ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency: National Marine Fisheries Service, Greater Atlantic Regional Fisheries Office, through its Sustainable Fisheries Division

Activity: Endangered Species Act Section 7 Consultation on the:
(a) Authorization of the American Lobster, Atlantic Bluefish, Atlantic Deep-Sea Red Crab, Mackerel/Squid/Butterfish, Monkfish, Northeast Multispecies, Northeast Skate Complex, Spiny Dogfish, Summer Flounder/Scup/Black Sea Bass, and Jonah Crab Fisheries and
(b) Implementation of the New England Fishery Management Council's Omnibus Essential Fish Habitat Amendment 2
[Consultation No. GARFO-2017-00031]

Consulting Agency: National Marine Fisheries Service, Greater Atlantic Regional Fisheries Office, through its Protected Resources Division

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Approved by: ________________________________
Michael Pentony
Regional Administrator

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Correction Note

In estimating interactions with and serious injury/mortality of North Atlantic right whales (page 217 of the Biological Opinion), we stated “In one case, the gear was described as a bridle, configuration unclear. Bridles are used in trap/pot fisheries.” This case was described in the database as "Initially with open bridle, changed to full configuration unclear, likely a closed bridle." When describing fishing gear, bridle (or becket) refers the line attached to the trap in the shape of a triangle. It is tied into each corner on the short width side of the trap. The gangion ties into this bridle, and the other end of the 6-foot or so gangion is tied or spliced into the ground line. Bridle is also used to refer to the configuration of line on an entangled animal. An open bridle is when line(s) of gear run through the mouth, exiting both sides and trail along each side of the animal without any rostrum wraps; a closed bridle is when line(s) of gear run through the mouth, exiting both sides and rejoin together without any rostrum wraps. For the case referenced on page 217, we misinterpreted the use of the term bridle to refer to the former when it was the latter. With this note, we are reclassifying this one case from trap/pot to unknown line gear.

As described in the Biological Opinion, all other cases when gear was present and the entanglement case was classified as unknown gear were as described as lines, sometimes with associated buoys or polyballs. Without identifying marks, we cannot know whether the line is from gillnet gear, trap/pot gear, or another source. Similarly, since bridle in this case referred to the configuration of the line rather than the gear itself, we cannot know whether the gear was pot/trap gear as originally stated in the Biological Opinion. However, as described in the Biological Opinion: (1) The records indicate line was the predominant gear involved in cases with unknown gear and the majority of the cases involved unknown gear; (2) interactions with net panels may result in less severe injuries as the animal may be able to break free from the gear; and (3) interactions with vertical lines are more likely to be trap/pot gear given the co-occurrence of the right whales and trap/pot gear. Based on this, we determined that it was reasonable to apportion unknown mortality and serious injury to trap/pot gear. The reclassification of this one case from trap/pot to unknown line gear does not change this conclusion.

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List of Abbreviations and Acronyms

ACA	Atlantic Coastal Fisheries Cooperative Management Act
ACL	annual catch limit
ACOE	Army Corps of Engineers
ACT	annual catch target
ALWDN	Atlantic Large Whale Disentanglement Network
ALWTRP	Atlantic Large Whale Take Reduction Plan
ALWTRT	Atlantic Large Whale Take Reduction Team
AM	accountability measure
AMAPPS	Atlantic Marine Assessment Program for Protected Species
AOC	area of occurrence
ARA	Assistant Regional Administrator
ASM	at-sea monitor
ASMFC	Atlantic States Marine Fisheries Commission
ASSRT	Atlantic Sturgeon Status Review Team
ATBA	area to be avoided
AUV	autonomous underwater vehicle
BDTRP	Bottlenose Dolphin Take Reduction Plan
BOEM	Bureau of Ocean Energy Management
CA	closed area
CB	Chesapeake Bay
CCB	Cape Cod Bay
CCL	curved carapace length
CCS	Center for Coastal Studies
CFR	Code of Federal Regulations
CGOM	central Gulf of Maine
CI	confidence interval
CLF	Conservation Law Foundation
CMP	coastal migratory pelagics
COASTSPAN	Cooperative Atlantic States Shark Pupping and Nursery
COLREG	International Regulations for Preventing Collisions at Sea
COP	construction and operation plan
CPH	confirmation of permit history
CPUE	catch per unit effort
CSE	conservation spawning escapement
CV	coefficient of variation
DAM	dynamic area management
DAS	days-at-sea
DFO	Department of Fisheries and Oceans, Canada
DHRA	dedicated habitat research area
DMA	dynamic management area
DMIS	data matching and imputation system
DPS	distinct population segment
DST	decision support tool
DWH	Deep Water Horizon
EEAA	exempted educational activity authorization

EEZ	exclusive economic zone
EFH	essential fish habitat
EFP	exempted fishing permit
EGI	East Greenland and West Iceland
EGOM	eastern Gulf of Maine
EOO	extent of occurrence
EPA	Environmental Protection Agency
EPAct	Energy Policy Act of 2005
EPR	eggs per recruit
EPU	ecological production unit
ESA	Endangered Species Act
FERC	Federal Energy Regulatory Commission
FKNMS	Florida Keys National Marine Sanctuary
FLUPSY	floating upweller system
FMP	fishery management plan
FR	Federal Register
FWRI	Fish and Wildlife Research
GAM	generalized additive model
GAR	Greater Atlantic Region
GARFO	Greater Atlantic Regional Fisheries Office
GB	Georges Bank
GCRU	Greater Caribbean Recovery Unit
GLM	generalized linear model
GOM	Gulf of Maine
GPS	global positioning system
GSC	Great South Channel
HAPC	habitat areas of particular concern
HMA	habitat management area
HPTRP	Harbor Porpoise Take Reduction Plan
IEc	Industrial Economics, Inc.
IHA	incidental harassment authorization
IMO	International Maritime Organization
INBS	index nesting beach survey
IPCC	Intergovernmental Panel on Climate Change
ISFMP	interstate fishery management plan
ITS	incidental take statement
IUCN	International Union for the Conservation of Nature
km	kilometer
kN	kilonewton
lb	pound
lbf	pounds-force
LMA	lobster management area
LOA	letter of authorization
LOF	List of Fisheries
m	meter
M/SI	mortality and serious injury

MAFMC	Mid-Atlantic Fishery Management Council
MARAD	Maritime Administration
MDDMR	Maryland Department of Marine Resources
MDMR	Maine Department of Marine Resources
MMAP	Marine Mammal Authorization Program
MMEP	Marine Mammal Exemption Program
MMHSRP	Marine Mammal Health and Stranding Response Program
MMPA	Marine Mammal Protection Act
MRIP	Marine Recreational Information Program
MSA	Magnuson-Stevens Fishery Conservation and Management Act
msb	Atlantic mackerel/squid/butterfish
MSR	mandatory ship reporting
mt	metric tons
N.J.S.A.	New Jersey Statutes Annotated
NA	North Atlantic
NADW	North Atlantic Deep Water
NAO	North Atlantic Oscillation
NASA	National Aeronautics and Space Administration
NASCO	North Atlantic Salmon Conservation Organization
NAST	National Assessment Synthesis Team
NE	northeast
NEAMAP	Northeast Area Monitoring and Assessment Program
NED	Northeast Distant
NEFMC	New England Fishery Management Council
NEFOP	Northeast Fisheries Observer Program
NEFSC	Northeast Fisheries Science Center
NEIT	Northeast Implementation Team
NFMA	Northern Fishery Management Area
NGO	non-governmental organization
NGOM	Northern Gulf of Mexico recovery unit
NLS	Nantucket Lightship
NMFS	National Marine Fisheries Service
nmi	nautical miles
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
NRDA	national resource damage assessment
NRU	Northern recovery unit
NWA	Northwest Atlantic
NYB	New York Bight
OBIS-SEAMAP	Ocean Biodiversity Information System Spatial Ecological Analysis of Megavertebrate Populations
OCS	outer continental shelf
PAIS	Padre Island National Seashore
PBF	physical and biological features
PCB	polychlorinated biphenyls
PCE	primary constituent elements

PFRU	Peninsula Florida recovery unit
PIT	passive integrated transponder
PRD	Protected Resources Division
PVA	population viability analysis
RA	Regional Administrator
READ	Resource Evaluation and Assessment Division
REEF	Reef Environmental Education Foundation
RGA	restricted gear area
RMU	regional management unit
ROV	remotely operated vehicle
RPA	reasonable and prudent alterative
RPM	reasonable and prudent measure
RSA	research-set-aside
RV	reproductive value
RWSAS	Right Whale Sighting Advisory System
SA	South Atlantic
SAM	seasonal area management
SAP	special access program
SAR	stock assessment report
SARA	Species at Risk Act
SARBO	South Atlantic Regional Biological Opinion
SBRM	standardized bycatch reporting methodology
SCL	straight carapace length
SCUTES	Students Collaborating to Undertake Tracking Efforts for Sturgeon
SEIT	Southeast Implementation Team
SERO	Southeast Regional Office
SFMA	Southern Fishery Management Area
SHRU	Salmon habitat recovery unit
SMS	small mesh multispecies
SMA	seasonal management area
SNBS	statewide nesting beach survey
SNE	Southern New England
SRP	scientific research permit
SSB/R_{max}	stock biomass per recruit
SSC	Scientific and Statistical Committee
SSSRT	Shortnose Sturgeon Status Review Team
SST	sea surface temperature
STDN	Sea Turtle Disentanglement Network
STSSN	Sea Turtle Stranding and Salvage Network
T&C	terms and conditions
TAL	total allowable landings
TDD	turtle deflector dredge
TED	turtle excluder device
TRP	take reduction plan
TSS	traffic separation scheme
U.S.C.	United States Code

UME	unusual mortality event
USCG	United States Coast Guard
USFWS	United States Fish and Wildlife Service
USN	United States Navy
UXO	unexploded ordinance
VIMS	Virginia Institute of Marine Science
VMS	vessel monitoring system
VSP	viable salmonid populations
VTR	vessel trip report
WGOM	western Gulf of Maine
YOY	young-of-the-year

1. INTRODUCTION

Section 7(a)(2) of the Endangered Species Act (ESA) (16 U.S.C. 1531 *et seq*.) requires that each federal agency shall ensure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When the action of a federal agency may affect species listed as threatened or endangered, that agency is required to consult with either NOAA's National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the species that may be affected. In instances where NMFS or USFWS are themselves proposing an action that may affect listed species, the agency must conduct intra-service consultation. Since the actions described in this document are authorized by the NMFS Greater Atlantic Regional Fisheries Office (GARFO), this office has requested formal intra-service section 7 consultation.

Per an October 17, 2017, memorandum (memorandum from Kimberly Damon-Randall, ARA for Protected Resources, to Michael Pentony, ARA for Sustainable Fisheries), NMFS GARFO has reinitiated formal intra-service consultation[1] on the authorization of fisheries managed by NMFS under their respective fishery management plans (FMP) issued under the authority of the Magnuson-Stevens Fisheries Conservation and Management Act (MSA) and, for the American lobster and Jonah crab fisheries, permitted and operated through implementing regulations compatible with the interstate fishery management plans (ISFMP) issued under the authority of the Atlantic Coastal Fisheries Cooperative Management Act (ACA). Fisheries included in the reinitiation are the following: (1) American lobster, (2) Atlantic bluefish, (3) Atlantic deep-sea red crab, (4) mackerel/squid/butterfish, (5) monkfish, (6) Northeast multispecies, (7) Northeast skate complex, (8) spiny dogfish, and (9) summer flounder/scup/black sea bass fisheries. In addition, we have initiated consultation on the Jonah crab fishery (see section 2.3), which has not previously undergone section 7 consultation. These fisheries are collectively referred to as "the fisheries," hereinafter.

The Atlantic herring (primarily purse seine and mid-water trawl gear), surfclam/ocean quahog (primarily hydraulic clam dredge gear), and tilefish (primarily bottom longline and rod/reel gear) FMPs are not included in this biological opinion because interactions with ESA-listed species in these fisheries have not been documented, are extremely unlikely, or the gear is not known to interact with protected species. Given this information, reinitiation was not triggered for these fisheries. In addition, the Atlantic Sea Scallop FMP was considered in a biological opinion dated July 12, 2012 and most recently amended on November 27, 2018. On February 14, 2020, NMFS reinitiated consultation on the Atlantic Sea Scallop FMP (memorandum from Jennifer Anderson, ARA for Protected Resources, to Sarah Bland, ARA for Sustainable Fisheries, February 19,2020).

As described in the October 2017 Memorandum, reinitiation of the consultations was necessary given new information on the status of the North Atlantic right whale (Pace et al. 2017). In addition, the North Atlantic right whale reinitiation trigger for the biological opinion on the Northeast multispecies, monkfish, spiny dogfish, Atlantic bluefish, Northeast skate complex, mackerel/squid/butterfish, and summer flounder/scup/black sea bass fisheries ("Batched Opinion")

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[1] Consultation No. GARFO-2017-00031

was exceeded, and preliminary information in the draft 2017 Marine Mammal Stock Assessment Report (SAR) (Hayes 2018) indicated the right whale reinitiation trigger for the biological opinion for the American lobster fishery would also be exceeded (memorandum from Kimberly Damon-Randall, ARA for Protected Resources, to Michael Pentony, ARA for Sustainable Fisheries, October 17, 2017). The red crab fishery biological opinion does not contain reinitiation triggers for large whales; however, it uses trap/pot gear equipped with vertical lines similar to that used in the lobster fishery. In accordance with section 7 of the ESA, as amended, this document represents NMFS' biological opinion (Opinion) on the authorization of these fisheries and their effects on ESA-listed species under NMFS jurisdiction.

2. CONSULTATION HISTORY

2.1. Overview of Past Fishery Consultations

All fisheries, with the exception of Jonah crab, included in the action have previously undergone formal consultation (Table 1). A formal consultation is conducted when an action may affect and is likely to adversely affect an ESA-listed species or critical habitat. The product of a formal consultation is a biological opinion that determines if the action is likely to jeopardize the continued existence of any ESA-listed species or result in the destruction of adverse modification of critical habitat. The table below lists the formal consultations previously completed on these fisheries. If an opinion determines that the proposed action is likely to jeopardize listed species or destroy of adversely modify critical habitat, it must include a "reasonable and prudent alternative (RPA)" that avoids the likelihood of jeopardy or adverse modification or otherwise indicate that to the best of the agency's knowledge, there are no RPAs. If the analysis concludes with a determination that the proposed action is not likely to jeopardize a listed species or destroy or adversely critical habitat and incidental take of listed species is reasonably certain to occur, then the biological opinion includes an incidental take statement (ITS) with the anticipated level of take of the listed species and "reasonable and prudent measures (RPM)" to avoid and minimize the take. Additional information on the conclusions is included in the individual opinions. In addition to the formal consultations outlined below, the effects of FMP Amendments or Framework Adjustments (Frameworks), as well as other management measures (e.g., marine mammal take reduction plans (TRP)) were evaluated to determine if reinitiation was triggered with any of these actions. If these actions did not trigger reinitiation of ESA consultation, they are not included in the list of completed consultations (Table 1) below.

Table 1: History of formal consultations completed on FMPs or marine mammal TRPs and the jeopardy determinations of those consultations When jeopardy was found, the species that was jeopardized is indicated in parentheses.

FMPS/ISFMP	Formal Consultation Actions	Year	Jeopardy?
American Lobster	Initial consultation on implementation of Marine Mammal Exemption Program (MMEP)	1989	No
	Reinitiation to consider measures designed to prevent overfishing	1994	No
	Reinitiation on lobster fishery due to right whale deaths off FL/GA	1996	Yes (for right whale)
	Supplemented 1996 RPA	1997	No
	Consultation on ISFMP (management authority transferred Jan 2000)	1998	No
	Reinitiation to consider new information on right whale status and changes to the ALWTRP.	2001	Yes (for right whale)
	Reinitiation due to fishery management actions	2002	No
	Reinitiation due to changes to the ALWTRP	2010	No
	Reinitiation due to listing of Atlantic sturgeon distinct population segments	2012	No
	Reinitiation due to new marine mammal stock assessment and modifications to the ALWTRP	2014	No
ALWTRP	Consultation on ALWTRP	1997	No
Multispecies[1]	Initial consultation	1986	No
	Reinitiation due to effort reduction programs in the fishery	1993	No
	Reinitiation due to fishery management actions	1996 (Feb)	No
	Reinitiation due to fishery management actions	1996 (Dec)	Yes (for right whale)
	Reinitiation due action implementing a gillnet prohibition	1997	No
	Reinitiation concurrent with the initial formal consultation on the ALWTRP	1997	No
	Reinitiation due to multiple entanglements of right whales	2001	Yes (for right whale)
	Reinitiation due to changes to the ALWTRP and sea turtle bycatch estimates	2010	No
Monkfish[1]	Initial consultation, including gillnet modifications under the ALWTRP	1998	No
	Reinitiation due to new information on the status of right whales and exceedance of the ITS for sea turtles	2001	Yes (for right whale)

FMPS/ISFMP	Formal Consultation Actions	Year	Jeopardy?
	Reinitiation due to specifications deferring measures that would have effectively eliminated the directed monkfish fishery	2002	No
	Reinitiation due to new fishery management measures	2003	No
	Reinitiation due to the elimination of the SAM and DAM programs and new information on effects to sea turtles	2010	No
Spiny Dogfish[1]	Initial consultation	1999	No
	Reinitiation due to new information on entanglement of right whales	2001	Yes (for right whale)
	Reinitiation due to changes in the ALWTRP, including replacing the SAM and DAM programs	2010	No
Bluefish[1]	Initial formal consultation	1999	No
	Reinitiation due to new information on whale interactions and sea turtle bycatch	2010	No
Skate Complex[1]	Initial consultation	2003	No
	Reinitiation due to new information on whale and turtle interactions and sea turtle bycatch	2010	No
Atlantic Mackerel/Squid/Butterfish[1]	Initial consultation for implementation of MMEP	1990	No
	Formal consultation due to information on possible sea turtle interactions	1999	No
	Reinitiation due to new information on sea turtle bycatch	2010	No
Summer Flounder/Scup/Black Sea Bass[1]	Imitation of consultation on the summer flounder FMP	1988	No
	Reinitiated due to new information on sea turtle takes	1991	Yes (for Kemp's ridley)
	Reinitiation due to proposal to include scup and black sea bass in FMP	1996	No
	Reinitiation due to proposed increased landing limits in each fishery	2001	No
	Reinitiation due to loggerhead bycatch estimates and new information on the ALWTRP/List of Fisheries on scup/black sea bass trap/pot fishery	2010	No
Red crab	Initial formal consultation on FMP	2002	No
Batched Fisheries	Reinitiation due to listing of Atlantic sturgeon distinct population segments	2013	No

[1] The bluefish, Atlantic mackerel/squid/butterfish, monkfish, Northeast multispecies, Northeast skate complex, spiny dogfish, and summer flounder/scup/black sea bass fisheries were combined into a “batched fisheries” opinion in 2013.

2.2. Cause for Reinitiating

Reinitiation of formal consultation is required where discretionary control over the action has been retained or is authorized by law and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in the opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action (50 CFR 402.16). On September 18, 2017, new information indicated North Atlantic right whale abundance has been in decline since 2010 (Pace et al. 2017). This differs from the information analyzed in the previous biological opinions listed above and may reveal effects from the fisheries analyzed in the Batched Fisheries, Lobster, and Red Crab Biological Opinions that may not have been previously considered.

In addition, the Lobster and Batched Fisheries Opinions contain different numerical mortality and serious injury (M/SI) reinitiation triggers for large whales. Based on information in the 2016 SAR (Hayes et al. 2017a), the reinitiation trigger for Batched Fisheries Biological Opinion was exceeded. At the time of reinitiation, preliminary information in the Draft 2017 SAR (Hayes 2018) indicated that the reinitiation trigger for the American Lobster Biological Opinion would also be exceeded. The Red Crab Biological Opinion does not contain reinitiation triggers for large whales; however, the red crab fishery uses trap/pot gear with vertical lines similar to that used in the lobster fishery. Therefore, we are including it in this analysis to more comprehensively evaluate impacts from trap/pot gear. Taking into consideration the above information, we reinitiated formal section 7 consultation on the fisheries due to the availability of new information that may reveal effects of the action that may not have been previously considered.

2.3. Initiation of Consultation

Jonah crab was not managed under a coast-wide management plan until 2015, and therefore, NMFS has not conducted an ESA section 7 consultation on this fishery. In 2015, the Atlantic States Marine Fisheries Commission (Commission) approved the ISFMP for Jonah crab. Since that time, the Commission has made slight modifications to the management measures (see: http://www.asmfc.org/species/jonah-crab). There is a high degree of overlap between the Jonah crab and American lobster fisheries with approximately 95 percent of Jonah crab landings caught in lobster traps (ASMFC 2015a, b). Based on this, the Jonah crab ISFMP restricts the targeted harvest of Jonah crab to lobster trap harvesters and relies on trap regulations in place for the lobster fishery to regulate the targeted Jonah crab trap fishery. Under the umbrella of the Commission, states and federal government collaboratively manage the Jonah crab fishery through the ISFMP for Jonah crab. Federal regulations that complement the Commission's Plan were proposed on March 22, 2019 (84 FR 10756). NMFS has implemented regulations to limit directed Jonah crab fishing access and harvest to those harvesters who have an existing limited-access American lobster permit. Other gears may land an incidental amount of Jonah crabs (84 FR 61571, November 13, 2019). As the regulations represent a federal action and because the gear type used in the fishery is the same as gear known to interact with ESA-listed species, we are including the federal Jonah crab fishery in this consultation.

The proposed action also includes management measures (see section 3.3) recently implemented in multiple New England Fishery Management Council FMPs (Atlantic deep sea red crab Atlantic herring, Atlantic salmon, Atlantic sea scallop, monkfish, Northeast multispecies, and skate) under the Council's Omnibus Essential Fish Habitat Amendment 2 (Habitat Amendment) (68 FR 15240, April 9, 2018). NMFS has consulted under the ESA on these FMPs. In addition to the formal consultations listed in Table 1, NMFS has formally consulted on the Atlantic sea scallop FMP (NMFS 2012b, amended 2018) and informally on the Atlantic herring (NMFS 2010b). Conservation Law Foundation challenged the rule implementing approved measures of the Omnibus Essential Fish Habitat Amendment 2 (68 FR 15240, April 9, 2018) and, on October 28, 2019, the court enjoined NMFS from allowing gillnet fishing within the former boundaries of the Nantucket Lightship and Closed Area 1 Groundfish Closure Areas (CLF v. Ross, Civil Action No. 18-1087 (JEB)) until NMFS fully complied with the ESA and MSA. Per this Order, NMFS is including the implementation of the Habitat Amendment so that we can comprehensively analyze impacts to ESA-listed species resulting from implementation of this Amendment.

3. DESCRIPTION OF THE PROPOSED ACTION

3.1. Authorization of the Fisheries

The proposed action is the authorization of the following fisheries, two managed under ISFMPs and eight managed under MSA FMPs: (1) American lobster, (2) Atlantic bluefish, (3) Atlantic deep-sea red crab, (4) Jonah crab, (5) mackerel/squid/butterfish, (6) monkfish, (7) Northeast multispecies, (8) Northeast skate complex (9) spiny dogfish, and (10) summer flounder/scup/black sea bass fisheries. Analysis of these fisheries will include all vessels with one or more federal permits for species included under these FMPs and ISFMPs. Through these permits, NMFS would authorize fishing in federal waters (> 3 nmi from shore) in the action area.

Previous biological opinions completed on the fisheries have assessed the effects of the fisheries on ESA-listed species and designated critical habitat (NMFS 2002a, 2013b, 2014b). As the data available at the time the biological opinions were written did not allow us to separate out the effects from federally-permitted vessels fishing in state waters from those fishing in federal waters, these past opinions considered the operation of the fisheries to include all federally-permitted vessels fishing in state and federal waters in the action area. With new tools and data available, we are able to refine our effects analysis to evaluate fishing operations in federal waters. As NMFS does not authorize, fund, or carry out fishing activities in state waters, these activities are not considered part of the proposed action in this Opinion. Dually permitted vessels (i.e., possessing both a state and federal permit) can still operate in state waters without federal authorization. Consequently, this Opinion is evaluating effects from fishing activities (i.e., entanglement/bycatch) by vessels with federal permits in federal waters only. The effects analysis will consider the effects to ESA-listed species of transits through state and federal waters to the fishing grounds in federal waters.

In assessing the authorization of these fisheries, we consider how they operate under current requirements of the MSA, ACA, ESA, and the Marine Mammal Protection Act (MMPA). In addition, NMFS is proposing regulations as part of the Atlantic Large Whale Take Reduction Plan (ALWTRP) that will modify the American lobster and Jonah crab trap/pot fisheries. Changes in the operation of these fisheries resulting from the proposed ALWTRP measures are included in our analysis in this Opinion.

