 3rd whale in 2008 showed evidence of entanglement scars (Carretta et al. 2021). The most recent model-based estimates for the DGN fishery during 2017-2021 are that 1.58 whales were entangled and seriously injured or killed by the fishery, or 0.32 sperm whales per year (Carretta 2022). The total mean annual fishery-related serious injury and mortality reported in the most recent SAR is 0.64 whales per year (Carretta et al. 2022a).

2.4.1.2 Vessel Collisions

Ship strikes were implicated in the deaths of seven fin whales along the U.S. west coast during 2015-2019, along with one additional serious injury to an unidentified large whale attributed to a ship strike (Carretta et al. 2022a). Additional mortality from ship strikes probably goes unreported because the whales do not strand or, if they do, they do not always have obvious signs of trauma. Fourteen humpback whales (13.2 deaths or serious injuries) were reported struck by vessels between 2016 and 2020 (Carretta et al. 2023a, b). One sperm whale reportedly died as the result of a ship strike in Oregon in 2007, and another sperm whale was struck by a 58-foot sablefish longline vessel in 2007 while at idle speed (Carretta et al. 2022). For the most recent 5-year period of 2013 to 2017 for which data are available, no ship strikes of a sperm whale was documented (Carretta et al. 2022a).

Whale carcasses can sink and ships may not detect a whale strike, although this is more likely to be the case with large container vessels and tankers. As a result, most vessel strikes are likely undetected/unreported, and the true number of vessel collisions that occur is unknown. Vessel strike mortality has been estimated for several whale species (including humpback, fin, and blue whales) in the U.S. West Coast EEZ by Rockwood et al. (2017) using an encounter theory model

(Martin et al. 2015) combining species distribution models of whale density (Becker et al. 2016), vessel traffic characteristics (size + speed + spatial use), and whale movement patterns obtained from satellite-tagged animals in the region to estimate whale/vessel interactions resulting in mortality. The most conservative model estimated number of annual ship strike deaths along the U.S. West Coast in this study was 43 fin whale and 22 humpback whales, though these estimates includes only the period July – November when whales are most likely to be present in the California Current Ecosystem. We note that these estimates may not encompass all the potential risk of vessel collisions for humpback whales, since based on the last 5 years (2016-2020), five out of 14 (36%) reported vessel strikes occurred off or within Washington waters (e.g., Strait of Juan de Fuca, Puget Sound, where they are vulnerable to vessel traffic, including ferries). A comparison of average annual vessel strikes observed over recent historical time periods with these new estimates suggest that the detection rate for fin whale and humpback whale ship strikes could be 5% and 12%, respectively (Carretta et al. 2023a, b). Based on estimates of 22 annual deaths due to vessel strikes, the number attributed to the Central America / Southern Mexico - CA-OR-WA and Mainland Mexico-CA/OR/WA humpback whale stocks during 2016-2020 by Carretta et al. (2023a, b) are 6.45 and 10.15 humpback whales struck by vessels/year, respectively.

There is a large amount of uncertainty surrounding what the true number of ship collisions and mortalities are for these species. NMFS has determined that recent stock assessments of the blue, fin, and humpback whale stocks off the U.S. West Coast do not support an assertion that the current level of vessel strikes (whether reflected by the Rockwood et al. estimates or not) are impeding the recovery of these stocks (NMFS 2022c). In addition, we note that Rockwood et al. (2017) used a probability of avoidance of 55 percent based on existing data. However, other studies have shown a greater avoidance behavior by large whales (Lesage et al. 2017; Garrison et al. 2022), including humpbacks (Schuler et al. 2019), so the encounter rate may be lower than estimated in Rockwood et al. (2017). The populations of these species continue to show signs of recovery with increasing abundance on the U.S. West Coast despite the ongoing and persistent nature of this threat, although there have been actions in recent years that may be helping to reduce the number of vessel collisions that occur. There has been a long-term trend of decreasing vessel speeds across all U.S. West Coast waters (Moore et al. 2018), likely due to several factors that include response to increasing fuel costs and air pollution regulation. In addition, for over 10 years, NOAA has established seasonal voluntary Vessel Speed Reduction (VSR) zones off of California that have requested that all vessels 300 gross tons (GT) or larger decrease speeds to 10 knots or less to reduce the risk of vessel strikes on endangered whales. Cooperation rates with VSR have been increasing over this time, although they are still modest overall (Morten et al. 2022), and the potential changes (reductions) in risk associated with VSR have been explored (Rockwood et al. 2020, 2021).

The Port and Waterways Safety Act (PWSA) authorizes the Commandant of the Coast Guard to designate necessary fairways and traffic separations schemes (TSSs) to provide safe access routes for vessels proceeding to and from United States ports. The USCG completed Port Access Route Studies for the Santa Barbara Channel and the approaches to San Francisco and made recommendations to the IMO that the TSSs be modified, in part, to reduce the co-occurrence of large ships and whales. In February 2017, NMFS completed section 7 consultation on the U.S.

Coast Guard's codification of the shipping lanes that vessels use to approach the ports of Los Angeles/Long Beach and San Francisco. Following formal consultation under ESA section 7, NMFS concluded that the proposed TSS lanes were not likely to adversely affect or jeopardize ESA-listed humpback, blue, and fin whales. On December 7, 2022 the United States District Court issued an order in *Center for Biological Diversity, et al. v. NOAA Fisheries, et al.,* Case No. 4:21-cv-00345-KAW (N.D. Cal.), vacating the biological opinion. On June 5, 2023, the Coast Guard announced the availability of the study results of the Pacific Coast Port Access Route Study (88 FR 36607). This study evaluated safe access routes for the movement of vessel traffic proceeding to or from ports or places along the western seaboard of the United States.

2.4.1.3 Whale Watching Operations and Scientific Research

Whale watching boats and research activities directed toward whales may have direct or indirect impacts on fin, humpback, and sperm whales as harassment may occur, preferred habitats may be abandoned, and fitness and survivability may be compromised if disturbance levels are too high. Specifically, whale watching companies throughout the U.S west coast, especially areas of Southern California and Monterey Bay, are the beneficiaries of the large amount of whale activity occurring in nearshore coastal waters. Individuals of all these whale species are known to visit the action on an annual basis during migrations. To date, there have been no indications or scientific studies suggesting that whale watching activities are significantly affecting any of these whale populations along the U.S. west coast. A review of the NMFS Authorizations and Permits for Protected Species (APPS) database indicates that currently 17 scientific research projects that include directed research on fin, humpback, and/or sperm whales off the U.S. west coast. Most of these projects include some level of harassment for close approach, photography, acoustic monitoring, and/or sampling for biological data or deployment of tags. These activities are intended to be non-injurious, with only minimal short-term effects, although risks of more significant injuries or impacts do exist. A recent biological opinion (NMFS 2020b) analyzing the effects of NMFS' SWFSC research program concluded that there were no adverse effects to ESA-listed large whales covered under this opinion (sperm whale, humpback whale and fin whale).

2.4.1.4 Other Threats and Strandings

Other threats or sources of harm for ESA-listed whales in the action exist, although often times strandings occur where the source of injury or mortality is unknown. These events may also include natural mortalities with no human cause. With respect to fin whales, there were 9 strandings reported from 2017-2021 along the coast of California where researchers could not determine whether or not the strandings could be attributed to human interaction (NMFS unpublished stranding data). For example, one report described trauma to the body, with no apparent cause. Therefore, we could not confirm or attribute the stranding to ship strike or fisheries interactions. From 2017-2021 a total of 35 humpback whale strandings were reported off the U.S. West Coast (NMFS unpublished stranding data), most of them found in areas that were difficult to access, or when accessed, the whales were too decomposed to examine or necropsy, or they were floating offshore, often with moderate decomposition. Similarly, these strandings weren't typically attributed to any specific causes, which could include natural mortality, but a few were suggestive of possible trauma or had old entanglement scars. From 2017-2021, six out of 10 sperm whales stranded off the U.S. West Coast (NMFS unpublished stranding data) where human-caused mortality could not be determined. These also were not attributed to specific causes, which could include natural mortality. The potential effects to ESA-listed marine mammals from oil spills and other activities associated with oil and gas development off Southern California have been evaluated in previous consultations with Bureau of Ocean Energy Management (BOEM) and the Bureau of Safety and Environmental Enforcement (BSEE), including most recently in 2017 (NMFS 2017b). Up to this point, NMFS has concluded these activities are not likely adversely affect ESA-listed marine mammals. A new consultation with BOEM/BSEE on oil and gas development in Southern California is in process resulting from a settlement reached in Center for Biological Diversity v. Haaland, et al., Case No. 2:22-cv-555-SPG-AS (C.D. Cal.).

2.4.2 Sea Turtles

As described above in the status section, East Pacific DPS green, leatherback, North Pacific DPS loggerhead, and olive ridley sea turtles are all exposed to threats associated with fisheries bycatch, including U.S. fisheries that occur in the action area. Other impacts to ESA-listed sea turtles that may occur while present along the U.S. West Coast include vessel collisions, scientific research, ingestion or entanglement in plastics or marine debris, as well as changes in climate or oceanographic conditions. Historically, entrainment in coastal power plants was an issue, although this risk has been reduced through mitigation measures and changes in power plant operations over time.

2.4.2.1 Fisheries Interactions

Along the west coast of the U.S. in the California Current Ecosystem, the four sea turtle species considered in this biological opinion are occasionally reported and/or observed interacting with fishing gear, including pot/trap gear, recreational hook and line gear, and gillnets. All four species have been observed taken in the DGN fishery (see the *Effects of the Action* section 2.5), although sea turtle interactions are now considered rare events in this fishery since the Pacific Sea Turtle Conservation Areas have been put in place (section 1.3). Since 2001 in the DGN fishery, two loggerheads have been observed taken and released alive (one in 2001 and one in 2006), and two leatherbacks have been observed taken and released alive (one in 2009 and one in 2012). Only one green and one olive ridley have been documented interacting with the DGN fishery (both in 1999; Carretta 2022).

As shown in Figure 8, in over 40 years (1975-mid-2016), only four loggerheads, 10 leatherbacks, and 14 green turtles, were documented stranded with fishing gear in California. Note that this may include recreational fishing gear and does not include information reported from observer programs, including the DGN observer program.

In other commercial fisheries along the U.S. West Coast, sea turtle bycatch has only rarely been documented. In 2008, one leatherback was found entangled (dead) in sablefish trap gear fishing offshore of Fort Bragg (NMFS 2012a). No leatherbacks have been observed entangled in this gear since 2008, through 2022 (data from 2002-2022; Benson et al. 2021; NMFS-WCR groundfish observer program, unpublished data). Under a 2012 biological opinion for the West Coast groundfish fishery, NMFS anticipated a maximum allowable level of entanglements across five years to be 1.9. Over the most recent 5-year period analyzed for this fishery, Benson et al. (2021) estimated that zero leatherbacks had been caught by the fleet. One leatherback was found dead entangled in unidentified pot/trap gear in 2015 off central California, and one leatherback was found entangled in Dungeness crab pot gear and released alive in 2016 (NMFS WCR stranding database, unpublished). More recently, in 2018, a dead leatherback was found floating offshore in Ventura County entangled in lines attached to two buoys (unknown fishery), which was subsequently identified as rock crab gear (NMFS WCR unpublished stranding data, 2022).

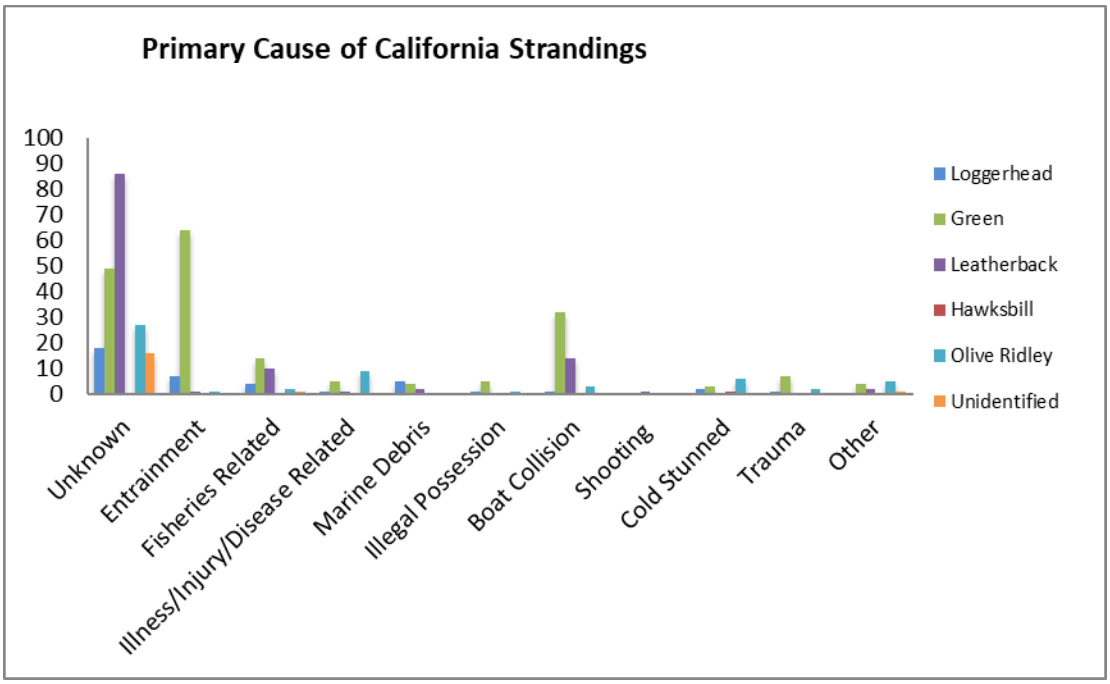

Figure 8. Known cause of sea turtle strandings in California, 1975-2016 (R. LeRoux, NMFS-SWFSC, unpublished data).

A review of the most recent stranding records (2017-2021) reveal two reports of loggerhead interactions with fishing gear, one off Oxnard in Ventura County (unidentified netting reported) and one entangled off Warrenton, Oregon with unidentified netting and fishing line in its mouth (NMFS WCR unpublished stranding data).

Green turtles have been found entangled and hooked in fishing gear; but most, if not all, have been documented interacting with recreational fishing gear. Of 116 green turtles that stranded off California between 2017 and 2021, 15 animals were found hooked (including ingested hooks) or entangled (or ingested) in fishing gear, all of it appearing to be recreational fishing gear (NMFS WCR unpublished stranding data). All were found within bays and estuaries or in the nearshore coastal areas, which further suggest that the likely interactions were with hook and line (recreational) gear. Most (n=9) were found alive, and most were able to be released either following the removal of gear, or following rehabilitation. No olive ridley turtles were found interacting with fishing gear, either commercial or recreation, during the same most recent time period (2017-2021).

When considering the impact of U.S. West Coast federal fisheries on ESA-listed species of turtles, recent biological opinions have found no jeopardy to any of these species (NMFS 2012a, 2013, 2016d). There are two state gillnet fisheries in California that may interact with sea turtles: the set gillnet fishery targeting halibut and white seabass; and the small mesh drift gillnet fishery targeting yellowtail, barracuda, and white seabass. No sea turtle interactions have been documented historically or recently, although observer coverage of these fisheries has been limited and irregular.

2.4.2.1.1 HMS Experimental Fisheries Permits

In 2018 and 2019, NMFS SFD consulted upon and/or issued 4 EFPs for HMS species recommended by the PFMC that may occur within the proposed action area. These EFPs include: DSBG issued in 2018 (NMFS 2018a); Deep-Set Linked Buoy Gear (DSLBG) issued in 2018 (NMFS 2018b); Longline Gear (LL), including deep-set longline gear (DSLL) and shallow-set longline gear (SSLL), issued in 2019 (NMFS 2018c); and Deep-Set Shortline (DSSL) consulted on in 2019 (NMFS 2019b). Through consultation NMFS ultimately determined that ESA-listed species, including all ESA-listed species considered in this biological opinion, would not be adversely affected by 3 of these EFPs: DSBG, DSLBG, and DSSL. Through formal consultation, NMFS determined that the LL EFP was likely to result in the take of ESA-listed sea turtles, including North Pacific DPS loggerhead, leatherback, and olive ridley sea turtles. Specifically, over the course of 2 years the LL EFP was expected to result in: as many as 2 loggerhead sea turtle entanglements, with 1 mortality; as many as 2 leatherback sea turtle entanglements, with 1 mortality; and no more than 1 olive ridley sea turtle entanglement and mortality (NMFS 2018c). The LL EFP was issued in April, 2019, and was set to expire after two years. Two fishermen fished DSLL and SSLL for around three months in 2019 with no interactions with sea turtles (100 percent coverage). On December 20, 2019, a federal court vacated the EFP, final EA, and biological opinion as a result of litigation on the issuance of the LL EFP. No SSLL or DSLL fishing activity has occurred within the West Coast EEZ under the EFP since the court's ruling, and both NMFS and the EFP permit applicants are considering options for how to proceed in the future.

2.4.2.2 Entrainment in Power Plants

In 2006, a biological opinion was completed and analyzed the effects of sea turtle entrainment in the two federally-regulated nuclear power plants located in California, the Diablo Canyon Power Plant found in San Luis Obispo County and the San Onofre Nuclear Generating Station found near San Clemente California (NMFS 2006a). While historically loggerheads, leatherbacks and olive ridleys were observed entrained in the power plants in very low numbers, since 2006, there have been only two reported entrainments, both in the San Onofre Nuclear Generating Station, one olive ridley (alive) in 2009, and one loggerhead (alive) in 2010. In addition, the San Onofre station began de-commissioning in 2014, although some cooling water is still drawn in to cool the reactors (D. Lawson, NMFS personal communication 2015). The incidental take statement covering both power plants estimates up to 6 loggerheads taken, 6 leatherbacks taken, and 6 olive ridleys taken (with two serious injuries each and two mortalities each for all three species) over a one year period (NMFS 2006a).

There are other coastal power plants in California (non-nuclear and state-managed) where sea turtle entrainment has occurred (typically green sea turtles). Although these facilities have all been required to install large organism excluder devices by the State of California (California State Water Resources Control Board (CASWRB) 2010), occasional instances of green turtle entrainments (typically alive) continue to be reported. As shown in Figure 8, only seven loggerheads were entrained in power plants over the last 40 years (1975-late-2016), and a review of the records from 2017-2021 showed no reports of entrained loggerheads. During that same time period (1975 through late-2016), 64 green turtles were entrained (most released alive). Since then, only three green turtles have been entrained in power plants, all released alive (2017-2021; NMFS-WCR unpublished stranding records). Over that same earlier time period, only one leatherback was entrained and no leatherbacks have been entrained in power plants based on stranding data from 2017-2021 (NMFS-WCR unpublished stranding data).

2.4.2.3 Scientific Research

NMFS issues scientific research permits to allow research actions that involve take of sea turtles within the California Current. Currently there are 2 permits that allow directed research on sea turtles, typically involving either targeted capture or sampling of individuals that may have stranded or incidentally taken in some other manner. These permits allow a suite of activities that include tagging, tracking, and collection of biological data and samples. These activities are intended to be non-injurious, with only minimal short-term effects. But the risks of a sea turtle incurring an injury or mortality cannot be discounted as a result of directed research. Prior to completing a section 7 ESA consultation on the Southwest Fisheries Science Center's programmatic research program, one leatherback was found during a scientific trawl net survey in 2011 and was released alive (NMFS 2015). The most recent biological opinion analyzed the effects of proposed SWFSC research surveys and estimated that one ESA-listed sea turtle found within the action area (any species of leatherback, North Pacific loggerhead, olive ridley and East Pacific green turtle) may be captured in CCE trawl surveys and one ESA-listed sea turtle may be captured/entangled in longline surveys, with both released alive (NMFS 2020b). The section 7 ESA consultation on the Northwest Fisheries Science Center's programmatic research program was completed in 2016 and estimated one loggerhead taken annually, one leatherback taken annually, and one olive ridley taken annually (no mortalities; NMFS 2016e).

2.4.2.4 Vessel Collisions

Vessel collisions are occasionally a source of injury and mortality to sea turtles along the west coast. A review of the strandings database for the U.S. west coast maintained by NMFS indicates that green sea turtles and leatherbacks are reported most often as stranded due to the impact by vessels strikes (Figure 8), although only approximately 15 leatherbacks were reportedly struck by vessels between 1975 and late 2016 (around 1 every 3 years), and many of these collisions occur off central California, when they are foraging in or near the approach to the ports of San Francisco and Oakland. A review of the stranding records from 2017-2021 indicated no reported vessel strikes off California and Oregon. As shown in Figure 8, one loggerhead was reportedly struck by a vessel in the last 40 years (1975-late 2016), although a review of the records from 2017-2021 revealed that two loggerheads were reportedly struck by vessels off Los Angeles (Long Beach) and San Diego County (Pacific Beach; NMFS WCR unpublished stranding data). In southern California (and including the state of California), green turtles are by far the most frequent species of sea turtles struck by vessels (including jet skis, small power boats, etc.). As shown in Figure 8, from 1975 through late 2016, 32 green turtles were suspected to be struck by vessels, with most resulting in mortality. In a review of the stranding records from 2017-2021, of 116 reported strandings of green turtles in California and Oregon, 29 of them were reported (suspected) struck by vessels, with almost all of them dead (28 animals; NMFS WCR unpublished stranding data). Most were in moderate to advanced decomposition, which often makes it difficult to determine a cause of death, although a cracked carapace or deep lacerations are usually a good indicator of blunt force trauma with a vessel's hull or propeller.

As described above, the PWSA authorizes the Commandant of the Coast Guard to TSSs to provide safe access routes for vessels proceeding to and from United States ports. In February 2017, NMFS completed section 7 consultation on the U.S. Coast Guard's codification of the shipping lanes that vessels use to approach the ports of Los Angeles/Long Beach and San Francisco. Following formal consultation under ESA section 7, NMFS concluded that the proposed TSS lanes were not likely to adversely affect or jeopardize ESA-listed sea turtles, including green, North Pacific DPS loggerhead, olive ridley, and leatherback sea turtles. On December 7, 2022 the United States District Court issued an order in *Center for Biological Diversity, et al. v. NOAA Fisheries, et al.,* Case No. 4:21-cv-00345-KAW (N.D. Cal.), vacating the biological opinion. On June 5, 2023, the Coast Guard announced the availability of the study results of the Pacific Coast Port Access Route Study (88 FR 36607). This study evaluated safe access routes for the movement of vessel traffic proceeding to or from ports or places along the western seaboard of the United States.

2.4.2.5 Contaminants

In southern California, green turtles forage in urbanized environments and therefore are more exposed to anthropogenic contaminants and pollutants. Sea turtles captured in Seal Beach and San Diego Bay in southern California were found to have higher trace metal concentrations (e.g., selenium and cadmium) than green turtles that inhabit non-urbanized areas (Barraza et al. 2019). A related study found that green sea turtles foraging in San Diego Bay had significantly higher total polychlorinated biphenyls (PCBs) than turtles in Seal Beach, and that these non-dioxin-like PCB congeners may be associated with neurotoxicity (Barraza et al. 2020).

2.4.2.6 El Niño/Changing Climate

El Niño events occur with irregularity off the U.S. west coast and are associated with anomalously warm water incursions. Sea turtles may be affected by El Niño event through a change in distribution or abundance of their preferred prey, which may result in a change in sea turtle distribution or behavior. These warm water events often bring more tropical marine species into normally temperate waters and therefore may affect the local ecosystem and normal predator-prey relationships. For example, North Pacific loggerheads have been encountered off the U.S. west coast in large numbers during an El Niño. Loggerhead presence in the SCB was first documented in the CA drift gillnet fishery during the 1990s, when they were taken by the fishery during years associated with El Niño events (1992-93 and 1997-98). Anomalously warm waters bring pelagic red crabs, a preferred prey item of loggerheads and may have brought loggerheads into the area, although they have also been documented associating with pyrosomes during the 2014 incursion of warm water into the waters off California.

We considered the effect of climate change on sea turtles foraging in the action area and/or migrating to and from their nesting beaches or other areas of the Pacific Ocean. While climate change effects have been documented extensively on sea turtle nesting beaches, there is less information available on the effects of climate change on sea turtles specifically within in the action area. Generally we suspect that some sea turtle species may shift their distribution north as sea surface temperatures increase, which could bring them into more contact with human activities that occur off the U.S. West Coast. The recent research described in Section 2.2.4 above suggest that the presence of loggerhead sea turtles should be expected to increase if warmer sea surface temperatures in the SCB occur and persist in the future (Eguchi et al. 2018; Welch et al. 2019). However, over limited duration of the proposed action over the next five years, it will be difficult to detect if shifts associated with climate change are happening.