We batched the ten fisheries into one comprehensive consultation to evaluate effects across the federally-permitted fisheries by gear type. This allows us to more accurately describe and evaluate interactions between NMFS-authorized fishing activities and ESA-listed species and to more holistically focus our efforts on reducing interactions and minimizing impacts resulting from interactions that do occur. Furthermore, we are implementing the recommendation of NMFS' Northeast Fisheries Science Center (NEFSC) to evaluate protected species impacts by fishing fleets in which multiple fisheries are prosecuted, rather than on a fishery-by-fishery basis (memorandum from Michael Simpkins, NEFSC READ Chief, to Jennifer Anderson, Acting ARA for Protected Resources, November 6, 2018).

3.2. North Atlantic Right Whale Conservation Framework for Federal Fisheries in the Greater Atlantic Region

NMFS has developed a North Atlantic Right Whale Conservation Framework for Fisheries in the Greater Atlantic Region (Framework) to further reduce M/SI in the federal fisheries (Appendix 1). Serious injury includes any injury that will likely result in mortality (50 CFR 229.2). This Framework outlines NMFS' commitment to use its authorities to implement measures that are necessary for the recovery of right whales, while providing a phased approach and flexibility to the fishing industry. It should be noted that while the MMPA and ESA have different objectives, they work together to protect and recover North Atlantic right whales, restoring stocks to sustainable levels. As the MMPA measures may also contribute to progress towards the ESA goals described below, they will be considered in implementing the Framework. The Framework will further modify how the federal fixed gear fisheries operate and, as such, these changes are considered as part of the proposed action.

The Framework reduces M/SI in federal fisheries over a 10-year period to an average of 0.136 M/SIs annually. The Framework identifies the level of reductions in M/SI that NMFS is committed to achieve in order to meet its mandates (Table 2). At this time, the Framework does not specify particular measures. If gear and operational measures cannot reach the targets of the Conservation Framework, NMFS has the authority to implement closures (partial/complete or seasonal) to reduce risk, if needed. The Framework is predicated on maximizing the likelihood of North Atlantic right whale recovery success. It recognizes that efforts to reduce M/SI from other sources are underway, that there is uncertainty associated with available data, and that environmental conditions are changing. To maintain the maximum likelihood of recovery success over time, the Framework utilizes an adaptive framework and allows for revisions as additional information becomes available or should any of the assumptions require revisions. To achieve this, a comprehensive evaluation will be completed in 2025/2026. If M/SI from sources other than the federal fisheries (e.g., U.S. vessel strikes, U.S. state fisheries, and/or Canadian vessel strikes and fisheries) are reduced, new information on the apportionment of M/SIs to source becomes available, or there are improvements in the species' status, NMFS will determine whether the Framework needs to be fully implemented to achieve its conservation goals. If specific criteria identified in the Framework are met, then measures required in the federal fisheries will be reduced (see Table 2). Any changes to the Framework as a result of this evaluation will still ensure that reductions in North Atlantic right whale M/SIs in the federal fisheries achieve the level needed to ensure that the fisheries are not impacting the survival and recovery of the species.

The Framework actions include the current ALWTRP rulemaking and three additional rulemakings over the next ten years. As described below, NMFS will evaluate population metrics and threats during the Framework implementation.

Table 2: Actions to be taken under the Framework

Phase	Year	Framework Action Description
	Annually	Provide updates, as appropriate, on the implementation of the Framework to the New England and Mid-Atlantic Fishery Management Councils, Atlantic States Marine Fisheries Commission, and ALWTRT.
1	2021	NMFS implements the MMPA ALWTRP rulemaking focused on 60 percent reduction in right whale M/SI incidental to American lobster and Jonah crab trap/pot fisheries. In federal waters, this action reduces M/SIs, on average annually, to 2.69. Implementation for certain measures will begin in 2021; others will be phased over time.
2	2023	NMFS implements rulemaking to reduce M/SI in federal gillnet and other pot trap (i.e., other than lobster and Jonah crab fisheries included in Phase 1) fisheries by 60 percent, reducing M/SI, on average annually, to 2.61. The ALWTRT will convene in 2021 to recommend modifications to the ALWTRP to address risk in the remaining fixed gear fisheries. This phase will consider how any changes to the ALWTRP contribute to achieving the target reduction under this Framework.
Evaluation	2023-2024	NMFS evaluates any updated or new data on right whale population and threats to assess progress towards achieving the conservation goals of this Framework. At this time, we will also assess measures taken by Canada to address M/SI in Canadian waters.
3	2025	NMFS implements rulemaking to further reduce M/SI by 60 percent in all federal fixed gear fisheries, reducing M/I, on average annually, to 1.04.
Evaluation	2025-2026	NMFS evaluates measures implemented in 2025 action as well as new data on right whale population and threats to assess progress towards achieving the conservation goals of this Framework. Based on the results of this evaluation, NMFS will determine the degree to which additional measures are needed to ensure the fisheries are not appreciably reducing the likelihood of survival and recovery. As described above, if actions outside the federal fisheries reduce risk to right whales by 0.5 M/SI on average annually (1 whale every two years), the M/SI reduction requirement in Phase 4 will be reduced from 87 percent to 39 percent. If M/SI from other sources is reduced by greater than one M/SI on average annually, we will evaluate whether further action in the federal fisheries is needed.
4	2030	In accordance with the goals identified in the 2025-2026 evaluation, NMFS implements regulations to further reduce M/SI (up to 87 percent) in fixed gear fisheries.

3.2.1. ***Proposed measures under the Atlantic Large Whale Take Reduction Plan***

The ALWTRP, last amended in 2015, includes fishing gear modifications and seasonal area closures to reduce the risk that large whales will die or be seriously injured as a result of entanglement in U.S. commercial fishing gear. The ALWTRP also includes requirements for marking gear to improve our understanding of where entanglement incidents occur. The nature and extent of the gear modification and seasonal closure requirements varies by jurisdiction (i.e. state waters, geographic regions, and within federal waters) such that risk reduction is distributed throughout the U.S. range (see section 5.4.5 for more information on the current requirements). NMFS is proposing to modify the requirements of the ALWTRP. Because these modifications will change the operations of the lobster and Jonah crab fisheries described above, we are considering how the proposed measures will alter the fisheries in this Opinion.

The Framework includes the current proposed ALWTRP measures (Table 3). These measures aim to reduce right whale entanglement risk posed by Northeast U.S. lobster and Jonah crab trap/pot gear by at least 60 percent[2]. Measures include operational requirements, seasonal closures, and gear modifications. "Trawling up" requirements increase the number of traps per trawl according to distance from shore (Table 3) to reduce the number of vertical lines fished. In Area 3, the proposed measures would increase the length of the trawl from 1.5 nmi (2.78 km) to 1.75 nmi (3.24 km) to accommodate the proposed trawling up requirements. If some vessels cannot accommodate 45 trap/trawls, the proposed measures allow consideration of permit conditions to vary trap/trawl requirements across the Area 3 fleet to achieve an average of 45 traps/trawl.

The proposed ALWTRP measures would establish two new seasonal restricted areas, the Lobster Management Area One Restricted Area Offshore of Maine and the Massachusetts South Island Restricted Area (Table 3, Figure 1). Fishing for lobster or crab with gear that uses persistent buoy lines would be prohibited during the restricted season. Seasonal restrictions in the existing Massachusetts Bay and Great South Channel restricted areas would also be modified to prohibit fishing with persistent buoy lines rather than a complete closure that prohibits the harvest of lobster and Jonah crab. While fishermen would still be required to get exemptions from the requirements for surface gear marking under the lobster ISFMP, with those exemptions commercial fishing using "ropeless fishing" technology would be permitted in these areas. In addition, state waters of the Massachusetts Bay Restricted Area would be closed an extra month, through May, with the potential to open earlier in May when surveys indicate whales have left the area. The Cape Cod Bay and Outer Cape State Water areas represent soft openings of the Massachusetts Bay Restricted Area as the closure would be continued until no more than three whales are left as confirmed by surveys.

The gear modifications proposed include weak inserts or weak rope. The depth at which the insert must be placed is based on distance from shore. Alternatively, full weak rope would be required to the same depth on the line (Table 3). Ropes retrieved from right whale entanglements from 1994 to 2010 had breaking strengths that were 1,700 pound-force (lbf) (7.56 kiloNewtons) (kN)) or higher; adult right whales were found in rope strengths of 4,496 lbf (20 kN) and higher

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[2] The ALWTRP proposed rule has a risk reduction greater than 60 percent as it accounts for risk reduction previously achieved in Massachusetts state waters. In this Opinion, we analyze only the future risk reduction measures which will achieve a 58.1 percent reduction in risk to right whales.

(Knowlton et al. 2016). This suggests right whales may be able to break free of rope that is weaker than 1700 lbf. This is consistent with estimates of the force that large whales are capable of applying, based on an axial locomotor muscle morphology study (Arthur et al. 2015). The authors suggested that the maximum force output for a large right whale is likely sufficient to break line at that breaking strength (Arthur et al. 2015). That study and others recognized that success breaking free is also somewhat dependent on the complexity of the entanglement (van der Hoop et al. 2017b). The proposed measures would also include additional gear marking requirements. Measures proposed by the states of Maine and Massachusetts are described in the *Cumulative Effects* section.

Table 3: Proposed measures under the ALWTRP proposed rule

Component	Area	Proposed Measures
Trawl up/Line Reduction	ME exempt area – 3 nmi	3 traps/trawl
	ME 3 – 6 nmi	8 traps/trawl
	Outside of ME 3-6 nmi	No change (10 traps/trawl maintained)
	Area 1 6 – 12 nmi	15 traps/trawl
	Area 2, Outer Cape Cod 3 – 12 nmi	15 traps/trawl
	Area 1, 2 over 12 nmi	25 traps/trawl
	MA state waters, all zones	No singles on vessels longer than 29' permits after 1/1/2020
	Area 3	45 traps/trawl, increase maximum trawl length to 1.75 nmi
Closures	Existing closures become buoy lineless	Allow EFPs for ropeless fishing, with conditions that might restrict areas and would include vessels speed and observer requirements, monitoring, and reporting requirements
	Lobster Management Area One Restricted Area	Fishing for lobster or Jonah crab with persistent buoy lines would be prohibited from Oct through Jan offshore of Maine at the Area 1/3 border and across Maine Zones C/D/E
	Massachusetts South Islands Restricted Area	South of Nantucket, Feb through Apr
	State waters of Massachusetts Bay Restricted Area	State water closed through May until no more than 3 whales remain as confirmed by surveys
Weak Link Modification	Northeast Region Trap/Trawl Management Area	Retain current weak link/line requirement at surface system but allow it to be at base of surface system or, as currently required, at buoy

Component	Area	Proposed Measures
Weak Line	ME exempt area – 3 nmi	2 weak insertions, at 25 percent and 50 percent down line
	NH/MA/RI Coast – 3 nmi	1 weak insertion 50 percent down the line
	All areas 3 – 12 nmi	2 weak insertions, at 25 percent and 50 percent down line
	Area 1, 2, Outer Cape Cod over 12 nmi	1 weak insertion 35 percent down the line
	Area 2	Same weak insertions as above based on distance from shore
	Area 3	One endline weak within the top 75 percent year round

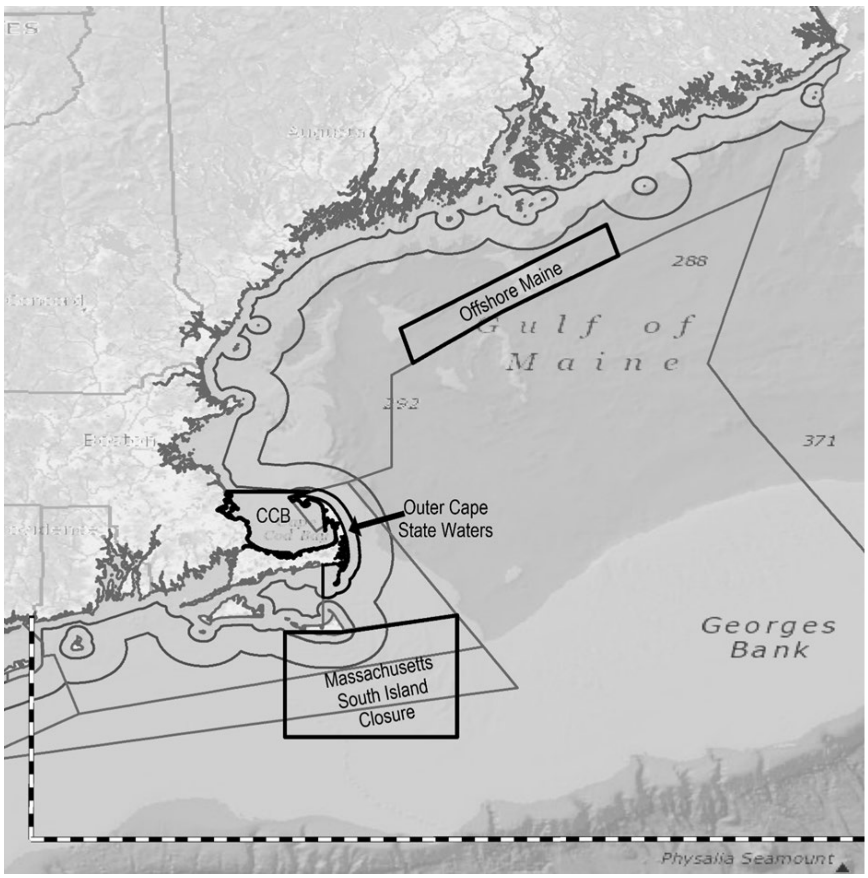

Figure 1: Proposed closures under the ALWTRP

3.3. NEFMC Omnibus Essential Fish Habitat Amendment 2

The proposed action also includes management measures recently implemented in multiple New England Fishery Management Council FMPs (Atlantic Deep-Sea Red Crab, Atlantic Herring, Atlantic Salmon, Atlantic Sea Scallop, Monkfish, Northeast Multispecies, and Skate) under the Council's Omnibus Essential Fish Habitat Amendment 2 (Habitat Amendment) (83 FR 15240, April 9, 2018). The Habitat Amendment updated the essential fish habitat (EFH) designations, designated habitat area of particular concern (HAPC), and updated prey species lists and non-fishing habitat impacts. It also revised the system of year-round closed areas that restrict some fishing gears in order to protect vulnerable habitat, and it established a system of Dedicated Habitat Research Areas (DHRA). Lastly, it implemented administrative measures for ongoing review of these areas.

NMFS approved the majority of the Council's Habitat Amendment recommendations (Letter from John Bullard, RA GARFO to Dr. Quinn, Chairman NEFMC, January 3, 2018). NMFS approved all the updated EFH designations, all the recommended HAPC designations, the majority of the habitat management area (HMA) recommendations, all the DHRA recommendations, all the seasonal spawning recommendations, and both of the framework and administrative recommendations. Because the EFH and HAPC designations are not codified, those updates became effective upon the Amendment decision. On April 9, 2018, NMFS published a final rule (83 FR 15240) implementing the approved management measures in the Omnibus Essential Fish Habitat Amendment 2.

In addition to the EFH designation updates, this action approved all of the Council's recommendations for HAPC, including the current Atlantic Salmon HAPC and the Northern Edge Juvenile Cod. In addition, the action approves the following new HAPCs: Inshore Juvenile Cod HAPC; Great South Channel Juvenile Cod HAPC; Cashes Ledge HAPC; Jeffreys Ledge/Stellwagen Bank HAPC; Bear and Retriever Seamount HAPC; and 11 canyon or canyon complexes.

Approved measures (see Figure 2 for areas) included:

- Establish the (Small) Eastern Maine HMA, closed to mobile bottom-tending gear;
- Maintain Cashes Ledge (Groundfish) Closure Area, with current restrictions and exemptions;
- Modify the Cashes Ledge Habitat Closure Area, closed to mobile bottom-tending gear;
- Modify the Jeffreys Ledge Habitat Closure Area, closed to mobile bottom-tending gear;
- Establish the Ammen Rock HMA, closed to all fishing, except lobster traps;
- Establish the Fippennies Ledge HMA, closed to mobile bottom-tending gear;
- Maintain the Western Gulf of Maine Habitat Closure Area, closed to mobile bottom-tending gear;
- Modify the Western Gulf of Maine Groundfish Closure Area to align with the Western Gulf of Maine Habitat Closure Area, with current restrictions and exemptions;
- Exempt shrimp trawling from the designated portion of the northwest corner of the Western Gulf of Maine Closure Areas;
- Add the Gulf of Maine Roller Gear restriction as a habitat protection measure;
- Remove the Closed Area I Habitat and Groundfish Closure Area designations;
- Remove the Nantucket Lightship Habitat and Groundfish Closure Area designations;
- Establish the Great South Channel HMA, closed to mobile bottom-tending gear throughout and clam dredge gear in the defined northeast section. Clam dredge gear would be permitted throughout the rest of the HMA for 1 year while the Council considers restrictions that are more refined;
- Establish the Stellwagen and Georges Bank DHRAs, with a 3-year review requirement, with the measures recommended;
- Establish the Winter Massachusetts Bay Spawning Closure, closed to gears capable of catching groundfish from November 1-January 31 of each year;

- Establish the Spring Massachusetts Bay Spawning Closure, in Block 125, closed to commercial and recreational gears capable of catching groundfish from April 15-30 of each year;
- Establish Closed Area I North and Closed Area II seasonal closures, closed to commercial gears capable of catching groundfish, except scallop dredges, from January 1-April 15 of each year;
- Remove the May Georges Bank spawning closure;
- Consider adjustments to the habitat management areas in framework adjustments; and
- Establish a system to review habitat management measures at least every 10 years.

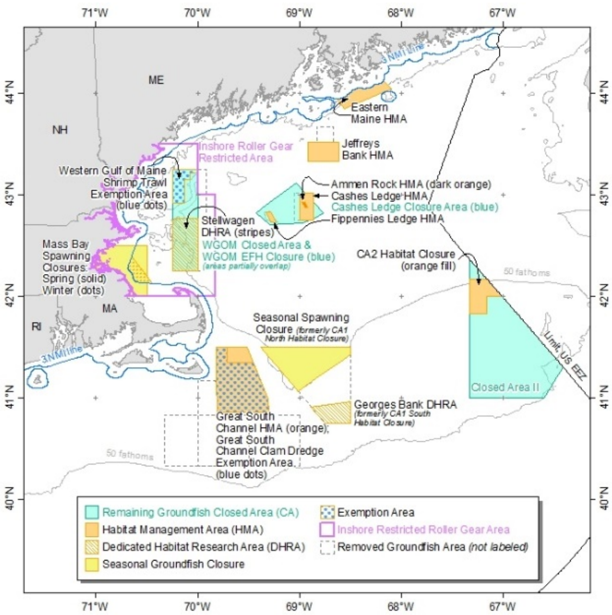

Figure 2: Omnibus Habitat Amendment 2 closures

As described above, the Conservation Law Foundation challenged the rule implementing approved measures of the Omnibus Essential Fish Habitat Amendment 2 (68 FR 15240, April 9, 2018) and, on October 28, 2019, the court enjoined NMFS from allowing gillnet fishing within the former boundaries of the Nantucket Lightship and Closed Area 1 Groundfish Closure Areas. The Amendment had resulted in the opening of these areas. Currently, some measures approved under this action remain in place, while others (i.e., the opening of the Nantucket Lightship and Closed Area 1 Habitat and Groundfish closure areas to gillnet gear) have been suspended and prior regulations that prohibited gillnet gear from fishing in these areas were restored (84 FR 68798, December 17, 2019). In order to fully evaluate the measures, this analysis will compare the impacts resulting from implementation of all approved measures to impacts occurring before the measures were approved and implemented.

3.4. Description of the Gear Used in the Fisheries Managed Under the FMPs

The level of data available on the fisheries considered in this Opinion varies depending on their characteristics and operation; therefore, the data presented below vary. In each case, this information is the best available for each FMP. For example, the following three fisheries provide a contrast in available information:

1. The Northeast Multispecies fishery requires most permit holders to use a vessel monitoring system (a GPS-type system) to track vessels and requires all permit holders to complete vessel trip reports (VTR). Thus, a high degree of information is available on the participation in and effort by this fishery.
2. The summer flounder fishery requires permit holders to complete VTRs. Thus, a moderate degree of information is available on the participation in and effort by this fishery.
3. The American lobster fishery does not have any reporting requirements for federal permit holders, though federal lobster permit holders may have reporting requirements if issued permits for other fisheries. As of 2018, approximately half of all lobster permit holders are not required to submit VTRs, the vast majority of whom fish in the Gulf of Maine which is responsible for the majority of landings. Thus, we must rely on other available information (e.g., number of permits, trap tag orders) as a proxy for participation in and effort by this fishery.

Specifics for each fishery are included in the following sections.

Sink gillnets, hook and line (i.e., handlines, bottom longline, and rod and reel), pots/traps, and bottom trawls are the predominant gears used in the fisheries included in this Opinion. The use of other gears (e.g., pound nets, pelagic longline, mid-water and paired trawls, haul and purse seines, and troll) occurs very infrequently or not at all. GARFO analysis of 2019 landings in the 10 FMPs indicated that mid-water and pair trawls in the mackerel, squid, butterfish fishery accounted for approximately 4 percent of landings under the FMP. All other gears accounted for no more than 0.1 percent of landings under the 10 FMPs. Therefore, we do not believe the limited use of these gears in the fisheries will have any effects on the ESA-listed species, and we will not discuss them further in this Opinion.

Sink gillnets are panels of net with a top rope, referred to as the head rope or floating line, and a bottom rope, referred to as the lead line. Floats are attached to floating line and the lead line is weighted to help maintain the vertical profile of the net in the water column. Multiple net panels are typically attached together to form a net string. Buoy lines attached to each end of a net string rise to the surface to mark the location of the gear. When fished in this configuration, these gillnets are referred to as ‘stand-up’ gillnets. In some areas, “tie-downs” (wire used between the floatline and the lead line as a way to create a pocket or bag of netting to trap fish) are used to reduce the vertical profile of gillnets. Fishermen may use tie-downs in order to better entangle bottom species (e.g., monkfish, flounder) in the gillnet. The minimum mesh size varies, depending on the species targeted. Vessels are also limited, as described below, in the number of gillnets that they can deploy, based on fishery. Based on where and when gillnet gear is set, gillnets must also comply with gear regulations in the ALWTRP (50 CFR 229.32), the Harbor Porpoise Take Reduction Plan (50 CFR 229.33, 50 CFR 229.34), the Bottlenose Dolphin Take Reduction Plan (50 CFR 229.35), and sea turtle regulations (50 CFR 223.206). Described more fully in section 5.4.5, 5.4.8, and 5.4.9, the take reduction plans include gear marking, pinger, buoy and groundline, storage, weak link requirements, and closures. Sea turtle requirements are described in section 5.4.4.

Hook and Line encompasses a variety of gears, but the defining characteristic of these gears is the use of artificial or natural bait placed on a hook, which is fixed to the end of a length of fishing line. The most basic hook and line gear types are handline and rod and reel, which use a single line with few or single baited hooks, brought to the appropriate depth by lead or other weights. Handline is spooled by hand, while line set out by a rod and reel is spooled mechanically. Both of these gears utilize a technique referred to as jigging, wherein a specialized lure is jerked vertically in order to attract and hook fish. Jigging can be conducted by hand or by automatic jigging equipment. Bottom longline have a mainline weighted to the seafloor with buoy lines marked by flags on either end, called high flyers. Leaders, called gangions or snoods, with baited hooks are attached to the mainline. Vessels are also limited in the number, shape, and size of hooks that can be set.

Trap/pot gear consists of the trap, buoy/surface line, groundline, buoys, and/or highflyers. The traps rest on the bottom and may or may not be baited. Buoy line(s) connect to the trap and rise vertically to the surface. Traps/pots may be set singly with each trap having its own surface line and buoy or fished in trawls consisting of two or more traps per trawl. A trawl consists of two or more traps attached to a single groundline, with at least one, but most often two, surface lines and buoys. The surface lines are typically at an end of a series of traps to mark the location of the gear. Trap gear configuration regulations differ based on jurisdiction. Offshore gear includes additional line at or near the surface to connect a radar reflector highflyer to one of the buoys to aid in the relocation and "visibility" of the gear. Traps/pots must also comply with the gear regulations in the ALWTRP (50 CFR 229.32), including buoy and groundline, storage, weak link, and traps per trawl requirements.

Bottom trawls are typically cone-shaped nets towed on the bottom. Large, rectangular doors attached to the two cables keep the net open while deployed. At the bottom of the trawl mouth is the footrope or ground rope that can bear many heavy steel weights (bobbins) to keep the trawl on the seabed. In addition, bottom trawls may have large rubber discs or steel bobbins (rockhoppers) that ride over structures such as boulders and coral heads that might otherwise snag the net. The constricted posterior netting of a bottom trawl that retains the catch is the codend. Nets are towed at a speed of 3 to 5 knots on average. Duration of tows varies, but averages 3 to 5 hours. The minimum mesh size for bottom trawls varies, depending on the target species. The summer flounder trawl fishery must also comply with gear requirements in the sea turtle regulations (50 CFR 223.206).