2.4.2.7 Other Threats and Strandings

Strandings of sea turtles along the U.S. west coast reflect in part the nature of interactions between sea turtles and human activities, as many strandings are associated with human causes. Sea turtles have been documented stranded off California (and Oregon and Washington, though in less frequent numbers) through their encounters with marine debris, either through ingesting debris or becoming entangled in the debris. Concentrations of plastic debris have been documented widely in the last decade, with the North Pacific Ocean showing similar patterns in other oceans, with plastics concentrating in the convergence zone of all five of the large subtropical gyres. Since the 1970s, the production of plastic has increased five-fold, with around 50% of it buoyant (summarized in Cozar et al. 2014). Studies documenting marine debris ingestion by sea turtles indicate impaired digestive capability, “floating syndrome,” or reduced ability to swim, in addition to death (Casale et al. 2016). In addition, studies of marine debris ingestion in green turtles (Santos et al. 2015) and loggerheads (Casale et al. 2016) indicated that the potential for death is likely underestimated, as is the magnitude of the threat worldwide, particularly for highly migratory species. The potential effects to ESA-listed sea turtles from oil spills and other activities associated with oil and gas development off Southern California have been evaluated in previous consultations with BOEM BSEE, including most recently in 2017 (NMFS 2017b). Up to this point, NMFS has concluded these activities are not likely adversely affect ESA-listed sea turtles. A new consultation with BOEM/BSEE on oil and gas development in Southern California is in process resulting from a settlement reached in Center for Biological Diversity v. Haaland, et al., Case No. 2:22-cv-555-SPG-AS (C.D. Cal.).

A study assessed the health of leatherbacks foraging off California and measured hematologic and plasma chemistry values. When these values were compared to nesting female leatherbacks in French Guiana and St. Croix, the foraging turtles were found to have elevated levels of Cadmium but Harris et al. (2011) note that biomagnification of trace elements via trophic transfer might be limited in this species due to their preference for cnidarian zooplankton. The authors note that hard-shelled turtles such as loggerheads, which have a more varied diet such as crustaceans and bivalves, have shown high levels of PCBs and dichlorodiphenyldichloroethene (DDE), when compared to more herbivorous consumers, such as green turtles. Domoic acid, which is a potent marine algal toxin that has been shown to cause neurologic disease in marine mammals and sea turtles was found in a stranded dead leatherback in 2008 (Harris et al. 2011). Other documented threats to sea turtles found off the U.S. west coast include illness, gunshot wounds, and unknown illnesses (usually cold-stunning, particularly for olive ridleys). Because not all dead stranded sea turtles are necropsied, the stranding database does not provide full documentation of the source of many threats to sea turtles, and the causes of a majority of strandings are unknown. This is especially true for leatherbacks, since they are often difficult to access and transport to a laboratory, given their size and rate of decomposition (Harris et al. 2011).

Figures 9 and 10 show the historical data on sea turtle strandings off the U.S. West Coast since 1958, including information on trends, species, and area along the coast. There are fewer strandings of sea turtles in the Pacific Northwest (Figure 10), although they do occur and are documented. A review of the most recent stranding information (2017-2021) for leatherbacks revealed four stranded turtles (one fishery-related stranding, described above). One juvenile leatherback stranded dead in Orange County, California in 2017 with evidence of trauma, but this may have been post-mortem. In 2020, a leatherback was found in San Francisco Bay but cause of death could not be determined as the animal was never recovered. Finally, in 2021 an adult leatherback stranded dead in Douglas County, Oregon with unknown cause of death, but the animal had markings indicative of a predation event and also had a puncture wounds and pieces of plastic in stomach/intestines (NMFS WCR unpublished stranding data, 2022).

A review of the most recent stranding information (2017-2021) for loggerheads, revealed 14 strandings off California and Oregon. All but one were identified as juveniles (one was unknown age class but likely a juvenile). Five loggerheads stranded in Oregon during the winter months (February/March) over this five-year period, mostly cold-stunned, although one showed signs of trauma/predation. One loggerhead stranded in northern California in February, so was likely also cold-stunned. One loggerhead stranded in Oregon and another in the SCB with signs of fishery interactions (described above), while two loggerheads stranded in southern California with signs of a vessel strike (described above). One loggerhead stranded with a string around its neck, was disentangled and released alive. Lastly, three loggerhead turtles stranded in San Diego County where cause of death could not be determined (NMFS WCR unpublished stranding data).

Strandings of olive ridleys increased in northern California and the Pacific Northwest since late 2014 (NMFS WCR stranding data, unpublished), with most of them cold-stunned (n=6 from 1975-late-2016), likely following the warm water incursion associated with a strong El Niño, which occurred during that time period through the fall of 2016. No olive ridleys were reported stranded in 2017-2021.

Many green turtles have reported stranded off California and Oregon where the cause of injury/death cannot be determined, especially when some are found with moderate to advanced decomposition. From 2017-2021, 66 green sea turtles stranded alive, injured and/or dead off California and Oregon, with the cause of death undetermined. In most cases, NMFS experts could not determine whether human interaction played a factor in the stranding, either because of the lack of details or the moderate to advanced decomposition of the animal.

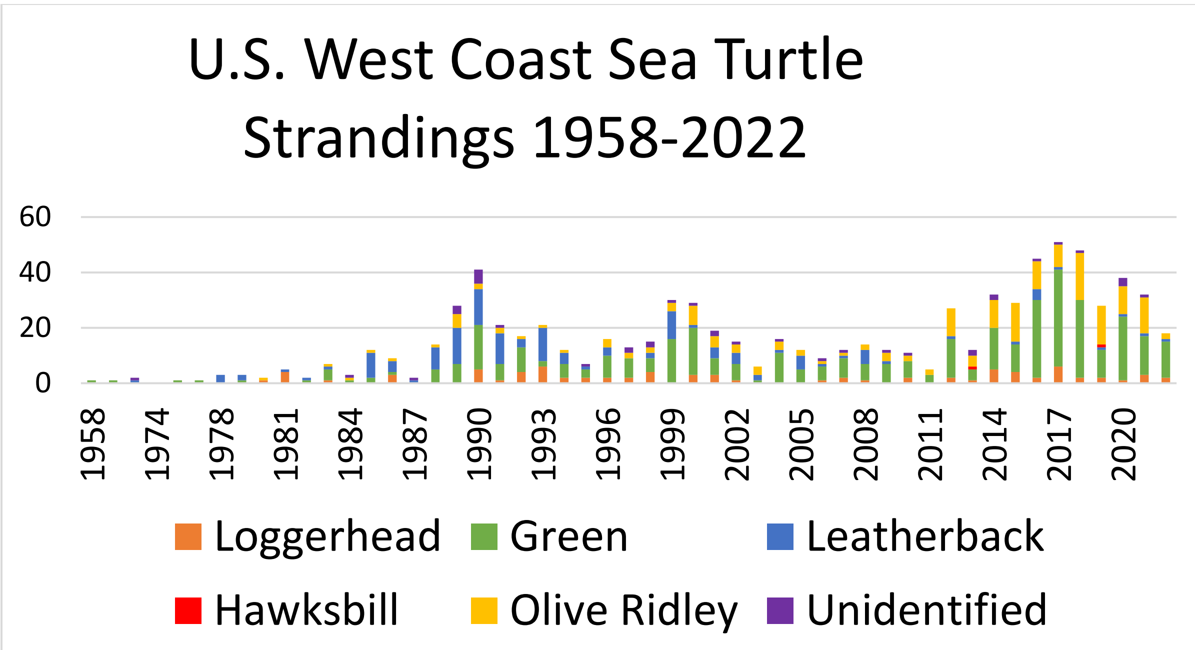

Figure 9. U.S. West Coast Sea Turtle Strandings, 1958 through mid-2022 (R. LeRoux, NMFS, unpublished data, 2022).

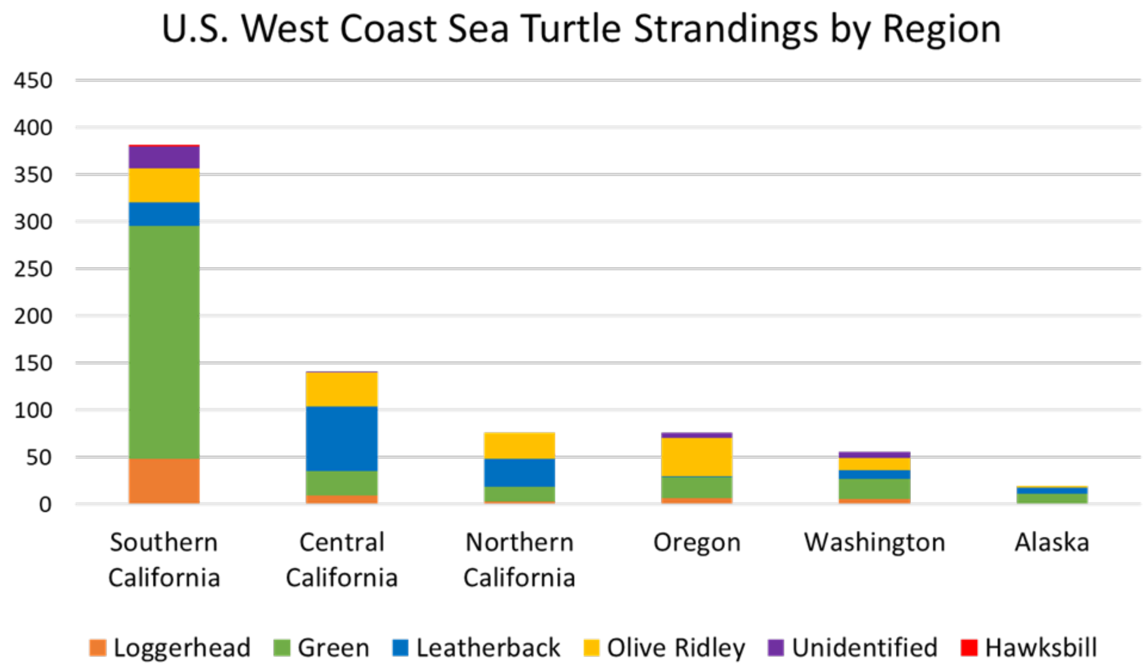

Figure 10. U.S. west coast sea turtle strandings by region and species, 1958-2021 (R.LeRoux, NMFS, unpublished data, 2022).

2.4.3 Giant Manta Ray

As mentioned, little is known regarding threats to the giant manta ray within the action area other than what has been documented in the CA DGN. As reviewed in the 2017 status review (Miller and Klimovich 2017), giant manta rays are occasionally observed as bycatch in the CA DGN, but in low numbers and primarily during El Niño events. From 1990-2006, only 14 giant manta rays were observed caught, with 36 percent released alive. The estimated (extrapolated) catch for that period was 90 individuals (95% CI: 26-182; CV=0.48) (Larese and Coan 2008). Since 2005, no giant manta rays have been observed in this fishery (Pacific Fisheries Information Network public data: https://reports.psmfc.org/pacfin). While purse seine fishing has posed a threat to giant manta rays in the eastern tropical Pacific Ocean, we have no records of this species being taken in our coastal pelagic species purse seine fishery.

Although giant manta rays are rarely found off the U.S. West Coast, and there have been no identified individuals or subpopulations within the EEZ, any manta ray foraging off the coast could be impacted by plastics ingestion or entanglement in marine debris. Climate change and ocean acidification could affect the distribution and abundance of zooplankton, which giant manta rays depend on, so individuals may have to travel further distances to find their preferred prey.

2.5 Effects of the Action

Under the ESA, "effects of the action" are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action (see 50 CFR 402.02). A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (see 50 CFR 402.17). In our analysis, which describes the effects of the proposed action, we considered the factors set forth in 50 CFR 402.17(a) and (b).

For the *Effects of the Action* analysis, we have identified the impact of capture or entanglement in DGN gear as the primary effect of the DGN fishery on ESA-listed species. In this effects analysis, the terms bycatch and entanglement are used interchangeably, as the primary mode of bycatch for ESA-listed species in the DGN fishery is entanglement in the net or any component such as buoy extender lines that could result in or contribute to an entanglement. There are other potential impacts that could occur as a result of the fishery, such as vessel collisions or impacts related to any pollution or marine debris generated by this action. It is also conceivable that impacts to prey might affect ESA-listed species,[17] or that avoidance of DGN gear could lead to increased energetic expenditure or temporary exclusion from important foraging resources. At this time, the available information does not suggest that any of these additional factors are affecting ESA-listed species as a result of the continued operation of the DGN fishery. Without evidence to support analyses of how these factors may affect ESA-listed species as a result of the proposed action, NMFS concludes these factors are insignificant or discountable. As a result, the effects analysis will concentrate on the impact of bycatch of ESA-listed species in the DGN fishery.

Exposure and Response

In order to determine the exposure of ESA-listed species to the DGN fishery, NMFS relies primarily upon data provided by fisheries observers. NMFS has been deploying observers in the DGN fishery since 1990, and the observer program represents an objective sampling scheme that constitutes the best available information regarding the frequency and trends in DGN fishery bycatch over time. Historically, annual estimates of bycatch were generated by NMFS using ratio estimators, although more recently Carretta and Moore (2014) noted that annual estimates of bycatch derived from ratio estimates for 'rare-event' cases like the bycatch of ESA-listed species in the DGN fishery were biased, volatile, and imprecise, particularly when observer coverage was low. Carretta and Moore (2014) also noted that strategies for pooling annual ratio estimates of bycatch in U.S. marine mammal stock assessments (5 years are typically pooled to calculate average annual bycatch) are insufficient to overcome these problems. In order to improve bycatch estimates, Carretta (2022) has applied a machine-learning approach of random forest trees (Breiman 2001) to generate annual estimates of bycatch in the DGN fishery from 1990-2021.

[17] See discussion of leatherback critical habitat in *Status of the Species* section 2.2.5 for a specific analysis.

In order to generate an estimate of the anticipated future bycatch of ESA-listed species in the DGN fishery, we relied upon the bycatch estimates from Carretta (2022) to generate an expected bycatch rate per unit of fishing effort (number of individuals caught per set of fishing) for each ESA-listed species of marine mammal and sea turtle. Similar estimates have not been generated for the giant manta ray, although we rely upon the available data on historical bycatch rate for this species to anticipate the future bycatch of this species. The data used includes bycatch estimates from 2001-2021 that represent data from the most recent past DGN effort that would be expected to be generally consistent with future effort including all of the current regulations and limitations on the DGN fishery. The total estimated take rate from 2001-2021 for each species calculated using the random forest tree estimates of bycatch per set during this time period was then projected into the future by anticipating that up to 305 sets may be fished each year on average in the DGN fishery for the remaining five years of the proposed action, although up to 573 sets could occur in any one year. In the previous biological opinion, we anticipated that up to 1,500 sets may occur (NMFS 2013). SFD proposed to revise this expectation to up to more accurately reflect current fishery operations and the outlook for the last five years of the DGN fishery. Finally, these annual estimates of bycatch were projected over a period of 5 years to produce total estimates of bycatch for each ESA-listed species over that time frame (see Table 8). We note here that although bycatch data from DGN fishery used in the generation of take estimates only incorporates data through 2021, there were no observed takes of ESA-listed species in the DGN fishery in 2022 (NMFS observer program data).

The exposure analysis below presents bycatch estimates from two perspectives: 1) what could be expected to occur in any 1 year; and 2) what would be expected to occur over the next five years. Both concepts are useful for monitoring the impact of the DGN fishery on ESA-listed species, especially given the prospect of approximately 20 percent observer coverage in this fishery, and both will be used to frame the ITS of this biological opinion. Given the data and projected effort, estimates of annual bycatch for each species are relatively low numbers (Table 8. Documented reports of ESA-listed species entanglements in this fishery occur vary rarely, especially during more recent time periods. As a result, we conclude that the anticipated bycatch that could occur in any year could approach the total anticipated bycatch anticipated to occur over a 5-year period.

In order to determine the response of individuals from bycatch in DGN gear, NMFS relies primarily upon the accounts of injury and mortality for each species provided from observer records, and assessments of mortality and serious injury that are conducted for the marine mammal SARS, as appropriate. The bycatch estimates produced by Carretta (2022) incorporate the most updated information regarding the response and likely outcomes of bycatch in DGN gear from the available data and SARs assessments, and we rely upon that information to help inform the expectation for response of each ESA-listed species in this biological opinion.

Risk

In order to measure the risk to the affected populations, NMFS calculates the expected mortality using the estimated rates of bycatch for each species and the expected mortality rates, from both the context of what could happen in a given year, and what would be expected to occur over a 5-

year period. For marine mammal species and giant manta rays, we assume that all individuals are of equal value to the population regardless of age or sex, in terms of lost reproductive capacity to the population. For sea turtles, additional calculations are made to convert mortalities to adult female equivalents. This is a standard approach to assessing impacts to sea turtles as the only metrics of sea turtle population abundance and trends generally available relate to the number of nesting females or nesting production that has been recorded over time, whereas population assessments of marine cetaceans (and other species) typically count all individuals and do not differentiate between sexes or age classes in population estimates.

Review of Other Data

As described in the *Environmental Baseline* (section 2.4), there have been substantial increases in the number of reports of entangled whales received by the NMFS WCR Marine Mammal Stranding Program, especially humpback whales. In addition to characterization of recent entanglement data as part of the *Environmental Baseline* (section 2.4), we reviewed all entanglement reports available from 2013-2022 that were associated or identified as potential gillnet entanglements based on the descriptive information and documentation provided in those reports. The purposes of the gillnet entanglement review were to evaluate general characteristics of gillnet entanglements (*e.g*., location of gear on animals) and general details commonly associated with gillnet entanglement reporting (*e.g.*, descriptions of gear and materials seen or photographed), as well as to identify any reports that appear to be consistent with DGN gear. In the *Effects Analysis* (section 2.5), we consider relevant results of this review along with fisheries observer data as part of the available information regarding DGN interactions with ESA-listed species.

2.5.1 Exposure and Response to Interactions with the DGN Fishery

All eight of the species/DPSs considered in this biological opinion have been documented by fishery observers as bycatch in the DGN fishery over the last two decades.[18] For this biological opinion, we have anticipated bycatch rates for eight ESA-listed species/DPSs based on the historical record of bycatch events observed and the expectation that up to 573 sets could be made in one year, and the effort will average 305 sets per year over the next five years, based on the methodology described above (Table 8). For giant manta rays, where no bycatch estimates have been made by Carretta (2022), we consider the fact that only one incidence of bycatch has been observed from 2001-2021 in nearly 3,300 observed sets. This equates to an anticipated bycatch rate of 0.09 takes per 305 sets (per year) for this species.

Table 8. Estimated future bycatch of ESA-listed species in the DGN fishery per year (assuming an average of 305 sets per year, with up to 573 sets occurring in any one years) and total over the next five years.

[18] In 2009, NMFS received a report of a humpback whale caught in DGN fishery off the coast of San Diego from a commercial fisherman during an unobserved trip. The report indicated that the whale was released alive and actively swimming away with some unknown quantity of gear remaining attached.

Species	Total DGN Fishing Effort (sets), 2001-2021	Total Estimated Bycatch, 2001 through 2021	Estimated Maximum Annual Bycatch (573 sets)	Estimated Average Annual Bycatch (305 sets)	Estimated Total Bycatch Over 5 Years[19]	Estimated Bycatch in 2013 Biological Opinion[20]
Fin Whale	16,634	0.95	0.03	0.02	1	2
Humpback Whale	16,634	10.6	0.37	0.20	1	4
Sperm Whale	16,634	12.2	0.42	0.22	2	8
Leatherback Turtle	16,634	15.6	0.54	0.29	2	10
Loggerhead Turtle	16,634	23.1	0.79	0.42	3	7
Olive Ridley Turtle	16,634	1.42	0.05	0.03	1	2
Green Turtle	16,634	2.03	0.07	0.04	1	2
Giant Manta Ray	16,634	n.a.	0.17	0.09	1	n.a.

As described previously, a number of significant actions have been taken to address bycatch of sea turtles and marine mammals, both ESA and non-ESA-listed species. As a result, the current DGN fishery is considerably different than the historical fishery (pre-2000); for example, gear modifications and time/area closures have been implemented to avoid areas prone to higher bycatch rates. In addition, the total effort by the fleet is much reduced and the spatial extent of the fishery has been largely constrained compared to historical effort as well. While overall bycatch rates for many species appear to be low, bycatch events still occur. However, these bycatch events are considered rare, making reliable estimates of bycatch challenging. For example, in the twenty-two years from 2001 through 2021 since the PLCA has been implemented, there have only been four observed interactions with sea turtles. A review of the effectiveness of the PLCA concluded that temporal extent of the current static closure period is the shortest and most effective for protecting the turtles while allowing fishing during low bycatch-risk periods (Eguchi et al. 2017).

[19] [1]All 5-year bycatch estimates are rounded up to a whole number to frame estimates as "up to" scenarios, as fractional numbers of bycatch are not realistic.
[20] NMFS 2013 biological opinion on the DGN fishery was based on annual effort up to 1,500 sets per year, using a different methodology than used current estimates in Carretta (2022).

2.5.1.1 Estimated Average Bycatch over the Next Five Years

The primary usefulness of the bycatch rate estimates described above (Table 8) is to describe the expected average annual bycatch rates of ESA-listed species over a more extended period of time (e.g., the next five years), as opposed to what might occur in any given year, given the small number and fractions of individual from each species that are estimated to be involved. Over the next five years, it is likely that bycatch rates for these species will more closely align with totals that reflect average bycatch rates generated by examining a long time series of empirical data, which may reflect annual variations in factors that influence the probability of bycatch, including distributions of species (target and non-target) that respond to changing environmental or oceanic conditions. The rarity of observed and reported interactions since 2001 does not support an assumption of consistent and/or multiple bycatch events each year, every year, for any or all of these species over time, although certainly we recognize that most bycatch goes unobserved given that the majority of fishing effort is unobserved each year. The bycatch levels presented below (Table 9) reflect an "up to" scenario for the bycatch expected over the next five years. The is generally consistent with how marine mammal stock assessment reports evaluate anthropogenic impacts to populations (average over most recent five years of data[21]), and is consistent with the general scaling of 20 percent observer coverage and likelihood of observing any event that occurs in the DGN fishery is approximately one out of five. Furthermore, while an average of 305 sets per year is anticipated over the next five years, and up to 573 sets may occur in a given year, as seen last in 2018, the annual estimated number of sets made in the DGN fishery has been considerably lower in recent years: with 323, 230, and 162 sets made in fishing seasons from 2019-2021, respectively. Over the most recent five years, the DGN fishery has averaged slightly less than 400 sets per year (362).

Although the estimates of total bycatch that has occurred over the most recent five-year period for most species are lower than what would generally be expected over the next five years in a 305-set DGN fishery using the historical bycatch rate estimated from 2001-2021, in some cases (e.g., humpback and sperm whales) the estimated bycatch that has occurred during the last five years is actually slightly higher than what would generally be expected (Table 9). This is a result of the influence of rare bycatch events such as the humpback whales observed in 2021 and recent variability in fishing effort patterns in contrast the "historical average." While these factors are understandable, they are difficult to predict.

Table 9. Expected total of bycatch in the DGN fishery for each species over the next five-year period, and the most recent bycatch estimates in the DGN for each species from the last five years.

ESA-listed species	Expected next 5 year bycatch totals	Most recent 5 year period (Carretta 2022)
Fin whale	**Up to 1**	0.0

[21] Changes to stock assessment for some marine mammal stocks such as sperm whales that are involved in rare bycatch events in fisheries like the DGN fishery have been evaluating even longer time series of impacts (Carretta et al. 2017).

Humpback whale	**Up to 1**	2.7
Sperm whale	**Up to 2**	2.3
Leatherback turtle	**Up to 2**	1.2
Loggerhead turtle	**Up to 3**	2.0
Olive ridley turtle	**Up to 1**	0.0
Green turtle	**Up to 1**	0.4
Giant manta ray	**Up to 1**	n.a.

2.5.1.2 Bycatch in a Given Year

In addition to looking at what the expected bycatch rate and total will be over time, we considered various scenarios of bycatch could occur in any given year for ESA-listed species in order to analyze the potential impact to affected populations. Since the full suite of bycatch reduction measures was implemented in 2001, there have been a total of 4 sea turtle, 5 ESA-listed marine mammal, and 1 ESA-listed marine fish interaction(s) recorded in almost 3,300 observed sets (Table 10).[22] If a generalized catch-per-unit-effort (CPUE) is calculated based on this level of observed bycatch and effort for each general species group (i.e., sea turtle, marine mammal, marine fish) and applied to an effort level of up to 573 sets in one year, the estimated level of bycatch is no more than about 0.7 sea turtles, 0.9 ESA-listed marine mammals, and 0.2 ESA-listed marine fish per year. In comparison, summing the maximum annual bycatch rate estimates in Table 10 across each species group results in cumulative estimates of 1.45 sea turtle, 0.82 ESA-listed marine mammal, and 0.17 ESA-listed marine fish entanglements in a 573 set DGN fishery.

Table 10. Observed bycatch of ESA-listed individuals in the DGN fishery 2001-2021 since bycatch reduction measures have been in place.

[22] While three gray whales have been observed taken in the DGN fishery from 2001 – 2021, based on their distribution and abundance it is most likely that they were from the non-ESA listed Eastern Pacific population and not the endangered Western North Pacific population (section 2.12.2 below).

Year	Sperm Whale	Humpback Whale	Leatherback Sea Turtle	Loggerhead Sea Turtle	Giant Manta Ray	Gray Whale
2001	0	0	0	1	0	0
2002	0	0	0	0	0	0
2003	0	0	0	0	0	0
2004	0	1	0	0	0	0
2005	0	0	0	0	1	1
2006	0	0	0	1	0	0
2007	0	0	0	0	0	0
2008	0	0	0	0	0	0
2009	0	0	1	0	0	0
2010	2	0	0	0	0	0
2011	0	0	0	0	0	0
2012	0	0	1	0	0	0
2013	0	0	0	0	0	1
2014	0	0	0	0	0	0
2015	0	0	0	0	0	0
2016	0	0	0	0	0	0
2017	0	0	0	0	0	0
2018	0	0	0	0	0	1
2019	0	0	0	0	0	0
2020	0	0	0	0	0	0
2021	0	2	0	0	0	0

These numbers illustrate the fact that observed bycatch rates of ESA-listed species in the modern DGN fishery have been low. Without question, fishing effort from the fleet as a whole has significantly decreased from the early 1990s when effort averaged over 4,000 sets annually (highest number was 5,442 in 1993), to an average of about 790 sets per year since 2001, with effort continuing to trend downward (see Tables 1 and 2 in *Proposed Federal Action* section 1.3). While it seems possible that this overall decrease in fishing effort is helping to further diminish the probability of bycatch for these protected species in comparison to historical reported bycatch prior to implementation of bycatch reduction measures, we acknowledge that the range of the DGN fishery has also been significantly affected by factors such as implementation of sea turtle conservation areas and diminishing participation among the DGN fleet, with effort more concentrated in the SCB during much of the season.

It is possible that changes in the DGN that have occurred since the 1990s, which include large reductions in fishing effort, implementation of minimum extender lengths, and seasonal closures during El Nino conditions for loggerhead sea turtle protection, have also worked to minimize the risk of giant manta ray bycatch. The extended absence of giant manta rays from the observer record and the likelihood that there have been changes in the exposure of this species to the DGN fishery over the last two decades, we conclude that the risk of bycatch in the current DGN fishery is low. Here we proceed with a conservative assessment of the bycatch of this species may occur for the duration of this action due to the uncertainty around the specifics whether any or all of these three preceding factors may have contributed to their reduced bycatch over the past two decades.

The overall relationships between changes in fishing effort, the distribution of fishing effort and protected species, implementation of bycatch reduction measures, and the observed bycatch, have yet to be well described. While it may not be possible to specifically attribute individual bycatch reductions measures with some species (e.g., the influence of pingers on reducing turtle bycatch, or the linkage between whale bycatch and the implementation of time/area closures to protect sea turtles), it may be reasonable to consider that the bycatch reduction measures in summation, working together in concert with the dynamics of fishing effort distribution and reduced total fishing effort, are combining to produce the low levels of bycatch observed (and estimated in total) in the current DGN fishery.

Given the information above, we conclude that looking to the bycatch estimates from the modern DGN fishery for each species from Carretta (2022) and looking at the observed rate of giant manta rays over this period provides the most reasonable and conservative estimates of what could be expected to occur in terms of bycatch for each species in the DGN fishery in any year given the current state of the fishery. Within any given year, it is likely that variable oceanographic or environmental conditions, as well other factors, impact the distribution of animals and fishermen and the relative likelihood of interactions by these species that may be affected by the DGN fishery. In essence, it is possible that the expectations for total bycatch that may occur over the next five years could largely occur during one year, as demonstrated by the estimated 2.05 entanglements that occurred recently for humpback whales in 2021 out of the estimated 2.7 humpback whale entanglements that occurred from 2017-2021. As a result, we assume that the bycatch that may occur in any one year for any species will not exceed the total bycatch for any species that we anticipate may occur over the next five years. Using this approach, we expect the following levels of bycatch could occur for each species in any given year, and across the last five years of the DGN fishery, under the proposed action (Table 11). The bycatch levels presented here offer an “up to” scenario from the annual bycatch perspective as a worst case scenario acknowledging that the probability and variability of rare events is difficult to predict in any one year.

Table 11. Expectations for maximum annual bycatch and total estimated bycatch over the last five years of the DGN fishery.

ESA-listed species	**Total bycatch**
Fin whale	**Up to 1**
Humpback whale	**Up to 1**
Sperm whale	**Up to 2**
Leatherback turtle	**Up to 2**
Loggerhead turtle	**Up to 3**
Olive ridley turtle	**Up to 1**
Green turtle	**Up to 1**

Giant manta ray	**Up to 1**

2.5.1.3 Review of Net Entanglements

In previous biological opinions and ongoing evaluation of incoming information relative to takes of ESA-listed species in the DGN fishery, we have acknowledged the risk for interactions between ESA-listed species and DGN gear that may go unobserved due to partial observer coverage of the fishery (i.e., observer coverage has typically been around 20% of fishing effort - so 80% of interactions would be expected to go unobserved). We have also acknowledged reports of incidents where large species such as whales may encounter DGN gear and break free or “blow through” the nets before being detected (NMFS 2013; NMFS 2017a). This consideration has been highlighted by two opportunistic reports of humpback whale entanglements (1 in both 2015 and 2016) that included documentation of gear that may have been associated with a large mesh drift gillnet, primarily based on assessment that includes the apparent mesh size and material type of the netting involved, along with other considerations (NMFS 2017a). While the specific type and origin of the gear could not be confirmed in these recent cases, we ultimately conclude that the documentation of a limited number of entanglements with gear that may have originated from the DGN fishery is consistent with our current understanding and interpretation of data and bycatch estimates in the DGN fishery (NMFS 2017a). In other words, sightings of whales entangled in gillnet outside of the observer reports is not inconsistent with our expectation that some DGN fishery takes go unobserved by federal fishery observers.

However, these recent entanglement reports that may have been associated with a large mesh drift gillnet have underscored an ongoing challenge related to identification of the origins of entangling gear. Unlike many entanglements that are associated with line and buoys that may have markings or tags that can be more easily identified, opportunistic reports of entanglements that involve net gear usually do not involve the kinds of features that can be readily identified. Instead, we must rely upon descriptions of the netting itself, especially mesh size and the netting material. Even then, interpretation remains difficult considering the number of gillnet fisheries that operate across the entire range of species that include multiple nations along the North American coast (NFMS 2017).

In order to improve the collection and interpretation of incoming reports in the future, and as part of the effects analysis of this biological opinion, we completed a comprehensive review of all entanglement reports received from 2013-2022 that involve some indication of netting as part of the entangling gear as part of ongoing forensic review of West Coast entanglements. This review could also identify any additional entanglements that may have involved gear associated with a large mesh drift gillnet. These records include opportunistic entanglement reports received by the NMFS Marine Mammal Stranding Program, along with fishery observer reports and Marine Mammal Assessment Program fisherman self-reports. Given the changes in gillnet fisheries that occurred along California during the 1990s (both DGN and state-managed gillnets) and the low quality of information and documentation associated with historical entanglement reports, we did not think review of older entanglement reports (dating back to 1982) would be any more informative.

In total, we reviewed 38 records of whale entanglements associated with netting. These records included:

- 18 gray whales - including one record from DGN fisheries observers
- 18 humpback whales – including two records from DGN fishery observers
- two unknown whales - zero from DGN fishery observers

During the review, we identified 20 records where at least some assessment of the mesh size could be made from the documentation or descriptions provided. These records included the three instances where DGN gear is known to be involved based on the observer report, as well as the two entanglements that were already flagged as potentially having involved gear associated with large mesh drift gillnets. Of the remaining 15 (of 20) records where some assessment of mesh size is possible, none of those assessments point to mesh sizes that would clearly be consistent with a large mesh drift gillnet (DGN fishery gear must be ≥ 14 inches). Where known (measured), estimated, or otherwise described, mesh sizes of netting involved in entanglement reports has been consistent with gillnets used in coastal set net fisheries all along the North American coast (6-8 inches is typical). There was one humpback whale entanglement report in 2021 that included a general description of "large mesh size" involving "twine" netting, although the available documentation does not permit and accurate assessment of the size of the netting of the mesh size. For the 18 other records where mesh size could not be assessed, several of the entanglements were described as involving monofilament. Considering that the majority of entanglement reports that involved netting did not include any information on mesh size, this is an area of documentation that needs to be emphasized as a priority for collecting during future entanglement reports that involve netting.

During the review, we identified 27 records where at least some assessment of the type of material could be made from the documentation or descriptions provided. These records included the three instances where DGN gear is known to be involved based on the observer report, as well as the two entanglements that were already flagged as maybe having involved gear associated with large mesh drift gillnets. Of the remaining 22 (of 27) records, only three of the assessments point to materials (twine) that that could be consistent with a large mesh drift gillnet (DGN fishery gear typically consists of multi-filament twine, and not monofilament). There was one gray whale entanglement with a twine gillnet reported in Southern California in 2017, but the mesh size was much smaller than DGN gear (6 inches measured). There was one humpback whale entanglement with twine gillnet reported in Southern California in 2018, but the mesh size from the gear recovered from the whale (reported dead) was described as "less than 12 inches stretched" which is smaller than DGN gear. In 2021, there was a humpback whale reported entangled in Southern California in what can be described as netting made of twine, although review of the available documentation does not permit an accurate assessment of the size of the netting. There were three entanglements reported from waters of the Salish Sea in the Pacific Northwest (1 gray whale and 2 humpback whale) that are involved with tribal gillnetting for salmon that are known to use monofilament or multifilament nets that typically are not constructed of the same type of twine that is used in large mesh drift gillnets. Although the mesh sizes weren't well described with those entanglement reports, salmon gillnet mesh sizes in the

Pacific Northwest are much smaller than gear used in the DGN fishery - ranging from 4.5-6 inches, which is relatively easy to distinguish from large mesh drift gillnet gear with good documentation. For the 11 records where gillnet material could not be assessed, none of them had any description of mesh size beyond the 2021 humpback whale entanglement described as "large". Similar to mesh size, obtaining better descriptions and documentation regarding the materials involved in entanglement reports that involve nets needs to be emphasized as a future priority.

During the review, we also looked at any information describing other aspects of the gear involved in the entanglement, including mesh color, presence/descriptions of floats, and associated lead or float lines. For example, the presence of small floats along with the netting would suggest gear associated with bottom set gillnets (or possibly smaller mesh drift gillnets from the Pacific Northwest). Some description of floats was associated with 19 of the 38 net entanglement reports reviewed. In nine of those cases, the netting was associated with small mesh (6-12 inches). However, in some of these instances it was not clear if the description of the float was directly related to the netting or possibly related to an attached buoy. Categorization of the other gear characteristics does not appear to be very informative at this time based on the sparseness of the information, although accumulation of these data over time, and improved descriptions of gear components in future entanglement reports, is likely to enhance our ability to assess and identify the gear origin of entanglements in the future.

In total, the result of the review did not identify any whale entanglement reports associated with netting from 2013-2022 that can be positively attributed to large mesh drift gillnets that had not already been identified before this comprehensive evaluation was completed. However, our review did highlight one report in 2021 that does share some characteristics with large mesh drift gillnets, and we cannot rule that possibility out of every one of these entanglement reports. We acknowledge there are numerous entanglements involving netting we reviewed where little to no information was available regarding mesh sizes and/or materials to evaluate relative to possible consistency with large mesh drift gillnet gear. Additional details from some of these entanglements may also point to gear origins other than large mesh drift gillnet, but are not definitive. However, all of the available information leads us to conclude that we likely have not overlooked information from historical entanglement reports that would indicate that large mesh drift gillnet gear, or the DGN fishery specifically, has been a common source of net entanglements that have been reported to NMFS over the last two decades. Instead, the available information leads us to conclude that most of these reported net entanglements have been associated with other fisheries such as smaller mesh gillnets commonly fished in coastal areas throughout the range of these species. Even if some small number of historical record of net entanglements were associated with large mesh drift gillnet gear or the DGN fishery in the past, this is consistent with the assumption that small numbers of interactions with the DGN fishery are expected to go unobserved/unreported. In addition to providing important consideration regarding the effects analysis in this biological opinion, the results of the review will be useful to help guide the future documentation, reporting and assessment entanglements associated with nets.

We also reviewed the NMFS Sea Turtle Stranding database to look for cases of sea turtle strandings that may have been associated with net entanglements. Since 2000, only one sea turtle stranding has been reported that involved any type or indications of involvement with netting. In this case, a green sea turtle was found dead in south San Diego Bay entangled in netting that was likely associated with a piece of derelict or discarded set gillnet webbing. As a result, we did not find any indication that the sea turtle stranding record would provide any additional information about unobserved or unreported sea turtle entanglements with DGN gear.

2.5.1.4 Review of Net Damage Observer Data

As mentioned above, we have previously acknowledged that there is a possibility that large animals such as whales could encounter and break free or "blow through" DGN gear, such that fishery observers onboard vessels could be completely unaware that an interaction has occurred other than the existence of damaged or missing pieces of net. As part of the reinitiation of the ESA consultation on the DGN fishery, data on the occurrence of net damage was requested from the NMFS observer program in order to examine the relative rates/occurrence of damaged nets as recorded by fishery observers. We acknowledge that incidents of net damage are not necessarily reflective of blow through events with large animals, and may have other causes or sources. The data that was provided indicated that net damage is something that is documented very rarely by observers. During almost 3,300 observed sets that occurred from 2001-2021, net damage was recorded only nine times, and including four since 2005 (one in 2013, 2017, and two in 2021). Historically, collection of this data has not been emphasized as a priority by the observer program compared to other sampling and data recording duties that observers are responsible for during fishing operations. It is likely that more damage to nets has occurred than what is reported, although more typical net damage may not be on a scale that is immediately obvious or at a level deemed worthy of note by observers. It is also likely that net damage is a rare event, and that incidents of damage that may be associated with bycatch that breaks free or blows through nets are also extremely rare. In the future, protocols for collection of net damage data could be reviewed and revised by the observer program in the context of other priority data collection needs.

2.5.1.5 Response

Marine Mammals

The probability that a marine mammal will initially survive an entanglement in fishing gear depends largely on the species and age or size of marine mammal involved. For instance, larger animals such as fin whales, humpback whales and sperm whales may encounter and even become entangled in gillnet gear, but often survive the initial contact with the gear by breaking some meshes and "punching" a hole through the gillnet webbing to continue swimming Fishermen have reported that large whales (e.g., blue and fin whales) break through drift gillnets without entangling, and that very little damage is done to the net (NMFS 2013). There are many variables to consider when evaluating how large whales are likely to respond when entangled. Marine mammals that become entangled and are either released by fishermen or release themselves, and may swim away with a portion of gillnet attached to their bodies. Observer records indicate that for large whales, there are generally three areas on their body where entanglement in a net occurs: 1) the gape of the mouth, 2) around the flippers, and 3) around the tail stock (although this area is often difficult to view, as most *balaenopterids* do not fluke frequently). Documented cases have indicated that entangled marine mammals may travel for extended periods of time and over long distances before either freeing themselves of gear, being disentangled by stranding network personnel, or dying as a direct result of the entanglement (Angliss and DeMaster 1998).

In most cases, it is unknown whether an entanglement results in an injury that is serious enough or debilitating enough to eventually lead to death.[23] If the debris fragments are heavy, the animal could become exhausted trying to repeatedly reach the surface to breathe and might eventually drown. Less heavy fragments may also lead the animal to exhaustion (not as quickly as expected with heavier gear), depletion of energy stores, and starvation due to the increased drag (Wallace 1985). Younger animals are particularly at risk if the entangling gear is tightly wrapped, for as they continue to grow, the gear will likely become more constricting. This is of particular concern as the majority of large cetaceans that become entangled in all types of fishing gear are juveniles (Angliss and DeMaster 1998). Data from the NMFS WCR Stranding Database do not provide conclusive information on the size or age of most whales that have been reported entangled, although reports of juvenile whale entanglements are certainly part of that record. NMFS assumes most marine mammals that die as a result of entanglement in drift gillnets have succumbed to drowning. With a typical soak time of 12-14 hours, the animal is unable to survive without oxygen, especially if it is entangled at the beginning of the set, or deep in the net. Marine mammals may also be affected in other sublethal ways as a result of being captured in a drift gillnet. If an animal's appendage is caught in the mesh, the debris can debilitate the animal, especially if it is constricting, causes lacerations, or impairs swimming or feeding ability (Scordino 1985), which may make the animal more susceptible to disease or predation (Angliss and DeMaster 1998). The lacerations themselves may become a source of infection. A sustained stress response, such as repeated or prolonged entanglement in gear or having gear left on the animal, may make marine mammals less able to heal and to fight infection or disease (Angliss and DeMaster 1998).

Sea Turtles

Potential impacts from the DGN on sea turtles will generally be related to injury or mortality, although any entanglement, whether or not it causes an injury or mortality, may also impact sea turtles. Injury of turtles entangled in a drift gillnet may result in mortality post-release due to impairment from debilitating effects of forced submergence, and/or wounds suffered as a result of net entanglement. Sea turtles are prone to entanglement as a result of their body configuration and behavior (Balazs 1985). Records of stranded or entangled sea turtles reveal that fishing gear can wrap around the neck, flipper, or body of a sea turtle and severely restrict swimming or

[23] The current criteria and process used for assessing the severity of injury to marine mammals come were published by NMFS in 2012 (77 FR 3233). In 2023, NMFS announced revisions to the process and intentions to revise the criteria using updated data that is currently being analyzed (88 FR 7957).

feeding. In drift gillnets, turtles are most likely to get entangled in the relatively large nylon mesh of the net, and may typically be released with little or no gear remaining on them. Large turtles such as leatherbacks do present a greater challenge to fishermen for safe release.

Once entangled, factors such as size, activity, water temperature, and biological and behavioral differences between species bear directly on metabolic rates and aerobic dive limits and will therefore also influence survivability in a gillnet. For example, larger sea turtles are capable of longer voluntary dives than smaller turtles, so juveniles may be more vulnerable to the stress of forced submergence than adults. During the warmer months, routine metabolic rates are higher, so the impacts of the stress due to entanglement may be magnified. In addition, disease factors and hormonal status may also play a role in anoxic survival during forced submergence. Any disease that causes a reduction in the blood oxygen transport capacity could severely reduce a sea turtle's endurance in a net, and since thyroid hormones appear to have a role in setting metabolic rate, they may also play a role in increasing or reducing the survival rate of an entangled sea turtle (Lutcavage and Lutz 1997). Sea turtles forcibly submerged for extended periods of time show marked, even severe metabolic acidosis as a result of high blood lactate levels. With such increased lactate levels, lactate recovery times are long (even as much as 20 hours; Lutcavage and Lutz 1997). Therefore, sea turtles need to have an adequate rest interval at the surface in order to successfully recover from forcible submergence (Stabenau and Vietti 2003).

We also consider whether sea turtles entangled and released from DGN gear could be at increased risk to entanglement again or perhaps exposed to an additional threat such as a ship strike or predation by sharks/killer whales. Presumably, a sea turtle recovering from a forced submergence would most likely remain resting on the surface, which would reduce the likelihood of being recaptured in a drift gillnet submerged at least 36 feet from the surface. Recapture would also depend on the condition of the turtle and the fishing pressure in the area, which is likely to be somewhat reduced from historical levels in the current DGN fishery. But the additional surface time could lead to an increase risk of a ship strike or predation. Although there may be some apparent risk that we can identify in general, we don't have enough information to further quantify or determine if the recapture of sea turtles by the DGN fishery or increased risk from any other activity has been realized.

Marine Fish

Effects from the DGN fishery on ESA-listed marine fish including the giant manta ray would be from entanglement in nets. Observed bycatch of giant manta ray from this fishery reported 14 observed caught from 1990-2006 with 36% released alive (Miller and Klimovich 2017), although the one giant manta ray caught since 2001 was reported dead. This species' physiology both increases their chance of entanglement, along with an increased chance of being released alive post an entanglement incident (up to ten hours for this gear type) when compared to species that require oxygen and surfacing for their survival. Their diamond shaped bodies, large wingspan (up to 6.8 m), and their long cephalic fins that help catch their prey make them more likely to be entangled if they encounter this gear. Historically, these species do not comprise a large component of bycatch from this fishery (Miller and Klimovich 2017). However, given that the majority of giant manta rays that have been observed captured in this fishery have been killed, including the most recent incident in 2005, we will conservatively assume a worst-case scenario, that any bycatch of giant manta rays would result in a mortality.

2.5.1.5.1 Mortality and Serious Injury Rates

In the DGN fishery, NMFS fishery observers record detailed information marine mammals, sea turtles, and other species that are entangled in the net. Animals that are released alive from the net, with netting attached, are classified as "injured." Animals that release themselves or are released from the net by fishermen and can swim normally, are recorded as "alive." Marine mammals and sea turtles that have been entangled in DGN gear and are released alive usually only have minor abrasions as a result of interaction with the net, but there have been no long-term studies to monitor the post-interaction effects of DGN bycatch on marine mammals and sea turtles. As part of the DGN bycatch estimates produced by Carretta (2022), the expectations for associated mortality and serious injury associated with these bycatch events for each species are also described in terms of the probability that bycatch events result in mortality or serious injury. Those expectations are based on the current criteria for such determinations for marine mammals in the SARS (NOAA 2012), and the known outcomes and information provided by fisheries observers during observed bycatch events for each species. In light of any further specific information regarding survival and mortality of ESA-listed species resulting from DGN bycatch being available, we rely upon this information to inform our anticipated response of ESA-listed species to DGN bycatch in this biological opinion (Table 12). We note that the variance in estimates of the probability that bycatch events result in mortality or serious injury may reflect the limited number of observed bycatch events upon which the estimates are made more than underlying differences between the species in terms of responses and outcomes of bycatch in the DGN fishery.

Table 12. The probability that a bycatch event results in mortality or serious injury, by species.

ESA-listed species	Probability of mortality or serious injury (Carretta 2022)
Fin whale	1.00
Humpback whale	0.25
Sperm whale	0.70
Leatherback turtle	0.68
Loggerhead turtle	0.25
Olive ridley turtle	0.00*
Green turtle	1.00
Giant manta ray	1.00**

* Although this value is cited in Carretta (2022), we assume that mortality could occur for this species.
** Not estimated by Carretta (2022)

As mentioned above, anecdotal evidence indicates that large whales will encounter DGN gear, but may avoid entanglement or may punch through the gear and thus not be observed by a

NMFS observer or reported by the fishermen. We acknowledge that these events are not reflected by the observer record of entanglement, but may be reflected in part by other reports of entangled whales received by the WCR Stranding Network, if any gear remains upon the whale after an encounter with gillnet gear. Not including reports from NMFS fisheries observers, there have not been any reports of fin and sperm whales found stranded that were entangled with some type of gillnet gear dating back to 2013, although large mesh drift gillnet gear is suspected to have been involved in at least two and maybe three cases of humpback whale entanglements since 2013 (see 2.5.1.3 Review of Net Entanglements; Saez et al. 2021). Historically, gillnet fisheries other than the DGN, including domestic and foreign set gillnet fisheries, are more commonly or likely to be associated with reported strandings of entangled whales (Saez et al. 2021). Without evidence from the stranding record that entanglements or events where large whales punch through gillnet gear are being significantly underestimated by the observer record, we will continue to rely upon the observer record as the primary information available to support analysis of the expected response and outcomes of DGN bycatch events.

2.5.1.6 Data Collection by Observers

As part of the data collection process as prescribed by the observer program and the *Terms and Conditions* (Section 2.9.4) of this biological opinion, fisheries observers will collect information and relevant biological samples from ESA-listed species that are caught in DGN gear. While observers carry long handled biopsy poles that may be able to take a biopsy of any live marine mammal if conditions are reasonably safe, it is unlikely that any ESA-listed whales will be sampled by observers as they are unlikely to be brought aboard the fishing vessel due to their large size and the general difficulty and safety concern of trying to handle large animals, especially if they are still alive. NMFS expects that fishing vessels will take appropriate measures to handle and release these individuals while minimizing injury to the animal and damage to their gear, per the regulations in 50 CFR § 223.206(d)(1) and the *Reasonable and Prudent Measures* (section 2.9.3) and *Terms and Conditions* (section 2.9.4) of this and prior biological opinions. Smaller species, such as hard-shelled sea turtles, are likely to be brought aboard the boat and should be available for sampling. Observers will, if practicable, measure, photograph, and apply flipper and passive integrated transponder (PIT) tags to any live sea turtle, and salvage any carcass or parts or collect any other scientifically relevant data from dead sea turtles, per authorization in 50 CFR § 222.310 and § 223.206 regarding the handling of endangered and threatened sea turtles by designated NMFS agents. In addition, observers will also collect skin tissue samples for genetic studies. Tissue biopsies would be taken using the antiseptic protocol described by Dutton and Balazs (1995). The biopsy site would be scrubbed with an isopropyl alcohol swab before and after sampling. The tissue biopsy would be obtained using a 4-mm sterile biopsy punch from the trailing edge of a rear flipper when possible, with the resulting plug less than the diameter of the punch. Following the biopsy, an additional antiseptic wipe would be used with modest pressure to stop any bleeding. A new sterile biopsy punch would be used on each animal. (This technique would only apply to live animals, primarily turtles. For dead mammals, observers are taking a sliver of tissue with their knife from the site of the blubber sample).

2.5.2 Risk for ESA-listed species Affected

In this section, we relate the anticipated exposure and response of individuals to the proposed action with the expected impacts to the population and/or species. In evaluating the risk to ESA-listed species that is associated with anticipated bycatch resulting from the maximum fishing effort associated with the proposed action, NMFS considers the possibility that up to 573 sets could occur in any one season, and that on average 305 sets will occur in the DGN fishery over the next five years.