3.4.1. *Description of the Current American Lobster and Jonah Crab Fisheries*

The states and federal government manage American lobster through the Atlantic States Marine Fisheries Commission under Amendment 3 to the ISFMP and its Addenda (I - XXIV). The Plan identifies seven Lobster Management Areas (Areas 1, 2, 3, 4, 5, 6, and the Outer Cape) and two stocks (Gulf of Maine/Georges Bank (GOM/GB) and Southern New England (SNE)), as depicted in Figure 3. Each management area has different effort control restrictions, such as trap limits, minimum/maximum sizes, gear requirements, and closed seasons (Table 4).

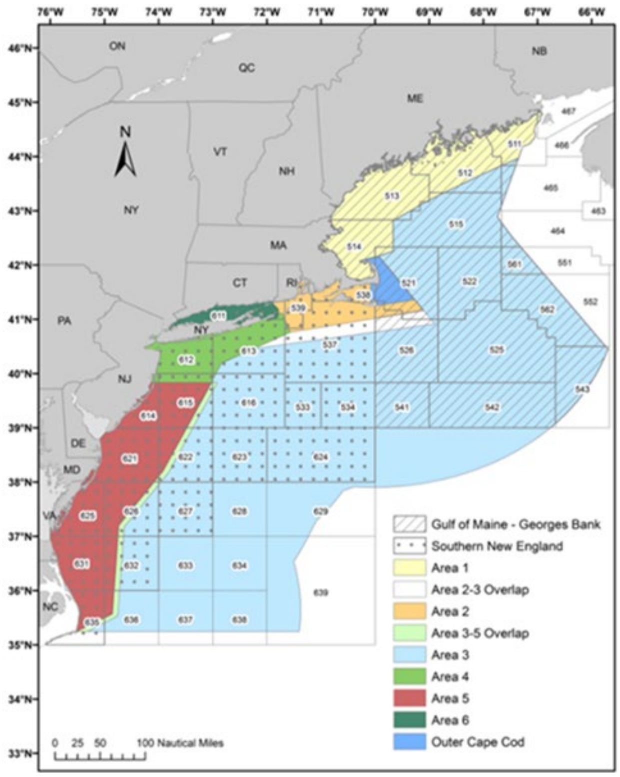

Figure 3: Lobster management and stock areas

Table 4: Summary of lobster trap limits in management areas

Management Measure	Area 1	Area 2	Area 3	Area 4	Area 5	Area 6	Outer Cape
Trap Limits	800	Permit-specific, not to exceed 800	Permit-specific, not to exceed 1,945	Permit-specific, not to exceed 1,440	Permit-specific, not to exceed 1,440	State waters only	Permit-specific, not to exceed 800

Lobsters occur in coastal waters from Maine south through North Carolina and are caught at depths of 15-1,000 ft (4.6-304.8 m). The lobster fishery is active year round, with greater effort inshore during the spring/summer and offshore in the fall/winter. Landings typically follow a seasonal pattern that is associated with the biological cycle of the American lobster, much of which is temperature-dependent. There are four restricted gear areas (RGAs) that are alternatively closed to either trap or mobile gear on a seasonal basis. Mobile gear vessels and trap harvesters agreed upon these closures to reduce gear conflicts (Table 5, Figure 4). These areas run west to east along the 50-fathom contour, south of Rhode Island.

Table 5: Summary of restricted gear areas

Restricted Gear Area	Area Closed to Mobile Gear	Area Closed to Lobster Fixed Gear
1	October 1st – June 15th	June 16th – September 30th
2	November 27th – June 15th	June 16th – November 26th
3	June 16th – November 26th	January 1st – April 30th
4	June 16th – September 30th	N/A

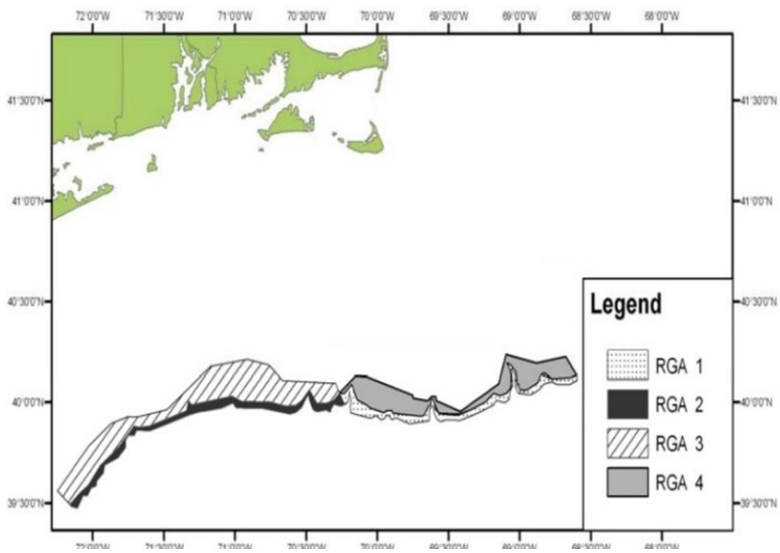

Figure 4: Restricted gear areas

The 2020 American Lobster Benchmark Stock Assessment and Peer Review Report outlined the status of lobster stocks. The Gulf of Maine/Georges Bank stock is at near record high abundance, above the abundance threshold, and overfishing is not occurring. The Southern New England stock is severely depleted and below the abundance threshold, but overfishing is not occurring. The poor stock condition in Southern New England is thought to be attributed to rising ocean temperatures (ASMFC 2015a, 2020a).

The states and federal government manage Jonah crab through the Commission's Jonah Crab ISFMP. This plan recognizes the interdependence of the American lobster and Jonah crab fisheries because available landings data indicate that Cancer crabs (Jonah crab and rock crab) were incidentally caught in lobster traps prior to becoming a targeted species (ASMFC 2015b). Jonah crabs are found in waters of the Atlantic Ocean from Newfoundland, Canada to Florida, United States (Haefner 1977). Very little is known about the life history and stock status of Jonah crabs, and, to date, no stock assessment has been completed.

Federal regulations, under the ACA in 50 CFR 697, were approved and became effective on December 12, 2019 complement the Commission's ISFMP. Measures include a minimum size, a prohibition on retaining egg-bearing females, incidental catch restrictions, requiring vessels to have a lobster permit to land Jonah crabs, and dealer permitting and report. By requiring a vessel landing Jonah crabs to have a lobster permit, the Jonah crab fishery is managed through effort controls, including permits, trap-limit, and area requirements from the American lobster fishery. The fishery takes place year-round. At present, the Jonah Crab Plan contains no closures.

However, due to the linkage with the lobster fishery, harvesters must abide by closures in the ISFMP for American Lobster, including restricted gear areas and ALWTRP closures.

The ALWTRP also regulates the American lobster and Jonah crab fisheries distinctly from the fishery management measures of the ISFMPs. Changes to the operation of these fisheries from the proposed ALWTRP regulations are considered as part of the proposed action in this Opinion.

Description of Gear Usage
Lobster fishing is conducted primarily using lobster pots/traps. A lobster trap, as regulated in 50 CFR 697.2, is a structure or device other than a net that is fished on the ocean bottom by a lobster permit holder and is designed for or capable of catching lobsters. Traps must be marked with a trap tag and identified by either the federal or state permit number. Trap trawls (multiple traps linked together by sinking groundline) cannot exceed a length of 1.5 nmi. The Interstate Lobster Plan does not regulate the number of traps per trawl, however, the ALWTRP includes minimum traps per trawl requirements.

Lobsters and Jonah crabs are also harvested by non-trap permit holders using methods other than trap gear such as trawls and gillnets. The 2020 lobster stock assessment estimated that these gear types accounted for approximately 2 percent of all landings between 1981 and 2018 (ASMFC 2020a). Non-trap landings in the Jonah crab fishery are also a minimal component of the fishery. These non-trap gears are authorized and regulated in other fishery management plans.

Fishing Effort
Due to the lack of mandatory harvester reporting requirements in the lobster fishery, effort is difficult to quantify. Furthermore, fishing effort is difficult to define in the American lobster fishery, because there is not a linear relationship between the number of traps fished and fishing effort. Many factors affect the catch rates of lobsters in traps including location, bait, trap design, soak time, temperature, and the presence of other animals (Cobb 1995 as cited in ASMFC 2020). This complicates the relationships between catches or catch per unit effort and abundance and/or densities, as well as between effort and mortality (ASMFC 2020a). Effort in the federally-permitted lobster fishery is controlled by limiting the number of eligible participating vessels or permits. States and the federal government have since further qualified and authorized harvesters using trap gear to fish in particular management areas using a specific number of traps. NMFS issued 3,068 trap permits for the 2018-fishing year (Table 6). This includes active permits and permits in confirmation of permit history (CPH), an inactive status. CPH status retains a permit's eligibility in the event the vessel has sunk or is sold. The permit in CPH may then be placed on a vessel at a later date.

Table 6: Fishing Year 2018 federal trap fishery permits and traps by state and area

State*	Area 1 Permits	Area 1 Traps	Area 2 Permits	Area 2 Traps	Area 3 Permits	Area 3 Traps	Area 4 Permits	Area 4 Traps	Area 5 Permits	Area 5 Traps	OCC Permits	OCC Traps
ME	1,305	1,044,000	2	741	10	4,681	1	1,200	0	0	1	645
NH	49	39,200	3	319	18	21,462	2	2,540	1	1,440	0	0
MA	276	220,000	72	24,397	49	47,543	2	1,580	1	500	17	9,034
RI	14	11,200	100	48,473	32	34,399	5	2,784	0	0	0	0
CT	1	800	6	3,020	1	589	3	2,725	1	875	0	0
NY	0	0	2	1,031	3	2,066	21	19,423	1	600	0	0
NJ	3	2,400	2	742	12	11,058	40	46,050	21	13,959	0	0

State*	Area 1 Permits	Area 1 Traps	Area 2 Permits	Area 2 Traps	Area 3 Permits	Area 3 Traps	Area 4 Permits	Area 4 Traps	Area 5 Permits	Area 5 Traps	OCC Permits	OCC Traps
DE	0	0	0	0	0	0	0	0	6	6,730	0	0
MD	1	800	0	0	0	0	0	0	6	4,700	0	0
VA	1	800	1	1	1	6	1	400	2	2,000	0	0
NC	0	0	0	0	0	0	1	800	0	0	0	0
FL	1	800	0	0	0	0	1	900	0	0	0	0
Total	1,651	1,320,000	188	78,724	126	121,804	77	78,402	39	30,804	18	9,679

*State is identified based on the permit holder's mailing address

Similarly, vessels using non-trap gear must have qualified into the fishery to be authorized to fish in any management area. Non-trap gear is not regulated by the lobster regulations, but could include trawl, gillnet, and dredge gear, as regulated by other FMPs. NMFS issued 1,041 non-trap permits for the 2018 fishing year. This includes active permits and permits in CPH status (see above). Table 7 summarizes both active and inactive non-trap permits.

Table 7: Federal non-trap fishery permits by state

State*	Total Non-Trap Permits
ME	111
NH	50
MA	464
RI	99
CT	28
NY	68
NJ	140
DE	2
MD	1
VA	42
NC	32
FL	4
Total	1041

*State is identified based on the permit holder's mailing address

Landings of lobster have increased over the past 35 years from 30 million lb in 1975 to peaking at 159 million lb in 2016. With a total ex-vessel value of approximately $667 million in 2016, the lobster fishery is one of the most valuable fisheries on the Atlantic coast. The greatest percentage of landings (97.7 percent) comes from the Gulf of Maine/Georges Bank stock. Landings from Southern New England have declined due to changing conditions making this region less productive. Figure 5 summarizes landings by area. There are no landing limits for harvesters using trap gear; harvesters using other gears (e.g., gillnets, trawls) are subject to a limit of 100 lobsters per 24-hour period, not to exceed 500 lobsters for a trip five days or longer.

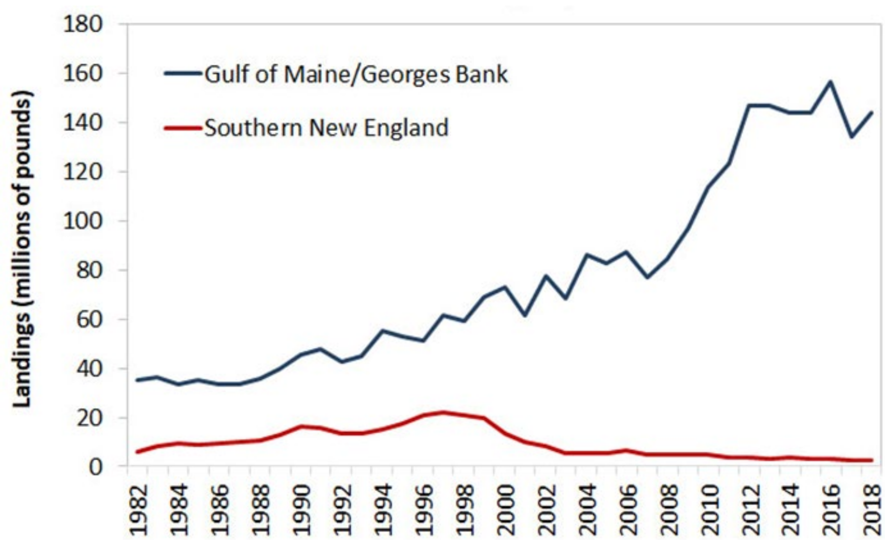

Figure 5: American lobster landings by area, 1981-2018 (see http://www.asmfc.org/species/american-lobster) (ASMFC 2015a)

Despite an increase in landings, the 2020 stock assessment (ASMFC 2020a) notes a historic increase in traps fished[3] followed by a more recent decrease, with variation by stock area. In the Gulf of Maine, the number of traps fished in the 1980s and early 1990s was fairly stable, averaging approximately 2.3 million traps. From 1993 until 2005, the number of traps reported fished steadily increased to over 3.5 million traps and remained there until 2008. Between 2009 and 2013, the number of traps fished has decreased slightly to 3.3 million traps and has since dropped to approximately 3.2 million traps in 2018. Available information suggests that traps have continued to decrease across the fishery since 2018. The number of traps fished on Georges Bank is not as well characterized. Using Massachusetts data to characterize a trend, the number of traps fished on Georges Bank increased by roughly 30 percent from 1982 to 1992. From 1993 to 2009, the number of traps varied, without trend, around a mean of 43,000 traps. Since 2010, the number of traps increased and fluctuated 44,000 traps and 50,000 traps. In Southern New England, the number of traps fished increased six-fold between 1981 and 1998, reaching a high of approximately 600,000 traps. Between 1999 and 2018, the number of traps fished declined by 75 percent, reaching a low of 147,860 traps in 2018 (ASMFC 2020a).

In addition to these trends, other actions adopted by the Commission's Lobster Plan and implemented by the states and federal government have and will continue to affect the number of traps authorized in the fishery. To address the declining abundance of the Southern New England lobster stock, NMFS implemented trap reductions to all permit holders in Areas 2 and 3, following the reduction schedule outlined in Table 8. These trap reductions are ongoing.

[3] The 2020 stock assessment estimates the total number of traps reported fished by state (or trap tags issued for Maine) within each stock are presented. Data from some states was not included as it was either confidential or not available. Thus, trap data should be considered an estimate.

Table 8: Area 2 and 3 trap reduction schedule

Effective Year (fishing year)	Area 2 Reductions (%)	Area 3 Reductions (%)
2016	25	5
2017	5	5
2018	5	5
2019	5	5
2020	5	5
2021	5	NA

Upcoming actions approved by the Commission and slated for federal implementation in 2020 include a reduction in the Area 3 trap limit, as described in Table 8. The Commission has recommended a gradual reduction schedule over a number of years, to a maximum of 1,548 traps (ASMFC 2013).

Federal regulations recently restricted Jonah crab harvest to lobster permit holders because 95 percent of historic cancer crab, including Jonah crab, landings were from lobster traps fished by lobster permit holders. As a result of this requirement, no additional traps are authorized in the fishery beyond what is authorized in the lobster fishery. Table 9 summarizes the number of Jonah crab harvesters by state from 2010-2015, prior to full state implementation of the Commission's Plan (NMFS 2018d).

Table 9: Jonah crab harvesters by state, 2010-2015

<table>
<tr><th>State</th><th>2010</th><th>2011</th><th>2012</th><th>2013</th><th>2014</th><th>2015</th></tr>
<tr><td>ME/NH</td><td>514</td><td>427</td><td>318</td><td>221</td><td>186</td><td>225</td></tr>
<tr><td>MA</td><td>103</td><td>86</td><td>94</td><td>122</td><td>126</td><td>124</td></tr>
<tr><td>RI</td><td>83</td><td>62</td><td>69</td><td>80</td><td>63</td><td>62</td></tr>
<tr><td>CT/NY</td><td>70</td><td>31</td><td>26</td><td>33</td><td>32</td><td>15</td></tr>
<tr><td>NJ</td><td>21</td><td>34</td><td>25</td><td rowspan="2">20</td><td>26</td><td>25</td></tr>
<tr><td>DE/MD/VA/NC</td><td>9</td><td>7</td><td>3</td><td>5</td><td>9</td></tr>
<tr><td>Total*</td><td>786</td><td>639</td><td>531</td><td>471</td><td>431</td><td>452</td></tr>
</table>

Jonah crab landings have increased dramatically since 1990 (Figure 6). The trend of increasing Jonah crab landings in the late 1990s coincides with the collapse of the Southern New England lobster stock and a decrease in lobster landings, suggesting harvesters turned to Jonah crab to supplement their income. While landings have increased overall, the majority of this increase was observed in federal waters, as reported by NMFS' Fisheries of the United States, 2010-2017 (NMFS 2018d).

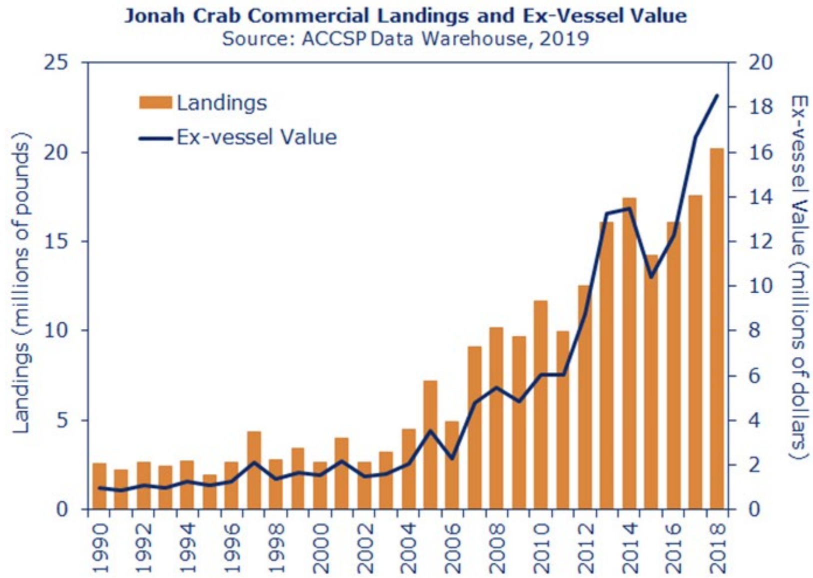

Figure 6: Jonah crab landings 1990-2018 (see http://www.asmfc.org/species/jonah-crab)

3.4.2. *Description of the Current Northeast Multispecies Fishery*

The New England Council manages the Northeast (NE) multispecies fishery through the NE Multispecies FMP. Sixteen species of groundfish are managed under the NE Multispecies FMP. Groundfish are found throughout New England waters, from the Gulf of Maine to southern New England. The NE multispecies fishery operates year-round. For management purposes, the fishing year runs from May 1 through April 30.

Large Mesh Multispecies: Thirteen species (20 stocks) are managed as part of the large-mesh complex, based on fish size and the type of gear used to harvest the fish. These species are fished both as target species (Atlantic cod, haddock, pollock, yellowtail flounder, witch flounder, winter flounder, American plaice, Atlantic halibut, redfish, and white hake) and as non-target species (windowpane flounder, ocean pout, and Atlantic wolffish).

The commercial NE multispecies fishery is divided between the sector program and the common pool. Vessels voluntarily choose to enter into the sector program as part of a groundfish sector, each of which are allocated a quota of Northeast multispecies stocks based on the collective fishing history of the sector's members. Each sector may determine how participating vessels fish that quota, also known as an Annual Catch Entitlement. Vessels that do not choose to participate in the sectors program are placed in the common pool fishery. Common pool vessels are subject to possession limits and days-at-sea (DAS – the number of days that can be fished per year), as well as quotas managed in four-month trimesters.

Vessels participating in the commercial fishery must have a permit; either a limited access permit that qualified into the fishery or an open access permit that typically allows only a small amount of NE multispecies to be harvested. All NE multispecies vessels must also comply with seasonal and year-round closed and habitat management areas (Figure 7 and Figure 8), gear size and modification restrictions, and minimum fish sizes. There is a large recreational fishery comprised of private vessels and for-hire (charter and party) vessels, which predominately fish for cod, haddock, pollock, redfish, and winter flounder with hook and line gear.

For all participants in the fishery, a system of annual catch limits (ACL) and accountability measures is in place to ensure that catches remain below desired targets for each stock in the complex. Accountability measures (AMs) are management controls to prevent ACLs from being exceeded and to correct or mitigate overages of the ACL, if they occur.

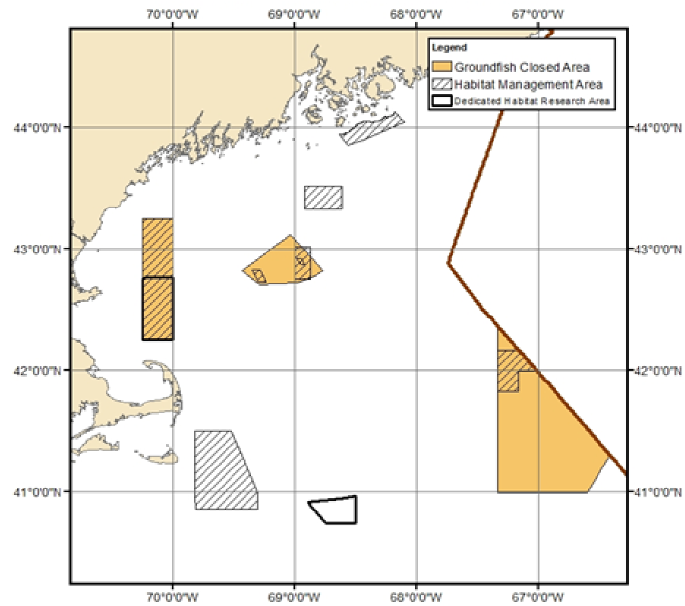

Figure 7: Overview of groundfish closure, habitat management, and dedicated habitat research areas

Large-mesh stock status is summarized below (Table 10). Overfishing is when the annual rate of catch of the stock is too high. A stock is considered overfished when the population size of the stock is too low.