2.5.2.1 Fin Whale

Under the proposed action, we expect that up to one fin whale could be caught in the DGN fishery in any given year; although no more than one individual would be expected to be caught over the last five years of the DGN fishery (Table 11). Because there has only been one observed entanglement of a fin whale since 1990, it is difficult to determine what age-class or sex of fin whales may be most vulnerable to entanglement in the DGN fishery based on observer records. For the purposes of this biological opinion, we will assume all individuals in the population are vulnerable to entanglement and of equal significance; that is, an entangled fin whale could be male or female of any age class. Based on the expected probability of mortality or serious injury in Table 12, we assume that this one fin whale that could be taken at any time over the next five years could be seriously injured or killed by the DGN fishery.

The best estimate of the CA/OR/WA stock of fin whales is about 11,065 whales, and the most recent PBR level for this stock is 80 animals (Carretta et al. 2022a). Consistent with the approaches typically used in the SAR to compare known mortalities and serious injuries to PBR and impacts that occur over a broader period of time to gauge effects, the loss of one animal over the next five years (0.2 per year) represents about 0.3% percent of PBR (80) for this stock on an annual basis. In any given year (over the next five years), the loss of one fin whale represents about 0.01% of the total estimated population of the stock that occurs within the action area. There is no estimate of the total population of ESA-listed fin whales globally, although that number is likely much higher as the population ranges widely throughout the world.

2.5.2.2 Humpback Whale

Under the proposed action, we expect that up to one humpback whale could be entangled in the DGN fishery in any given year; although no more than one individual would be entangled over the last five years of the DGN fishery (Table 11). For the purposes of this biological opinion, we assume that all age-classes are vulnerable to entanglement and of equal significance and males are as vulnerable as females. Based on the expected probability of mortality or serious injury in Table 12, we assume that this one humpback whale that could be taken at any time over the next five years would be seriously injured or killed by the DGN fishery. Below we evaluate the risk of humpback whale bycatch that is anticipated in the DGN fishery that is associated with each ESA-listed humpback whale DPS that may be affected.

2.5.2.2.1 Mexico DPS Humpback Whale

Given that the DGN fishery almost exclusively operates off the southern and central coast of California, we assume that the most likely DPS origin of any humpback whale that may interact with the DGN fishery will be the Mexico DPS. In fact, given that an estimated 58% of the humpback whales that occur off California may be associated with the Mexico DPS, it is likely that any humpback whale that may be entangled during a given year, and the one humpback whale that is expected to be entangled during the last five years of the DGN fishery (Table 11), would be associated with this DPS. As a result, we conclude it is reasonable to expect that the one mortality or serious injury that may be associated with DGN bycatch during any year, and over the next five years, could be associated with the Mexico DPS of humpback whales.

As described in *Status of the Species* (section 2.2.2.1), the most recent estimate of the abundance of Mainland Mexico-CA/OR/WA stock is 3,477 (minimum population size of 3,185). Consistent with the approaches typically used in the SAR to compare known mortalities and serious injuries to PBR and impacts that occur over a broader period of time to gauge effects, the loss of one animals over the next five years (0.2 per year) represents about 0.5% percent of PBR (43) for this stock on an annual basis. While the current total abundance for the entire Mexico DPS is uncertain, the population size is highly likely to be at least 6,000-7,000 or higher based on a range of different historical (Calambokidis et al. 2008; Barlow et al. 2011; Bettridge et al. 2015) and more current estimates (NMFS 2021a). These estimates are conservative given the increasing abundance of humpback whales along the U.S. West Coast since that time. Considering the prospect of losing one individual from the population of at least 6,000 in any given year (over the next five years), this represents less than 0.1 percent (0.017) of the total Mexico DPS population, at most. This is a very small proportion of the total population.

2.5.2.2.2 Central America DPS Humpback Whale

Given that the DGN fishery almost exclusively operates off the southern and central coast of California, and the fact that up to 42% of humpback whales in this area may belong to the Central America DPS, we must assume that it is possible that any given humpback whale that may interact with the DGN fishery during any year, and the one humpback whale expected to be entangled over the last five years of the DGN fishery (Table 11), could be associated with this DPS (we will assume it could occur). As a result, we conclude it is reasonable to expect that the one mortality or serious injury that that may be associated with DGN bycatch during any year, and over the next five years, could be associated with the Central America DPS of humpback whales.

As described in the *Status of the Species* (section 2.2.2.1), the Central America/Southern Mexico-CA/OR/WA stock of humpback whales is estimated to be approximately 1,496 individuals (1,284 minimum population estimate). This is a small proportion of the total population. Consistent with the approaches typically used in the SAR to compare known mortalities and serious injuries to PBR and impacts that occur over a broader period of time to gauge effects, the loss of one animal over the next five years (0.2 per year) represents about six % percent of PBR (3.5) for this stock on an annual basis. While NMFS will continue to evaluate the relationship between the humpback whale DPSs and recognized DIPs moving forward, at this time we consider the inclusion of southern Mexico humpbacks and the abundance and trend estimates recently published by Curtis et al. (2022) as being reflective of the current status of the Central America DPS. Considering the prospect of potentially losing one individual from the population in any given year (over the next five years), this represents less than one percent (0.07) percent of the total Central America DPS population. This is a small proportion of the total population. Consistent with the approaches typically used in the SAR to compare known mortalities and serious injuries to PBR and impacts that occur over a broader period of time to gauge effects, the loss of one animal over the next five years (0.2 per year) represents about 6% percent of PBR (3.5) for this stock on an annual basis.

2.5.2.3 Sperm Whale

Under the proposed action, we expect that up to two sperm whales could be entangled in the DGN fishery in any given year; although no more than two individuals would be entangled over the last five years of the DGN fishery (Table 11). For the purposes of this biological opinion, we assume that all age-classes are vulnerable to entanglement and of equal significance and males are as vulnerable as females. Based on the expected probability of mortality or serious injury in Table 12, we expect that in any given year, and over the next five years, up to two sperm whales could be entangled in a manner which could be a lethal (up to 2 individuals x 0.70 mortality or serious injury rate = ~ 2 [1.4 rounded up] mortalities or serious injuries).

The best estimate of the CA/OR/WA stock of sperm whales is about 2,000 whales, and the most recent PBR level for this stock is 2.5 animals (Carretta et al. 2022a). In any given year (over the next five years), the removal of two sperm whales from the population equates to an average of 0.1 % of the population being removed from the stock that occurs within the action area. Consistent with the approaches typically used in the SAR to compare known mortalities and serious injuries to PBR and impacts that occur over a broader period of time to gauge effects, the loss of two animals over the next five years (0.4 per year) represents about 16% percent of PBR (2.5) for this stock on an annual basis. The best available information suggests there are at least several hundred thousand sperm whales globally. A loss of two from the global population represents far less than one percent (~.001 percent).

2.5.2.4 North Pacific DPS Loggerhead Sea Turtles

Under the proposed action, we expect that up to three North Pacific DPS loggerhead sea turtles could be entangled in the DGN fishery in any given year; although no more than three individuals would be entangled over the last five years of the DGN fishery (Table 11). At this time, there is no specific available information about the sex ratio of loggerheads foraging in the Pacific that may be found off the U.S west coast, so we rely upon information used by Martin et al. (2020a) about sex ratios for loggerheads that may be vulnerable to Hawai'i longline fisheries that assume females constitute 65% of the juvenile population. Based on data collected from loggerheads observed caught in the DGN fishery and other known information, we assume that loggerheads that are likely to be present off the U.S. west coast consist mainly of juveniles (NMFS 2013). Based on information in the previous biological opinion, we also assume an expected survival rate of juvenile loggerheads to adulthood of 0.80 (Snover 2008; Conant et al. 2009; Martin et al. 2020a).

Based on the expected probability of mortality or serious injury in Table 12, the anticipated survival rate for juveniles to adulthood, and the anticipated sex ratio for juvenile loggerhead sea turtles in the action area, we expect that in any given year, and over the next five years, up to one mortality or serious injury could occur to a juvenile female loggerhead sea turtle that could have survived to adulthood (up to 3 individuals x 0.25 mortality or serious injury rate x 0.59 survival rate for juveniles to adulthood x 0.65 sex ratio = ~ 1 [0.39 rounded up]).

The removal of one adult female (equivalent) in a year, and over the next five years, constitutes less than 0.1 percent (0.011) of the estimated adult female population (8,733). The removal of up to three juvenile turtles in a year or over the next five years within a total estimated population of loggerheads foraging off Southern California when conditions are optimal for them given anomalously warm sea surface temperatures (70,000,) is less than 0.01 percent (0.004) of the local population.

2.5.2.5 Leatherback Sea Turtles

Under the proposed action, we expect that up to two leatherback sea turtles could be entangled in the DGN fishery in any given year; although no more than two individuals would be entangled over the last five years of the DGN fishery (Table 11). The leatherbacks that are typically foraging in the waters off coastal California and likely to interact with the DGN fishery are expected to be adult or sub-adult, based upon curved carapace lengths from leatherbacks observed caught in the DGN fishery and tagging studies being conducted in Central California by the SWFSC (Benson et al. 2007b; Benson et al. 2011). Data from these studies have suggested that females constitute about 73 percent of the individuals found in the action area (Benson et al. 2011, applied in Martin et al. 2020a). As noted above in the status section, the leatherbacks exposed to the proposed action are most likely to originate from beaches in the Western Pacific, with the majority comprised of summer nesters from Papua, Indonesia.

Based on the expected probability of mortality or serious injury in Table 12, and the anticipated sex ratio for leatherback turtles in the action area, we expect that in any given year, and the next five years, up to one adult female leatherback sea turtle could be entangled in manner which could be a lethal (up to 2 individuals x 0.68 mortality or serious injury rate x 0.73 sex ratio = ~ 1 [0.99 rounded up] mortality or serious injury).

The prospect of removing up to one adult female (equivalent) in a year, and over the next five years, represents less than 0.1 percent (0.095) of the total Western Pacific adult female population (1,443). Given an estimated 100,000 adults and juveniles in the Western Pacific subpopulation, the loss of two individuals in a year or over five years represents 0.002 percent of the population.

2.5.2.6 Olive Ridley and Green Sea Turtles

Under the proposed action, NMFS expects that up to one olive ridley sea turtle from the endangered Mexican nesting beach population and one East Pacific DPS green sea turtle could be entangled in the DGN fishery in any year; although no more than 1 individual of either species would be expected to be caught over the last five years of the DGN fishery (Table 11). The lack of observed bycatch history of these two species makes it difficult to distinguish the age-class, size, or sex of individuals that are likely to interact with the DGN fishery. As a result, we will conservatively assume that adult females from each species could be caught. In any given year and over the next five years, we assume this could result in mortality for that individual even though Carretta (2022) currently estimates the probability of an olive ridley mortality or serious injury is zero (Table 12). As a result, we assume that at most one adult female from each species could be killed or seriously injured over the next five years as a result of bycatch in the DGN fishery.

In the olive ridley Mexican nesting beach population that numbers over one million at a minimum, the loss of one individual, including an adult female, in any given year would have a very small impact on that population (loss of < 0.0001%).

Given the abundance of nesting females in Mexico, Ecuador, and Costa Rica, Seminoff et al. (2015) estimated the adult female population of East Pacific DPS green turtles to be 20,062 females. The loss of one adult female from the population would represent a loss of about 0.005 percent of the total number of adult females from the population, which would be a very small impact on that population.

2.5.2.7 Giant Manta Ray

Under the proposed action, NMFS expects that up to one giant manta ray could be entangled in the DGN fishery in any year, although no more than one individual would be expected to be caught over the last five years of the DGN fishery (Table 11). Given what we know about the historical documentation, we assume this one individual will be killed or seriously injured. It is unclear what life stage of giant manta ray may be most vulnerable, although we assume all life stages and sexes are of equal value.

The abundance of the global population of giant mays, or the regional population that may be exposed and vulnerable to bycatch in the DGN fishery is unknown. Manta rays are listed as threatened under the ESA with limited known global distribution and population with a regional population size between 500 and 1,500 individuals. However, ongoing research including mark-recapture analyses suggests that typical subpopulation abundances are more likely in the low thousands (e.g., Beale et al. 2019) and in rare cases may exceed 22,000 in areas with extremely high productivity, such as in coastal Ecuador (Harty et al. 2022). Assuming the population that may be impacted by the DGN over the next five years may have around 1,000 individuals at least, this results in the potential removal of up to 0.1% of the total population, which is a very small impact on the population.

2.5.3 Relationship between Observer Coverage and the Effects Analysis

The observer program is an integral part of this proposed action, as the information that is necessary for monitoring current and estimating future impacts of the DGN fishery on ESA-listed species comes almost exclusively from observer records. The DGN fishery has generally maintained an observer coverage goal of 20 percent since the implementation of the DGN observer program in 1990, although there have been instances more recently where coverage goals were temporarily higher. In the last biological opinion, questions and concerns were raised about several aspects related to observer coverage and potential biases in data (NMFS 2013). As described above in the *Proposed Federal Action* (Section 1.3) NMFS SFD has either implemented or is in the process of currently evaluating alternatives and assessing the feasibility of implementing measures to address these concerns. Where appropriate and possible, we provide updated consideration of how observer coverage is related to the effects analysis.

The proposed action intends to maintain the coverage goal in the DGN of targeting 20-30 percent given the primary challenge of funding limitations in the current observer program. There are also challenges in how observer coverage could be increased given that a relatively large portion of the total fishing effort takes place on vessels that are not observable for the reasons discussed in NMFS (2013). The percentage of the DGN fishery (both in terms of boats and total fishing effort) that is unobservable appeared to have been increasing over time, until about 2010, but has recently leveled off at approximately one-third of the annual fishing effort in the DGN fishery (Figure 11).

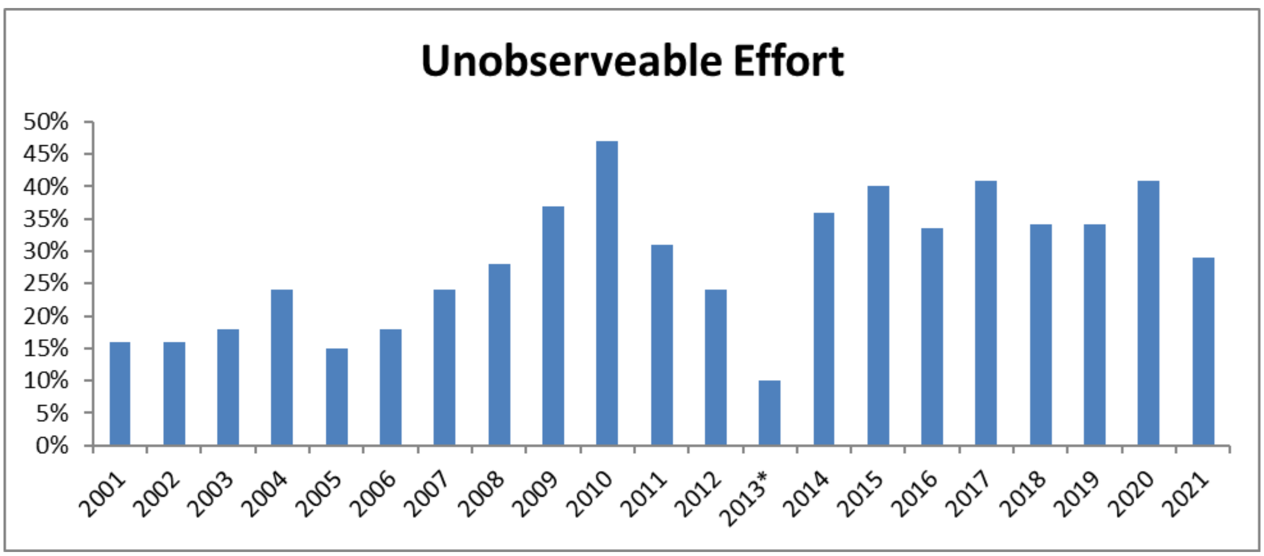

Figure 11. Proportion of DGN fishing effort conducted by unobservable vessels. *notes that data through 2012 are aggregated by fishing season, and starting in 2013 by calendar year. Also, note that 2013 was the year that observer coverage was mandatory in portions of the fishery as part of an emergency rule in response to concern over sperm whale bycatch.

The concept of using data gathered from observing ~20 percent of fishing effort is based on the concept of sampling and the fundamental assumption that the ~20 percent of the effort that is documented is proportionally representative of the other ~80 percent of the effort that is not, in terms of catch or encounter rates for target and non-target species. When certain portions of the fishery are never "sampled," in this case boats that are unobservable, it raises questions about whether the fishing effort of the unobservable vessels is represented by the observer data gathered from the rest of the fleet and the reliability or accuracy of bycatch estimates produced from data that may not represent the whole fleet. Factors such as any difference in compliance with bycatch mitigation measures and general fishing behaviors that could lead to increased encounters with protected species onboard DGN vessels without the presence of fisheries observers could bias the observed record of protected species bycatch rates compared to the bycatch rate of the entire fleet.

As part of the previous biological opinion, some analysis of fishing effort data was conducted to assess if any potential bias between observed and unobserved fishing effort was evident using logbook data. Results of that analysis did not detect any obvious bias (NMFS 2013), although the opinion continued to express that the uncertainty surrounding the potential bias of unobserved fishing effort was not resolved. Ultimately, several of the *Reasonable and Prudent Measures* and *Terms and Conditions* of the NMFS 2013 biological opinion were developed to help address this uncertainty in the future, including notably the requirement for implementation of a VMS program which was implemented in 2015.

As described above/in the consultation initiation request, the PSMFC published a report analyzing and comparing the characteristics of observed and unobserved DGN fishing trips (Suter et al. 2021). One purpose of this analysis was to assess the presence of an "observer effect or bias" where fishing behaviors and operations might be different when an observer is on board versus on unobserved trips. The report found "there were few statistically significant differences in fishing metrics between observed and unobserved trips on periodically observed vessels, or between unobservable and periodically observed vessels."

This analysis looked at key variables of fishing effort, including some identified in Carretta (2022) as being influential to the relative to the bycatch of ESA-listed marine mammals and sea turtles, including sea surface temperature (SST), latitude, longitude, and depth. One influential variable that was not examined explicitly by statistical analysis included day of the year (seasonality), although analysis of all fishing variables was differentiated between May 1 - Nov 15 (Season 1) and November 15 - January 31 (Season 2) to explore an element of environmental and regulatory seasonality.

Although there were few statistically significant differences in fishing metrics between observed and unobserved trips on periodically observed vessels, or between unobservable and periodically observed vessels, there were some interesting differences noted. Notably, the periodically observed vessels fished about 45% deeper and farther from shore than the unobservable vessels at the trip- and set-level during Season 1. However, there was no significant difference during Season 2. Likewise, the observed versus unobserved trips for the periodically observed vessels showed a similar pattern, where observed trips fished 23% deeper when an observer was onboard during Season 1. This pattern was less evident in the distance-from-shore metrics. The pattern reversed for both depth and distance from shore in Season 2 for the set-level metrics and for depth in the trip-level metrics.

Overall, across the entire DGN fishery, the average fishing behavior and operations of observed and unobserved effort appear to be similar, especially with respect to some key variables that influence the bycatch risk for ESA-listed species. As a result, we conclude there is no obvious or apparent discrepancy with unobserved effort that suggests bycatch estimates of ESA-listed in the DGN fishery are conclusively biased, and that the observer data is representative and the estimates from Carretta (2022) derived from it are likely reflective of the total fishery. In order to ensure this remains the case, it is imperative that observer coverage be deployed in a representative manner of all fishing effort. Qualitative review of the observer coverage data used for the statistical analysis indicate that observer coverage is not necessarily deployed in a uniform fashion across the entire season (NMFS unpublished observer data), as influenced by the availability of observer resources, the participation of unobservable vessels, and other factors. There have been certain times (months) and fishing areas where the observer coverage rates have been lower than the "20-30% target" that is generally used to describe coverage goals across the entire fishery. We encourage the Observer Program to maximize the uniformity of observer coverage in the DGN fishery in order to maintain the representative nature of the data collected from observed effort across the entire fishery.

NMFS views the ~20 percent observer coverage that has generally been achieved since 1990 as offering a representative sample of what has happened in the fishery over a long period of time that is more informative than focusing on the observer record in any single year, regardless of the specific observer coverage level achieved in any single year. NMFS concludes that the estimates of bycatch expected to occur in the DGN fishery produced from this long-term data set and anticipated in this biological pinion are reasonably certain.

In this biological opinion, the effects analysis has estimated the total number of ESA-listed individuals that will be affected by the DGN fishery. Because fishery observers will not be present on all boats at all times, we do not expect all bycatch events to be observed, recorded, and reported. The most recent estimates of ESA-listed bycatch in the DGN fishery (considered rare events) are based on an updated methodology that incorporates data from a long time series of observation of the DGN fishery and overall observed bycatch rates over time, as opposed to relying upon generating ratio estimates of bycatch from observed proportions of the fishery each year. In line with our use of the observer data to generate estimates of anticipated bycatch in the future DGN fishery, we will track both the annual estimates and the most recent 5-year cumulative totals that will be generated using observer data and the updated methodology by Carretta (2022). These estimates are published each year, although there is currently a time lag between the bycatch events and ultimate generation of bycatch estimates (published estimates for 2020 were released during 2022, for example). In this respect, the exact observer coverage rates achieved each year are not as influential on annual estimates of bycatch for any given year, although over the next five year the relative levels of observer coverage that is achieved will ultimately influence the confidence and accuracy of bycatch estimates each year and over time. As a result, this biological opinion relies upon the proposed action and the assumption that the

20-30% percent observer coverage goal will be maintained over the last five years of the DGN fishery to sustain (at a minimum) the current level of confidence in the bycatch estimates produced from observer data. The biological opinion also assumes that bycatch rates between observed and unobserved DGN vessels are similar.

In the interim time lag of published bycatch estimates by Carretta et al. (2022b), we will track incidents of observed (or otherwise detected or reported) bycatch in the DGN for consistency with the anticipated bycatch levels described in Table 11. Although the actual probability of observing any event that occurs one time in a year is one out five (20%) if you observe 20 percent of the total fishery, it is possible that expected bycatch of ESA-listed species could occur in the observed portion of the fishery. However, over the next five years, you would not expect to see a multitude or series of events where any of these ESA-listed species are caught consistently in the observed portion of the fishery given the historical observer data that suggests bycatch rates for these species are events that occur only a few times a year at most. In other words, although we expect that up to three loggerheads may be entangled in the DGN fishery in any year, and over the next five years, we would not expect that all three loggerheads entanglements would be observed during a single year,[24] or that loggerheads bycatch will be observed every year based upon 20 percent observer coverage. If such a series of observations were to occur given ~20% observer coverage, it is highly likely that interaction rates during this time would be higher than expected, and we would assume that overall impacts to ESA-listed species were greater than what has been anticipated in this biological opinion. Ultimately, we expect that over the next five years the observed record of DGN fishery bycatch will remain consistent with the anticipated total take levels analyzed in the *Effects of the Action* (section 2.5) and *Integration and Synthesis* (section 2.7) of this biological opinion and the acknowledgement that observation of rare event bycatch during proportional sampling should remain rare.

2.5.4 Significance of Net Entanglement Reports and DGN Net Damage Data

In Section 2.5.1 above, we evaluated two additional sources of information that are relevant to the overall understanding of potential effects of the DGN fishery on ESA-listed fisheries: reports that do not originate from fishery observers of whale entanglements along the U.S. West Coast that involve netting, and damage to DGN nets as reported by fisheries observers that could be indicative of escape or "blow through" events. Both of these sources of information relate to interpretation of the observer bycatch record and generation of take estimates based on that data. Above, we concluded that even if some small number of historical records of net entanglements were associated with large mesh drift gillnet gear or the DGN fishery in the past, this is consistent with the assumption that small numbers of interactions with the DGN fishery are expected to go unobserved. Additionally, the available information regarding net damage did not indicate that obviously damaged nets appear to be regular occurrences during observed fishing effort. As a result, these data are not dispositive that encounters between large organisms such as whales and DGN gear go beyond the scale of the bycatch estimates produced from observer data. Further, this information does not inform modification of the estimates, or their underlying

[24] Probability of observing all 3 takes of loggerheads that occur if 3 occur during 20% observer coverage is less than 1%.

methodology, that we rely upon in characterizing the reasonably certain effects of the DGN fishery on ESA-listed species.

2.5.5 Impact of Data Collection

NMFS routinely authorizes biological sampling of sea turtles captured in directed research that includes tissue sampling, as well as more invasive sampling techniques. Based on the described methods of cleansing and disinfection, infection of the tissue biopsy site would not be expected. At most, NMFS expects turtles would experience brief, minimal discomfort during the process. It is not expected that individual turtles would experience more than short-term stress during tissue sampling. Researchers who examined turtles caught two to three weeks after sample collection noted the sample collection site was almost completely healed. During a more than 5-year period of tissue biopsying using sterile techniques, NMFS researchers encountered no infections or mortality resulting from this procedure (NMFS 2006b). Bjorndal et al. (2010) investigated the effects of repeated skin, blood and scute sampling on juvenile loggerhead growth. Turtles were sampled for each tissue type three times over a 120-day period. The authors found that repeated sampling had no effect on growth rates; growth rates of sampled turtles were not significantly different from control animals. Turtles exhibited rapid healing at the sampling site with no infection or scarring. Further, all turtles increased in body mass during the study indicating that sampling did not have a negative impact on growth or weight gain. The authors conclude that the sampling did not adversely impact turtle physiology or health (Bjorndal et al. 2010). Consequently, NMFS believes the impact of collecting tissue samples is minor and will not have any significant effect on any species of sea turtle that may be captured in the DGN fishery.