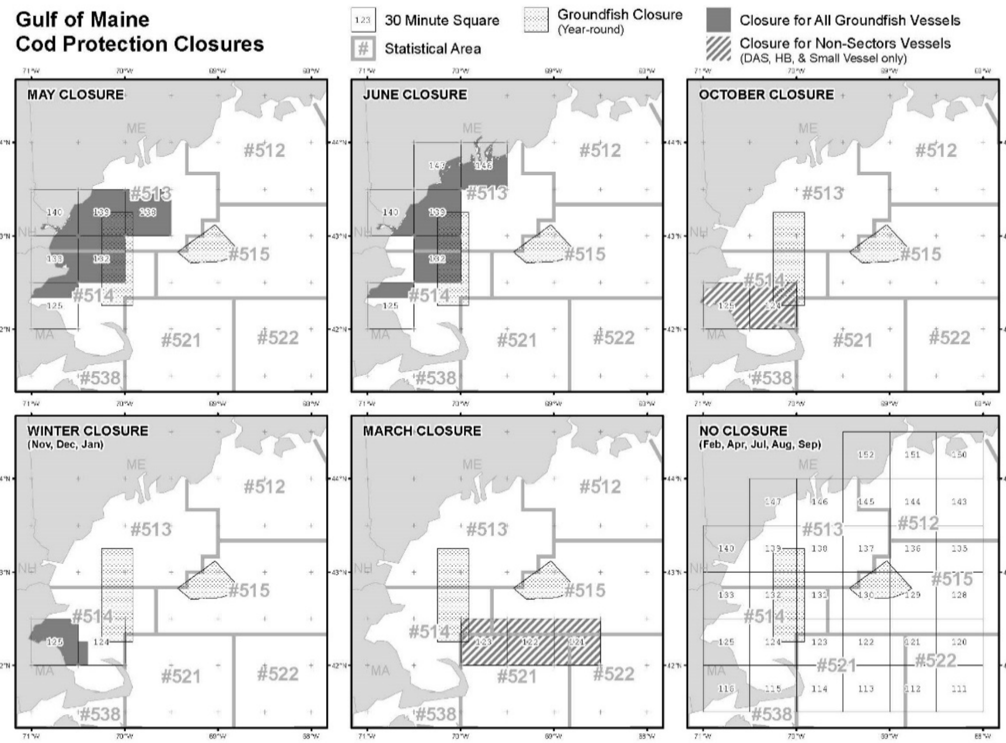

Figure 8: Seasonal Gulf of Maine cod protection closures

Table 10: Status of large-mesh Northeast multispecies stocks for fishing year 2020

Stock	Overfishing?	Overfished?
Georges Bank Cod	Yes	Yes
Gulf of Maine Cod	Yes	Yes
Georges Bank Haddock	No	No
Gulf of Maine Haddock	No	No
Georges Bank Yellowtail Flounder	Yes	Yes
Southern New England/Mid-Atlantic Yellowtail Flounder	No	Yes
Cape Cod/Gulf of Maine Yellowtail Flounder	No	No
American Plaice	No	No
Witch Flounder	Unknown	Yes
Georges Bank Winter Flounder	No	Yes
Gulf of Maine Winter Flounder	No	Unknown
Southern New England/Mid-Atlantic Winter Flounder	No	Yes
Acadian Redfish	No	No
White Hake	No	Yes
Pollock	No	No
Northern Windowpane Flounder	No	Yes
Southern Windowpane Flounder	No	No
Ocean Pout	No	Yes

Stock	Overfishing?	Overfished?
Atlantic Halibut	No	Yes
Atlantic Wolffish	No	Yes

Small-mesh multispecies: Three species (silver hake/whiting, red hake, and offshore hake) are included in the FMP as the small-mesh complex but are managed under a separate program through a series of exemptions to the NE Multispecies FMP. The small-mesh fishery operates under exemptions that allow vessels to fish for these species in designated areas, called exemption areas, using mesh sizes smaller than the minimum mesh sizes otherwise allowed under the NE Multispecies regulations (Figure 9).

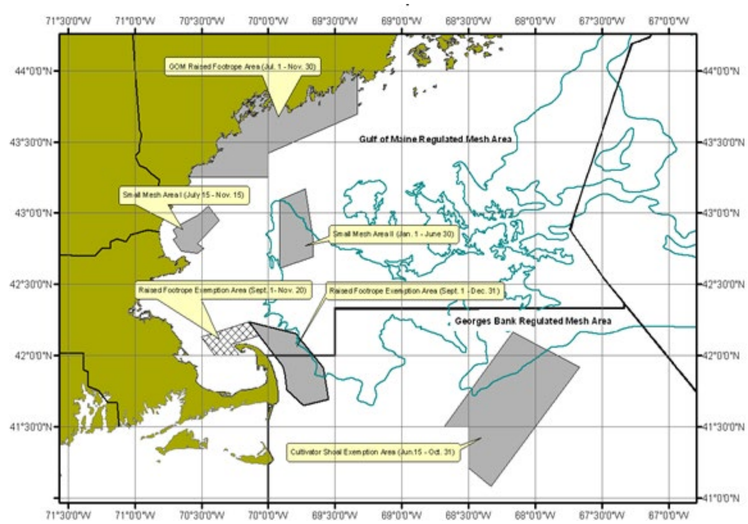

Figure 9: Whiting small mesh exemption areas

Each small-mesh exemption area has its own open season, depending on the target species and other factors considered when the exemption area was approved and implemented. Some areas can be open for several months, while others are open year round. Small-mesh stock status is summarized below (Table 11).

Table 11: Status of small-mesh Northeast multispecies stocks for fishing year 2018

Stock	Overfishing?	Overfished?
Northern Silver Hake (Whiting)	No	No
Southern Silver Hake	No	No
Offshore Hake	Unknown	Unknown
Northern Red Hake	No	No
Southern Red Hake	Yes	Yes

Description of Gear Usage

Large-mesh multispecies: A variety of gears are used in the multispecies fishery. Groundfish vessels fish for target species with trawl, gillnet, and hook and line gear (including jigs, handline, and non-automated demersal longlines). General minimum mesh sizes for trawl and gillnet are specified in Table 12, by area, though some caveats apply (see 50 CFR 648.80). In general, gillnets may not exceed 300 ft (91.4 m) in length and NE multispecies vessels may not possess more than 150 gillnets.

Table 12: Minimum mesh size and number of net requirements in the Northeast Multispecies FMP

Area	Trawl	Gillnet
Gulf of Maine	Body: 6 in. diamond or 6.5 in. square Codend: 6.5 in. diamond or square	Size: 6.5 in. Number: 50 standup or 100 tie-down gillnets
Georges Bank	Body: 6 in. diamond or 6.5 in. square Codend: 6.5 in. diamond or square	Size: 6.5 in. Number: 50 gillnets
Southern New England	Body: 6 in. diamond or 6.5 in. square Codend: 6.5 in. diamond or square	Size: 6.5 in. Number: 75 gillnets
Mid-Atlantic	Body: 5.5 in. diamond or 6.0 in. square Codend: 6.5 in. diamond or square	Size: 6.5 in Number: 75 gillnets

The NE Multispecies FMP requires that a vessel intending to fish with gillnet gear to obtain an annual designation as either a day or trip gillnet vessel. A vessel with a day gillnet designation may set its gear and return to port leaving the gear in the water to actively fish. A day gillnet vessel must abide by the size and net limits outlined above and tagging requirements. In contrast, a trip gillnet vessel sets and actively tends its gear. Because such a vessel is limited by the number of nets it can actively tend, it has no specific net limit.

In many cases, these minimum mesh sizes also regulate gear usage in other fisheries. The NE multispecies regulations also allow for large- and small-mesh exemptions, which typically provide seasonal access to an area-specific fishery using gear that would otherwise be prohibited, including the small mesh exemption program, which uses modified bottom trawls and gillnets.

Small-Mesh Multispecies: The small-mesh multispecies fishery is managed primarily through a series of seasonal exemptions from the Northeast Multispecies FMP. The directed commercial fishery is conducted with small-mesh bottom trawl gear with a number of specific requirements to reduce bycatch of large-mesh groundfish species. For the most part, the gear requirements for the small-mesh multispecies fishery are determined by the exemption or regulated mesh area being fished.

Vessels fishing in the Raised Footrope Trawl Exempted Whiting Fishery, Gulf of Maine Grate Raised Footrope Trawl Exempted Whiting Fishery, Cape Cod Exemption Area, and Small-Mesh Areas 1 and 2 must use a raised footrope trawl with a minimum mesh size of at least 2.5 inches (6.4 cm) that must be configured in such a way that, when towed, the footrope is not in contact with the ocean bottom.

Vessels fishing in the Cultivator Shoals Small Mesh Exemption area must adhere to regulations requiring all nets to have a minimum mesh size of 3-inch (7.6-cm) square or diamond mesh applied to the first 100 meshes (200 bars in the case of square mesh) for vessels greater than 60 ft (18.3 m) in length and applied to the first 50 meshes (100 bars in the case of square mesh) for vessels less than or equal to 60 ft (18.3 m) in length.

Fishing Effort

Because the Northeast Multispecies FMP includes 25 large and small mesh stocks, fishing effort is incorporated by reference from other publically available documents.

Large-Mesh Multispecies: A summary of recent large-mesh catch specifications can be found on the Council's website (see https://www.nefmc.org/management-plans/northeast-multispecies). Table 13 summarizes major landings from 2010 to 2018. In general, there has been a decreasing trend in the fishery over this period. For additional information on how this data was generated, please see Framework Adjustment 59 to the Northeast Multispecies FMP. While the total landed pounds have decreased somewhat, the value of the groundfish fishery has declined from nearly a $140-million fishery in 2011 to less than $70 million in 2017 and 2018. This is reflected in the average price for groundfish, which has declined from $1.64 per pound in 2010 to $1.12 per pound in 2018. Table 14 summarizes the level of effort broken down by the gear used on groundfish trips.

Table 13: Summary of major trends in the Northeast multispecies fishery

	2010	2011	2012	2013	2014	2015	2016	2017	2018
Groundfish lb landed	57,415,923	61,372,883	47,093,911	42,072,677	42,998,382	41,440,576	33,827,147	37,237,816	44,271,347
Non-groundfish lb landed	21,683,247	27,705,817	27,458,707	19,987,155	24,916,795	22,874,953	23,678,927	24,065,322	22,515,434
Active vessels	428	414	398	342	304	277	268	252	233
Groundfish trips	12,860	15,695	14,466	10,582	9,766	8,326	7,323	7,351	7,693
Days absent from port on groundfish trips	17,943	21,233	19,881	17,364	16,709	15,038	12,620	11,646	10,904

Notes: Data includes all vessels with a valid limited access multispecies permit that made at least one groundfish trip (declared into the fishery and landed >1 pound of any stock). "Trips" refer to commercial trips in the northeast EEZ. Source: GARFO Data Matching and Imputation System (DMIS) Database. Accessed August 13, 2019.

Table 14: Number of trips and gear types used while fishing under a groundfish limited access permit Multiple gear types may be used on a single trip (GARFO DMIS Database. accessed August 14, 2019).

Fishing Year	Trawl	Gillnet	Hook	Pot	Other Gear
2010	4,876	7,674	823	22	2
2011	6,073	9,142	1,298	24	0
2012	6,258	7,988	939	41	0
2013	5,001	5,695	289	12	0
2014	4,591	5,750	224	2	2
2015	4,744	4,186	301	19	26
2016	3,943	3,953	502	8	0
2017	4,009	3,687	522	11	0
2018	4,130	3,842	477	15	0

Small-Mesh Multispecies: A summary of recent small-mesh catch specifications can also be found on the Council's website (https://www.nefmc.org/management-plans/small-mesh-multispecies) and is summarized below in Table 15. Data was generated from the 2017-2019 SAFE report (NEFMC 2020e). In general, there has been a decreasing trend in the number of permits and landings. Price per pound ranged from a low of $0.63 2013 to a high of $0.81 in 2018. While the number of permits and landings have decreased, there were over 8,000 trips with small-mesh multispecies landings in 2019, indicating that effort may have increased despite the decrease in active boats and landings. Revenue was at its lowest for 2019 at $9.0 million, down $2.3 million from 2012 where revenue totaled $11.3 million (NEFMC 2020e).

Table 15: Summary of small mesh multispecies (SMS) trends 2012-2019 (NEFMC 2020e)

	2012	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Number of Permits*	356	357	352	321	324	334	317	304
Trips w /SMS Landings	8,726	8,098	7,903	6,589	6,299	6,912	7,722	8,426
SMS Landings mil lbs (dealer)	17.6	14.7	17.5	15.1	15.1	12.6	12.4	12.4
SMS Price/lb	$0.64	$0.63	$0.69	$0.73	$0.74	$0.75	$0.81	$0.73
SMS Revenue (mil $)	11.3	9.2	12.0	11.0	11.2	9.4	10.0	9.0

*Number of permits represents active boats with SMS landings

3.4.3. *Description of the Current Monkfish Fishery*

The New England and Mid-Atlantic Fishery Management Councils jointly manage the monkfish fishery (NEFMC 2017). It is managed primarily with a DAS management system with corresponding trip limits per DAS. The fishery also employs a total allowable landings limit, within an ACL and AM framework. The fishery takes place year-round. Monkfish occur from Maine to North Carolina out to the continental margin and are generally found at depths from 82-656 ft (25-200 m). However, there are two separate management areas (Figure 10): the Northern (NFMA) and Southern (SFMA) Landings in the SFMA peak in the late spring/early summer months when fish are migrating from deeper water, while landings in the NFMA peak in January through March. The Fishery Management Areas are based on differing fishing activity/operations in each area. A separate offshore program area, which operates under its own regulations, spans the two management areas.

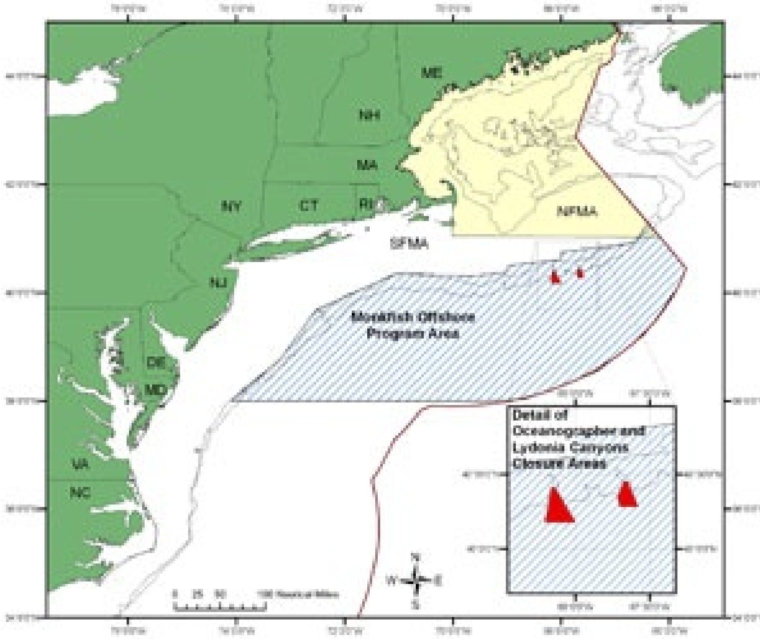

Figure 10: Monkfish fishery management areas

There are two closed areas affecting commercial monkfish vessels: Lydonia Canyon Closed Area and Oceanographer Canyon Closed Area (Figure 10). A vessel using a monkfish DAS is prohibited from fishing in these areas regardless of gear used. These areas are not closed to recreational anglers or vessels with a monkfish permit that are not fishing on a monkfish DAS.

A 2019 stock assessment updated commercial fishery statistics, fishery-independent survey indices, and fishery performance indices, among other indicators. However, because an empirical assessment was conducted based on estimates of recent catch, the 2019 assessment was only used to set catch advice, as was the case in 2016. The stock status results from the 2013 assessment, which indicated that monkfish are not overfished and no overfishing is occurring in the NFMA or the SFMA, remain valid. See Table 16 for the current monkfish specifications.

Table 16: Monkfish 2020-2022 specifications

Specification	NFMA	SFMA
Overfishing Limit	17,805 mt	23,204 mt
Acceptable Biological Catch	8,351 mt	12,316 mt
Annual Catch Limit	8,351 mt	12,316 mt
Total Allowable Landings	6,624 mt	5,882 mt

Description of Gear Usage
In the commercial fishery, bottom trawl, gillnet, longline, dredge, and trap/pot gear are authorized. In 2018, trawl gear accounted for 46 percent of landings, gillnet gear accounted for 45 percent, and dredge and other gears accounted for the remaining 9 percent. Dredge gear has accounted for a large proportion of discards in recent years (NEFMC 2020c). In the NFMA, landings are primarily by bottom trawl gear, with gillnet gear landings making up a small proportion during the winter months and a much larger proportion in the summer. In the SFMA, gillnet gear accounts for the majority of the landings. A vessel fishing with gillnet gear in the

monkfish fishery (without a NE multispecies permit) is presumed to set its gear and return to port leaving the gear in the water to actively fish. Although there is no known recreational fishery for monkfish, recreational fishing is authorized using a rod and reel or spears. Monkfish-specific gear requirements are summarized in Table 17.

Table 17: Monkfish gear requirements

Gear Type	Minimum Mesh Size	Maximum number of nets that can be set, hauled, fished, or possessed onboard	Other Requirements
Dredge	Dredge gear prohibited on a monkfish DAS		
Trawl	10 in. square or 12 in. diamond mesh throughout the codend for at least 45 continuous meshes forward of the terminus of the net. The minimum mesh size for the remainder of the trawl net is the regulated mesh size specified by the regulated mesh area fished as outlined in the multispecies regulations. **Exception:** Vessels fishing with trawl gear under both a monkfish and multispecies DAS, are subject to the minimum mesh size determined by the multispecies fishery.	n/a	The maximum roller size in the SFMA is 6 in. diameter.
Gillnet	10 in. diamond mesh **Exception:** Vessels fishing under both a monkfish and multispecies DAS or switch from a multispecies DAS to a monkfish DAS may continue to use gillnet gear with less than 10-inch diamond mesh. However, the vessel must go by the more restrictive mesh sizes as outlined in the multispecies regulations	150-160 gillnets at any time, depending on permit category **Note:** If vessel is also fishing on a multispecies DAS, it must go by the more restrictive net limits of the multispecies regulated mesh areas.	Each gillnet must be tagged and cannot be longer than 300 ft.

Fishing Effort

The monkfish fishery differs regionally. The NFMA has significant overlap with the NE multispecies fishery, as evidenced by the gear requirements in Table 17 and the permit categories listed below in Table 18. The fishery in the SFMA operates more independent of other fisheries.

Table 18: Monkfish permits (NMFS Permit Data, https://www.fisheries.noaa.gov/permit/monkfish)

Permit Category	Description	Permits Issued in 2017	Number of Permits in CPH
Category A	Commercial limited access DAS permit that does not also have a Northeast Multispecies or scallop limited access permit	20	173
Category B		38	
Category C	Commercial limited access DAS permit that has either a Northeast Multispecies or scallop limited access permit	260	
Category D		220	
Category F (offshore)	Commercial limited access offshore fishery	18	
Category H	Commercial limited access DAS for use in the Southern Fishery Management Area.	7	

Permit Category	Description	Permits Issued in 2017	Number of Permits in CPH
Category E	Commercial open-access incidental permit	1,470	Not Applicable

Landings in both areas combined peaked in 2003 but then declined to reach a relatively stable level between 2011 and 2014 (Table 19). Landings in 2015 showed a slight increase in the NFMA and a slight decrease in the SFMA (NEFMC 2017, NMFS 2012a). Since that time, landings in the SFMA have remained relatively stable in the SFMA. Landings in the NFMA between 2016-2018 increased due to management actions that allowed increased trip limits.

Table 19: NFMA and SFMA landings, 1999-2018 (NEFMC 2020c)

Year	NFMA (mt)	SFMA (mt)
1999	9,720	14,311
2000	11,859	7,960
2001	14,853	11,069
2002	14,491	7,478
2003	14,155	12,198
2004	11,750	6,193
2005	9,533	9,656
2006	6,677	5,909
2007	5,050	7,180
2008	3,528	6,751
2009	3,344	4,800
2010	2,834	4,484
2011	3,699	5,801
2012	3,920	5,184
2013	3,596	5,088
2014	3,403	5,415
2015	4,080	4,733
2016	5,443	4,280
2017	6,850	3,723
2018	5,961	4,581

3.4.4. *Description of the Current Spiny Dogfish Fishery*

The New England and Mid-Atlantic Councils jointly manage the Atlantic spiny dogfish fishery under the federal Spiny Dogfish FMP. The FMP was implemented in 2000, when spiny dogfish were determined to be overfished. However, the spiny dogfish stock was declared successfully rebuilt in a 2010 assessment and continues to remain above its threshold biomass with no overfishing occurring. The Atlantic States Marine Fisheries Commission also manages the spiny dogfish fishery in state waters from Maine to North Carolina through its Interstate Fishery Management Plan for Spiny Dogfish. This Opinion considers the federal component of the fishery managed by the Councils.

The spiny dogfish fishery is managed using a coastwide annual quota and possession limits. There is very limited directed recreational fishing for spiny dogfish and no federal recreational permit or quota. An annual catch limit and commercial quota are established through the specifications process for up to three years at a time, and AMs are used should overages occur. Each year, in addition to U.S. discards, deductions are made from the acceptable biological catch to account for Canadian landings, as a small portion of the spiny dogfish fishery takes place in Canadian waters during the summer. Up to 3 percent of the annual quota can be set aside for research purposes, but this program has not been utilized in recent years. The current federal possession limit for spiny dogfish is 6,000 lb per trip, and only one trip may be landed each calendar day. The current spiny dogfish fishery specifications are shown below in Table 20.

Table 20: Spiny dogfish 2019-2021 specifications, in metric tons

	2019	2020	2021
Overfishing Limit	21,549	N/A	N/A
Acceptable Biological Catch	12,914	14,126	16,043
Annual Catch Limit	12,865	14,077	15,994
Commercial Quota	9,309	10,521	12,438

Both Councils and the Commission reviewed and approved Scientific and Statistical Committee (SSC) and Monitoring Committee recommendations at their respective meetings in October and December 2020, and all recommended revised and projected 2021 and 2022 spiny dogfish specifications to reflect the Mid-Atlantic Council's updated risk policy. On May 4, 2021, NMFS finalized the spiny recommendations (88 FR 23633). The revisions increased the commercial quota 8 percent from what was originally projected. Table 21 shows the specifications.

Table 21: Comparison of original and revised spiny dogfish specifications for 2021 and 2022, in metric tons (mt)

	Original 2021	Revised 2021 and 2022	Percent Change
Overfishing Limit	16,043	17,498	9
Acceptable Biological Catch	15,994	17,453	9
Annual Catch Limit	12,519	13,461	8
Commercial Quota	12,438	13,408	8

Spiny dogfish are a migratory species in the North Atlantic and are most abundant from Nova Scotia to Cape Hatteras, North Carolina. They move northward in the spring and summer and southward in the fall and winter, with a preferred temperature range from 7.2 °C to 12.8 °C. This places peak abundance in mid-Atlantic waters during winter and spring months, with the bulk of the stock migrating as far north as Canada by mid-summer. Spiny dogfish also tend to congregate further offshore (near the shelf break) in the winter and move inshore (sometimes up into bays and estuaries) in the summer. The highest concentrations of spiny dogfish migrate to Southern New England, Georges Bank, and the Gulf of Maine in the fall.

The spiny dogfish fishery is active year-round, although, there is some seasonality in the distribution of landings due to the migratory nature of the species. In general, fishing effort follows the north-south seasonal migratory pattern. Spiny dogfish fishing is concentrated in the north Atlantic around Georges Bank, the Gulf of Maine, and Massachusetts state waters from

May through October. Effort shifts further south (e.g., Virginia and North Carolina) in late fall and early winter. Overall, the highest landings of spiny dogfish typically occur between June and October in Massachusetts. There are no closed areas specifically under the spiny dogfish FMP. However, permit holders are subject to the regulations and restrictions of the other permits they may be fishing under in conjunction with spiny dogfish (i.e. NE Multispecies).

Description of Gear Usage
In the commercial spiny dogfish fishery, gillnet, trawl, hook and line, rod and reel, spear, and dredge are all authorized gear; though gillnets, hook gear (longline, handline), and bottom trawls are most commonly used. Gillnets are the primary gear in the directed spiny dogfish commercial fishery, responsible for approximately 66 percent of landings annually. The other most prevalent gears in the spiny dogfish fishery are bottom longline (25 percent of catch) and bottom trawl (4 percent). The remaining spiny dogfish (about 4 percent annually) are caught with other or unknown gear. There is a small spiny dogfish recreational fishery (less than 1.5 percent of total catch annually) where handline, rod and reel, and spear are all authorized recreational gears (MAFMC 2019c).

Vessels participating in the spiny dogfish fishery must abide by the minimum mesh sizes and gear limits for gillnet and trawl gear required by the NE multispecies regulations in the four RMAs shown in Figure 11. There are also nine exempted fishing areas from the Gulf of Maine through the mid-Atlantic where spiny dogfish may be caught; some of which allow the use of smaller mesh sizes. Incidental harvesters may land spiny dogfish with gear authorized and regulated through these fishery exemption areas and programs. These exemption areas and the type of gear used are outlined below in Table 22. For a map of these exemptions areas, see: https://www.fisheries.noaa.gov/species/atlantic-spiny-dogfish.