2.6 Cumulative Effects

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

Some continuing non-Federal activities are reasonably certain to contribute to the overall environmental health and habitat quality within the action area. In section 2.4 *Environmental Baseline*, we described the current and ongoing impacts associated with other activities that affect ESA-listed species along the U.S. West Coast. We are reasonably certain that these activities and impacts will continue to occur while this proposed action occurs. Some continuing non-Federal activities are also reasonably certain to contribute to climate effects within the action area. However, it is difficult if not impossible to distinguish between the action area's future environmental conditions caused by global climate change that are properly part of the environmental baseline vs. cumulative effects. Therefore, all relevant future climate-related environmental conditions in the action area are described earlier in the discussion of environmental baseline (Section 2.4).

We did not identify additional state or private activities that are reasonably certain to occur within the action area, do not involve Federal activities (including permitting), and could result in cumulative effects to ESA-listed species and designated critical habitat within the action area. Activities that may occur in these areas will likely consist actions related to ocean use policy and management of public resources, such as commercial and recreational fishing, aquaculture, energy development that includes offshore wind, and other spatial planning/management projects. Changes in ocean use policies as a result of non-Federal government action are highly uncertain and may be subject to sudden changes as political and financial situations develop. Examples of actions that may occur include changes to state fisheries which may alter fishing patterns or influence the bycatch of ESA-listed species; installation of wind/wave energy projects or aquaculture projects near areas where ESA-listed species are known to migrate through or congregate; designation or modification of marine protected areas that include habitat or resources that are known to affect marine mammals and sea turtles; and coastal development which may alter patterns of shipping or boating traffic. However, none of these potential state, local, or private actions, can be anticipated with any reasonable certainty in the action area at this time, and most all actions in federal waters of the EEZ of those described as examples would likely involve federal involvement (e.g., permitting) of some type.

2.7 Integration and Synthesis

The Integration and Synthesis section is the final step in assessing the risk that the proposed action poses to species and critical habitat. In this section, we add the effects of the action (Section 2.5) to the environmental baseline (Section 2.4) and the cumulative effects (Section 2.6), taking into account the status of the species and critical habitat (Section 2.2), to formulate the agency's biological opinion as to whether the proposed action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) appreciably diminish the value of designated or proposed critical habitat as a whole for the conservation of the species. This biological opinion does not perform an analysis of destruction or adverse modification of critical habitat because NMFS has determined that the proposed action is not likely to adversely affect designated critical habitat.

Using the projected fishing effort that could occur in the DGN fishery and the bycatch rates that have been observed since implementation of measures to reduce interactions between protected species and the DGN fishery, we have estimated the capture and resulting mortalities and serious injuries of ESA-listed marine mammals and sea turtles that may occur as a result of this proposed action. The rest of this biological opinion will be focused on describing how this anticipated level of effect, when added to the status and environmental baseline for each of these species, affects the likelihood of both the survival and recovery of each species.

2.7.1 Marine Mammals

When assessing the impact of proposed or ongoing projects on marine mammals under the MMPA, NMFS relies upon the concept of PBR level to assist or guide decision making about acceptable or appropriate levels of impact that marine mammal stocks can withstand. As described in the MMPA, PBR[25] is defined as "the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population (OSP; 16 U.S.C. 1362 (20)." In addition, the MMPA states that PBR is calculated as the product of three elements: the minimum population estimate (Nmin) of the stock; half the maximum net productivity rate (0.5R max) of the stock at a small population size; and a Fr. PBR is an approach developed to assess incidental take of marine mammals under the MMPA. It uses conservative minimum population estimates and a Fr based on the population status and is also comprehensive because it calculates take (total take) per stock. The underlying analysis supporting the PBR concept examined the impact of population removals for a period of 100 years in terms of the time delay in populations reaching carrying capacity. These simulations evaluated the robustness of each case over a range of bias or uncertainty in productivity rates, abundance estimation, and mortality estimation (Wade 1998). Given this long term simulation approach used to support this concept, the levels established under the PBR are most appropriate for examining the impact of annual average removals over a long period of time, and are not an indicator of some point beyond which the stock could not reach OSP at all, over shorter time periods, or within a given year.

It is important to note that while PBR serves as a useful metric for gauging the relative level of impact on marine mammal stocks as defined in the MMPA, PBR by itself does not equate to a species or population level assessment under the ESA where analyses are conducted at the level of the species as listed as threatened or endangered. The concept of managing impacts to marine mammal populations to levels that do not significantly affect recovery times shares the general intent of the jeopardy standard of the ESA in terms of looking at both the continued existence and recovery of a population. However, the ESA does not rely specifically on the same metrics or directly relate the likelihood of recovery to potential delay of recovery. In this biological opinion, the ESA-listed marine mammals also are not necessarily protected at the same scale as stocks under the MMPA. For example, fin whales are listed under the ESA as a global population whereas the area protected under the MMPA in stocks such as the CA/OR/WA stock of fin whales that occurs off the west coast of the United States. Therefore, we use the PBR concept from the MMPA to help characterize the relative impact of the DGN on the MMPA stocks of ESA-listed marine mammals likely to be adversely affected by the DGN fishery, and then relate those findings to the species as a whole under the jeopardy standard of the ESA.

2.7.1.1 Fin Whales

In this biological opinion, we have identified the CA/OR/WA stock of fin whales as the population of fin whales that may be affected by the DGN fishery occurring off the U.S. west coast. We anticipate that up to one fin whale may become entangled or captured in DGN gear in any year, although no more than one individual would be expected to be caught over the next five years (Table 13). It is possible this incident may result in a mortality or serious injury, so we consider the worst-case scenario that this would occur to any individual in the population.

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[25] Included in the 1994 amendments to the MMPA.

The best estimate of fin whale abundance of this stock is about 11,065 whales, and the most recent PBR level for this stock is 80 animals (Carretta et al. 2022a). In any given year, the loss of one fin whale represents about 0.001% of the total estimated population of the stock. Consistent with the approaches typically used in the SAR to compare known mortalities and serious injuries to PBR and impacts that occur over a broader period of time to gauge effects, the loss of one animal over a 5-year period represents <1 percent of PBR on an annual basis, well below any level that would be expected to impact the timing of the CA/OR/WA stock of fin whales recovering to OSP. As mentioned in the *Environmental Baseline* section 2.4.1, significant threats to this stock include ship strikes and incidental entanglement in commercial fishing gear. The most recent SAR for fin whales estimates the 5-year annual average mortality and serious injury to the CA/OR/WA stock of fin whales from all human-caused sources, including commercial fisheries (0.64 animals) + ship strikes (1.4 animals), is 2.24 animals, which is about three percent (2.8%) of this stock's PBR. Although Rockwood et al. 2017 estimates that annual ship strike deaths of fin whales along the U.S. West Coast may be 43 fin whales, NMFS recognizes that there is uncertainty surrounding what the true number of ship collisions and mortalities are for fin whales, and concludes that current level of vessel strikes (whether reflected by the Rockwood et al. estimates or not) do not appear to be impeding the recovery of these stocks (NMFS 2022c).

In this biological opinion, we consider that the DGN fishery is expected to occur each year for the next five years, and then will no longer be prosecuted, with the effects that have been described in Table 13 occurring (section 2.5.2 *Risk for ESA-listed species Affected*). As described in section 2.4 *Environmental Baseline* and section 2.6 *Cumulative Effects*, we anticipate that most of the factors that have been affecting fin whales along the U.S. West Coast such as ship strikes are likely to continue over the next five years as well. While climate change may be influencing fin whale migrations and the distributions of prey, this factor is unlikely to substantially affect the relative exposure of fin whales to the DGN fishery over the next five years. In lieu of any information that suggests the magnitude of impacts resulting from all sources of mortality and serious injury to this stock will change due to climate change over the next five years, we anticipate that the magnitude of impacts on fin whales that have occurred in the past are expected to continue for the next five years, including one take in the DGN fishery. Since receiving protection from whaling, the stock is likely increasing, as indicated most recently from abundance estimates from four surveys conducted off the U.S. west coast from 1996 through 2014. During the past 18 years, only one fin whale has been observed taken by the DGN fishery (1999), indicating that the likelihood that a fin whale would be taken in the DGN fishery is very low. In combination with the impacts of ship strikes and other known fishery interactions that lead to mortality and serious injury, we expect that the proposed action will not contribute to sources of mortality at a level that would threaten the ability or timing of this stock of whales to recover in the future, including after the DGN fishery has concluded.

In this biological opinion, we must consider the impacts from the DGN fishery on the globally-listed population of fin whales. The trend in the global population of fin whales is not definitive, yet there is some evidence of increased abundance from 1991 to 2018 (Moore and Barlow 2011, Nadeem et al. 2016, Becker et al. 2020a). The additional protection from the threat of whaling is believed to have relieved the major source of mortality for this species, particularly in the Northern Hemisphere. Based on the relatively small level of impact expected from the proposed action on the affected fin whale population (CA/OR/WA stock), there is no reason to expect these anticipated impacts would lead to effects on the global population that would be significant or detectable, especially given the effects will cease within the next five years. As a result, we conclude that the incidental take and resulting mortality of one fin whale associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of fin whales.

2.7.1.2 Humpback Whales

In this biological opinion, we have identified two DPSs of humpback whales that may be affected by the DGN fishery occurring off the U.S. west coast, although both DPSs are associated with the former CA/OR/WA stock of humpback whales, and with new stocks designated using DIPs that share both wintering and feeding areas. First, we will generally consider relative impact of the proposed action on the affected stock of humpback whales, and then we will consider the potential impacts on the ESA-listed DPS.

Under the proposed action, we expect that up to one humpback whale could be entangled and killed or seriously injured in the DGN fishery in any given year over the next five years, after which the fishery will no longer be prosecuted. We also anticipate that no more than one individual would be entangled over the next five years, leading to no more than one mortality or serious injury over that 5-year period. For the purposes of this biological opinion, we assume that all age-classes are vulnerable and of equal significance and males are as vulnerable as females. We expect that these mortalities or serious injuries could occur to any individual in the population. Over the next five years, this amounts to an average of 0.2 individuals being removed from the population every year.

The best estimate for the abundance of the former CA/OR/WA stock, which combines members of several different humpback whale DPSs, is 4,973 (CV=0.048) animals Calambokidis and Barlow 2020), with a minimum population estimate of 4,776 whales. In the most recent final SAR (2021; Carretta et al. 2022a) the PBR level for this in U.S. waters is 29.4 whales per year. Since the 2021 SAR was published, NMFS has published a draft SAR for the two new DIPs related to the ESA-listed DPSs that forage along the U.S. west coast. Since the draft SAR was published and available for public comment (Carretta et al. 2023a), the most recent draft SAR included revised estimates of abundance and trend for the two new stocks, as well as a revised apportionment of anthropogenic threats to them (Carretta et al. 2023b). These will be summarized in sections specific to the two DPSs.

Although Rockwood et al. (2017) estimates that annual ship strike deaths of humpback whales along the U.S. West Coast may be 22 humpback whales, NMFS recognizes that there is uncertainty surrounding what the true number of ship collisions and mortalities are for humpback whales, and concludes that current level of vessel strikes (whether reflected by the Rockwood et al. estimates or not) do not appear to be impeding the recovery of these stocks (NMFS 2022c). In this biological opinion, we consider that the DGN fishery is expected to occur each year for the next five years, and then will no longer be prosecuted, with the effects that have been described in Table 13 occurring (section 2.5.2 *Risk for ESA-listed species Affected*). As described in section 2.4 *Environmental Baseline* and section 2.6 *Cumulative Effects*, we anticipate that most of the factors that have been affecting humpback whales along the U.S. west coast such as ship strikes are likely to continue over the next five years as well. Climate change could influence humpback whale migrations, the distributions of prey, and the relative exposure of humpback whales to the DGN fishery in the future. However, this factor is unlikely to substantially affect the relative exposure of humpback whales to the DGN fishery over the next five years. In lieu of any information that suggests the magnitude of impacts resulting from all sources of mortality and serious injury to this stock will change due to climate change over the next five years, we anticipate that the magnitude of impacts on humpback whales that have occurred in the past are expected to continue for the next five years, including one take in the DGN fishery.

One humpback whale was observed incidentally taken in the DGN fishery in 1994, 1999, and 2004, and recently two humpbacks were observed incidentally taken in 2021 (Carretta 2022). In addition, one additional report of a humpback whale take was reported by a fisher and received under the MMAP in 2009. In recent years, there have also been three entanglements reported that potentially were associated with large mesh drift gillnet gear, although it is not possible to confirm the origins. These data are consistent with the understanding that occasional bycatch of humpback whales will occur in the DGN fishery. As described in the section 2.5 (*Effects of the Action*), we anticipate that up to one humpback whale might be taken (with a 25% chance of being killed). Because we do not know whether a humpback whale may originate from the threatened Mexico DPS or the endangered Central America DPS, we will assume it could come from either DPS; and thus, we will synthesize and integrate the information and sources of all threats, including the proposed action, on both DPSs (below).

As described in the *Environmental Baseline,* although humpbacks continue to be entangled, the number of confirmed reports in the last four years (through 2022) has shown a decline since the number of confirmed reports from 2015-2018, likely due in part to conservation efforts, as described below. Serious injury determinations for humpbacks entangled in 2021 and 2022 have not been made or incorporated into a SAR. A summary of information on the human-caused threats specific to each of the two listed DPSs within the Action Area will be provided below.

2.7.1.2.1 Mexico DPS Humpback Whales

As described in the *Status of the Species* (section 2.2.2), the most recent estimate of the abundance of Mainland Mexico-CA/OR/WA stock humpback whales is 3,477, (minimum population size of 3,185). Based on this information, the most recent 2022 draft SAR (Carretta et al. 2023b) calculated PBR of 65 whales for this stock, and given the residency time of this DPS yields a PBR in U.S. waters of 43 whales per year. The loss of one animal over the next five years (0.2 per year) as a result of the proposed action represents about 0.5% percent of PBR (43) for this stock on an annual basis.

While the current total abundance for the entire Mexico DPS is uncertain, the population size is highly likely to be at least 6,000 (likely more) based on an array of estimates which are conservative given the increasing abundance of humpback whales along the U.S. West Coast since that time. Considering the prospect of losing one individual from the population in any given year, this represents less than 0.1 percent (0.017) of the total Mexico DPS population during a single year. This is a very small proportion of the total population.

In this biological opinion, we know the DGN fishery will only be prosecuted through the next five years, with an anticipated effort of on average 305 sets (up to 573 sets in a given year), with effects that have been described in Table 13 occurring (section 2.5.2 *Risk for ESA-listed species Affected*). Over the next five years, we expect that the Mexico DPS humpback whale population will lose up to 1 individual as a result of this proposed action. After 5 years, the DGN fishery will no longer be prosecuted and any threats will be nonexistent. The Mexico DPS population is increasing at healthy levels, and this level of impact is expected to be undetectable for such a robust population that has been showing signs of improvement in recent decades, as indicated by the 2016 listing as threatened as opposed to the formal global listing as endangered.

Following along a similar approach for cataloging current threats and impacts for the individual DPS as is done for the former CA/OR/WA stock of humpback whales, in the *Status of the Species* (section 2.2.2) and *Environmental Baseline* (section 2.4.1) we summarized information from the most recent SAR for the portion of the Mexico DPS that occurs in the action area (Mainland Mexico - CA/OR/WA stock), which estimated that mortality and serious injury from commercial fisheries amounted to a total of 11.4 Mexico DPS whales per year (not including the estimated mortality/serious injury due to the DGN fishery, as estimated through 2020). These estimates are specific to all of the commercial fisheries, including the many unidentified fisheries, and take into account *unidentified whales*, which are prorated to humpback whales based on location, depth, and time of year (Carretta 2018). Serious injuries/mortalities due to marine debris, recreational and tribal fisheries were estimated to be 0.56 Mexico DPS whales per year.

As summarized in the *Environmental Baseline* (section 2.4.1), a total of fourteen vessel strikes involving humpback whales were observed off the U.S. west coast (2016-2020), totaling 13.2 serious injuries/mortalities, or 2.6 whales/year. Given that 58% of these whales could be from the Mexico DPS, approximately 1.5 whales per year represents the annual average of recent known vessel strikes of the Mexico DPS within the action area. As noted earlier, NMFS recognizes that there is uncertainty surrounding what the true number of ship collisions and mortalities are for humpback whales, and concludes that current level of vessel strikes (whether reflected by the Rockwood et al. estimates or not) do not appear to be impeding the recovery of these stocks (NMFS 2022c). If we consider the Rockwood et al. (2017) values, the prorated value apportioned to the Mexico DPS within the action area would be 10.2, accounting for the fraction of observations in the state of WA and CA/OR (Carretta et al. 2023b).

The most recent draft SAR for the Mainland Mexico-CA/OR/WA stock of humpback whales (Carretta et al. 2023b) has calculated a PBR of 43 whales per year. Given the estimated human-caused serious injury/mortality due to commercial fisheries (11.4), plus the serious injury/mortality due to marine debris, recreational, and tribal fisheries (0.56) and known vessel strikes (1.5), yields 13.5 humpback whales removed from a population of at least 3,185 whales (minimum) or the best estimate of 3,477 animals (CV=0.101). When compared to PBR for the portion of the Mexico DPS that occurs within the action area (43 animals), the serious injury/mortality of 13.5 animals equates to 31% of PBR. Although uncertain, including the 10.2 prorated mortalities from estimated vessel strikes from Rockwood et al. (2017) results in a total of 22.1 animals, or 51% of the PBR for the portion of the Mexico DPS that occurs within the action area. As acknowledged previous, the Mexico DPS is distributed widely across the North Pacific. The extent of similar threats across the entire DPS has not been described to the same detail, but we assume they are occurring at some level throughout.

Although humpbacks continue to be entangled, the number of confirmed reports in the last four years (< 20 animals per year during 2019-2022) has shown a decline since the number of confirmed reports from 2015-2018, where between 30 and nearly 50 humpbacks were entangled (NMFS 2023e). We know some of these humpbacks likely are represented by the Mexico DPS, and that some may be serious injuries or mortalities. However, the low observed bycatch rates and most recent total bycatch estimates suggest that the loss of one individual from this population in the next five years as a result of bycatch in the DGN fishery will not result in a detectable impact on the Mexico DPS of humpback whales, particularly considering the positive growth rate of this DPS (at least 5-6% growth per year) that has been occurring for the last 15 years, and especially given the effects of the proposed action will cease within the next five years.

As with most large whales, removal of the threat of whaling has relieved the primary source of mortality that resulted in reduced population sizes and the listing of this species as threatened. In addition, the work of the California Dungeness Crab Fishing Gear Working Group and the Risk Assessment Mitigation Program has developed a strategy to assess circumstances where entanglement risk of humpbacks (and blue whales and leatherbacks) may be elevated and, as needed, identify possible management measures for CDFW. Such actions may include fleet advisories, fishing depth constraints, vertical line reductions, fishery closures, and the use of approved alternative gear. Given that 34 humpbacks (out of 136, or 25%) were reported entangled in CA Dungeness pot gear from 2016-2020 (Carretta et al. 2023a b), these strategies are likely reducing the risk of entanglements in this gear in concert with other conservation actions taken in Oregon and Washington to address entanglement risks in their Dungeness crab fisheries.

Although a number of threats facing humpback whales remain, this species does not appear to be at significant risk of extinction, especially in the North Pacific. While a recovery plan for the newly listed Mexico DPS has not yet been developed, we would expect that plan to address the recent increase in reported entanglements that have been observed along the U.S. West Coast, which include both U.S. and international sources of entanglements. In combination with what is known about the impacts from ship strikes and other known fishery interactions that lead to mortality and serious injury, we expect that the proposed action is not contributing additional sources of mortality at a level that would threaten the ability of this stock of whales to recover, especially considering the effects of the action will cease within the next five years. As a result, we conclude that the one incidental take and resulting mortality of a humpback whale associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the Mexico DPS of humpback whales.

2.7.1.2.2 Central America DPS Humpback Whales

As described in the *Status of the Species* (section 2.2.2), Central America/Southern Mexico-CA/OR/WA stock of humpback whales is estimated to be approximately 1,496 individuals, and is showing a positive growth rate of 1.6 % (Curtis et al. 2022). Based on this information, the most recent 2022 draft SAR (Carretta et al. 2023b) calculated PBR of 5.2 whales for this stock, which given the residency time of this stock in U.S. waters yields a PBR of 3.5 whales per year. The loss of one animal over the next five years (0.2 per year) as a result of the proposed action represents about six percent of PBR (3.5) for this stock on an annual basis. While NMFS will continue to evaluate the relationship between the humpback whale DPSs and recognized DIPs moving forward, at this time we consider the inclusion of southern Mexico humpbacks and the abundance and trend estimates recently published by Curtis et al. (2022) as being reflective of the current status of the Central America DPS.

Following along a similar approach for cataloging current threats and impacts for the portion of the Central America DPS that occurs within the action area (effectively the entire Central America DPS) in the *Status of the Species* (section 2.2.2) and *Environmental Baseline* (section 2.4.1), we summarized information from the most recent SAR for the Central America DPS which estimated that mortality and serious injury from commercial fisheries amounted to a total of 8.1 Central DPS whales per year (not including the estimated mortality/serious injury due to the DGN fishery, as estimated through 2020). These estimates specific to all of the commercial fisheries, including the many unidentified fisheries, takes into account *unidentified whales*, which are prorated to humpback whales based on location, depth, and time of year (Carretta 2018). Serious injuries/mortalities due to marine debris, recreational and tribal fisheries were estimated to be 0.35 Central America DPS whales (Carretta et al. 2023b).

As summarized in the *Environmental Baseline* (section 2.4.1), a total of fourteen vessel strikes involving humpback whales were observed off the U.S. west coast (2016-2020), totaling 13.2 serious injuries/mortalities, or 2.6 whales/year. Given that 42% of these whales could be from the Central America DPS, approximately 1.1 whales per year represents the annual average of recent known vessel strikes of the Central America DPS within the action area. As noted earlier, NMFS recognizes that there is uncertainty surrounding what the true number of ship collisions and mortalities are for humpback whales, and concludes that current level of vessel strikes (whether reflected by the Rockwood et al. estimates or not) do not appear to be impeding the recovery of these stocks (NMFS 2022c). If we consider the Rockwood et al. (2017) values, the prorated value apportioned to the Central America DPS within the action area would be 6.5, accounting for the fraction of observations in the state of WA and CA/OR (Carretta et al. 2023b).

The most recent draft SAR for the Central America-CA/OR/WA stock of humpback whales (Carretta et al. 2023b) has calculated a PBR of 3.5 whales per year. Given the estimated human-caused serious injury/mortality due to commercial fisheries (8.1), plus the serious injury/mortality due to marine debris, recreational, and tribal fisheries (0.35) and known vessel strikes (1.1), yields 9.6 whales removed from a population of at least 1,284 whales (minimum) or the best estimate of 1,496 animals (CV=0.171). When compared to PBR for the portion of the

Central America DPS that occurs within the action area (3.5 animals), the serious injury/mortality of 9.6 animals equates to 274% of PBR, or ~2.7 times greater that PBR for this stock. Although uncertain, including the 6.5 prorated mortalities from estimated vessel strikes from Rockwood et al. (2017), results in a total of 14.9, which is 4.3 times greater than the PBR for the portion of the Central America DPS that occurs within the action area. As described earlier, the portion of the Central America DPS that occurs within the action areas is effectively the entire population, so the levels of impact described for the Central America-CA/OR/WA stock effectively represent to the totals in U.S. waters.

Considering the prospect of potentially losing one individual from the population in any given year, this represents about 0.07 percent of the total Central America DPS population during a single year. This is a small proportion of the total population. In this biological opinion, we know the DGN fishery will only be prosecuted through the next five years, with an anticipated effort of on average 305 sets (up to 573 sets in a given year), with effects that have been described in Table 13 occurring (section 2.5.2 *Risk for ESA-listed species Affected*). Over the long-term, we expect that the Central America DPS humpback whale population could lose one individual as a result of this proposed action. After five years, the DGN fishery will no longer be prosecuted and any threats from the DGN fishery will be nonexistent. The Central America DPS population is increasing, although not apparently at the same rate as the Mexico DPS. However, this remains an area of uncertainty that will need additional study moving forward given the underlying context of the observed increases in humpback whales off the U.S. West Coast, making it difficult to draw conclusions about the factors that might explain why different populations of humpback whales that have significant presence off the U.S. West Coast are growing at different rates at this time.

Although humpbacks continue to be entangled, the number of confirmed reports in the last four years (< 20 animals per year during 2019-2022) has shown a decline since the number of confirmed reports from 2015-2018, where between 30 and nearly 50 humpbacks were entangled (NMFS 2023e). We know some of these humpbacks likely are represented by the Central America DPS, and that some may be serious injuries or mortalities. However, the low observed bycatch rates and most recent total bycatch estimates suggest that the loss of one individual from this population in the next five years as a result of bycatch in the DGN fishery will not result in a detectable impact on the Central America DPS of humpback whales, particularly considering the positive growth rate of this DPS (estimated 1.6% growth per year) that has been occurring for the last 15 years, and especially given the effects of the proposed action will cease within the next five years.