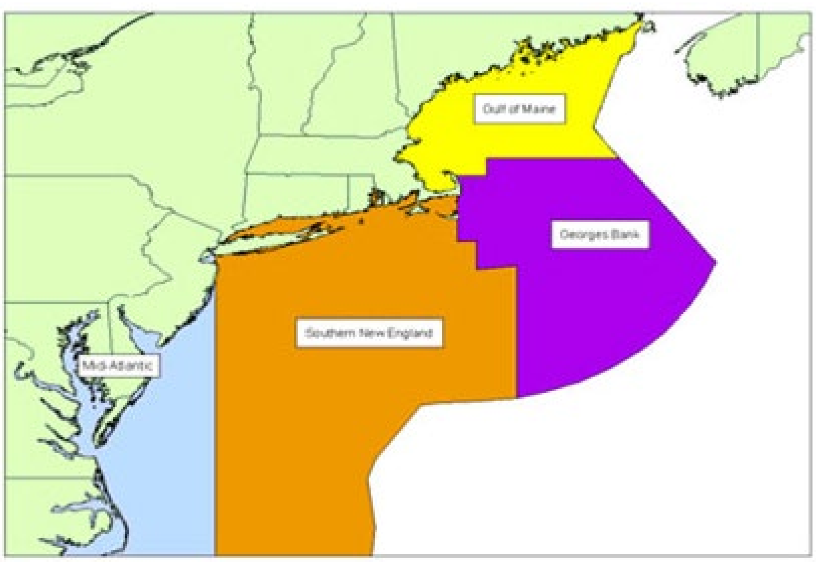

Figure 11: Regulated Mesh Areas (RMAs) in the North Atlantic

Table 22: Exemption areas in the spiny dogfish fishery

Exemption Area	Gear Allowed	Season	Letter of Authorization Required
Nantucket Shoals Dogfish Fishery	Trawl, Gillnet	June 1 – Oct 15	Yes
Cultivator Shoals Whiting Fishery	Trawl	June 15 – Oct 31	Yes
Small Mesh Areas 1 and 2	Trawl	1: July 15 – Nov 15, 2: Jan 1 – June 30	No
Raised Footrope Trawl Whiting Fishery	Trawl	Sept 1 – Dec 31	Yes
GOM/GB Dogfish Gillnet	Gillnet	July 1 – Aug 31	No
Cape Cod Spiny Dogfish	Gillnet, Longline, Handgear	June 1 – Dec 31	No
Southern New England	Trawl	Year round	No
SNE Dogfish Gillnet	Gillnet	May 1 – Oct 31	No
Mid-Atlantic	Trawl, Gillnet	Year round	No

Fishing Effort

As previously stated, no significant directed recreational fishery exists for spiny dogfish. All federal permits for the spiny dogfish fishery are open access commercial permits, so the number of active permits and vessels participating in the fishery can fluctuate on an annual basis. The best available data shows that NMFS issued 2,305 commercial spiny dogfish permits in 2019. However, of the 2,259 vessels with open access permits in 2017, only 244 actively contributed to overall landings that year.

While there is some seasonality in effort within the spiny dogfish fishery due to the migration of the stock as described above, it is still active in the United States year round. The vast majority (60-70 percent) of commercial landings each year are made in Massachusetts, with North Carolina and Virginia landing the next highest with approximately 14 and 10 percent, respectively. Most spiny dogfish are caught closer inshore, with some vessels venturing further offshore in the first half of the year to follow stock distribution. Figure 12 shows the general areas of spiny dogfish commercial fishing from 2016-2018 from dealer and VTRs. Landings from January-June are on the left and account for 67.24 percent of the total landings reported for these months. Landings from July-December are in the right panel and account for 85.78 percent of total landings for these quarters (MAFMC 2018b).

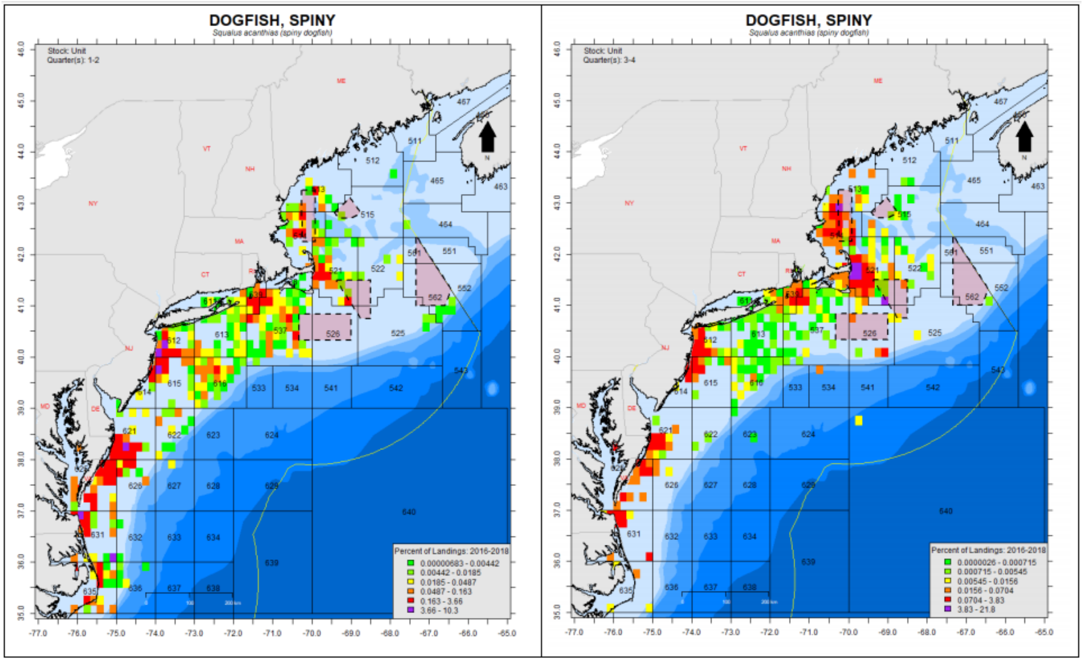

Figure 12: Spiny dogfish commercial landings 2016-2018 (data queried on July 22, 2019. Green and yellow colors represent a smaller percent of landings. Red and purple colors represent a larger percent.

3.4.5. *Description of the Current Atlantic Bluefish Fishery*

The Atlantic States Marine Fisheries Commission and the Mid-Atlantic Fishery Management Council jointly manage the Atlantic bluefish fishery in state and federal waters under the Bluefish FMP. Overall, the fishery is managed with annual catch limits, catch targets, and total allowable landings for the recreational and commercial sectors, which are then translated into quotas. The Atlantic bluefish fishery is primarily a recreational fishery, with 83 percent of the overall annual total allowable landings allocated to the recreational fishery quota and 17 percent allocated to the commercial fishery. Up to 3 percent of the total annual quota can be set aside for research purposes, but this program has not been utilized in recent years. Current fishery specifications and state commercial quota allocations for Atlantic bluefish are described in Table 23 and Table 24). The state commercial quota allocation shares were set in 2000 through Amendment 1 to the Bluefish FMP (65 FR 45844). The FMP authorizes quota to be transferred from the recreational to commercial sector as well as between states.

The recreational fishery is managed to the annual recreational harvest limit (recreational quota) using a federal bag limit and seasonal closures. In the commercial fishery, the annual coastwide commercial quota is allocated into state-specific quotas based on historic percentages specified in the FMP (Table 23). There is no federal commercial possession limit for bluefish. Each state must develop its own regulations to manage landings within its allocated commercial quotas. Though there are no closed areas under this FMP; NMFS will close the commercial fishery within a state when its commercial quota has been harvested.

Bluefish are a migratory schooling species found from Maine to Florida. They typically spend the colder winter months in the south, with larger bluefish remaining in the Mid-Atlantic Bight off North Carolina through March and smaller fish farther south, closer to Florida. Bluefish migrate north in spring as water temperatures increase. In summer, bluefish abundance centers around the New York Bight and Southern New England, but distribution can reach as far north as Maine and Nova Scotia. Starting in late fall, they begin migrating south to warmer waters. Juveniles and adults are found primarily in waters less than 65 ft (20 m) deep.

The bluefish fishery is active year-round. There is seasonality to both the commercial and recreational fisheries due to the migratory nature of the species. In general, fishing effort follows the north-south seasonal migratory pattern. Fishing is concentrated in the south Atlantic in January and February, moves north to the mid-Atlantic in the early spring, to New England in the summer and fall, back to the mid-Atlantic in late fall, and in the south Atlantic for the winter. The majority of recreational activity occurs between March and October, with peak activity in May and June and again in September and October. Most recreational fishing for bluefish is conducted by private anglers from or near shore (>75-95 percent), although, there is a small portion of the for hire community that catch bluefish recreationally. North Carolina, Rhode Island, and New York have been the states with the highest commercial bluefish harvest for the past several years.

Based on the most recent stock assessment in August 2019, bluefish are overfished, but overfishing is not occurring (NEFSC 2020). This is a change from the 2015 assessment (NEFSC 2015). That assessment indicated the stock was not overfished and overfishing was not occurring. Both assessments used an age-structured assessment program model, and while the status determination criteria did not change between the two assessments, the 2019 assessment incorporated recently calibrated recreational catch data from the Marine Recreational Information Program (MRIP). The next rebuilding plan is scheduled for 2021, and a rebuilding plan is in development for the overfished stock. Table 23 and Table 24 include the current bluefish specifications and state commercial quota allocations.

Table 23: Atlantic bluefish 2021 specifications, in lb

Fishing Year	2021
Overfishing Limit	32.98
Acceptable Biological Catch = Annual Catch Limit	16.28
Commercial Annual Catch Target (ACT)	2.77
Recreational ACT	13.51
Commercial Total Allowable Landings (TAL)	2.77
Recreational TAL	8.3
Sector Quota Transfer	0
Commercial Quota	2.77
Recreational Harvest Limit	8.34

Table 24: Atlantic bluefish 2021 specifications by state, in lb

State	FMP Percent Share	Initial* Quota
Maine	0.67	18,503
New Hampshire	0.41	11,473
Massachusetts	6.72	185,904
Rhode Island	6.81	188,434
Connecticut	1.27	35,049
New York	10.39	287,438
New Jersey	14.82	410,082
Delaware	1.88	51,985
Maryland	3.00	83,084
Virginia	11.88	328,800
North Carolina	32.06	887,377
South Carolina	0.04	974
Georgia	0.01	263
Florida	10.06	278,432
Total	100.00	2,767,793

*Quota may be transferred between states through in-season actions. These are the initial allocations and any changes may not be reflected in this table. See the quota monitoring page for updates.

Description of Gear Usage

Gillnets are the primary gear types used in the commercial bluefish fishery, accounting for approximately 64 percent for commercial catch in 2019. Hook and line gear (i.e. longline, handline, rod and reel, etc.), pound nets, seines, pots/traps, and trawls are also authorized gears. In the past five years, gillnets have accounted for around 65 percent of the commercial directed bluefish catch, with the next most common gear used various types of trawls (bottom, beam, midwater, etc.) (23 percent), and handline (8 percent). The combination of all other gear types, including traps, seines, and cast nets, comprised the remaining 4 percent. In the recreational fishery, rod and reel, and handline are the most commonly used gear to catch bluefish. There are no gear-specific requirements identified in the Bluefish FMP; but states have the option to implement their own regulations on gear that would apply to vessels and private anglers from shore in their area.

Fishing Effort

The bluefish fishery has two available open access permits; one for the commercial fishery, and one for charter/party vessels in the recreational fishery. Because these permits are open access, the number of active permits and vessels participating in the bluefish fishery can change on an annual basis. The most recent permit data from 2020 is shown in Table 25.

Table 25: Atlantic bluefish permits in 2020

Permit Category	Type	Number of Permits Issued
1	Commercial	2,100
2	Charter/Party (Recreational)	826

Most recreational harvest in the bluefish fishery comes from inland private anglers on or near the shore. Based on 2019 recreational harvest data, approximately 60 percent of coastwide recreational landings of bluefish came from shore, followed by 36 percent private/rental and 4 percent for hire (Figure 13) (MAFMC 2020). Over the last five years (2015-2019), 60 percent of the total bluefish landings came from shore, 35 percent from private/rental boats, and 5 percent from for-hire boats. The states with the highest recreational landings in 2019 were New York, North Carolina, Florida, and New Jersey (Table 26). Over 75 percent of commercial landings in 2019 came from the six statistical areas surrounding Connecticut, New York, and Rhode Island, (Table 27, Figure 14) (MAFMC 2020).

Table 26: MRIP estimates of 2019 recreational harvest and total catch for bluefish

State	Harvest			Catch
	Pounds	Number	Average wt (lb)	Number
ME	0	0	0	0
NH	0	0	0	0
MA	719,130	265,628	2.7	736,761
RI	210,033	119,801	1.75	271,594
CT	340,666	312,022	1.09	817,150
NY	1,399,517	1,203,567	1.16	3,905,614
NJ	2,007,110	1,421,477	1.41	3,933,439
DE	315,105	75,703	4.16	611,903
MD	493,192	274,834	1.79	692,643
VA	264,534	443,112	0.6	870,958
NC	2,630,685	3,304,587	0.8	11,216,797
SC	403,141	765,113	0.53	2,295,592
GA	70,284	90,991	0.77	386,195
FL	4,525,038	2,052,080	2.21	5,212,593
Total	13,270,862	10,245,711	1.3	30,928,703

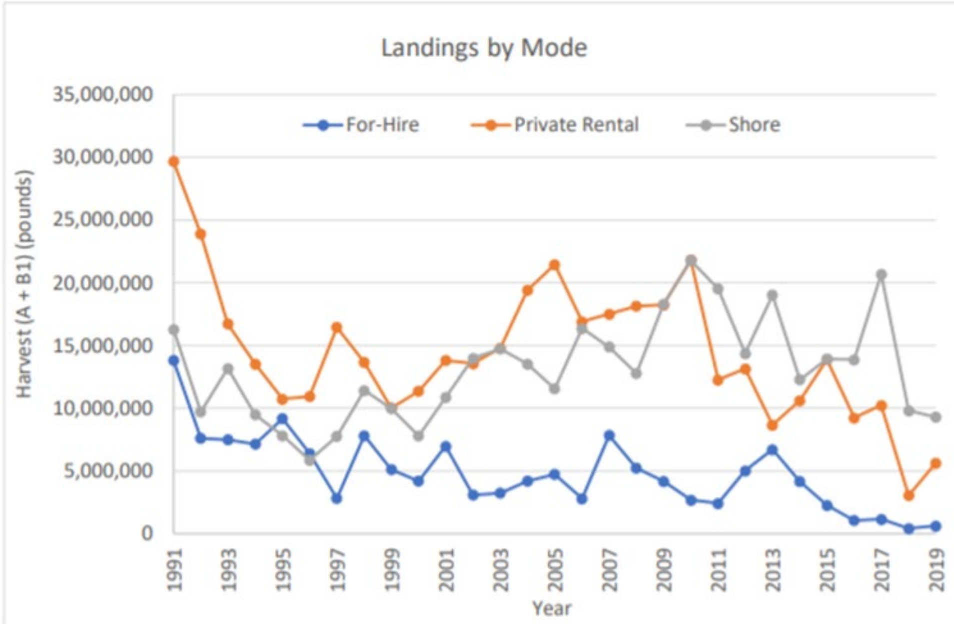

Figure 13: Bluefish recreational harvest (lb) by mode, Atlantic Coast, 1991-2019, from MRIP data (MAFMC 2020)

Table 27: Statistical areas with at least 5 percent of the total commercial bluefish landed in 2019 (MAFMC 2020)

Statistical area	Bluefish Landings (lb)	Percent of 2019 Commercial Bluefish Catch	Number of Trips	Percent of 2019 Commercial Bluefish Trips that Caught Bluefish
611	169,338	18	1,667	31
539	166,201	18	1,051	20
613	130,350	14	727	14
626	80,566	9	84	2
632	53,364	6	27	< 1
612	37,076	4	287	5

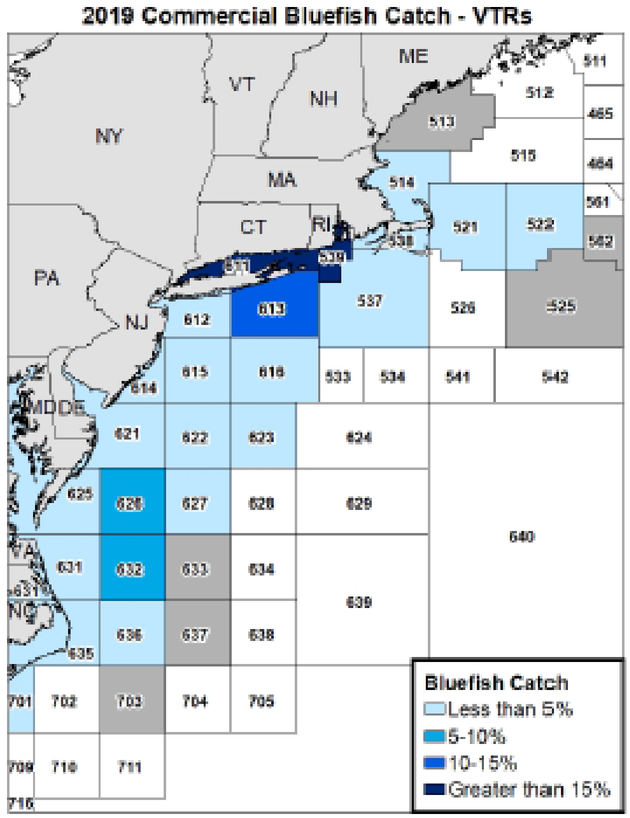

Figure 14: Commercial bluefish catch, 2019, by statistical area (MAFMC 2020)

3.4.6. *Description of the Current Northeast Skate Complex*

The New England Council manages the skate fishery under the NE Skate Complex FMP. The fishery operates from Maine to Cape Hatteras, North Carolina. Skates are mostly harvested incidentally in trawl and gillnet fisheries targeting groundfish, monkfish, and sometimes scallops. The FMP manages a complex of seven different skate species: barndoor (*Dipturus laevis*), clearnose (*Raja eglanteria*), little (*Leucoraja erinacea*), rosette (*Leucoraja garmani*), smooth (*Malacoraja senta*), thorny (*Amblyraja radiata*), and winter (*Leucoraja ocellata*) skates. Each of the seven skate stocks has overfishing and overfished status determination criteria that are based on a NMFS trawl survey index of abundance. None of the seven skate stocks are subject to overfishing. Thorny skate is overfished, and the other six skate stocks are not overfished.

The seven species in the skate complex are distributed along the coast of the northeast United States from near the tide-line to depths exceeding 2,300 ft (700 m). Within the complex, the ranges of the individual species vary. In general, barndoor skate are found along the deeper portions of the Southern New England continental shelf and the southern portion of Georges Bank, extending into Canadian waters. Clearnose skates are caught by the NMFS surveys in shallower water along the Mid-Atlantic coastline, but are known to extend into unsurveyed shallower areas and into the estuaries, particularly in Chesapeake and Delaware Bays. Little skate are found along the Mid-Atlantic, Southern New England, and Gulf of Maine coastline, in shallower waters than barndoor, rosette, smooth, thorny, and winter skates. Rosette, smooth, and thorny are typically deep-water species. NMFS' survey catches rosette skate along the shelf edge in the Mid-Atlantic region, while smooth and thorny are found in the Gulf of Maine and along the northern edge of Georges Bank. Winter skate are found on the continental shelf of the Mid-Atlantic and Southern New England regions, as well as Georges Bank and into Canadian waters. Winter skate are typically caught in deeper waters than little skate, but partially overlap the distributions of little and barndoor skates. Skates are not known to make large-scale migrations, but they do move seasonally in response to changes in water temperature, moving offshore in summer and early fall and returning inshore during winter and spring.

Skates are harvested for two different markets – skate wings for human consumption and whole skates for use as bait in other fisheries, such as lobster and Jonah crab. Use of skate as bait is growing in importance with declines in the availability of herring. The fishery is primarily managed with fishery specific (wing vs. bait) total allowable landings, possession limits, seasons, in-season possession limit adjustments, and other AMs. Each fishing year (May 1-April 30), the skate wing fishery is allocated 66.5 percent of the federal TAL for skates, and the skate bait fishery is allocated 33.5 percent of the federal TAL. There are no closed areas identified with the Northeast Skate Complex FMP. However, area management within the Northeast Multispecies, Scallop, and Monkfish FMPs would impact the harvest of skates.

The skate wing fishery evolved in the 1990s as skates were promoted as underutilized species. Attempts to develop domestic markets were short-lived, and the bulk of the skate wing market remains overseas. Winter skate is the dominant component of the wing fishery. The Southern New England sink gillnet fishery targets winter skates (a primary component of the wing fishery) seasonally along with monkfish. Highest catch rates are in the early spring and late fall when the boats are targeting monkfish, at about a 5:1 average ratio of skates to monkfish. Little skates are also caught incidentally year-round in gillnets and sold for bait.

The skate bait fishery is more of a directed and historical fishery, compared to the wing fishery. The skate bait fishery has three seasons, with about 68 percent of total allowable landings allocated to seasons 1 and 2 (May 1st to October 31). This is designed to accommodate the amplified effort in the spring through fall lobster fishery. Small, whole skates are among the preferred baits for the lobster fishery. The skate bait fishery involves vessels from primarily Southern New England ports that target a combination of little skates (>90 percent), and to a lesser extent juvenile winter skates (<10 percent). The 2019-2021 Northeast skate complex specifications and seasonal quota allocations are described in Table 28 and Table 29.

Table 28: NE skate complex 2019-2021 specifications, in metric tons

Fishing Year	2019	2020	2021
Acceptable Biological Catch = Annual Catch Limit	31,327	32,715	32,715
Annual Catch Target	28,194	29,444	29,444
Overall Total Allowable Landings	15,788	17,864	17,864
Wing Fishery Total Allowable Landings	10,499	11,879	11,879
Bait Fishery Total Allowable Landings	5,289	5,984	5,984

Table 29: NE skate complex 2019-2021 seasonal TAL allocations

Fishing Year		2019	2020	2021
Skate Wing Fishery	Season 1 (May 1 – August 31)	5,984	6,771	6,771
	Season 2 (September 1 – April 30)	4,515	5,108	5,108
Skate Bait Fishery	Season 1 (May 1 – July 31)	1,629	1,843	1,843
	Season 2 (August 1 – October 31)	1,962	2,220	2,220
	Season 3 (November 1 – April 30)	1,698	1,921	1,921

Description of Gear Usage

Trawl, gillnet, longline, handline, dredge, and rod and reel are all authorized gears in the skate fishery. In general, skates are mostly harvested incidentally in otter trawl and gillnet fisheries targeting groundfish, monkfish, and, sometimes, scallops. In 2018, otter trawl was the primary gear used in the bait fishery (99 percent of bait-only landings), while more skates were landed in the wing fishery with gillnet gear (81 percent of wing-only landings). Overall, gillnets are responsible for approximately 66 percent of skate catch, and trawls comprise about 32 percent. Skates are also consistently caught with traps, hook gear, and scallop dredges; although landings from these gears are relatively insignificant; about 2 percent of all catch combined (NEFMC 2020d). All vessels fishing for skates using a DAS are subject to the gear regulations of whichever limited access fishery it has declared into for that DAS. Otherwise, vessels fishing for skates must abide by the gear requirements of the NE Multispecies FMP.

Fishing Effort

Total skate landings have fluctuated over the years (Table 30) (NEFMC 2020d). The fluctuations in landings is largely attributable to the wing fishery, as landings in the bait fishery have remained relatively stable.

An open access permit is required to land skates. Both a permit and a skate bait letter of authorization (LOA) is required to land whole skate for the bait fishery. Vessels fishing for skate wings must be on a New England multispecies, scallop, or monkfish DAS to land more than the incidental limit of 500 lb of skate wings. In general, vessels fishing for skate bait under a bait LOA must also be on a DAS, unless the vessel is fishing in a DAS exemption area.

The number of skate permits peaked in fishing year 2007 at 2,686 permits and has declined since; the number of skate permits in 2019 was 2,028. The number of active federally-permitted vessels (i.e., federal fishing vessels landing more than 1 lb of skate) has decreased as well, with 567 active permits in 211 to 357 active permits in 2019 (NEFMC 2020d).

Table 30: Skate landings in live weight lb (i.e., the weight of a whole skate) by fishery type. A conversion factor is applied to all wing landings in order to estimate weight of the entire skate.

Fishing Year	Bait	Wing	Total
2010	9,698,695	23,000,058	32,698,753
2011	10,837,172	30,465,414	41,302,586
2012	10,766,626	22,427,119	33,193,745
2013	11,176,451	19,720,311	30,896,762
2014	9,386,666	24,704,030	34,090,696

Fishing Year	Bait	Wing	Total
2015	10,513,990	22,943,092	33,457,082
2016	10,148,571	20,228,685	30,377,
2017	12,495,542	20,057,874	32,553,416
2018	10,625,319	21,164,021	31,789,340
2019*	8,424,659	19,019,727	27,444,386
2020*	4,468,490	10,315,403	14,783,893

*Preliminary data as of August 2020. Fishing year 2020 ends April 30, 2021 so fishing year 2020 is incomplete.

The skate bait fishery involves vessels from primarily Southern New England ports that target a combination of little skates (>90 percent), and to a lesser extent juvenile winter skates (<10 percent). The bait fishery is largely based out of Rhode Island (primary ports in Point Judith and Newport) and other secondary ports (Sea Isle City, New Jersey; New London, Connecticut; and Montauk, New York) also identified as participants in the directed bait fishery (NEFMC 2020d).

The majority of skate wings are landed in Massachusetts, Rhode Island, and New Jersey. New Bedford, Massachusetts emerged early on as the leader in production, both in landed and processed skate wing, although skate wings are landed in ports throughout the Gulf of Maine and extending down into the Mid-Atlantic. In 2016, Chatham surpassed New Bedford for the most skate wings landed, New Bedford still processes the greatest share of skate wings. As of August 2020, the three primary ports for skate wings are Chatham, Massachusetts; New Bedford, Massachusetts, and Point Judith, Rhode Island (NEFMC 2020d).

3.4.7. *Description of the Current Mackerel/squid/butterfish Fishery*

The Mid-Atlantic Council manages Atlantic mackerel, chub mackerel, longfin squid, *Illex* squid, and butterfish through a single FMP called the Mackerel, Squid, and Butterfish (MSB) FMP. All species use quotas and AMs. Various permitting systems, mesh requirements, time-area closures, and trip limits are used in these fisheries to help achieve optimum yield. The Atlantic mackerel and longfin squid fisheries are managed separately through incidental catch permits and a tiered limited access permit system, which includes different possession limits for the different permit categories. Atlantic mackerel catch is controlled by an annual mackerel quota and a bycatch quota for river herring and shad. The *Illex* and butterfish fisheries are managed by limited access and incidental catch permits, mesh size and area restrictions, annual quotas, and trip limits. Atlantic chub mackerel was integrated into the MSB FMP in 2020 and is managed by an annual quota and AMs (85 FR 47103, August 4, 2020).

Even though the overfishing limit is unknown, the Mid-Atlantic Council's SSC concluded that long-term average landings by the directed longfin and *Illex* squid fleet appears to be sustainable. Due to the limited fishery dependent and independent data available for Atlantic chub mackerel, there is no stock assessment for this species to specify status determination criteria. The December 2017 Atlantic mackerel stock assessment concluded that the stock is overfished and subject to overfishing. NMFS has implemented measures to establish a stock rebuilding program for Atlantic mackerel (84 FR 58053, October 30, 2019). An updated stock assessment for Atlantic mackerel is expected in 2021. The 2020 management track assessment determined the status of butterfish is not overfished with no overfishing occurring. The assessment discovered that biomass is 69 percent of its target. Given butterfish's short life history and variable recruitment substantial fluctuations in biomass are not unexpected. Fishing mortality appears to have been low in recent years, which means recent declines are not a result of overfishing but poor recruitment. If recruitment returns to average levels, then the stock is predicted to build above the SSBmsy target quickly. The 2021 quotas for the MSB FMP is provided in Table 31.

Table 31: Squid, mackerel, and butterfish 2021 specifications

Stock	Atlantic Chub Mackerel	Atlantic Mackerel	Butterfish	*Illex*	Longfin
Overfishing Limit	3,026	N/A	22,053	Unknown	Unknown
Allowable Biological Catch	2,300	19,184	11,993	30,000	23,400
Annual Catch Limit	2,261.7	19,184	11,993	N/A	N/A
Commercial Annual Catch Target	2,171.2	17,387	11,393	N/A	N/A
Recreational ACT/Recreational Harvest Limit	N/A	1,270	N/A	N/A	N/A
Domestic Annual Harvest	N/A	17,312	6,350	28,644	22,932

The five species in the MSB FMP are available and harvested in varied distribution ranges at various times of the year along the eastern seaboard of the United States, from the coast to the continental shelf break. Atlantic mackerel are generally associated with, or related to (inversely), the distribution of herring, with some years off the Mid-Atlantic Bight, other years off Cape Cod, and others on Georges Bank. Longfin squid may aggregate from waters just south of Cape Cod to off the Mid-Atlantic, generally inshore in the summer and offshore in the winter. Butterfish are widely distributed and may aggregate in various locations throughout their range. A summary of the distribution and seasonality of the fishery is summarized in Table 32.

Table 32: Spatial distribution of mackerel, squid, and butterfish stocks

Species	Seasonality of Fishery	Spatial Distribution of Stock
Atlantic mackerel	November-April	Distributed between Labrador and North Carolina, with catch between Maine and North Carolina.
Longfin squid	Year round, peaks in the spring and fall	During the early spring and late fall, catch occurs near the shelf break, with summer and early fall catch primarily occurring nearshore.
Illex squid	May-October (dependent on aggregation and market)	Along the continental shelf break
Atlantic chub mackerel	May-October	Along the continental shelf break and the east coast of Florida
Butterfish	Year-round, but historically mostly in winter	Southern New England shelf break areas, in and around Long Island Sound.

Atlantic mackerel, squid, and butterfish permitted vessels cannot fish with bottom trawl gear in the Oceanographer or Lydonia Canyons or the Frank R. Lautenberg Deep-Sea Coral Protection Area. The area extends from the continental shelf/slope break off the Mid-Atlantic states (New York to North Carolina) to the border of the EEZ (Figure 15). The use of bottom-tending commercial fishing gear in the designated deep-sea coral zone is prohibited in this area. Gear restrictions in this area do not apply to recreational fishing, commercial gear types that do not contact the sea floor, or the American lobster trap fishery. An exemption is also provided for the deep-sea red crab commercial trap fishery.

Figure 15: Frank R. Lautenburg Deep-Sea Coral Protection Area

Description of Gear Usage

Longfin squid, *Illex* squid, and butterfish are primarily harvested with bottom-tending otter trawl gear. Vessels fishing with otter trawl gear that possess 5,000 lb or more of butterfish must use nets that have a minimum codend mesh of 3 inches (7.62 cm). Vessels targeting longfin squid must comply with seasonal gear requirements described in Table 33. During closures of the longfin squid fishery resulting from the butterfish mortality cap, vessels can still fish for longfin squid using jigging gear, but this has not been common. Vessels fishing for *Illex* squid with otter trawl gear during June-September in the *Illex* squid exemption area are exempt from the longfin squid minimum mesh size requirements.

Table 33: Longfin squid seasonal mesh requirements

Trimester	Minimum Mesh Size
Trimester I (Jan-April)	2 1/8 inches (54 mm)
Trimester II (May-Aug)	1 7/8 inches (48 mm)
Trimester III (Sept-Dec)	2 1/8 inches (54 mm)
Net strengtheners must be 5 inches (12.7 cm) or greater square or diamond mesh	

The primary participants are generally larger vessels, though a wide range of vessels participate, especially in the longfin squid fishery. Larger vessels often either freeze their catch on board or keep it in refrigerated seawater and process it on shore. The squid fisheries are predominantly bottom otter trawl. Both mackerels are harvested with a variety of gears but mostly bottom otter trawl, single midwater trawls, and paired midwater trawls. Atlantic chub mackerel is usually targeted by vessels that also target *Illex* squid when *Illex* squid is not available. Midwater otter trawls and paired midwater trawls have become increasingly important for mackerel in recent years. While there is no permit for Atlantic chub mackerel, any MSB permit is needed to possess Atlantic chub mackerel for sale.

Fishing Effort
The number of active vessels in the Atlantic mackerel, squid, and butterfish fisheries has fluctuated over the past decade, but has generally decreased for all fisheries. A more detailed breakdown of federal permit holders in 2020 is in Table 34.

Table 34: MSB permit holders in 2020

Permit Type	Permit Category	Description	Count
Moratorium Permits	SMB 1A	Longfin	230
	SMB 1B	Longfin	49
	SMB 1C	Longfin	23
Moratorium Permits	SMB 5	*Illex*	69
	SMB 6	Butterfish Moratorium	279
	SMB T1	T1 Mackerel	31
	SMB T2	T2 Mackerel	23
	SMB T3	T3 Mackerel	73
	SMB 2	SMB Charter/Party	736
Open Access	SMB 3	Squid/Butterfish Incidental	1,515
	SMB 4	Mackerel Incidental	1,643

Landings for Atlantic mackerel, longfin squid, *Illex* squid, and butterfish vary over the years due to availability of the resource and market conditions. The following descriptions are based on landings from 1996-2016. Atlantic mackerel landings ranged from a high of 124,868,012 lb in 2006 to a low of 1,169,156 lb in 2011. Longfin squid landings range from a high of 42,095,370 lb in 1999 to a low of 13,396,792 in 2004. Landings of *Illex* squid are also variable, ranging from a high of 50,965,858 lb in 1998 to a low of 589,598 lb in 2002. Butterfish landings range from a high of 9,777,854 lb in 2001 to a low of 1,032,754 lb in 2005. Atlantic chub mackerel landings in 1999-2018 range from a high of 5,250,807 in 2013 to a low of 117 in 2009.

3.4.8. ***Description of the Current Summer Flounder, Scup, and Black Sea Bass Fishery***
The Mid-Atlantic Council and the Atlantic States Marine Fisheries Commission jointly manage summer flounder, scup, and black sea bass. These species are managed under a single FMP because these species occupy similar habitat and are often caught at the same time. All three

species are highly sought after by commercial and recreational fishermen. Although managed under one FMP, permits for summer flounder, scup, and black sea bass are issued separately based on having met that fishery's limited access eligibility requirements. Each of these three fisheries also issues open access charter/party permits.

NMFS implements ACLs and AMs for the summer flounder, scup, and black sea bass fisheries. Specific landing limits (i.e., quotas) are derived for the commercial and recreational sectors after accounting for scientific and management uncertainty. The commercial quota for summer flounder is managed on a state-by-state basis in both federal and state waters. For scup, the commercial quota is divided into three harvest periods. Federal waters are managed on a coastwide basis for each quota period. In state waters, the Commission manages the fishery with individual state quotas during the summer quota period and coastwide during the winter quota periods. The black sea bass commercial quota is managed on a coastwide basis in federal waters while the Commission manages the fishery in state waters using individual state quotas. Quota specifications for the three species regulated under the FMP are generally set on an annual basis, but may be proposed for a 3-year period.

Summer Flounder: Summer flounder are targeted in waters from Cape Cod, Massachusetts to Cape Hatteras, North Carolina. The commercial fishery is managed using a quota system along with size limits and gear requirements. The recreational fishery is managed using size, season, and bag limits. The commercial fishery for summer flounder runs year round with two major trawl fisheries concentrating offshore in the winter and inshore in the summer. The recreational fishery occurs primarily in the spring, summer, and early fall. There is a commercial closure in place in Delaware, due to an overharvest from a previous year. The 2019 stock assessment concluded that summer flounder is not overfished and no overfishing is occurring.

Scup: Scup are typically found in offshore waters in the winter from New Jersey to Cape Hatteras and in the warmer months move north to areas in southern New England and Long Island, New York. Commercial management methods include a coastwide seasonal quota, size limit, seasonal possession limits, and gear restrictions. The recreational fishery is managed using size, season, and bag limits. The commercial scup fishery occurs year round in waters from Massachusetts through North Carolina, with a significant portion of the 2018 landings coming from Rhode Island (34 percent), New York (25 percent), and New Jersey (18 percent). The recreational fishery occurs mostly during spring and fall.

Black Sea Bass: Black sea bass are distributed from the Gulf of Maine to the Gulf of Mexico, but fish north of Cape Hatteras, North Carolina are considered part of a single unit stock. Over the past decade, the distribution of sea bass has expanded into the Gulf of Maine (Bell et al. 2015) as far as eastern coastal Maine (M. McMahan, pers. comm. as cited in NEFSC 2017). Within the stock area, distribution changes on a seasonal basis and the extent of the seasonal change varies by location. In the northern end of the range (New York to Massachusetts), black sea bass move offshore crossing the continental shelf, then south along the edge of the shelf (Moser and Shepherd 2009). By late winter, northern fish may travel as far south as Virginia, but most return to the northern inshore areas by May. Black sea bass originating inshore along the Mid-Atlantic coast (New Jersey to Maryland) head offshore to the shelf edge during late autumn, travelling in a southeasterly direction. They return inshore in spring to the general area from which they originated. Black sea bass in the southern extent of the stock (Virginia and Carolina) move (Bell et al. 2015) offshore in late autumn/early winter. Given the proximity of the shelf edge, they transit a relatively short distance, due east, to reach over-wintering areas.

The commercial fishery occurs in two seasons: the spring-fall inshore season and the winter offshore season. The commercial fishery is managed using an annual coastwide quota, size limits and gear restrictions. The recreational fishery uses size, season and bag limits. The 2019 stock assessment concluded that black sea bass is not overfished and not experiencing overfishing (NEFSC In Press). See Table 35 and Table 36 for 2020 information on ABCs, ACLs, quotas and recreational harvest limits (NMFS 2018g).

Table 35: Commercial 2020 specifications for summer flounder, scup, and black sea bass

Species	ABC (millions of lb)	ACL (millions of lb)	Quota (millions of lb)
Summer Flounder	25.03	13.52	11.53
Scup	35.77	27.90	22.23
Black Sea Bass	15.07	6.98	5.58

Table 36: Recreational 2020 specifications for summer flounder, scup, and black sea bass

Species	ABC (millions of lb)	ACL (millions of lb)	Harvest Limit (millions of lb)
Summer Flounder	25.03	6.2	5.2
Scup	35.77	7.87	6.51
Black Sea Bass	15.07	8.09	5.81

Description of Gear Usage
For the commercial fishery, trawl, longline, handline, trap/pot, gillnet, and dredge are all authorized gears. For the recreational fishery, rod and reel, handline, pot, trap, and spear are authorized gears. Otter trawls are the predominant gear type used in the commercial fisheries for all three species. Pots/traps are also used to catch black sea bass and scup in the commercial fishery. For summer flounder, VTR data indicate that 96 percent of commercial 2018 landings were caught with bottom otter trawls. All other gear types each accounted for less than 1 percent of the landings. For scup, about 97 percent of the commercial 2018 scup landings reported on VTRs were caught with bottom otter trawls. Pots and sink gillnets each accounted for about 1.7 percent of commercial landings. All other gear types each accounted for less than 1 percent of commercial landings. Although bottom otter trawl is the dominant gear type overall and in federal waters, other gear types such as pots/traps, hand lines, floating traps, and pound nets play a larger role in the summer in some state waters. For black sea bass, VTR data indicate that 72 percent of the black sea bass caught in 2018 was caught with bottom otter trawl gear. About 18 percent were caught with fish pots and traps, 4 percent in offshore lobster traps, and 3 percent with hand lines. Other gear types accounted for less than 1 percent of total commercial catch. As is the case with scup, pots/traps may play a larger role in state waters. Minimum mesh size requirements are summarized in Table 37.

Table 37: Minimum mesh size requirements for summer flounder, scup, and black sea bass

Fishery Management Plan	Minimum Trawl Mesh Size
Summer Flounder	5.5 in diamond or 6.0 in square
Scup	5.0 in diamond
Black Sea Bass	4.5 in diamond

Summer flounder trawler vessels fishing within the Summer Flounder Fishery-Sea Turtle Protection Area are required to use a turtle excluder device as detailed at 50 CFR part 223. Vessels fishing north of Oregon Inlet, NC, are exempted from this requirement from January 15 through March 15 (Figure 16).

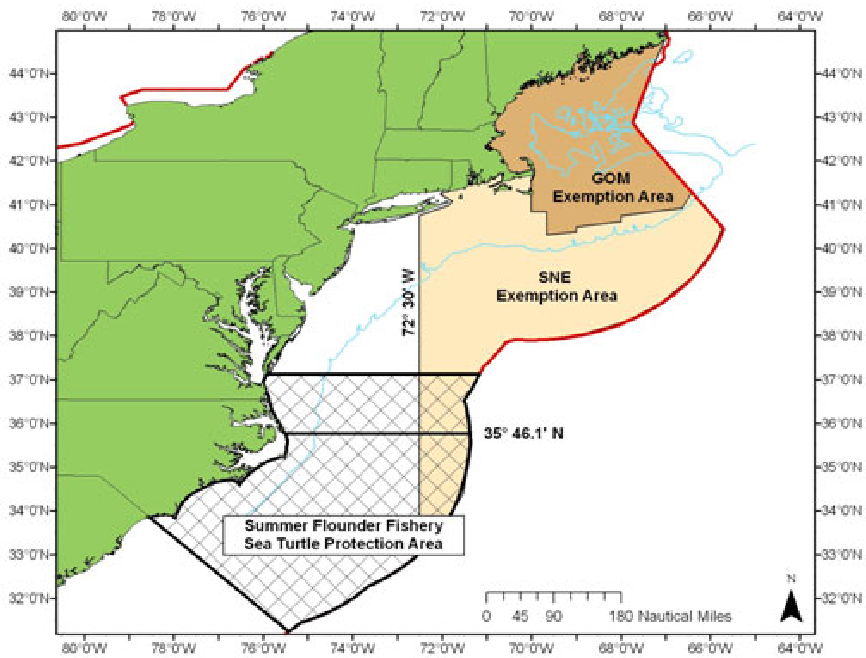

Figure 16: Summer flounder small mesh exemption and sea turtle protection areas

Regulations also restrict certain gear types in two areas off the mid-Atlantic (Figure 17) in order to minimize the mortality of juvenile scup caught as incidental bycatch. Small-mesh gear (i.e., less than 5-inch diamond mesh is prohibited for vessels fishing for longfin squid, black sea bass, or whiting (the primary small-mesh species) in the Northern Gear Restricted Areas from November 1 through December 31 and in the Southern Gear Restricted Areas from January 1 through March 15.

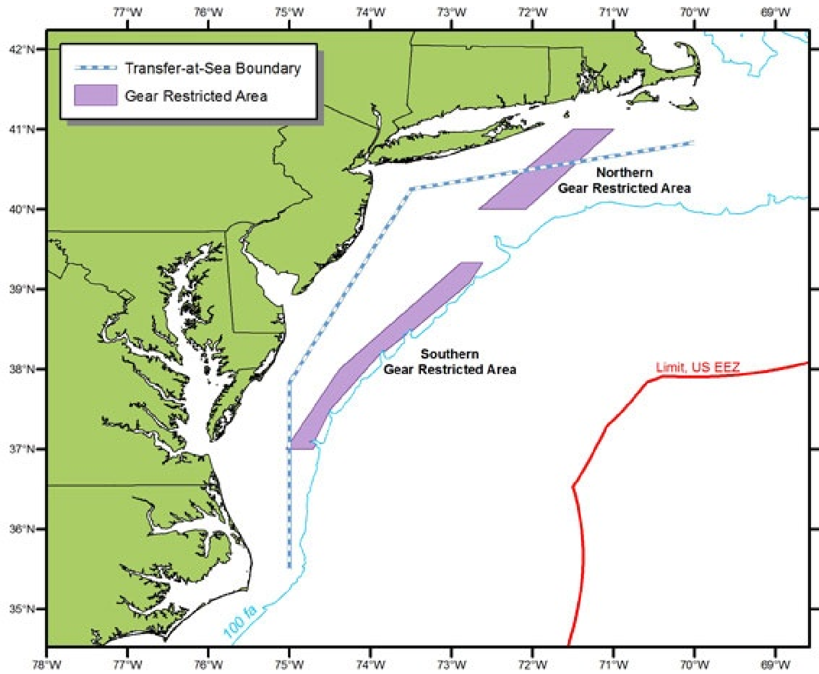

Figure 17: Scup gear restricted areas

Fishing Effort

Summer Flounder: Following the implementation of a coastwide quota in 1993, commercial summer flounder landings have fluctuated between 6 and 18 million lb. Commercial landings of summer flounder peaked in 1984 at 37.77 million pounds and reached a low of 5.83 million pounds in 2017. In 2019, commercial fishermen from Maine through North Carolina landed 9.06 million pounds of summer flounder, about 83 percent of the commercial quota (10.98 million pounds after deductions for prior year landings and discard overages. Recreational landings have varied more with catch peaking in 2010 with 59 million lb to a high of 38 million lb in 1980 to a low of three million lb in 1989. In July 2018, MRIP released revisions to their time series of recreational catch and landings estimates based on adjustments for a revised angler intercept methodology and a new effort estimation methodology (i.e., a transition from a telephone-based effort survey to a mail-based effort survey). The revised estimates of catch and landings are several times higher than the previous estimates for shore and private boat modes, substantially raising the overall summer flounder catch and harvest estimates. On average, the new landings estimates for summer flounder (in pounds) are 1.8 times higher over the time series 1981-2017, and 2.3 times higher over the past 10 years (2008-2017). In 2017, new estimates of landings in pounds were 3.16 times higher than the previous estimates.

Revised MRIP estimates indicate that recreational catch for summer flounder peaked in 2010 with 58.89 million fish caught. Recreational harvest peaked in 1983, with 25.78 million fish landed, totaling 36.74 million pounds. Recreational catch reached a low in 1989 with 5.06 million fish caught. Recreational harvest in numbers of fish reached a low in 2019 with 2.38 million fish landed (7.80 million pounds), while recreational harvest in pounds was lowest in 1989 at 5.66 million pounds. See Figure 18 for more information on commercial and recreational summer flounder landings.

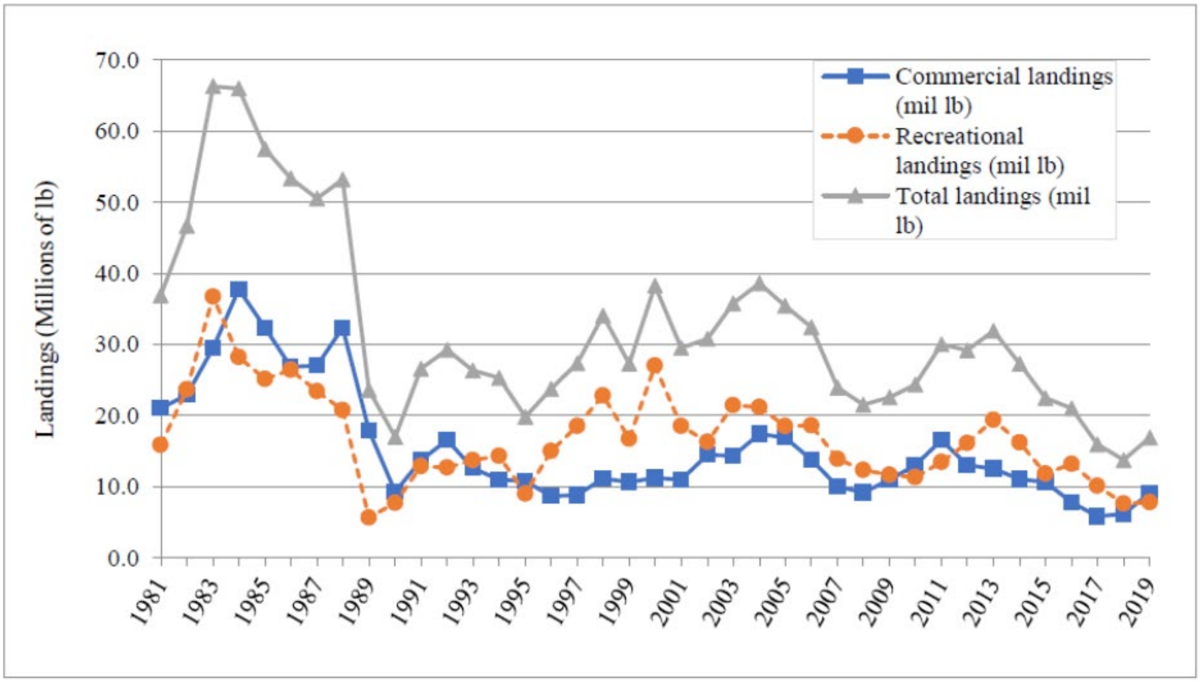

Figure 18: Commercial and recreational summer flounder landings in millions of pounds, Maine- North Carolina, 1981-2019. Recreational landings are based on revised MRIP data

Scup: Commercial scup landings peaked in 1981 at 21.73 million pounds and reached a low of 2.66 million pounds in 2000 (Figure 19). In 2019, commercial fishermen landed 13.78 million pounds of scup, about 57 percent of the commercial quota. Landings over the last decade have ranged from about 5 million lb (2008) to 17 million lb (2013). The recreational fishery accounts for a large portion of the catch of scup. From 1981-2019, recreational catch of scup peaked in 2017 at 41.20 million scup and landings peaked in 1986 with an estimated 30.43 million scup landed by recreational fishermen from Maine through North Carolina. Recreational catch was lowest in 1998 when an estimated 6.86 million scup were caught and 2.74 million scup were landed. Recreational anglers from Maine through 14 North Carolina caught an estimated 28.67 million scup and landed 14.95 million scup (about 14.12 million pounds) in 2019. See Figure 19 for more information on scup total landings (MAFMC 2019b).