As with most large whales, removal of the threat of whaling has relieved the primary source of mortality that resulted in reduced population sizes and the listing of this species as threatened. In addition, the work of the California Dungeness Crab Fishing Gear Working Group and the Risk Assessment Mitigation Program has developed a strategy to assess circumstances where entanglement risk of humpbacks (and blue whales and leatherbacks) may be elevated and, as needed, identify possible management measures for CDFW. Such actions may include fleet advisories, fishing depth constraints, vertical line reductions, fishery closures, and the use of approved alternative gear. Given that 34 humpbacks (out of 136, or 25%) were reported entangled in CA Dungeness pot gear from 2016-2020 (Carretta et al. 2023a, b), these strategies are likely reducing the risk of entanglements in this gear in concert with other conservation actions taken in Oregon and Washington to address entanglement risks in their Dungeness crab fisheries.

One of the primary factors in the recent listing decision to retain an endangered status for this DPS is that the population is estimated to be at risk of extinction due to such a relatively small population size. Since that time, NMFS has determined the population has grown, although this population should still be considered relatively small. While a recovery plan for the newly listed Central America DPS has not been developed, we would expect that plan to address the recent increase in entanglements that have been observed along the U.S. west coast, which include both U.S. and international sources of entanglements. In combination with what is known about the impacts from ship strikes and other known fishery interactions that lead to mortality and serious injury, we expect that the proposed action is not contributing additional sources of mortality at a level that would threaten the ability of this stock of whales to recover, especially considering the effects of the action will cease within the next five years. As a result, we conclude that the limited incidental take and resulting mortality of one humpback whale associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the Central America DPS of humpback whales.

2.7.1.3 Sperm Whales

In this biological opinion, we have identified the CA/OR/WA stock of sperm whales as the population of sperm whales that may be affected by the DGN fishery occurring off the U.S. west coast. We anticipate that up to two sperm whales may become entangled or captured in DGN gear in any year, and that both may end up as mortalities or serious injuries. We also anticipate that up to two individuals would be entangled over the next five years, both of which may end up as mortalities or serious injuries. We expect that these mortalities or serious injuries could occur to any individual in the population. In any given year over the next five years, this level of mortality and serious injury to about 0.1 % of the population being removed from the stock that occurs within the action area.

As mentioned in the *Environmental Baseline* (section 2.4.1), significant threats to this stock include ship strikes and incidental entanglement in commercial fishing gear. The 2021 SAR for sperm whales estimates that the most recent 5-year annual average of mortality and serious injury for the CA/OR/WA stock of sperm whales from all human-caused sources, including commercial fisheries (2013-2017 for DGN and 2012-2016 for the WA/OR/CA groundfish, bottomfish longline/set line) is ≥0.64 animals (Carretta et al. 2022a). Of this total, the DGN fishery has accounted for about 57 percent of the estimated mortality and serious injury related to commercial fisheries in the past. We note this SAR does incorporate the updated bycatch methodology employed by Carretta (2022) that is currently used as the basis to estimate recent and future bycatch in the DGN fishery in this biological opinion. In addition to what has been evaluated in the most recent SAR for sperm whales, we recognize that additional recent impacts include one report of an unidentified fishery interaction determined to be a serious injury off Ventura, CA in 2020 (Carretta 2022).

In this biological opinion, we consider that the DGN fishery is expected to occur each year for the next five years, and then will no longer be prosecuted, with the effects that have been described in Table 13 occurring (section 2.5.2 *Risk for ESA-listed species Affected*). As described in section 2.4 *Environmental Baseline* and section 2.6 *Cumulative Effects*, we anticipate that most of the factors that have been affecting sperm whales along the U.S. West Coast such as ship strikes and fishery bycatch are likely to continue over the next five years as well. While climate change may be influencing fin whale migrations and the distributions of prey, this factor is unlikely to substantially affect the relative exposure of fin whales to the DGN fishery over the next five years. In lieu of any information that suggests the magnitude of impacts resulting from all sources of mortality and serious injury to this stock will change due to climate change over the next five years, we anticipate that the magnitude of impacts on sperm whales that have occurred in the past are expected to continue for the next five years, including up to two entanglements in the DGN fishery.

Sperm whale abundance estimates in the action area have been variable in recent decades, which likely reflect inter-annual variability in movement of animals into and out of the study area more than fluctuations in population size. Although populations are expected to have increased due to the cessation of whaling, the ability to detect trends in abundance for the CA/OR/WA stock of sperm whales has been limited by the variable population estimates. This is in part because sperm whale migration patterns are not well understood (patterns seem to vary with age and sex) and because sperm whales occur in larger groups and tend to range more widely, making abundance estimates more variable than those of other large whales with similar population sizes. The distribution and relative abundance of sperm whales in relation to key environmental features may also influence the distribution of their prey and thus, sperm whale relative abundance. While some threats to sperm whales do remain despite the advent of whaling protections, we do not believe this stock has experienced any dramatic decline in abundance.

One sperm whale was observed incidentally taken in the DGN fishery in 1998, but the net did not have a full complement of pingers; therefore, it is difficult to evaluate whether pingers have an effect on sperm whale entanglement. However, pingers have been shown to have a positive effect on other odontocetes (i.e., lower entanglement rates; Barlow and Cameron 2003). Two more sperm whales were observed taken in 2010 (one killed; one released seriously injured) in the DGN fishery and the net did have a full complement of pingers (NMFS 2013). In any given year, the loss of two sperm whales would represent a sizeable portion (80 percent) of the PBR (2.5) for this stock. Consistent with the approaches typically used in the SAR to compare known mortalities and serious injuries to PBR and impacts that occur over a broader period of time to gauge effects, the loss of two animals over a 5-year period represents about 16% percent of PBR on an annual basis (0.4 per year). This is below the level that would be expected to impede the speed of recovery for the CA/OR/WA stock of sperm whales. In combination with limited impacts from ship strikes and other known fishery interactions that lead to mortality and serious injury, we expect that the proposed action is not contributing to sources of mortality at a level that would threaten the ability and timing of this stock of whales to recover.

In this biological opinion, we consider the impacts from the DGN fishery on the globally-listed population of sperm whales. Similar to the CA/OR/WA stock, trends in the abundance estimates of other Pacific populations, or the species globally, are not clear; but it is more likely sperm whale populations are showing signs of increasing and the overall population of sperm whales has increased worldwide since it was listed under the ESA in 1973. As mentioned in the *Status* of *the Species* (section 2.2.3), the estimate of worldwide sperm whale abundance is in the hundreds of thousands of individuals. Protection from whaling has eliminated the primary source of mortality that occurred historically. Based on the relatively small level of impact expected from the proposed action on the sperm whale population (CA/OR/WA stock of sperm whales), there is no reason to expect these anticipated impacts would lead to effects on the global population that would be significant or detectable. As a result, we conclude that the occasional incidental take and resulting mortality of sperm whales associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of sperm whales.

2.7.2 Sea Turtles

As described in the *Status of the Species* for leatherback sea turtles, there have been recent efforts to derive a quantitative approach for establishing turtle bycatch management frameworks for U.S. and international fisheries (Curtis et al. 2015). However, the implications of applying such a framework into the ESA management and policy decision making process have not been fully evaluated. It is clear that there will have to be reconciliation between the mandates of the ESA along with any further development of a bycatch management program for sea turtles before NMFS can implement any such framework. In the future, it does seem possible that such approaches could yield insight into appropriate short and long-term limits for sea turtle bycatch based on variable population status or conditions. Until then, NMFS will continue to rely on the best available information relating impacts to ESA-listed sea turtles and other species under the existing ESA standards.

2.7.2.1 North Pacific DPS Loggerhead Sea Turtles

The North Pacific Ocean DPS of loggerhead sea turtles is listed as endangered under the ESA. This DPS nests nearly exclusively on the mainland and offshore islands of Japan. There was a documented decline in the number of nesting turtles comprising this DPS between 50 and 90 percent since the 1950s (Kamezaki et al. 2003). However, since 2003/2004, an increasing trend of approximately 9 percent annual growth in the number of nests has been documented for the entire nesting assemblage, through 2015 (i.e., all nesting beaches combined) (Y. Matsuzawa, S, personal communication, 2017). This positive trend can likely be attributed to a complete ban on egg harvest in the mid-1970s, bycatch reduction implementation measures in pelagic longline fisheries in the North Pacific (e.g., circle hooks; TurtleWatch), increases in survival through safe handling, sea turtle bycatch resolutions in regional fishery management organizations, among other conservation efforts.

Recent abundance estimates suggest that there are approximately 342,000 North Pacific loggerheads, given nesting female abundance estimates, as well as adult male/juvenile abundance estimates (T. Jones, NMFS-PIFSC personal communication, 2019; Martin et al. 2020a). A recent PVA indicates that the adult female portion of the Yakushima subpopulation is increasing at a rate of approximately 2.3%/year and is comprised of 4,538 adult females. Assuming that the index beaches represent 52% of all nesting in this DPS, there are an estimated 8,276 adult females in this DPS.

Off Southern California, aerial surveys conducted during 2015, when anomalous warming of the North Pacific and El Niño conditions occurred, resulted in an estimated 70,000 loggerheads foraging in the area (Eguchi et al. 2020). A similarly designed aerial survey was conducted during a non-El Niño year (2011) that yielded zero observed loggerheads in Southern California. This indicates while most of the loggerheads observed taken in the DGN fishery occurred during anomalously warm ocean conditions, there may be high variability of loggerhead density in the area where the majority of the effort occurs.

As described in the *Status of the Species* (section 2.2.4), North Pacific loggerheads nesting in Japan are threatened by coastal development as well as coastal fisheries, which likely threaten the later life stages, such as large juveniles and adults. Fisheries bycatch from pelagic longlining are the greatest threat to this DPS throughout the North Pacific, with hundreds (if not thousands, depending on the source) of loggerheads killed each year. In the Western and Central Pacific Ocean, participating countries reporting to the Western and Central Fisheries Commission, given observed interactions from 2013 to 2020 produced an estimate of approximately 2,387 interactions per year, with an estimated 390 loggerhead mortalities per year. With low observer coverage in these international fleets (~3%), confidence in these estimates are low (i.e., only 69 loggerheads were observed during that time period). Nonetheless, we have more confidence in our domestic longline fishery, given 100% observer coverage in the Hawai'i-based shallow-set fishery and approximately 20% observer coverage in the Hawai'i-based deep-set fishery, along with variable coverage in the American Samoa longline fishery with zero loggerheads observed taken. The most biological opinions indicate that approximately 17 loggerheads are incidentally captured in these two fisheries combined each year, with around five mortalities each year assuming historical mortality rates from the injuries that have occurred in the past. Given that most of these turtles are taken in their oceanic phase, most loggerheads taken in the pelagic longline fisheries are likely juveniles/subadults. Thus, if we consider high 95% CI from the WCPFC estimates (452 loggerheads), we can assume that around 460 loggerheads may be removed from the population each year by pelagic longline fisheries. These could be an underestimate, considering the estimated number of loggerheads taken by 16 countries in this same area from 1989 to 2015 (10,980 animals = 610 loggerheads/year), but we do not know the variability of captures over decades, and with loggerheads showing an increasing trend, there may be more animals in the north Pacific Ocean interacting with longline gear.

Under the proposed action, we anticipate that up to three juvenile loggerhead sea turtles could be entangled in the DGN fishery in any given year, one of which may end up as a mortality or serious injury of a juvenile female that could have survived to adulthood. We also anticipate that up to three juveniles would be entangled over the next five years, one of which may end up as a mortality or serious injury of a juvenile female that could have survived to adulthood, after which the fishery will no longer be prosecuted.

The removal of one adult female (equivalent) in a year or over the next five years constitutes less than 0.1 (0.011) percent of the estimated adult female population (8,733). The removal of up to three juvenile turtles in a year or over the next five years within a total estimated population of loggerheads foraging off southern California when conditions are optimal for them given anomalously warm sea surface temperatures (70,000,) is less than 0.01 percent (0.004) of the local population. This is a very small proportion of the total population. This level of impact is essentially undetectable compared to the variations in nesting patterns that have been seen in the last two decades. With respect to the local juvenile population that may be present off the Southern California coast numbering in the tens of thousands primarily during anomalously warm ocean conditions, which may occur during the next five years,[26] the potential removal of up to three juveniles in a year or over the next five years is also a relatively small impact (<.01%). Currently, the dynamics in place are suggesting that recruitment is outpacing removals, despite the fact that known sources of mortality are quite large, particularly from bycatch in fisheries. While accurate or reliable totals of bycatch interactions or mortality for loggerheads across the Pacific are not available, based on what is known about threats off Baja California, Mexico and Japan, it seems likely that annual loggerhead bycatch totals can be measured in at least the hundreds, if not thousands. Other impacts associated with threats to nesting beach activity and nesting habitat in Japan are not easily quantified, but it appears that conditions at some primary and secondary beaches are improving.

The 2013 biological opinion on the DGN fishery also concluded that the loss of one adult female every five years from the DGN fishery presented negligible additional risk to survival and recovery of the North Pacific DPS loggerhead population. Considering an equivalent level of impact anticipated from this proposed action, these results are consistent with other modeling results (e.g., Martin et al. 2020a, which assessed the removal of loggerheads in the Hawai'i-based shallow-set longline fishery, given PVA projections) that have yet to quantitatively conclude discernible changes in the risk of extinction to loggerheads in the North Pacific to the low level of impact anticipated under this proposed action.

At this time, there seems to be little change in the outcome of analyses using the tools or approaches available to quantitatively assess the impact of removing very small numbers of juvenile loggerheads from the North Pacific DPS. The most current information suggests that North Pacific loggerheads are increasing at around 2.3%/year (Martin et al. 2020a). In comparison, expected impacts to this loggerhead population from the DGN fishery currently estimated in this biological opinion are less (in some cases substantially) than what was analyzed in the models and analyses presented above. As a result, the resultant impact of repeating any of these modeling exercises with a smaller level of adult female removal would be expected to be even less. These results are consistent in that none of the modeling or analytical results have yet to quantitatively describe changes in extinction risks to North Pacific DPA attributed to the type of low level of impact anticipated under this proposed action.

[26] El Niño conditions are forecast to occur in the summer of 2023, with a transition to El Niño persisting into the northern hemispheric winter (NOAA's National Weather Service Climate Prediction Center, accessed May 11, 2023).

In addition to the risk of extinction for loggerhead populations, NMFS must also consider the impact of the proposed action on the prospects for recovery of ESA-listed species under the jeopardy standard. The recovery tasks and goals identified by NMFS and USFWS (1998a) for loggerheads in the U.S. Pacific are very similar to those for leatherbacks. NMFS has been actively engaged in research and conservation efforts that are directed towards facilitating recovery. As with leatherbacks, it seems likely that any abundance goals for populations, including the North Pacific DPS, rest on factors of productivity and mortality throughout their range that are not likely to be affected by the limited removal of a of no more than a few juveniles, including one adult female equivalent with eventual reproductive potential over the next five years. As a result, the limited mortality of one adult female equivalent over the next five years would present negligible additional risk to this DPS of loggerheads. This would not be expected to prohibit this nesting population from increasing or maintaining a stable population in perpetuity, nor would it substantially impair or prohibit increases to the foraging population at key foraging grounds, especially considering the effects of the action will cease within the next five years. As a result, it seems unlikely that this anticipated effect from this proposed action will appreciably affect the chances of survival or recovery of this population.

Given the best available information, we conclude that the anticipated occasional incidental take and resulting mortality of North Pacific DPS loggerhead sea turtles associated with the proposed action is not likely to appreciably reduce the likelihood of survival or recovery of this species.

2.7.2.2 Leatherback Sea Turtles

As discussed in the *Status of the Species* (section 2.2.5), leatherback sea turtles are globally listed as endangered. The species is composed of seven populations, and the DGN fishery may adversely affect the West Pacific Ocean and the East Pacific populations, although we have assumed that the potential risk to the East Pacific population to be extremely low. In addition, as described, critical habitat was revised in 2012 (77 FR 4170) to include additional areas within the U.S. West Coast, which provide added protection of their principal biological feature, primarily scyphomedusae.

In the Western Pacific, the primary nesting beaches are located in Papua Barat, Indonesia and have provided us with long-term monitoring surveys in order to understand the abundance of nesting females, trends, hatchling success, and threats due to coastal inundation, predation, harvest of eggs and sea turtles, and other factors that may be impeding its recovery. A recent discovery of a previously undocumented nesting area on Buru Island, Indonesia and relatively new sites on the Solomon Islands suggest that additional undocumented nesting habitats may exist on other remote or infrequently islands of the western Pacific Ocean (NMFS and USFWS 2020b). In the eastern Pacific, researchers have been surveying primary leatherback nesting sites for decades, particularly in Costa Rica, so we have more confidence in abundance estimates, trends, and threats to that subpopulation, although there are secondary beaches that may not be surveyed as regularly and thus the abundance and trends may be more uncertain.

As summarized in the *Status of the Species* (section 2.2.5), the most recent estimated number of nesting females at Jamursba Medi and Wermon beaches, where 50%-75% of nesting believed to occur in the western Pacific, is 790 nesting females (Martin et al. 2020a). Applying the conservative estimate of 75% to the Martin et al. (2020a) estimate yields 1,053 females in this subpopulation, as of 2017. Assuming a 73% female sex ratio yields an estimated 1,443 adult leatherbacks in the West Pacific subpopulation. Using assumptions in Jones et al. (2012) and life history parameters, survival rates, etc. in Martin et al. (2020a), NMFS estimates the juvenile and adult population size of the West Pacific subpopulation to be 100,000 leatherbacks (NMFS 2023c).

Recent preliminary data from the 2 index beaches indicates that the nest numbers were relatively stable from 2017 to 2021 (Lontoh et al. in prep; Figure 3); however, the data are not available in sufficient detail to update the Martin et al. (2020a) model. Hence, we acknowledge that there is uncertainty associated with the current status of West Pacific leatherbacks, as represented by the two index beaches. Using the median trend in annual nest counts from Jamursba Medi (2001-2017) and Wermon (2006-2017), Martin et al. (2020a) estimated the combine trends for the two index beaches to be -6% annually. Until we receive the detailed raw monthly data from the nesting beaches, the growth trend analysis of Martin et al. (2020a) through 2017 cannot be updated.

Notably, however, we are seeing a positive growth trend using a newly established monitoring program (since 2017) on Buru Island, Indonesia which estimates approximately 103 adult female nesters. This would constitute an addition to the modeled estimate of 790 adult females nesting at the two index beaches on Papua Barat. Over the last six years (which may span two remigration intervals for leatherbacks), Buru Island is showing an increasing trend of 10.1% per year, which is a positive sign for this subpopulation.

Using the best available data to assess the status of the East Pacific subpopulation, NMFS and USFWS (2020b) estimated that there are approximately 1,007 nesting females, given what is known through monitoring the index beaches, which compromise approximately 75% of the total nesting in the population. Assuming a sex ratio of 79% females, suggests a total of 1,274 adults (males and females). Based on data from Jones et al. (2012), we expected that adults comprise a mean of 2.1% of the total population size, which would suggest an estimated 60,611 individuals in the subpopulation. The trend at the nesting beaches in Mexico and Costa Rica show a decline of -4.3%/year for Mexico (given the worst-case scenario) and a decline of -15.5% per year for Costa Rica (Las Baulas; NMFS and USFWS 2020b).

Given the proposed action and a projected 305 sets per year over a 5-year period (up to 573 sets in a given year) in the DGN fishery, we anticipate that up to two leatherback sea turtles could be entangled in the DGN fishery in any given year, although no more than two individuals would be entangled over the last five years of the DGN fishery. We anticipate that one of these may end up as a mortality or serious injury of an adult female (or equivalent), which equates to approximately 0.2 adult females removed from the population per year over the next five years. Based on genetic analyses of leatherbacks taken in the DGN fishery and captured for research in central California, all of the turtles originated from the West Pacific subpopulation. Therefore, we shall focus the majority of our integration and synthesis on this population, considering the effects of the DGN and status and environmental baseline. The prospect of removing up to one adult female in a year (over the next five years) represents less than 0.1 percent (0.095) of the total Western Pacific adult female population (1,443). Given an estimated 100,000 adults and juveniles in the West Pacific subpopulation, the loss of two individuals in a year or over five years represents 0.002 percent of the population.

We add the DGN effects to the status (and environmental baseline, albeit minimal threats to leatherbacks in the action area) of interactions and mortalities from other fisheries, including international and domestic fisheries occurring primarily outside of the U.S. West Coast EEZ. These fisheries and their effects are likely to continue and may increase over time due to the effects of increased human population and consumption of fish products, although the DGN fishery will only continue for another five years. However, we note that resolutions implementing sea turtle bycatch reductions (e.g., use of circle hooks and/or fish bait) throughout the western central Pacific Ocean have likely significantly reduced leatherback bycatch, depending on which countries are implementing the measures. Swimmer et al. (2017) found that the mean bycatch rates of leatherbacks in the Hawai'i-based shallow-set longline fishery declined by 84% for the post-regulation period, which has required large circle hooks and the use of mackerel-type bait since 2004.

Fisheries bycatch from pelagic longlining are the greatest known threat to Pacific leatherbacks, with hundreds of leatherbacks killed each year, but these estimates are uncertain. In the Western and Central Pacific Ocean, participating countries reporting to the Western and Central Fisheries Commission, given observed interactions from 2013 to 2020, produced an estimate of approximately 722 interactions per year, with an estimated 76 leatherback mortalities per year (this included a small portion of Hawai'i reports from the deep-set longline fishery described below). With low observer coverage in these international fleets (~3%), confidence in these estimates are low (i.e., only 18 leatherbacks were observed during that time period). Nonetheless, we have more confidence in our domestic longline fishery, given 100% observer coverage in the Hawai'i-based shallow-set fishery and approximately 20% observer coverage in the Hawai'i-based deep-set fishery and variable coverage in the American Samoa longline fishery. The most biological opinions indicate that approximately 48 leatherbacks are anticipated to be incidentally captured in these fisheries combined each year, with around 16 mortalities each year assuming historical mortality rates from the injuries that have occurred in the past. Thus, if we consider high 95% CI from the WCPFC estimates (mortality of 136 individuals per year), we can assume that around 150 leatherbacks may be removed from the population each year by pelagic longline fisheries.

In the South China and Sulu-Suluwesi Seas, less is known of leatherback interactions with coastal and artisanal fisheries. Traditional harvest of adult and subadult leatherbacks in the Kei Islands have been significantly reduced. In addition conservation efforts at the two major nesting beaches in Papua Barat have increased nest success rates and hatchling production, due in part to increased efforts to protect nests from predation, tidal inundation, erosion and high sand temperatures. Similar protections at Buru Island have reduced poaching and predation of nests and killing of nesting females. By 2022, less than 1% of nests were poached, and no nesting females were taken. Conservation efforts to address many of these threats have been significant, although measurable increases in nesting females may not be realized for some years, given the long-lived slow-to-mature nature of sea turtles.

Climate change may be affecting leatherbacks already but will likely increase in the future. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years and by one to four feet by 2100 (Wuebbles et al. 2017). This could affect migration and feeding patterns by changing ocean circulation. Leatherback turtles were predicted to gain core habitat area by Hazen et al. (2012). Such range shifts could affect foraging success and sea turtle reproductive periodicity (e.g., remigration intervals). Increased sand temperatures can also cause decreased egg survival and increase the proportion of female hatchlings, skewing sex ratios and affecting the reproductive capacity of the populations. Without aggressive and sustained conservation actions such as relocating nests threatened by rising seas and inundation, shading nests, and protecting nests from domestic and wild predators, as is being conducted in Papua Barat, hatchling success and the health of the nests will be compromised.

Using satellite tracking of post-nesting females and foraging males and females as well as genetic and stable isotope analyses indicate that leatherback found off the U.S. West Coast are from the Western Pacific nesting populations, specifically boreal summer nesters. Approximately 38-57 percent of summer-nesting females from Papua Barat migrate to distant foraging grounds off the U.S. West Coast, including the neritic waters off Central California. Using 28 years of aerial survey data, Benson et al. (2020) estimate that leatherback abundance off the U.S. West Coast has declined at an annual rate of 5.6%. Within the action, leatherbacks have been documented interacting with coastal pot/trap fisheries and the DGN fishery (albeit rarely), have been killed by vessel strikes, entrained in power plants (rarely), and taken through scientific research (i.e., pelagic trawls, hand captures, etc.). Off the U.S. West Coast, Pacific leatherbacks face much fewer threats than they do outside the action area.

Based on studies involving large-scale movements of leatherbacks into the California Current Ecosystem, fewer turtles would be found in the action area during the DGN fishing season, since leatherbacks have been documented typically arriving in the SCB in the springtime, traveling in the nearshore area as they approach the central/northern California areas (Benson et al. 2011). As described, the loss of one adult female (equivalent) and up to two individuals overall, represents a very small proportion of the total population. Locally, the potential mortality of individuals from a group of less than 200 individuals during a single year that may be foraging off the U.S. west coast would be more significant.