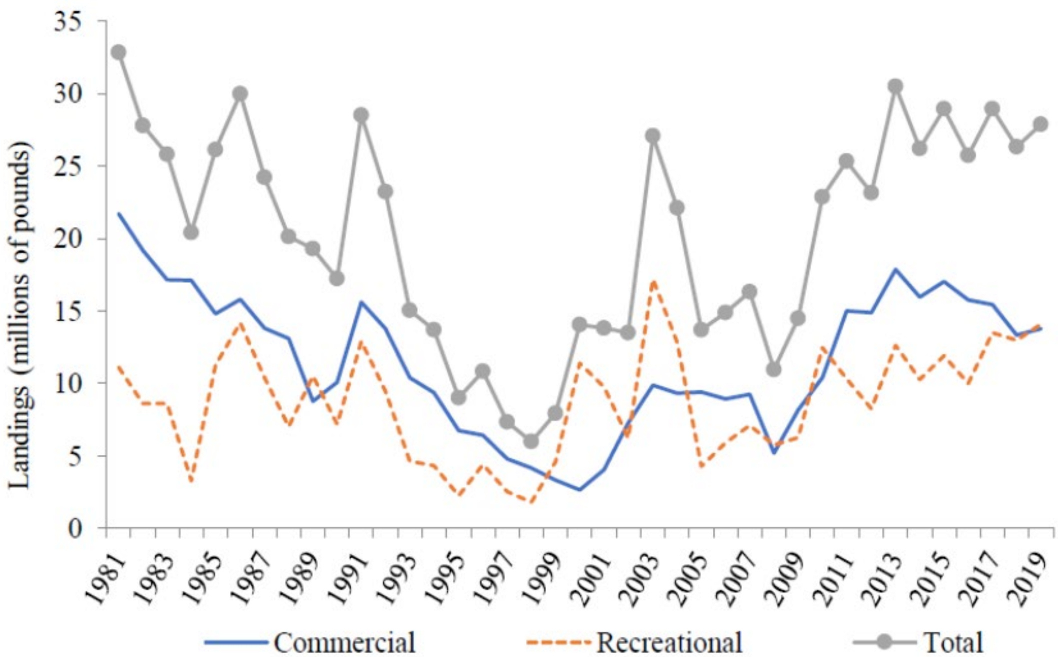

Figure 19: Commercial and recreational scup landings, Maine-North Carolina, 1981-2019. Recreational landings are based on the new MRIP numbers.

Black Sea Bass: Commercial black sea bass landings peaked at 22 million lb in 1952, following the emergence of the trap fishery. Following the implementation of quotas, average landings have ranged from 1.2 to above 3.99 million lb (in 2017). About 3.53 million pounds of black sea bass were landed by commercial fishermen in 2019, very close to the commercial quota of 3.52 million pounds. In recent years, the recreational harvest of black sea bass was highest in 2016 (12.05 million lb). In 2018, an estimated 7.92 million lb were harvested by recreational anglers. In 2019, an estimated 4.38 million black sea bass, at about 8.61 million pounds, were harvested by recreational anglers from Maine through Cape Hatteras, North Carolina (Table 38, Figure 20). Harvest prior to 2020 should not be compared against the respective recreational harvest limits as the recreational harvest limits prior to 2020 do not account for the recent changes in the MRIP estimation methodology. In 2019, 62 percent of black sea bass harvested by recreational fishermen from Maine through North Carolina (in numbers of fish) were caught in state waters and about 38 percent in federal waters. Most of the recreational harvest in 2019 was landed in New York (36 percent), followed by New Jersey (19 percent), Massachusetts, Rhode Island, and Connecticut (12 each percent). See Table 38 and Figure 20 for more information on black sea bass landings.

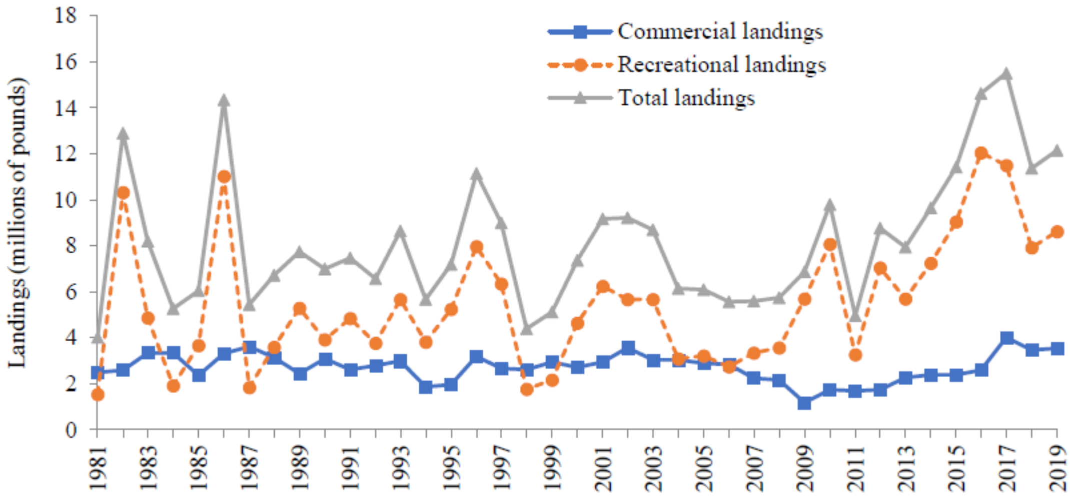

Figure 20: Commercial and recreational black sea bass landings in millions of pounds from Maine through Cape Hatteras, North Carolina, 1981-2019. Recreational landings are based on the revised MRIP estimates

Table 38: Commercial landings of summer flounder, scup, and black sea bass, 2019

Species	Landings (millions of lb)
Summer Flounder	9.06
Scup	13.4
Black Sea Bass	3.53

3.4.9. *Description of the Current Atlantic Deep-Sea Red Crab Fishery*

The New England Fishery Management Council manages the Atlantic deep-sea red crab fishery through the Atlantic Deep-Sea Red Crab Fishery Management Plan. This FMP uses a quota system comprised of a total allowable landings limit, within an annual catch limit and AM framework. Very little is known about the life history and stock status of deep-sea red crab, but the New England Council's SSC has indicated that the long-term average landings by the directed red crab fleet appears to be sustainable. Even though the overfishing limit is unknown, the SSC considers long-term average landings to be sufficiently below whatever that value is likely to be. Unlike most fisheries, no reliable discard estimate could be determined for red crab. Historically, the acceptable biological catch, annual catch limit, and total allowable landings are currently equal to the long-term average landings of 3.91 million lb of male crabs (NMFS 2017g). More recently, NMFS approved an increase in the acceptable biological catch for fishing years 2020-2023, as recommended by the SSC and approved by the Council, as summarized in Table 39.

Table 39: Atlantic deep-sea red crab 2011-2023 specifications

	2011-2019	2020-2023
Maximum Sustainable Yield	Undetermined	Undetermined
Overfishing Limit	Undetermined	Undetermined
Optimum Yield	Undetermined	Undetermined
Acceptable Biological Catch	1,775 mt	2,000 mt
Annual Catch Limit	1,775 mt	2,000 mt
Total Allowable Landings	1,775 mt	2,000 mt

The fishery takes place year-round. While the average landings vary seasonally and can be limited by market demand, landings are typically highest in summer and fall. Red crabs occur in a patchy distribution from Nova Scotia to Florida, primarily at depths of 400-1800 m along the continental shelf and slope. Figure 21 displays the statistical area groupings used to describe regions where Atlantic deep-sea red crabs are caught (Georges Bank/southern New England (1), New Jersey (2) and Delmarva (3) areas) (NEFMC 2020a).

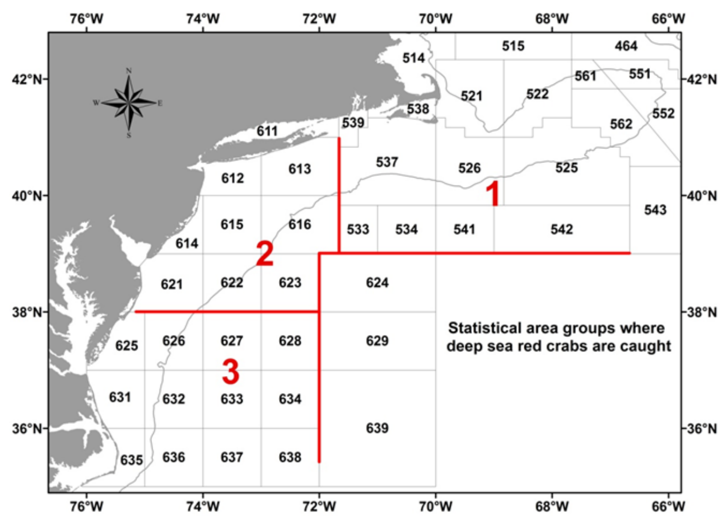

Figure 21: Atlantic deep-sea red crab harvest regions

At present, the Red Crab FMP contains no closures. However, the New England Council has submitted an omnibus FMP action to NMFS that would protect coral. This action would prohibit the use of trap gear in the Georges Bank Deep-Sea Coral Protection Area, and the red crab fishery is exempt from these restrictions. The Council has no immediate plans to revisit this exemption.

Description of Gear Usage

Red crab traps/pots (as defined in 50 CFR 648.2) are "any structure or other device, other than a net or parlor trap/pot, that is placed, or designed to be placed, on the ocean bottom and is designed for, or is capable of, catching red crabs." Each trap may not be larger than 18 cubic feet, and they may be rectangular, trapezoidal, or conical only, unless otherwise allowed by the Regional Administrator. The most common trap used is conical in shape (NEFMC 2002). Traps are set in trawls of typically 150 traps per trawl. Red crab fishing vessels are restricted to 600 crab pots. This equates to a maximum of approximately 40 vertical lines used in the fishery if all five limited access permitted vessels were active. More recent information suggests that the red crab fishery has deployed either 24 or 32 vertical lines annually, over the last 10 years (NEFMC 2020a) There is some amount of gear loss or damage on every trip. The reported average for pot loss or damage is just over 10 pots per trip. The average soak time of the baited traps is approximately 24 hours (NEFMC 2020a). Traps are hauled one at a time, and the catch sorted immediately (NEFMC 2002). Incidental harvesters may land red crab with gear authorized and regulated in other fishery management plans.

Fishing Effort

The majority of permits issued in for the red crab fishery are an open access, incidental permit category that allows a small amount of red crabs to be landed while participating in other fisheries. While these account for the vast majority of permits issues, these vessels account for less than 1 percent of landings (NEFMC 2016a, 2020a). The targeted red crab fishery is comprised of a small number of vessels (Table 40) with limited access permits (Category B and C permits). The majority of these vessels' revenue is generated from red crab landings. Limited access red crab vessels may fish with or carry on board up to 600 traps/pots when fishing for red crab, for 3,000 traps total authorized for use in the fishery.

Table 40: Number of permits in the Atlantic deep-sea red crab fishery in 2018

Permit Category	Description	Permits Issued
Category A	Open Access	~1,300
Category B	Limited Access	4
Category C	Limited Access	1

Updated information indicates that landings have increased since 2013, and in 2018, the landings were among the highest since implementation of the Red Crab FMP in 2002 (Figure 22). Incidental landings by vessels not targeting red crabs were nearly zero. Annual landings by region are one measure of the spatial extent of the fishery over the year. Recent data indicate that landings have increased from Region 2, compared with early years in the fishery when most of the landings were concentrated from Region 1. Landings from Region 3 have stabilized since 2013 after being highly variable in previous years (Figure 23) (NEFMC 2016a, 2020a).

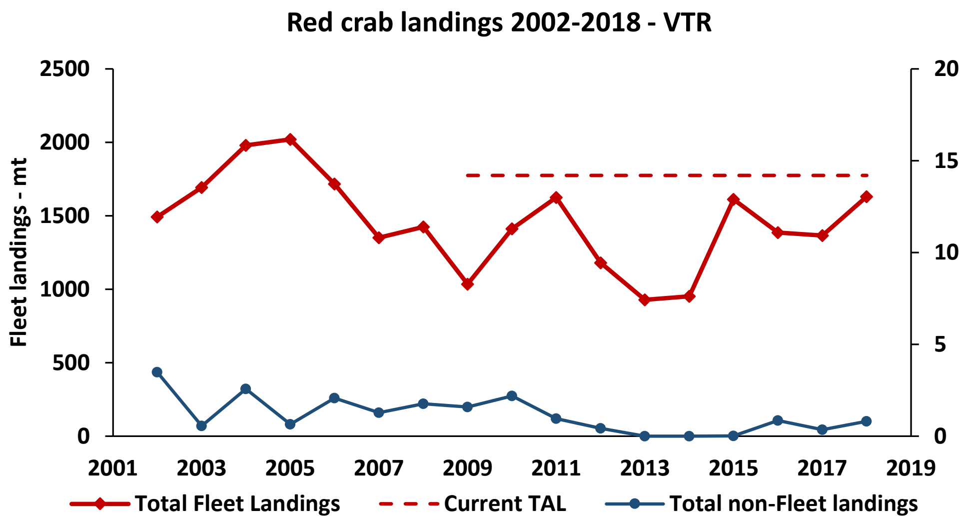

Figure 22: Atlantic deep sea red crab landings (mt), 2002-2015

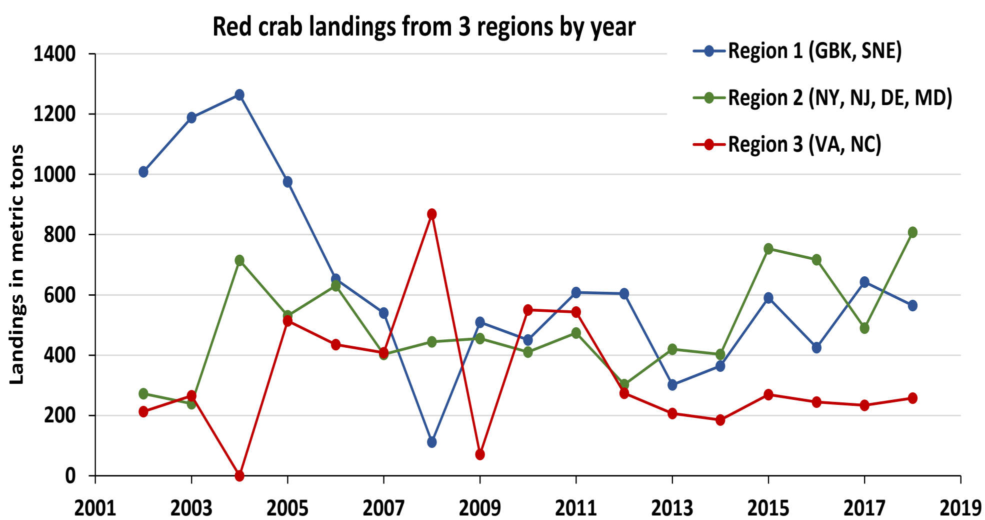

Figure 23: Atlantic deep-sea red crab landings by region, 2002-2015

3.4.10. ***Exempted Fishing, Education, and Research Permits***

Regulations at 50 CFR 600.745 allow the Regional Administrator to authorize the targeted or incidental harvest of species managed under an FMP or fishing activities that would otherwise be prohibited for scientific research, limited testing, public display, data collection, exploration, health and safety, environmental cleanup, hazardous waste removal purposes, or for educational activities. Every year, GARFO may issue a small number of exempted fishing permits (EFP), scientific research permits (SRP), and/or exempted educational activity authorizations (EEAA) exempting the collection of a limited number of species from Northeast federal waters from regulations implementing the appropriate FMP. EFPs and EEAAs involve fishing by commercial or research vessels that use similar or identical fishing methods as the fisheries that are the subject of this Opinion. The only differences with these projects are typically (a) the use of modified gear, which was not authorized under the specific FMP at the time, or (b) requests for additional DAS or trips to closed areas beyond what the annual specifications for the fishery allowed. A SRP covers similar types of activities as an EFP in terms of authorizing minor changes to the regulations, but they are usually issued to scientific research vessels. Table 41 shows the number of EFPs, EEAAs, and SRPs for each fishery issued by GARFO from 2014 to 2020.

Table 41: Number of EFPs, EEAAs and SRPs issued by GARFO (2014-2020)

FMP	EFPs	EEAAs	SRPs
Northeast Multispecies	52	17	18
Summer flounder/Scup/Black sea bass	16	0	3
Spiny dogfish	0	0	1
Squid/mackerel/butterfish	2	0	1
Bluefish	0	0	0
Skate	3	0	1
Monkfish	16	0	0
Lobster	26	0	0
Jonah Crab	2	0	0
Total	**117**	**17**	**24**

These research activities have previously been determined to be small in both scale (number of participating vessels, amount of gear, etc.) and effort (number of trips) compared to the overall fisheries. For the EFPs, EEAAs, and SRPs examined between 2014 and 2020, we were able to conclude that, in all cases, the types and rates of interactions with listed species from the EFP, EEAA, and SRP activities would be similar to those analyzed in their respective biological opinions. Given our past experience with and knowledge of the usual applicants (and when and where they fish), we expect that future EFPs, EEAAs, and/or SRPs would propose fishing types and associated fishing effort similar to previous EFPs/EEAAs/SRPs and, therefore, not introduce a significant increase in effort levels for the ten fisheries considered in this Opinion. For example, issuance of an EFP to an active commercial vessel that is similar to the ones described above likely does not add additional effects compared to those that would otherwise accrue from the vessel's normal commercial activities. Similarly, issuance of an EFP, EEAA, or SRP to a vessel to conduct a minimal number of tows/trips with gear used in the fisheries likely would not add sufficient fishing effort to produce a detectable change in the overall amount of fishing effort in a given year. Therefore, we consider the future issuance of most SRPs, EFPs and EEAAs by GARFO to be within the scope of this Opinion. If an SRP, EFP or EEAA is proposed which modifies this agency action in a manner that causes an effect to listed species or critical habitat not considered in this Opinion (i.e., is beyond the scope of the fishery activity considered), then additional section 7 consultation would be necessary.

3.5. Action Area

Action area means all areas affected directly, or indirectly, by the federal action, and not just the immediate area involved in the action (50 CFR. §402.02). For the purposes of this Opinion, the action area encompasses the area in which the ten fisheries operate, broadly defined as all U.S. EEZ waters from Maine through Key West, FL. This includes state waters (0-3 nmi) as vessels fishing in the federal fishery transit to the fishing grounds through these waters.

4. STATUS OF THE SPECIES

ESA-listed species and designated critical habitat occur in the action area of the proposed action. Table 42 summarizes those species and critical habitat that occur in the action area[4] and that may be adversely affected (e.g., there have been observed or documented interactions in the fisheries or with gear type(s) similar to those used in the fisheries). Section 4.1 details which species and critical habitat are not likely to be adversely affected by the proposed action because the effects of the proposed action are deemed insignificant, discountable, or completely beneficial. Section 4.2 summarizes the biology and ecology of those species that may be adversely affected by the proposed action and details information on their life histories in the action area, if known.

Table 42: ESA-listed species and designated critical habitat in the action area of the proposed action

Species	Status	Potential to be adversely affected by the proposed action?
Marine Mammals: Cetaceans		
Blue whale (*Balaenoptera musculus*)	Endangered	No
North Atlantic right whale (*Eubalaena glacialis*)	Endangered	Yes
Fin whale (*Balaenoptera physalus*)	Endangered	Yes
Sei whale (*Balaenoptera borealis*)	Endangered	Yes
Sperm whale (*Physeter macrocephalus)*	Endangered	Yes
Marine Reptiles: Sea Turtles		
Green sea turtle (*Chelonia mydas*), North Atlantic DPS	Threatened	Yes
Green sea turtle (*Chelonia mydas*), South Atlantic DPS	Threatened	No
Hawksbill sea turtle (*Eretmochelys imbricata*)	Endangered	No
Kemp's ridley sea turtle (*Lepidochelys kempii*)	Endangered	Yes
Loggerhead sea turtle (*Caretta caretta*), Northwest Atlantic Ocean DPS	Threatened	Yes
Leatherback sea turtle (*Dermochelys coriacea*)	Endangered	Yes
Fish		
Atlantic salmon (*Salmo salar*)	Endangered	Yes
Atlantic sturgeon *(Acipenser oxyrinchus)*		
Gulf of Maine DPS	Threatened	Yes
New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs	Endangered	Yes

[4] The previous biological opinion on the lobster fishery (NMFS 2014b) found that while there was a potential for lobster fishing activity to occur within the GOM DPS of Atlantic salmon critical habitat, it was not likely to adversely affect the designated critical habitat for the GOM DPS of Atlantic salmon. This fishing would have been by dually-permitted (i.e., state and federally permitted) vessels. Given that this Opinion is only considering fishing activity in federal waters, it does not overlap with the designated critical habitat. Vessels participating in the fisheries in this Opinion are also not expected to transit through the critical habitat. The GOM DPS of Atlantic salmon critical habitat is outside the action and area and will not be considered here. Critical habitat for the five listed DPSs of Atlantic sturgeon, which is found in rivers, is similarly outside the action area.

Species	Status	Potential to be adversely affected by the proposed action?
Giant manta ray (*Manta birostris)*	Threatened	Yes
Nassau grouper (*Epinephelus striatus*)	Threatened	No
Oceanic whitetip shark (*Carcharhinus longimanus*)	Threatened	No
Shortnose sturgeon (*Acipenser brevirostrum*)	Endangered	No
U.S. DPS of smalltooth sawfish (*Pristis pectinata*)	Endangered	No
Seagrass		
Johnson's seagrass (*Halophila johnsonii* Eiseman)	Threatened	No
Coral		
Elkhorn coral (*Acropora palmata*)	Threatened	No
Staghorn coral (*Acropora cervicornis*)	Threatened	No
Lobed star coral (*Orbicella annularis*)	Threatened	No
Mountainous star coral (*Orbicella faveolata*)	Threatened	No
Boulder star coral (*Orbicella franksi*)	Threatened	No
Pillar coral (*Dendrogyra cylindrus*)	Threatened	No
Rough cactus coral (*Mycetophyllia ferox*)	Threatened	No
Critical Habitat		
North Atlantic right whale	Designated	No
Northwest Atlantic DPS of loggerhead sea turtle	Designated	No
U.S. DPS of smalltooth sawfish	Designated	No
Johnson's seagrass	Designated	No
Elkhorn and staghorn corals	Designated	No

4.1. Species and Critical Habitat Not Likely to be Adversely Affected by the Proposed Actions

As indicated in Table 42, we have determined that the actions considered in this Opinion are not likely to adversely affect a number of species that are listed as threatened or endangered under the ESA. Additionally, we have determined that the proposed actions are not likely to adversely affect any designated critical habitat found in the action area (Table 42). Destruction or adverse modification of critical habitat is a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species (50 CFR 402.2). Below, we present our rationale for our "not likely to adversely affect" determinations.

4.1.1. *Blue Whale*

Blue whales do not regularly occur in waters of the U.S. EEZ (Lesage et al. 2018, Waring et al. 2010). Over the last 48 years, there have only been 42 sightings of blue whales in waters of the U.S. EEZ from Maine to Key West, Florida reported in OBIS SEAMAP (http://seamap.env.duke.edu/). This is less than one blue whale sighting per year within the action area. In a recent study on the seasonal acoustic occurrence of whales in the New York Bight, researchers detected blue whales, using passive acoustic monitoring, on 11 percent of the survey days (Muirhead et al. 2018). The whales were detected from January to March, and detections increased with recorder distance from shore, suggesting that the individuals occurred to the seaward, offshore end of the recording array (which extended to the shelf edge) or beyond. A single blue whale was also tracked moving south-southwest along the shelf edge (Muirhead et al. 2018). Given the limited co-occurrence between blue whales and the fisheries in this Opinion, effects to blue whales from the operation of any of the ten fisheries are extremely unlikely. This conclusion is further supported by the information on observed and documented U.S. Atlantic fishery-related interactions. In 1986, a blue whale was documented on Stellwagen Bank with gear around its flipper; the gear type was not confirmed and its origin was uknown (Waring et al. 1999). There have been other records since then and no observed or documented U.S. Atlantic fishery-related M/SIs to blue whales to date (Henry et al. 2017, Henry et al. 2015, 2016, Henry et al. 2019, Waring et al. 2010). Based on this information, effects of the fisheries on blue whales are extremely unlikely and, therefore, discountable[5].

4.1.2. ***Green Sea Turtle, South Atlantic DPS***

The green sea turtle is the largest of the hardshell marine turtles. Two DPSs occur within the action area, the North Atlantic and South Atlantic DPSs. Both DPSs are listed as threatened under the ESA (81 FR 20058, April 6, 2016). While all of the mainland U.S. nesting individuals are part of the North Atlantic DPS, the U.S. Caribbean nesting assemblages are split between the North Atlantic and South Atlantic DPS. Differences in mitochondrial DNA properties of green sea turtles from different nesting regions indicate there are genetic subpopulations (Bowen et al. 1992, FitzSimmons et al. 2003). Despite the genetic differences, sea turtles from separate nesting origins are commonly found mixed together on foraging grounds throughout the species' range. Within U.S. waters of the Atlantic, individuals from both the North Atlantic and South Atlantic DPSs can be found on foraging grounds. There are currently no in-depth studies available to determine the percent of North Atlantic and South Atlantic DPS individuals in any given location. However, one small-scale study on the Atlantic coast of Florida (off Hutchinson Island) found that approximately 5 percent of the turtles sampled came from the Aves Island/Suriname nesting assemblage, which is part of the South Atlantic DPS (Bass and Wayne 2000). All of the individuals in both studies were benthic juveniles. Available information on green turtle migratory behavior indicates that only juvenile turtles display long distance dispersal, suggesting that larger adult-sized turtles return to forage within the region of their natal rookeries, thereby limiting the potential for gene flow across larger scales (Monzón-Argüello et al. 2010).