In this biological opinion, we consider that the DGN fishery is expected to occur each year for the next five years, after which the fishery will no longer be prosecuted and the potential effects of the DGN fishery will cease. Over this time period, we expect that the Western Pacific leatherback population may lose up to one adult female over the next five years as a result of this proposed action, and up to two individuals overall from the population.

The major index beaches of the Western Pacific population have been declining through 2017, at around -6%, indicating reproductive females were not being replaced as nesting counts continued to decline. The major threats identified to leatherbacks in this region are related to activities on nesting beaches (e.g., coastal erosion, feral pigs, environmental perturbations in the marine environment, directed take, and bycatch in fisheries. Conservation actions to address many of these threats has been significant, and there is optimism that some of the efforts may be beginning to show measurable increases in productivity on the nesting beaches, as suggested by recent nesting counts. This type of recovery, allowing for the lag in population dynamics for long-lived and slow-maturing species, has been shown by leatherback populations on other beaches such as St. Croix, USVI, where nesting females increased at 13 percent per year, following approximately 10 years after protection of nesting beaches (Dutton et al. 2005).

Previous consultations on the DGN fishery or similar actions that affect Western Pacific leatherbacks have considered the impact of small numbers of leatherback mortality. The 2004 biological opinion concluded that up to three deaths of leatherbacks per year was likely below a level that would appreciably affect survival and recovery (NMFS 2004b). This was supported by some demographic modeling simulation and the qualitative considerations of this small level of impact. Other actions looking at the effect of losing one female considered the prospect that conservation actions in recent years were likely to facilitate the chance that increases in young turtles would act as a buffer to provide more recruits into the adult population, in context with the very small level of impact expected (NMFS 2008). In the NMFS (2012b) biological opinion on the Hawai'i-based shallow-set longline fishery, two different modeling approaches considered the impact of annually removing four adult females from the population per year using analyses by Van Houtan (2011), and neither of these models offered evidence that an appreciable difference of relative extinction risk was detectable from the removal of four adult females. The 2013 biological opinion on the DGN fishery also concluded that the loss of one adult female per year from the DGN fishery presented negligible additional risk to survival and recovery of the western Pacific leatherback sea turtle population. Recent PVAs that have assessed the removal of leatherbacks in the Hawai'i and Samoa-based longline fisheries (Martin et al. 2020a, 2020b; Siders et al. 2023; NMFS 2023c, d) have yet to quantitatively conclude discernible changes in the risk of extinction to leatherbacks in the Western Pacific as a result of the levels of impact considered in those longline fisheries.

In comparison, expected impacts to this leatherback population from the DGN fishery currently estimated in this biological opinion are less (in some cases substantially) than what has been quantitatively analyzed in the models and analyses described above. As a result, we conclude the resultant impact of repeating any of these modeling exercises considering the removal of only more female over the next five years would predictably also conclude that no discernable risk of extinction would be evident from analysis of the impact of the DGN fishery over the next five years. These results are consistent in that none of the modeling or analytical results have yet to quantitatively describe changes in extinction risks to leatherback in the Pacific attributed to the type of low level of impact anticipated under this proposed action. While questions have been raised regarding the impacts of climate change on leatherback sea turtles, uncertainty remains related to future nesting beach forecasts and correlations with climate indices such as the PDO.

In order for the Western Pacific population of leatherback sea turtles to remain viable, it is reasonable to expect that the dominant factors currently (and historically for such a long-lived species) affecting survival must improve. As mentioned previously, leatherbacks are vulnerable to international fisheries across the Pacific which are likely responsible for the mortalities of hundreds of juvenile, sub-adult, and adult mortalities. In addition, there have been threats documented on the nesting beaches, including the directed harvest of adults and eggs, as well as other major threats to egg and hatchling survival from predators and coastal erosion. As described in the *Status of the Species* (section 2.2.5), significant conservation actions have been taken throughout the range of Western Pacific leatherbacks to address and reduce these threats from historical levels that were driving the significant population declines that have been documented. Recent data from the nesting beaches may be pointing to early signs that conservation actions are having some positive influence as survival rates appear to be improving.

In addition to the risk of extinction for leatherback populations, NMFS must also consider the impact of proposed actions on the prospects for recovery of ESA-listed species under the jeopardy standard. The NMFS and USFWS (1998b) recovery plan for leatherback sea turtles in the U.S. Pacific Ocean contains a number of goals and criteria that should be met to achieve recovery for this species. A number of these goals are being addressed through the research efforts determining stock structure of populations and monitoring their status, at least for populations that range into U.S. waters. It seems likely that any abundance goals for leatherback populations, including the Western Pacific Ocean, rest on the productivity of nesting beaches in concert with increased survival rates of individuals throughout their range and life-cycle.

In the face of these large threats and variation in natural and human-induced survivability rates, we conclude it does not seem reasonable or possible to detect how the small impact of the removal of an adult leatherback (or two at most) from the population has any effect on this trajectory for the foreseeable future. At some point, however, if the leatherback population does not stabilize and declines continued, the population will reach a critically low level where the fate of each and every individual has a significant influence on the survival and/or recovery of this population and the leatherback species as a whole. At this time, such a critical point has not been identified for this population or for any sea turtle populations in general.

The optimal chance of leatherback sea turtle recovery in the Pacific rests in the reproductive capability and the relatively high fecundity of sea turtles. Each female leatherback produces around 400 eggs each season they reproduce (Tapilatu and Tiwari 2007; Hitipuew et al. 2007; Dutton et al. 2007). Regardless of how many times a female does reproduce, only one out of all these offspring hatchlings needs to survive as an adult female to achieve replacement, although we should not discount the importance of male survival to ensure reproductive capacity into the future. The current sex ratio of this population has been estimated at 73 percent female. While skewed sex ratio could be a problem in general, it may also underlie the potential for relatively high productivity and population growth rates should other factors affecting survival across their life-cycle become more favorable. The mating system of sea turtles is both polyandrous (1 female fertilized by more than 1 male) and polygynous (1 male mates with more than 1 female) and occurs in areas where turtles congregate near natal home ranges (see Bell et al. 2010 review). Males from some sea turtle species have been found to return to waters adjacent to some nesting beaches more often than females, but it is unclear whether potentially reduced males due to climate change variability (hotter sand temperatures produce more female hatchlings) may impact the maintenance of breeding rates (Hays et al. 2010). It seems possible that fewer males than females may be needed for adequate mating, with the added benefit that increased percentage of females could lead to more nesting activity and egg production.

A recent study concluded that there was no evidence for depensation (reduced fertility due to small population size) for various green and loggerhead sea turtle populations that were examined, even for very small turtle populations (Bell et al. 2010). These factors suggest that recovery potential is definitely there for small turtle populations of turtles that are much smaller than the current Western Pacific leatherback population, and a number of small populations of turtles have shown signs of recovering fairly quickly after conservation efforts have been implemented (see Bell et al. 2010 for review). It has also been documented that much smaller populations of much less productive species have rebounded quickly given the right conditions (e.g., Mediterranean monk seals; Martinez-Jauregui et al. 2012).

The limited mortality of one adult female (equivalent) over the next five years, and up to two individuals overall, would present negligible additional risk to the chances of survival and recovery of the Western Pacific leatherback sea turtle population. After this time, the DGN will cease, and no further impact in terms of captured or killed leatherbacks will occur. Consequently, we would not expect the proposed activity to prohibit leatherback nesting populations from increasing or maintaining a stable population in perpetuity, nor would it substantially impair or prohibit increases to leatherback foraging populations at key foraging grounds. As a result, it seems unlikely that the effects of the proposed action on the survival and recovery of this population would be detected.

Given the best available information, we conclude that the limited incidental take and resulting mortality of up to two leatherback sea turtles associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of this species.

2.7.2.3 Olive Ridley Sea Turtles

In this biological opinion, we have identified that olive ridley sea turtles from the eastern Pacific nesting population are most likely to be affected by the DGN fishery occurring off the U.S. west coast. We anticipate that no more than one olive ridley may become entangled or captured in DGN gear in any year, although no more than one individual would be expected to be caught over the next five years (Table 13). It is possible this may result in a mortality or serious injury, so we will consider the worst case scenario that this take would lead to a removal from the population.

Olive ridley sea turtles from Mexican breeding populations are listed as endangered, although the available nesting data suggest the population is increasing substantially, presumably in response to the significant efforts to reduce nesting beach harvest across the region. Annual nesting in Mexico alone is estimated to be more than one million nests, with well over one million individual olive ridleys estimated to inhabit eastern tropical Pacific waters. Olive ridleys are generally a tropical species, and based on stranding records, they are likely more occasional visitors to the offshore water of the U.S. EEZ, and seem to only be very rarely encountered by most activities, including the rare, chance entanglement with DGN gear. Only one adult female olive ridley turtle was documented interacting with commercial or recreational fishing gear within the U.S. West Coast from 2017-2021. There are two state gillnet fisheries in California that may interact with sea turtles: the set gillnet fishery targeting halibut and white seabass; and the small mesh drift gillnet fishery targeting yellowtail, barracuda, and white seabass. No other olive ridley sea turtle interactions have been documented recently, although observer coverage of these fisheries has been limited and irregular. In a population that numbers over one million at a minimum, the loss of one individual, male or female, in any given year would not result in a detectable effect.

In addition to the risk of extinction, we must also consider the impact of proposed actions on the prospects for recovery of ESA-listed species under the jeopardy standard. The recovery tasks and goals identified by NMFS and USFWS (1998c) for U.S Pacific populations of olive ridley sea turtles are focused on the research and conservation activities that NMFS has been actively engaged in. As with other ESA-listed sea turtle species in the Pacific it seems likely that any abundance goals for populations, including the populations in the eastern Pacific, rest on factors of productivity and mortality throughout their range that are not likely to be affected by the removal of one adult female in the next five years. In any one year, this small impact will be insignificant to the future recovery potential of the species. In 2004 and 2013, we concluded that the periodic loss of one olive ridley as a result of the DGN fishery at that time was not likely to appreciably reduce the likelihood of survival or recovery of this species due to the low level of expected impact. Considering there is no evidence that the impact of the DGN fishery on olive ridley sea turtles has increased since that time, and the status of this species in the eastern Pacific has likely improved, the results of that analysis appear to remain valid.

Given the best available information, we conclude that the anticipated occasional incidental take and resulting mortality of olive ridley sea turtles associated with the proposed action is not likely to appreciably reduce the likelihood of survival or recovery of this species.

2.7.2.4 East Pacific DPS Green Sea Turtles

The East Pacific DPS green turtle is listed as threatened under the ESA. The IUCN (2021) assessed the East Pacific green sea turtles as “vulnerable,” resulting with the downlisting of their endangered status (IUCN 2021). Seminoff et al. (2015) ranked the DPS as having a low risk of extinction based on the abundance of nesting females. NMFS and USWFS are in the process of designating critical habitat for the East Pacific green sea turtle DPS within U.S. jurisdiction and both agencies shall propose a determination on or before June 30, 2023.

Given the abundance of nesting females in Mexico (13,664 nesters), Ecuador (3,603 females in the Galapagos and 15 on the mainland) and Costa Rica (2,826 females), Seminoff et al. (2015) estimated the adult female population to be 20,062 females. Given that this population is likely increasing owing in part to the significant conservation efforts around the region, NMFS recently estimated of over 3,500,000 individuals over one year old (NMFS 2023c).

In this biological opinion, we have identified that green sea turtles from the East Pacific DPS are most likely to be affected by the DGN fishery occurring off the U.S. west coast. We anticipate that no more than one green turtle may become entangled or captured in DGN gear in any year, although no more than one individual would be expected to be caught over the next five years (Table 13). It is possible this may result in a mortality or serious injury, so we will consider the worst-case scenario that this take would lead to a removal from the population.

Although the significance of the northern foraging aggregations off Southern California is not fully understood, it is possible that healthy and robust groups of green turtles living at the relative edge of their home range is indicative of a population showing some signs of recovery as opposed to being on the verge of extinction. Threats to green turtles within the U.S. west coast EEZ include occasional bycatch in some coastal fisheries and exposure to boating and vessel traffic, especially in dense population centers such as southern California. With the exception of occasional boat strikes and entrainment in power plants historically (section 2.4.2.2), we have not identified any other serious threat to the population of green turtles in the action area.

The potential bycatch and loss of one adult female green turtle in the DGN fishery during a year from a population of over 20,000 adult females in the East Pacific DPS equates to less than 0.01 percent (0.005 percent) of the population. We believe the effect on the population would be undetectable, particularly considering the natural variation in factors such as environmental productivity and survival rates for all sea turtles, including green turtles, in addition to the evidence of an increasing trend.

In addition to the risk of extinction, we must also consider the impact of proposed actions on the prospects for recovery of ESA-listed species under the jeopardy standard. The recovery tasks and goals identified by NMFS and USFWS (1998d) for eastern Pacific green sea turtles are focused on the research and conservation activities that NMFS has been actively engaged in. As with other ESA-listed sea turtle species in the Pacific it seems likely that any abundance goals for populations, including the populations in the eastern Pacific, rest on factors of productivity and mortality throughout their range that are not likely to be affected by the occasional removal of one adult female every few years. In any one year, this small impact will be insignificant to the future recovery potential of the species. In 2004 and 2013, we concluded that the periodic loss of one green turtle in the eastern Pacific as a result of the DGN fishery at that time was not likely to appreciably reduce the likelihood of survival or recovery of this species due to the low level of expected impact. Considering there is no evidence that the impact of the DGN fishery on green sea turtles has increased since that time, and the status of the East Pacific DPS has likely improved, the results of that analysis appear to remain valid.

Given the best available information, we conclude that the limited anticipated incidental take and resulting mortality of East Pacific DPS green sea turtles associated with the proposed action is not likely to appreciably reduce the likelihood of survival or recovery of this species.

2.7.3 Giant Manta Rays

In this biological opinion, we have identified that giant manta rays are most likely to be affected by the DGN fishery occurring off the U.S. West Coast. We anticipate that no more than one giant manta ray may become entangled or captured in DGN gear in any year, although no more than one individual would be expected to be caught over the next five years (Table 13). It is possible this may result in a mortality or serious injury, so we will consider the worst-case scenario that this take would lead to a removal from the population.

The abundance of the global population of giant mays, or the regional population that may be exposed and vulnerable to bycatch in the DGN fishery is unknown. Manta rays are listed as threatened under the ESA with limited known global distribution and population with a regional population size between 500 and 1,500 individuals. However, ongoing research including mark-recapture analyses suggests that typical subpopulation abundances are more likely in the low thousands (e.g., Beale et al. 2019) and in rare cases may exceed 22,000 in areas with extremely high productivity, such as in coastal Ecuador (Harty et al. 2022). Assuming the population that may be impacted by the DGN over the next five years may have around 1,000 individuals at least, this results in the potential removal of up to 0.1% of the total population, which is a very small impact on the population. We believe the effect on the population would be undetectable, particularly considering the natural variation in factors such as environmental productivity and survival rates, and that there will no continued impacts from the DGN fishery after the next five year period.

Since giant manta rays range in the Pacific Ocean includes equatorial tropical and sub-tropical water it does extend north to southern California where their range overlaps with the action area (Miller and Klimovich 2017). Due to their fragmented populations, low fecundity, and primary threat from commercial fishing, their likelihood of recovery is low (NOAA https://www.fisheries.noaa.gov/species/giant-manta-ray). While manta ray bycatch has been documented historically in the DGN fishery, it has been in low numbers and only during El Niño events (Miller and Klimovich 2017). Since 2000, only one giant manta ray has been observed caught in the California drift gillnet fishery (NMFS observer program data). We understand that those caught incidentally to the fishery may have been misidentified to other *Mobula* species that look similar (NMFS observer program). We consider the possibility that changes in the DGN fishery since the 1990s, that included large reductions in fishing effort, implementation of minimum extender lengths, and seasonal closures during El Nino conditions for loggerhead sea turtle protection, have also worked to minimize the risk of giant manta ray bycatch. The extended absence of giant manta rays from the observer record and the likelihood that there have been changes in the exposure of this species to the DGN fishery over the last two decades, we conclude that the risk of bycatch in the current DGN fishery is extremely low. Here we proceed with a conservative assessment of the bycatch of this species may occur over the next five years due to the uncertainty around the specifics whether any or all of these three preceding factors may have contributed to their reduced bycatch over the past two decades.

Given the best available information, we conclude that the limited incidental take and resulting mortality of giant manta rays associated with the proposed action is not likely to appreciably reduce the likelihood of survival or recovery of this species.

2.8 Conclusion

After reviewing and analyzing the current status of the listed species that may be affected by the proposed action, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of the following ESA-listed species: fin whales, Mexico DPS and Central America DPS humpback whales, sperm whales, leatherback sea turtles, North Pacific Ocean DPS loggerhead sea turtles, olive ridley sea turtles, East Pacific DPS green sea turtles, and giant manta rays.

2.9 Incidental Take Statement

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. "Harm" is further defined by regulation to include significant habitat modification or degradation that actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering (50 CFR 222.102). "Harass" is further defined by interim guidance as to "create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering." "Incidental take" is defined by regulation as takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the Federal agency or applicant (50 CFR 402.02). Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this ITS.
A marine mammal species or population which is listed as threatened or endangered under the ESA is, by definition, also considered a strategic stock and depleted under the MMPA. Section 7(b)(4) of the ESA provides for an incidental take statement for threatened and endangered marine mammals only if authorized pursuant to section 101(a)(5) of the MMPA. On May 11, 2022, NMFS issued a permit to authorize the incidental, but not intentional, take of ESA-listed marine mammal species or stocks under the MMPA, in the DGN fishery (87 FR 28811). This permit is effective for three years from the date of issuance.

2.9.1 Amount or Extent of Take

In the biological opinion, NMFS determined that incidental take is reasonably certain to occur as a result of continued operation of the DGN fishery off the U.S. West Coast within the current regulatory framework governing the fishery for five more years, anticipating that up to 573 sets could be made in one year, and the effort will average 305 sets per year over the next five years, as follows:

Table 1. Amount and extent of take on individuals expected in the DGN fishery. Expectations for maximum annual bycatch and total estimated bycatch (along with the number of mortalities or serious injuries) over the last five years of the DGN fishery.

ESA-listed species	Total bycatch (mortality or serious injury)
Fin whale	Up to 1 (1)
Humpback whale	Up to 1 (1)
Sperm whale	Up to 2 (2)
Leatherback turtle	Up to 2 (1)*
Loggerhead turtle	Up to 3 (1)*
Olive ridley turtle	Up to 1 (1)
Green turtle	Up to 1 (1)
Giant manta ray	Up to 1 (1)

*** one adult female equivalent**

The bycatch and mortality rates were estimated based on the record of recorded observations of ESA-listed species bycatch in the DGN fishery from historical data that is considered to be consistent with the manner of current and future operation of this fishery over the next five years, including the measures that have previously been implemented to avoid protected species interactions. Although this biological opinion acknowledges that there are underlying issues that could affect the reliability of estimating bycatch in the entire fishery from observer data, we have yet to identify any definitive gap in observer coverage that may be influencing or biasing bycatch estimates. The proposed action indicates that NMFS will continue target an observer coverage level of 20-30 percent for the last five years of the DGN fishery. While it is unlikely that observer coverage will equal target levels each year, we expect the target levels to represent the general average over the next five years, that is, in some years overall observer coverage levels may be slightly below or above target levels.

As described previously in this biological opinion, we will primarily rely upon bycatch estimates (both annually and over the next five years) generated using updated bycatch estimation methodology from Carretta (2022) each year. In the interim time lag while bycatch estimates are updated each year, we will also evaluate interaction rates in the context of less than 100 percent observer coverage of the DGN fishery. As a result, assuming the current observer coverage levels are maintained at similar levels, NMFS expects the observer record to comply with the anticipated incidental take described in Table 13. Exceedances of the level of take reported by fishery observers for any species likely will lead to a conclusion that take levels are higher across the entire DGN fishery than what has been assumed and analyzed in this biological opinion, unless observer coverage rate are dramatically changed.

Table 13 reflects the conclusion that anticipated bycatch could occur in the observed portion of the fishery, but that we would not expect to see a multitude or series of events where any of these ESA-listed species are caught consistently in the observed portion of the fishery given the historical observer data that suggests bycatch rates for these species are events that occur only a few times a year at most. The table also reflects that some low level of take is (typically) estimated for each ESA-listed species evaluated using the updated bycatch estimation methodology by Carretta (2022) during years when no takes are observed. As a result, we should not generally expect to see all anticipated takes in the observed portion of the DGN fishery with ~20-30 percent observer coverage. However, for species with very rare bycatch and anticipated take levels annually and over the next five years that are very small numbers, we accept the possibility that these takes could be observed. This table also acknowledges that we will likely not know the DPS of origin for any observed humpback whales taken in DGN fishery, and can only effectively track the bycatch of humpback whales as one species against this incidental take statement.

2.9.2 Effect of the Take

In the biological opinion, NMFS determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to result in jeopardy to the species or destruction or adverse modification of critical habitat.

2.9.3 Reasonable and Prudent Measures

"Reasonable and prudent measures" are nondiscretionary measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take (50 CFR 402.02). The following reasonable and prudent measures must be implemented to allow continued operation of the DGN fishery along the U.S. west coast.[27]

1. NMFS shall monitor the DGN fishery to ensure compliance with the regulatory and conservation measures included in the proposed action, including but not limited to collection and evaluation of data on the capture, injury, and mortality of ESA-listed and other protected species, as well as life history information for species that may interact with the DGN fishery.

2. NMFS shall provide training to DGN fishery vessel operators and observers on the status and biology of ESA-listed and other protected species, and on methods that may reduce their injury or mortality during fishing operations.

2.9.4 Terms and Conditions

In order to be exempt from the prohibitions of section 9 of the ESA, the Federal action agency must comply (or must ensure that any applicant complies) with the following terms and conditions. NMFS has a continuing duty to monitor the impacts of incidental take and must report the progress of the action and its impact on the species as specified in this ITS (50 CFR

[27] The reasonable and prudent measures and associated terms and conditions in this Opinion are applicable only to the DGN fishery. The other HMS fisheries remain subject to the reasonable and prudent measures and associated terms and conditions of the 2004 Biological Opinion on the HMS FMP and the 2016 Biological Opinion on the deep-set longline fishery that remain in effect for all the other fisheries covered under the HMS FMP.

402.14). If the entity to whom a term and condition is directed does not comply with the following terms and conditions, protective coverage for the proposed action would likely lapse.

1) The following terms and conditions implement reasonable and prudent measure No. 1.

1A. NMFS shall continue to maintain an observer program to collect and disseminate data on the incidental take of marine mammals, sea turtles, and other protected species. Reports summarizing protected species bycatch data collected for the DGN fishery shall be prepared and disseminated to the West Coast Region Protected Resources Division. Annual reports from each fishing season should be submitted to PRD by April 1st each year. Information on any ESA-listed species bycatch shall be reported to the PRD and the Office of Law Enforcement by SFD as soon as possible after verification of report. This information should include species, condition, date of interaction, and location. A copy of the observer report shall be provided to both offices, following review by SFD staff.

1B. NMFS shall continue to collect life history information on ESA-listed and other protected species that are taken as bycatch in the DGN fishery, including species identification, measurements, condition, skin biopsy samples, and the presence or absence of tags. If feasible, NMFS observers shall directly measure or visually estimate tail length on all sea turtles captured by DGN gear.

1C. NMFS collected data and other available information shall be submitted to PRD upon receipt of any reports of ESA-listed species interactions to inform determinations of whether observed or estimated takes of ESA-listed sea turtles and/or marine mammals has exceeded the level of anticipated take over the course of one fishing season, and/or the last five years of the DGN fishery, as described in Table 13 in the *Incidental Take Statement* section 2.9.1. SFD will also review the annual report of protected species bycatch and confer with PRD on the current status of protected species and any management concerns prior to beginning of the fishing season May 1.

1D. NMFS shall regularly review and evaluate incoming VMS data to facilitate the ability of NMFS to monitor compliance with time/area closures such as the PLCA, provide OLE the opportunity to deploy enforcement personnel to inspect vessels for compliance with take reductions measures such as proper use of pingers, provide NMFS an opportunity to deploy observers to monitor catch in alternative platform, and provide NMFS the ability to more closely examine and compare the distribution of observed and unobserved fishing effort, as long as unobserved effort continues in the fishery.

2) The following terms and conditions implement reasonable and prudent measure No. 2.

2A. NMFS shall continue to provide DGN skipper education workshops, required for skippers of DGN vessels upon notification from NMFS as described in 50 CFR 229.31 (d), with a module on sea turtle handling, resuscitation, and release requirements, as outlined in 50 CFR § 223.206(d)(l), as well as appropriate handling and release procedures for marine mammals.

2B. NMFS shall also include in skipper education workshops a module of information on sea turtle biology and methods to avoid and minimize sea turtle impacts.

2.10 Conservation Recommendations

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Specifically, conservation recommendations are suggestions regarding discretionary measures to minimize or avoid adverse effects of a proposed action on listed species or critical habitat or regarding the development of information (50 CFR 402.02). The following conservation recommendations are provided for developing management policies and regulations, and to encourage multilateral research efforts, which would help in reducing adverse impacts to listed species in the in the DGN fishery and throughout the Pacific Ocean.

1. Unless or until additional requirements for 100% observer coverage using EM or other means are implemented, NMFS should seek to obtain as much observer coverage in the DGN fishery in excess of the current observer coverage target rate as resources and the ability to deploy observers on DGN fishing vessels will allow. NMFS should also explore options for reducing the amount of effort that is conducted by unobservable vessels.

2. NMFS should evaluate the utility and feasibility of additional or improved gear marking requirements for gillnet fisheries that could enhance the ability of NMFS to identify (or eliminate from further consideration) the origin of entanglements that may involve gillnets that are reported to the NMFS WCR Marine Mammal Stranding Network. This evaluation may be used as the basis of any recommendations regarding gear markings made to the Council or State agencies that may be interested, as well as internationally to Regional Fishery Management Organizations.

3. NMFS should explore the feasibility of using biological and oceanographic modeling outputs/measures to determine when ESA-listed marine mammals and sea turtles may be in an area where the DGN fishery occurs. For example, EcoCast (https://coastwatch.pfeg.noaa.gov/ecocast/about.html) was developed to help fishers and managers evaluate how to allocate fishing effort to maintain target fish catch while minimizing bycatch of protected or threatened species. However, implementation of this tool within the DGN or other potentially relevant fishery contexts has been limited. NMFS should work to develop specific utilities and guidance/directives for how to take advantage of near-real time ecological information to protect ESA-listed species while promoting sustainable fisheries.

4. NMFS should continue to promote the reduction of marine mammal and sea turtle bycatch in Pacific fisheries by supporting:

a. The Inter-American Convention for the Protection and Conservation of Sea Turtles

b. Any binding Regional Fishery Management Organizations’ marine mammal and sea turtle conservation, mitigation, and management measures for commercial fisheries operating in the eastern Pacific Ocean

c. Technical assistance workshops and research to assist other nations in reducing marine mammal and sea turtle bycatch in DGN.

d. A trans-Pacific international agreement that would include relevant Pacific Rim nations for the conservation and management of marine mammal and sea turtle populations.

2.11 Reinitiation of Consultation

This concludes formal consultation on the continued operation of the DGN fishery under the HMS FMP for five more years. Under 50 CFR 402.16(a): "Reinitiation of consultation is required and shall be requested by the Federal agency or by the Service where discretionary Federal agency involvement or control over the action has been retained or is authorized by law and: (1) If the amount or extent of taking specified in the incidental take statement is exceeded; (2) If new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) If the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in the biological opinion or written concurrence; or (4) If a new species is listed or critical habitat designated that may be affected by the identified action." In addition to the limits regarding observed or total takes of ESA-listed species specified in the incidental take statement of this biological opinion, there may be a need to reinitiate consultation based on any information related to the structure or conduct of the DGN fishery that raise concern or have implications regarding possible impacts to ESA-listed species and/or our ability to detect them that have not been evaluated in this biological opinion. This may include, but is not limited to, the amount and distribution of fishing effort that occurs, the deployment of fisheries observers, and development or implementation of other management measures in the DGN fishery.

2.12 "Not Likely to Adversely Affect" Determinations

The following ESA-listed species and designated critical habitats are not expected to be adversely affected by the proposed action, for the reasons explained below.

2.12.1 Southern Resident DPS Killer Whale

One killer whale was observed taken in the DGN fishery in early November 1995 south of Monterey Bay and prior to the implementation of the PCTRP and the PLCA. The animal was identified as a West Coast transient killer whale (formerly recognized as Eastern North Pacific transient killer whale stock), which is not listed under the ESA. It is reasonable to assume that any killer whales that interact with the DGN fishery would either be a transient or offshore killer whale and not an endangered Southern Resident DPS killer whale (SRKW) based on the number, distribution and behaviors of these three sub-species.

There are three ecoptypes of killer whales; transients, offshores, and residents - and each can be distinguished genetically, morphologically and behaviorally. Transients are most common worldwide and generally prey on marine mammals. Less is known about offshores, although they appear to be opportunistic feeders. Residents are generally piscavores and maintain stable family units and are often "resident" to a specific area. Along the U.S. West Coast, it has been estimated that there are at least 349 West Coast transients (Muto et al. 2021) and 276 offshore (Carretta et al. 2022a) killer whales, compared to 72 SRKWs (Carretta et al. 2022a).[28] Based on the relative population sizes, it is likely that any killer whale interaction with DGN gear would be a transient or offshore killer whale.

As noted previously, the majority of effort in the DGN occurs within southern California which is beyond the observed range of SRKWs. Offshore and transient killer whales have been observed along the entire U.S. West Coast, including southern California. SRKW spend a substantial amount of time within the inland water of waters of Washington state and Vancouver Island, Canada. There are a number of whale watch vessels along the U.S. West Coast and there have been reports of SRKWs off the coast of Northern and Central California in the winter and spring, but no whale watch company has reported seeing SRKWs south of Monterey Bay. This general trend in distribution is supported by recent tagging work. In order to better understand the winter distribution of the SRKWs, in early 2012, the NWFSC began satellite tagging individual whales from the SRKW. There have been limited tracks, but whales tracked into California waters have not traveled south of Monterey Bay (Carretta et al. 2022a). Based on the relative number of SRKWs and their distribution, it is extremely unlikely that any killer whales that interact with the DGN off of California would be a SRKW.

There is a potential for overlap of DGN fishing effort off of Washington and Oregon with killer whale distribution, but given the very low fishing effort in the area and the fact that offshore and transients significantly outnumber SRKWs, it is considered extremely unlikely that any interactions between the DGN fishery and killer whales would be with SRKWs. As a result, NMFS concludes the proposed action is not likely to adversely affect SRKWs.

2.12.2 Western North Pacific Gray Whales

Gray whales are presently recognized as two populations in the North Pacific Ocean and recent genetic studies using both mtDNA and nuclear markers have demonstrated significant differentiation between the western North Pacific (WNP) and eastern North Pacific (ENP) populations (LeDuc et al. 2002; Lang et al. 2011; Weller et al. 2013). The WNP gray whales are listed as endangered under the ESA. ENP and WNP gray whales were once considered geographically separated along either side of the ocean basin, but recent photo-identification, genetic, and satellite tracking data indicate WNP gray whales may be accompanying ENP gray whales along their U.S. West Coast migrations. Information from tagging, photo-identification and genetic studies show that some whales identified in the WNP off Russia have been observed in the ENP, including coastal waters of Canada, the U.S., and Mexico (Lang 2010; Weller et al. 2012; Mate et al. 2015; Urbán et al. 2019). Photographs of 379 individuals identified on summer feeding grounds off Russia (316 off Sakhalin; 150 off Kamchatka) were compared to 10,685 individuals identified in Mexico breeding lagoons, with a total of 43 matches found (Urbán et al. 2019). The number of whales documented moving between the WNP and ENP represents 14% of gray whales identified off Sakhalin Island and Kamchatka according to Urbán et al. (2019).

[28] Recent census data by the Center for Whale Research is that the population stands at 72 whales as of July, 2022. https://www.whaleresearch.com/orca-population

Cooke et al. (2018) note that the fraction of the WNP population that migrates to the ENP is estimated to be 45-80%.

In the fall, gray whales migrate from their summer feeding grounds, heading south along the coast of North America to spend the winter in their breeding and calving areas off the coast of Baja California, Mexico. Calves are born in shallow lagoons and bays from early January to mid-February. From mid-February to June, gray whales can be seen migrating northward with newborn calves along the West Coast of the U.S. The timing of the majority of effort in the DGN fishery overlaps with the gray whale southbound migration along the U.S. west coast (November to February), but there are a number of fishing restrictions during this time that may limit the overlap between migrating gray whales and DGN fishing. From August to November 15, fishing may occur outside of the PLCA, typically south of Point Conception; from December 15 to January 31, the fishery is restricted to areas outside of 25 nm from the coastline; fishing is closed in the California EEZ from Feb 1-April 30; and from May 1-August 14, the fishery is restricted to outside of 75 nm from the coastline. Northbound gray whales, which include all age classes, migrate from February to June and therefore are not expected to overlap with any DGN fishing. Since 2001, from November 1 to January 31, approximately half of the total DGN fishing effort occurs and is concentrated south of Point Conception, with the exception of some limited effort that occurs just north of Pt. Conception or inside the PLCA that opens in mid-November. Southbound gray whales typically migrate within 10 kilometers from shore during the southbound migration, but some individuals have been observed farther offshore, usually less than 50 kilometers from the coastline. In the SCB, gray whales do travel around and through the Channel Islands, in addition to a migratory route in between the mainland and the Channel Islands.

From 1998 to 2021, a total of five gray whales have been observed by the NMFS fishery observer program interacting with the DGN fishery, including the most recent case in 2018 (Carretta 2022). Historically, the assumption has been that these whales were ENP gray whales: one in 1998 (alive); one in 1999 (dead); one in 2005 (alive), one in 2013 (dead), and one in 2018 (dead). Although the total documented interactions with DGN gear may be a minimum, as some interactions may have been unobserved, the likelihood that a gray whale would interact with the DGN fishery is low. Historically, records suggest that gray whale strandings have been commonly associated with gillnet gear, although no positive identification of DGN gear on entangled gray whales can be made from those records outside the observer program (summary of review in section 2.5.1.3 *Review of Net Entanglements*). Using the latest methodology for estimating gray whale bycatch in the DGN fishery, we predict that up to two gray whales may be incidentally taken in the DGN fishery over the next five years, or about 0.23 gray whales per year (Table 14).

Table 14. Estimated gray whale bycatch in the DGN fishery.

Species	Total DGN Fishing Effort (sets), 2001-2021	Total Estimated Bycatch, 2001 through 2021	Estimated Maximum Annual Bycatch (573 sets)	Estimated Average Annual Bycatch (305 sets)	Estimated Total Bycatch Over 5 Years
Gray whale	16,634	12.8	0.44	0.23	2

Based on tagging data, we assume that when WNP gray whales migrate along the coast of North America to Baja California, they are likely slightly delayed from the ENP's "start date" by at least a couple of weeks based on distance and average swim speed (i.e.*,* they have to swim from Sakhalin Island, Russia before joining the ENP route). The first migratory ENP gray whales can be observed in California as early as October, depending on the year, but mid-to late November is typical and approximately 10 percent of the population is expected to have made the migration by the end of December. Thus, it is possible that a WNP gray whale's migratory route could overlap with the DGN fishing area, particularly from November to January during the southbound migration and most likely in the SCB region, based on the distribution of DGN fishing effort in that area. However, there is no evidence indicating that WNP gray whales behave differently than an ENP whale and are more susceptible to interaction with the DGN fishery.

The estimated population size from photo-ID data for Sakhalin and Kamchatka in 2016 was 290 whales (90% percentile intervals = 271 – 311; Cooke et al. 2018). Systematic counts of gray whales migrating south along the central California coast have been conducted by shore-based observers at Granite Canyon most years since 1967. The current minimum population estimate for non-ESA-listed ENP gray whales is 26,960 (Carretta et al. 2022a). The most recent minimum estimate of endangered WNP gray whale abundance is 271 individuals (Carretta et al. 2022a). At any given time during the migration, WNP gray whales could be part of the approximately 27,000 gray whales migrating through the CCE. However, the probability that any gray whale observed along the U.S. west coast would be a WNP gray whale is extremely small, i.e., less than 1%, even if the entire population of WNP gray whales were part of the annual gray whale migration in the ENP. In addition, the DGN fishery will only last for 5 more years, and the chance that either of the up to two gray whales that might be entangled in DGN during this last 5 years belongs to the WNP gray whale population is extremely small. As a result, we conclude the risk of WNP bycatch is discountable, and that WNP gray whales are not likely to be adversely affected by the proposed action.

2.12.3 Other ESA-listed Species

The following marine mammal species may be in the action area, but have never been observed interacting or entangled with DGN gear: blue whale, sei whale, North Pacific right whale, and Guadalupe fur seal. It is possible that some species, such as blue and sei whales, may occasionally encounter DGN gear. Due to their large size, however, these species are capable of bursting through nets making it unlikely that they would be observed entangled. A review of stranding and entanglement records do not indicate that positive identification of DGN gear on any of these whale species can be made from those records outside the observer program (summary of review in section 2.5.1.3 *Review of Net Entanglements*). North Pacific right whales are rarely found off the U.S. west coast and have primarily been documented foraging in the Bering Sea and the Gulf of Alaska, where critical habitat was designated in 2006. Guadalupe fur seals are known to occur near Guadalupe Island, Mexico, their primary breeding area, and more recently have been observed breeding in small numbers on San Miguel Island (NMFS-Alaska Fisheries Science Center (AFSC) unpublished data). In recent years, Guadalupe fur seals have been increasing in numbers in the Channel Islands and several strandings have been observed along central California coast, and in 2015 a UME was declared, which lasted through 2021.[29] While some of these recent strandings along the U.S. west coast have involved entanglements with fishing gear that includes gillnets (Norris et al. 2017), none of these have been identified as likely to have come from any specific fishery from U.S., Mexico, or anywhere else. Without documented records of DGN bycatch of these ESA-listed marine mammals and given the information described above, we conclude that these ESA-listed species are not likely to be adversely affected by the proposed action.

There are a number of ESA-listed fish species in the proposed action area, including: some salmonids, steelhead, eulachon, and green sturgeon. None of these species are likely to be caught in DGN gear, due to the relatively large size of mesh used (typically 18 inches). Also, the distribution of the DGN fishery is more offshore and southerly focused than the distribution of most of these species, including salmonids, eulachon, and green sturgeon. The eastern Pacific DPS of scalloped hammerhead sharks was listed as endangered in 2014. The range of this DPS does extend up into southern California, although the primary habitat for scalloped hammerhead sharks is found in waters warmer than 22°C south and west of the U.S. EEZ and throughout the Eastern Tropical Pacific region (78 FR 20718). Groupers are bottom-associated fishes found in tropical and subtropical oceanic waters and are commonly associated with coral reefs, estuaries, and rocky reefs (Heemstra and Randal 1993), and gulf grouper typically reside in reefs and seamounts in waters up to 30 - 45 m deep (Dennis 2015). The DGN does not typically occur in shallow waters near reefs where groupers may occur, and no gulf grouper have been observed caught in the DGN fishery (NMFS observer program data).

The bycatch of scalloped hammerhead sharks has never been documented in the DGN fishery by fisheries observers. From 1990-2012, a total of 50 hammerhead sharks have been observed caught in the DGN fishery, but none have been identified as a scalloped hammerhead (78 FR 20718). More recently, 28 hammerhead sharks have observed caught in the DGN fishery since 2012 (27 during the 2014/2015 fishing season) but all have been identified as smooth hammerhead sharks (NMFS observer program data[30]). While the range of the oceanic whitetip in the Eastern Pacific is noted as extending as far north as southern California waters, based on the available data, the distribution of the species appears to be concentrated in areas farther south,

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[29] https://www.fisheries.noaa.gov/national/marine-life-distress/2015-2021-guadalupe-fur-seal-and-2015-northern-fur-seal-unusual

[30] http://www.westcoast.fisheries.noaa.gov/fisheries/wc_observer_programs/sw_observer_program_info/data_summ_report_sw_observer_fish.html

and in more tropical waters (Young et al. 2017). Oceanic whitetip sharks have not been recorded by observers in the DGN fishery (NMFS observer program data).

Given that the available information and lack of documented bycatch in the current DGN fishery for all of the ESA-listed fish species described above suggests that the risks of the proposed action for these species are discountable, we conclude the proposed action is not likely to adversely affect any of these species.

White and black abalone are species confined to the sea floor bottom and would not be encountered by DGN gear operating in the mid-water column. As a result, we conclude the proposed action is not likely to adversely affect these species.

2.12.4 Humpback Whale Critical habitat: Central America and Mexico DPSs

Critical habitat for the endangered Western North Pacific DPS, Central America DPS and the threatened Mexico DPS of humpback whales was proposed for specific marine areas located off the coasts of California, Oregon, Washington, and Alaska on October 9, 2019 (84 FR 54354). In April 21, 2021(86 FR 21082) a final rule was published on May 21, 2022, to establish their critical habitat that has been designated for three DPS including Western North Pacific DPS, Central America DPS, and Mexico DPS. Within the geographic areas occupied by these DPS of humpback whales, the critical habitat review team (CHRT) identified nine specific areas ~~of~~ marine habitat for the Western North Pacific DPS, nine specific areas of marine habitat for the Central America DPS, and 19 specific areas of marine habitat for the Mexico DPS - all of which contain the identified essential prey feature (NMFS 2020c).

Specific areas designated as critical habitat for the Western North Pacific DPS of humpback whales contain approximately 59,411 nmi^2 of marine habitat in the North Pacific Ocean, including areas within the eastern Bering Sea and Gulf of Alaska. Specific areas designated as critical habitat for the Central America DPS of humpback whales contain approximately 48,521 nmi^2 of marine habitat in the North Pacific Ocean within the portions of the California Current Ecosystem off the coasts of Washington, Oregon, and California. Specific areas designated as critical habitat for the Mexico DPS of humpback whales contain approximately 116,098 nmi^2 of marine habitat in the North Pacific Ocean, including areas within portions of the eastern Bering Sea, Gulf of Alaska, and California Current Ecosystem.

The CHRT identified a prey biological feature that is essential to the conservation of the two humpback whale DPSs, defined as follows: "prey species, primarily euphausiids and small pelagic schooling fishes of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth." Humpback whales that may be affected from the action and within the action area include both Central America and Mexico DPSs, who travel to U.S. coastal waters to access energy-rich feeding areas, and a high degree of fidelity to specific locations indicates the importance of these feeding areas. Although humpback whales are generalist predators and prey availability can vary seasonally and spatially, substantial data indicate that the humpback whales' diet within the California Current marine ecosystem, which extends from British Columbia to southern Baja California Mexico, includes:

Pacific sardine (*Sardinops sagax*); northern anchovy (*Engraulis mordax*); Pacific herring (*Clupea pallasii*); euphausiids (specifically Euphausia, Thysanoessa, Nyctiphanes, and Nematoscelis) and occasionally juvenile rockfish (Sebastes) (Appendix A of NMFS 2020c). Humpback whales are also known to switch between target prey depending on what is most abundant or of the highest quality in the system; thus, their diet composition may vary spatially and temporarily. Because humpback whales only rarely feed on breeding grounds and during migrations, humpback whales must have access to adequate prey resources within their feeding areas to build up their fat stores and meet the nutritional and energy demands associated with individual survival, growth, reproduction, lactation, seasonal migrations, and other life functions. Essentially, while on feeding grounds, the whales must finance the energetic costs associated with migration to breeding areas, reproductive activities, as well as the energetic costs associated with their return migration to high-latitude feeding areas (NMFS 2020c).

The DGN fishery uses a large mesh (typically around 18-20 inches measured diagonally knot-to-knot) that is too big to capture and/or retain any of the primary prey species for humpback whales. None of these primary prey species are known to be common or even occasional bycatch in the DGN fishery. As a result, the continued operation of the DGN for the next five years is not expected to have any significant effects to humpback whale prey, and we conclude the proposed action is not likely to adversely affect designated critical habitat for Central America and Mexico DPS humpback whales that occurs within the action area.

2.12.5 Leatherback Critical Habitat

Critical habitat was designated off the U.S. west coast for leatherback sea turtles (77 FR 4170, January 26, 2012), which does include areas that are seasonally open to the DGN fishery off the central coast of California. In the final rule, NMFS identified one primary constituent element essential for the conservation of leatherbacks in marine waters off the U.S. West Coast: the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (e.g., *Chrysaora, Aurelia, Phacellophora,* and *Cyanea*), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks. However, the critical habitat designation does not specifically define or develop standards or measurable criteria for any of these particular aspects of prey occurrence. Observers in the DGN fishery do occasionally report the bycatch of invertebrate species (Table 3 *in* Larese and Coan 2008). This bycatch includes pelagic tunicates (likely salps) and other unidentified invertebrate species, presumed to be passively snagged by these nets as the gear drifts in the water or is being hauled through the water and onboard the net reel of fishing vessels. The fate of these species upon release is very difficult to judge. The critical habitat designation emphasizes that the preferred prey of leatherbacks off the California coast is jellyfish, with other gelatinous prey, such as salps (a pelagic tunicate), considered of lesser importance (77 FR 4170). While jellyfish bycatch may occur in the DGN fishery, the extent is believed to be rare and cannot be quantified (NMFS 2012c). In addition, significant portions of the designated critical habitat are not open to DGN during the PLCA restriction when leatherbacks would be expected to be foraging on prey (i.e., summer and early fall). As a result, we do not anticipate any significant effects of the proposed action to designated critical habitat for leatherback sea turtles. Therefore, we conclude designated critical habitat for leatherback sea turtles is not likely to be adversely affected by the proposed action.

2.12.6 Southern Resident Killer Whale Critical Habitat

The critical habitat for the endangered Southern Resident killer whale DPS (SRKW; *Orcinus orca*) under the ESA was established on November 29, 2006, (71 FR 69054). On August 2, 2021, (86 FR 41668) this was revised to include the designation of six additional coastal critical habitat areas along the U.S. West Coast to establish their critical habitat effective on September 1, 2021. These newly designated areas along the U.S. West Coast include 41,207 square kilometers (km^2) of marine waters between the 6.1-meter (m) depth contour and the 200-m depth contour from the U.S. international border with Canada south to Point Sur, California with the exclusion of the Quinault Range Site (including a 10-km buffer around a portion of the site), comprising 3,627 km^2, from the critical habitat designation because we have determined that the benefits of exclusion outweigh the benefits of inclusion, and exclusion will not result in extinction of the species.

NMFS' final Biological Report (2021e) identified the following physical and biological features essential to the conservation of Southern Resident killer whales: 1) Water quality to support growth and development; 2) Prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and, 3) Passage conditions to allow for migration, resting, and foraging.

SRKW critical habitat may be affected by the action within the portion of the action area where designated critical habitat and the proposed action overlap within the 6.1 and 200 m depth contours of Oregon and California. The risk of overlap is very small, as the DGN fishery is typically prosecuted in offshore water much deeper than 200 m. There are not many active participants in the fishery, and most of the fishery occurs offshore Southern California which is not associated with SRKW critical habitat. In addition, Chinook salmon, the preferred prey of SRKW, are not known to be common or even occasional bycatch in the DGN fishery. As a result, the continued operation of the DGN for the next five years is not expected to have any significant effects to SRKW whale prey or other essential physical and biological features. Consequently, we conclude the proposed action is not likely to adversely affect designated critical habitat for SRKW.

2.12.7 Other Critical Habitat

Critical habitat for salmonids and eulachon on the U.S. west coast has not been designated in the marine environment, and consequently does not overlap with the DGN fishery. Green sturgeon critical habitat has been designated within marine waters out to 60 fm (110 meters) depth along and isobath from Monterey Bay to the U.S.-Canada border. The DGN fishery does not operate within the nearshore coastal waters where this critical habitat has been designated, and would not be expected to impact any of the important habitat features if it did. Black abalone critical habitat has been designated along intertidal and nearshore areas where the DGN cannot physically operate. As a result of the lack of occurrence of the DGN or potential impacts to their designated critical habitats, we conclude that salmonid, eulachon, green sturgeon and black abalone critical habitats will not be affected by the proposed action.

3. DATA QUALITY ACT DOCUMENTATION AND PRE-DISSEMINATION REVIEW

The Data Quality Act (DQA) specifies three components contributing to the quality of a document. They are utility, integrity, and objectivity. This section of the opinion addresses these DQA components, documents compliance with the DQA, and certifies that this opinion has undergone pre-dissemination review.

3.1 Utility

Utility principally refers to ensuring that the information contained in this consultation is helpful, serviceable, and beneficial to the intended users. . The intended users of this opinion are federal and state agencies and members of the PFMC that are directly involved in management the DGN fishery, as well as fishermen who participate in the DGN fishery that may be affected by the outcomes of the proposed action and this biological opinion. Other interested users include non-governmental organizations that monitor fishery management along the U.S. west coast, including regulatory actions that affect the DGN fishery. We anticipate that users will understand the impact of the drift gillnet fishery on ESA-listed species and the approach taken by NMFS to manage commercial fishing impacts on ESA-listed species. The document will be available within 2 weeks at the NOAA Library Institutional Repository [https://repository.library.noaa.gov/welcome]. The format and naming adhere to conventional standards for style.

3.2 Integrity

This consultation was completed on a computer system managed by NMFS in accordance with relevant information technology security policies and standards set out in Appendix III, 'Security of Automated Information Resources,' Office of Management and Budget Circular A-130; the Computer Security Act; and the Government Information Security Reform Act.

3.3 Objectivity

Information Product Category: Natural Resource Plan

Standards: This consultation and supporting documents are clear, concise, complete, and unbiased; and were developed using commonly accepted scientific research methods. They adhere to published standards including the NMFS ESA Consultation Handbook, ESA regulations, 50 CFR 402.01 et seq., and the MSA implementing regulations regarding EFH, 50 CFR part 600.

Best Available Information: This consultation and supporting documents use the best available information, as referenced in the References section. The analyses in this opinion contain more background on information sources and quality.

Referencing: All supporting materials, information, data and analyses are properly referenced, consistent with standard scientific referencing style.

Review Process: This consultation was drafted by NMFS staff with training in ESA, and reviewed in accordance with West Coast Region ESA quality control and assurance processes.

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