Of the ten fisheries in this Opinion, only the bluefish fishery operates in waters commonly used by green sea turtles from the South Atlantic DPS as it is the only fishery to extend south of North Carolina. The management unit for the Bluefish FMP extends from Maine through Key West, Florida (MAFMC and ASMFC 1998). In Georgia and South Carolina, effort in the bluefish fishery is de minimis (ASMFC 2019c). Therefore, our analysis of bluefish effort south of North Carolina is restricted to Florida. Takes of green sea turtles from the South Atlantic DPS (see section 4.2.2.1 for the North Atlantic DPS) in the bluefish fishery are considered extremely unlikely given:

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[5] When the terms "discountable" or "discountable effects" appear in this document, they refer to potential effects that are found to support a "not likely to adversely affect" conclusion because they are extremely unlikely to occur. The use of these terms should not be interpreted as having any meaning inconsistent with our regulatory definition of "effects of the action."

1. Commercial fishing effort in Florida for bluefish is limited. The majority of the commercial bluefish landings from 2012-2017 were in North Carolina, Rhode Island, New York, Massachusetts, and New Jersey (MAFMC 2013, 2014, 2015, 2016, 2017, 2018a). In 2018, North Carolina and New York continued to account for the majority of landings followed by Florida. Landings continued to be predominately in the mid-Atlantic and north, with 82 percent of landings from seven statistical areas (Figure 24). Approximately 13 percent (approximately 326,000 lb) of landings were from Florida (ASMFC 2019c, MAFMC 2019a). There was only one port in Florida where landings were greater than 100,000 lb, accounting for 6 percent of the commercial landings. At this port, the landings were from three vessels (MAFMC 2019a).

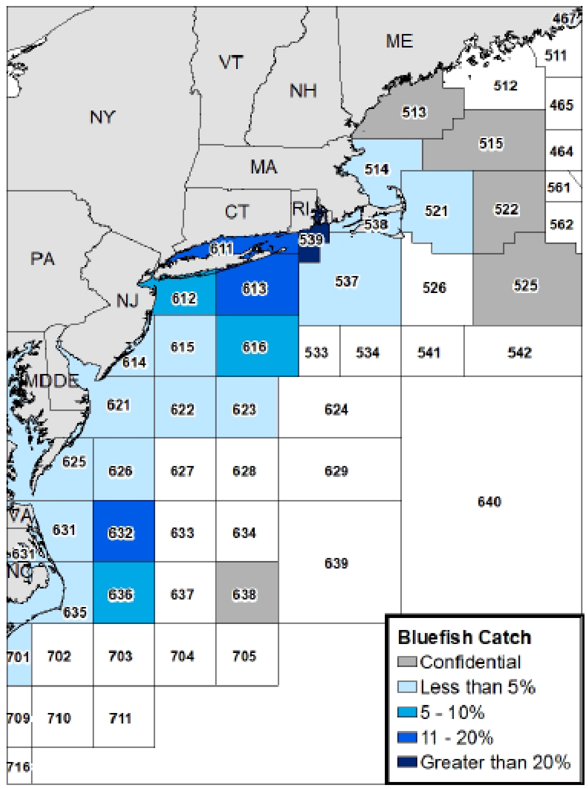

Figure 24: NMFS statistical areas accounting for a percentage of the commercial bluefish landings in 2018 (MAFMC 2019a)

2. Green sea turtles on the foraging grounds are associated with seagrass habitats, which are shallow water habitats. Bluefish are a migratory pelagic species (i.e., occupying the water column not near the bottom or coast) (Shepherd and Packer 2006). Fishing effort in this Opinion occurs in federal waters. This difference in habitat preferences and the spatial extent of the fishery limits the overlap of green sea turtles and the bluefish fishery.
3. Bluefish is a restricted species (Fla. Stat. §379.101(32), requiring an endorsement to harvest commercially, in Florida (68B-43.001) and subject to possession and gear restrictions (68B-43.005). Nets in federal waters adjacent to Florida state waters must be tended and soak times are limited to one hour (68B-43.005). When nets are tended, it is more likely that sea turtles at the surface will be detected.
4. Recreational effort targeting bluefish in waters off Florida is also very low. In 2016 and 2017, the majority of the recreational landings have come from New Jersey, North

Carolina, and New York (MAFMC 2018a, 2019a). In 2018, the greatest overall catches occurred in North Carolina, Florida, and New Jersey (MAFMC 2019a). Bluefish were primarily caught by hook and line from shore or private boats and are often landed while targeting other species. To better understand the recreational fishery targeting bluefish in Florida, we obtained data from the Marine Recreational Information Program (MRIP). From 2014-2018, MRIP conducted 54,943 interviews in Atlantic Florida. During these interviews, bluefish were recorded on approximately 2.5 percent (1,351) of all interviews. Of the interviews where bluefish were caught, only 6 percent identified bluefish as a primary or secondary target. This indicates that approximately 0.15 percent of the recreational effort is targeting bluefish. This very low effort limits the likelihood that the recreational fishery will incidentally capture a sea turtle.

5. Bluefish are fairly active and recreational fishermen catch them by casting the lure or bait into the water and retrieving it (jigging). They may cast into schools of bluefish. While sea turtles are vulnerable to capture on hook and line gear, the techniques used in the bluefish fishery makes the effects extremely unlikely. Foraging green sea turtles are unlikely to be snagged by jigged gear as it is deployed near the surface and constantly reeled back to the boat. It is possible a sea turtle could become snagged if it comes into contact with the jigged hook, but the chances of that occurring are extremely low.
6. Interactions with vessels operating in the fishery are also unlikely to occur given the limited overlap of the fishery and South Atlantic DPS of green sea turtles.

Due to the limited distribution of the South Atlantic DPS of green sea turtles in the action area, the commercial gear requirements, and the operations of the recreational fishery, it is extremely unlikely that this species would interact with fishing gear utilized in the bluefish fishery; therefore, effects are discountable.

4.1.3. ***Hawksbill Sea Turtle***

The hawksbill sea turtle is listed as endangered. This species is uncommon in the waters of the continental United States. Hawksbills prefer coral reef habitats, such as those found in the Caribbean and Central America. Within the U.S. territories, Mona Island (Puerto Rico) and Buck Island (St. Croix, U.S. Virgin Islands) contain especially important foraging and nesting habitat for hawksbills (NMFS and USFWS 1993). Within the continental United States, nesting is restricted to the southeast coast of Florida and the Florida Keys, but nesting is rare in these areas. Hawksbills have been recorded from all Gulf of Mexico states and along the U.S. east coast as far north as Massachusetts, but sightings north of Florida are rare. Many of the strandings in states north of Florida have been after hurricanes or offshore storms. Aside from Florida, Texas is the only other U.S. state where hawksbills are sighted with any regularity (NMFS and USFWS 1993). The Sea Turtle Stranding and Salvage Network (STSSN) can also provide some information on relative abundance of sea turtles as all species in a given area face similar threats. Therefore, the standings can provide some information on the proportion of sea turtles in an area. From 2008-2017, an annual average of 14.5 hawksbill sea turtles stranded from all causes in Atlantic Florida. This represented approximately 1.3 percent of all strandings in this area (NMFS STSSN, unpublished data).

Of the ten fisheries in this Opinion, only the bluefish fishery operates in waters commonly used by hawksbill sea turtles, as it is the only fishery to extend south of North Carolina. The management unit for the Bluefish FMP extends from Maine through Key West, Florida (MAFMC and ASMFC 1998). In Georgia and South Carolina, effort is de minimis (ASMFC

2019c). Therefore, our analysis of bluefish effort south of North Carolina is restricted to Florida. Takes of hawksbill sea turtles in the bluefish fishery considered extremely unlikely given:

1. As described above, commercial fishing effort in Florida for bluefish is low, and gear restrictions apply to vessels possessing bluefish (68B-43.05), making interactions between the bluefish fishery and hawksbill sea turtles unlikely.
2. Hawksbill sea turtles are commonly associated with coral reefs. In Florida, the coral reefs are shallow water reefs that extend from the Dry Tortugas in the Florida Keys to St. Lucie Inlet on the Atlantic coast (Rohmann and Monaco 2005). Bluefish are a migratory pelagic species (i.e., occupying the water column not near the bottom or coast) (Shepherd and Packer 2006). This difference in habitat preferences limits the overlap of hawksbills and the fishery.
3. Gillnet gear is the primary gear (50 percent of 2018 landings) used to commercially land bluefish. Gears used to a lesser extent are unknown (26 percent), bottom trawl (9 percent) and other (9 percent) (MAFMC 2019a). Gillnet fishing is prohibited in Florida state waters (Florida Administrative Code 688-4.0081), and nets in federal waters adjacent to Florida state waters must be tended and soak times are limited to one hour (68B-43.005), making it more likely that sea turtles at the surface would be detected.
4. Bottom trawl would not be fished in coral reef areas. This further limits the overlap of hawksbill sea turtles and the fishery.
5. Recreational fishermen targeting bluefish in Florida represent approximately 0.15 percent of the recreational fishery (see above). This extremely low effort limits the likelihood that the recreational fishery will incidentally capture a sea turtle.
6. As described above, bluefish are fairly active and fishermen catch them by casting the lure or bait into the water and retrieving it (jigging). Sea turtles are unlikely to be snagged by jigged gear as it is deployed near the surface and constantly reeled back to the boat. It is possible a sea turtle could become snagged if it comes into contact with the jigged hook, but the chances of that occurring are extremely low.
7. Interactions with vessels operating in the fishery are also unlikely to occur given the limited overlap of the fishery and hawksbill sea turtles.

Due to the species' tropical distribution, the lack of documented interactions in the fishery, the commercial gear requirements, and the operations of the recreational fishery, it is extremely unlikely that hawksbill sea turtles would interact with fishing gear utilized in the bluefish fishery; therefore, effects are discountable.

4.1.4. ***Nassau Grouper***

The Nassau grouper (*Epinephelus striatus*) is a reef fish, but it transitions through a series of habitats. As larvae, they are planktonic. As juveniles, they inhabit nearshore shallow waters in macroalgal and seagrass habitats. With increasing size and maturation, they shift primarily to reef habitat (Hill and Sadovy de Mitcheson 2013). They inhabit waters from the shoreline to about to 426 feet (Hill and Sadovy de Mitcheson 2013). Nassau grouper are mostly absent from the continental United States, except Florida, where larger juveniles and adults have been recorded. In the action area, its confirmed distribution is limited to southern Florida (Figure 25). A number of surveys have collected information on Nassau group in Florida waters. A Florida Fish and Wildlife Conservation Commission survey from 1999-2007 observed 79 Nassau grouper between 35 and 70 cm in length (Letter from J. McCawley, Director, Division of Marine Fisheries Management to NMFS Southeast Regional Office as cited in Hill and Sadovy de

Mitcheson, 2013). The Reef Environmental Education Foundation (REEF) reported 1,322 Nassau grouper in 9,706 surveys over a 10-year period (2003-2013). Surveys up the east coast of Florida to Jupiter Inlet reported 83 Nassau grouper in 6763 surveys (Hill and Sadovy de Mitcheson 2013). The most serious threats to Nassau grouper are fishing at spawning aggregations and inadequate law enforcement. No spawning aggregation sites have been reported in Florida (Hill and Sadovy de Mitcheson 2013).

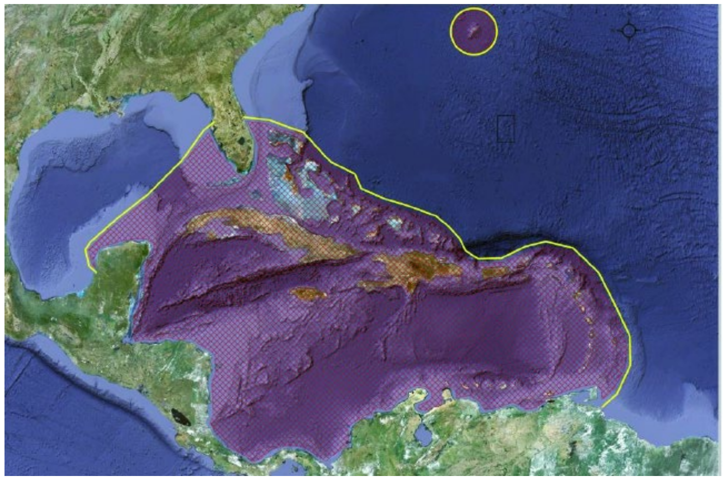

Figure 25: Range of Nassau grouper (Hill and Sadovy de Mitcheson 2013)

Of the ten fisheries, only the bluefish fishery may occur in waters typically used by Nassau grouper. The NEFSC Observer Program observes the fisheries in this Opinion, including the bluefish fishery, north of Cape Hatteras; given this, together with the geographical range of Nassau grouper, there is no information available on observed interactions with Nassau grouper in the bluefish fishery. Takes of Nassau grouper in the bluefish fishery are extremely unlikely for a number of reasons, including:

1. Nassau grouper and bluefish occupy and use the water column differently. As juveniles, Nassau grouper inhabit nearshore shallow waters in macroalgal and seagrass habitat. This area is outside the area where fishing activity considered in this Opinion is occurring. As adults, Nassau grouper are a relatively sedentary reef-fish species. In contrast, bluefish are a migratory pelagic species. The use of different parts of the water column and habitat by Nassau grouper and bluefish makes it unlikely that commercial or recreational fishermen targeting bluefish would capture Nassau grouper.
2. Nassau grouper occurs only in southeast Florida, which is the most southern extent of the bluefish fishery. This limits the overlap between the bluefish fishery and Nassau grouper. In the commercial fishery, the majority of landings (82 percent) in 2018 came from seven statistical areas, all of which were north of Cape Hatteras. While Florida had one port where more than 100,000 of bluefish were landed in 2018, these landings accounted for only 6 percent of the total commercial landings and were from three vessels (ASMFC 2019c, MAFMC 2019a). All of these factors limit the overlap of the commercial bluefish fishery and Nassau grouper.
3. As described above, from 2014-2018, MRIP documented bluefish on approximately 2.5 percent of the interviews conducted in Atlantic Florida. MRIP is a survey of recreational fishermen and does not include commercial vessels. Of these, only 6 percent of interviews had bluefish identified as a primary or secondary target. During this same

period, less than 0.5 percent of interviews that were positive for bluefish were also positive for Nassau grouper (includes all interviews where bluefish were documented, regardless of if they were targeted). In all cases where bluefish and Nassau grouper were recorded during the same interview, bluefish was not identified as a primary or secondary target species. Given the limited effort in the recreational bluefish fishery in this area and the limited overlap of the two species, it is unlikely bluefish recreational hook-and-line would interact with Nassau grouper.

4. Interactions with vessels transiting to the fishing grounds are also considered unlikely to occur. Nassau grouper are primarily demersal and would rarely be at risk from moving vessels which need sufficient water to navigate without encountering the bottom. When operating in areas with marginal clearance, vessels generally transit these areas slowly, allowing the species an opportunity to move out of the way. In addition, there is very limited overlap with vessels participating in the fisheries considered in this Opinion.

Given the limited overlap of the fishery and Nassau grouper; the different habitats used by bluefish and Nassau grouper, and the lack of documented interactions between this fishery and Nassau grouper, it is extremely unlikely that Nassau grouper would interact with fishing gear utilized in the bluefish fishery; therefore, effects are discountable.

4.1.5. ***Oceanic Whitetip Shark***

In the western Atlantic, oceanic whitetip sharks (*Carcharhinus longimanus*) occur from Maine to Argentina, including the Caribbean and Gulf of Mexico. It is a highly migratory species that is usually found offshore in the open ocean, on the outer continental shelf, or around oceanic islands (Bonfil et al. 2008, Young et al. 2017). The species can be found in waters temperatures between 15 °C and 28 °C, but it exhibits a strong preference for the surface mixed layer in water with temperatures above 20 °C (Bonfil et al. 2008) and is considered a surface-dwelling shark. Little is known about movements or possible migration paths (Young et al. 2017). Currently, the most significant threat to oceanic whitetip sharks is mortality in commercial fisheries, largely driven by demand of the international shark fin trade and bycatch-related mortality, as well as illegal, unreported, and unregulated fishing. Oceanic whitetip sharks are generally not targeted, but they are frequently caught as bycatch in many global fisheries, including pelagic longline fisheries targeting tuna and swordfish, purse seine, gillnet, and artisanal fisheries (Young et al. 2017).

Although some of these gear types known to interact with oceanic whitetip sharks are utilized in the fisheries considered in this Opinion, these sharks are found farther offshore in the open ocean, on the continental shelf or around oceanic islands in deep water greater than 600 ft (184 m). They have a strong preference for the surface mixed layers in waters warmer than 20 °C (Young et al. 2017). Given the more offshore distribution of oceanic whitetip sharks, little overlap between fishing gear and oceanic whitetip sharks is expected. For the fisheries that have a larger offshore component (e.g., lobster and red crab fisheries), interactions are extremely unlikely as these fisheries use trap/pot gear, a gear type not known to interact with this species. Other gear types (e.g., bottom trawls) and fisheries (e.g., squid, Northeast multispecies) may also operate in offshore waters (see https://www.northeastoceandata.org/). As a surface-dwelling species, oceanic whitetip sharks are unlikely interact with gears that are fished deeper in the water column. In addition, there have not been any observed interactions between the fisheries and oceanic whitetip sharks (NEFSC observer/sea sampling database, unpublished data) since the beginning of the observer program in 1989. Given their offshore distribution and the diffuse vessel traffic, of which a limited number of vessels are fishing vessels operating in the fisheries considered in this Opinion, it is also extremely unlikely that there will be interactions between oceanic white tip sharks and the vessels in this Opinion. Given this information and the pelagic surface-dwelling nature of oceanic whitetip sharks, it is extremely unlikely and, therefore, discountable that the fisheries would interact with oceanic whitetip sharks.

4.1.6. ***Shortnose Sturgeon***

Shortnose sturgeon are benthic fish that occur in large coastal rivers of eastern North America. They range from as far south as the St. Johns River, Florida (possibly extirpated from this system) to as far north as the Saint John River in New Brunswick, Canada. Shortnose sturgeon are a diadromous fish species and one of only two sturgeon species that occur in marine waters and estuaries from Canada to Florida. Tracking data indicate that shortnose sturgeon are capable of making coastal migrations, and fish have been tracked between several Maine rivers and down to the Merrimack River in Massachusetts (SSSRT 2010, Wippelhauser and Squiers 2015, Zydlewski et al. 2011). However, even in the Northeast where these coastal migrations have been documented, shortnose sturgeon do not appear to spend significant time in the marine environment and generally stay close to shore (SSSRT 2010, NMFS unpublished data).

We consider it extremely unlikely that the fisheries considered in this Opinion will interact with shortnose sturgeon given:

1. No interactions with shortnose sturgeon have been reported in the fisheries since 2005, and there have been only 12 observed (annual average of 0.4) since the inception of bycatch data collection by the Northeast Fisheries Observer Program in 1989. We reviewed these reports to determine whether Atlantic sturgeon may have been misreported as shortnose sturgeon. Three interactions occurred in 1992, and there were very little data to evaluate whether these may have been misreported. One shortnose sturgeon was captured in 2004. This sturgeon was 160 cm long, which is large for a shortnose sturgeon, but still possible. The remaining eight interactions were documented during two hauls on the same bottom trawl trip in April 2005 off New York Harbor and southwestern Long Island (the primary commercial species landed was windowpane flounder). While their estimated length puts them in the range for either shortnose or Atlantic sturgeon, the aggregation behavior where multiple sturgeon are caught in two hauls is more typical of the aggregation behavior of Atlantic sturgeons in ocean waters. In addition, recent trawl research in and around those waters where these takes have occurred has only led to captures of Atlantic sturgeon (Dunton et al. 2012, Dunton et al. 2015, Dunton et al. 2010, O'Leary et al. 2014). Therefore, we are skeptical that these identifications are correct. If the captures in the observer data were not misidentified, we consider them an anomaly and unlikely to reoccur.
2. Subsequent to these takes, additional information and training on sturgeon were provided to observers, and the observers began to collect samples for genetic analysis with the purpose of identifying the species. The fact that no shortnose sturgeon takes have been recorded since 2005 is likely a reflection of the additional information provided for sturgeon species identification. In addition, the Northeast Area Monitoring and Assessment Program (NEAMAP) surveys are conducted with bottom other trawl in nearshore waters from the Gulf of Maine to Cape Hatteras in the spring and fall. Initiated in 2006, the NEAMAP southern New England/mid-Atlantic near shore trawl fishery

collects data from Aquinnah, Massachusetts to Cape Hatteras, North Carolina. There are no records of shortnose sturgeon captured during these surveys.

3. Shortnose sturgeon only infrequently move along the coast (SSSRT 2010, NMFS unpublished data. These movements are generally limited by geographic distance between river mouths, with greater movement between geographically proximate rivers. Movement between larger groups of rivers at greater geographic distance rarely occurs. When coastal migrations have been documented, shortnose sturgeon do not appear to spend significant time in the marine environment and generally stay close to shore (SSSRT 2010). The fisheries in this Opinion do not generally operate along the shore where these migrations are taking place.
4. Vessel strikes are also considered unlikely to occur. Shortnose sturgeon are primarily demersal, occupying the bottom of the water column, and would rarely be at risk from moving vessels, which need sufficient water to navigate without encountering the bottom. Given the species distribution, there is very limited overlap with vessels participating in the fisheries considered in this Opinion.

Because the fisheries undergoing consultation occur in federal waters and generally do not overlap with the species, the lack of documented take in more than a decade in the fisheries under consultation, and the lack of documented take in nearshore trawl surveys, we have determined that it is extremely unlikely that shortnose sturgeon would interact with these fisheries; therefore, effects are discountable.

4.1.7. ***Smalltooth Sawfish and Designated Critical Habitat***

While distributed circumglobally, NMFS identified smalltooth sawfish from the southeast United States as a DPS (68 FR 15674, April 1, 2003). North of Florida, recent records of smalltooth sawfish are rare. Records in the mid-Atlantic are from the late 1800s and early 1900s. Recent records from North Carolina through Georgia are sparse. Since 1970, there was one record in North Carolina (1999) and two in Georgia (2002, 2015) (Wiley and Brame 2018). Most specimens captured along the Atlantic coast north of Florida are large adults (over 10 ft (3 m)) that likely represent seasonal migrants, wanderers, or colonizers from a historic Florida core population(s) to the south, rather than being members of a continuous, even-density population (Bigelow and Schroeder 1953a).

Current records of smalltooth sawfish from the east coast of Florida remain relatively scarce compared to the west coast, Florida Bay, and the Florida Keys (Wiley and Brame 2018). The largest numbers of smalltooth sawfish are found in south and southwest Florida from Charlotte Harbor through the Dry Tortugas (Wiley and Brame 2018). In this area, there is a resident reproducing population of smalltooth sawfish which is also the last U.S. stronghold for the species (Poulakis and Seitz 2004, Seitz and Poulakis 2002). Florida Bay and the west coast of Florida are outside the action area of this Opinion.

In Florida, smalltooth sawfish generally inhabit shallow coastal waters, estuaries, and rivers, down to a maximum depth rarely exceeding 328 ft (100 m) and are associated with mangrove, seagrass, and shoreline habitats (Wiley and Brame 2018, Wiley and Simpfendorfer 2010). When documented, the substrate associated with encounters included mud (61 percent), sand (11 percent), or seagrass (10 percent). Other habitat types reported included limestone hard bottom, rock, coral reef, and sponge bottom (Poulakis and Seitz 2004). Water temperatures (no lower than 16-18 °C) and the availability of appropriate coastal habitat (shallow, euryhaline waters and red mangroves) are the major environmental constraints limiting the northern movements of smalltooth sawfish in the western North Atlantic.

With the exception of the bluefish fishery, none of the fisheries considered in this Opinion overlap with smalltooth sawfish. The NEFSC Observer Program observes the fisheries in this Opinion, including the bluefish fishery, north of Cape Hatteras; given this, together with the geographical range smalltooth sawfish, there is no information available on observed interactions with smalltooth sawfish in the bluefish fishery. While the bluefish fishery uses gears known to be detrimental to smalltooth sawfish (i.e., gillnets , trawls, and hook-and-line), we believe that takes of smalltooth sawfish in the bluefish fishery are extremely unlikely given:

1. There is limited overlap between the commercial bluefish fishery and smalltooth sawfish. As described above, gillnets are the primary gear used to commercially harvest bluefish; however, the majority of the effort occurs farther north. In addition, Florida has banned gillnet fishing in state waters (Florida Administrative Code 688-4.0081). As gillnet is the primary gear type used in the bluefish fishery and smalltooth sawfish abundance is higher in state waters, this further limits the overlap.
2. Anecdotal information collected by NMFS port agents suggest smalltooth sawfish captures in commercial fisheries in the southeast are now rare (NMFS 2020c). Bluefish effort in the southeast represents only a small fraction of the overall effort in this area.
3. As described above, bluefish were only reported on 2.5 percent of MRIP interviews in Atlantic Florida. Of the 2.5 percent of interviews positive for bluefish, bluefish was the primary of secondary target species in only 6 percent of the interviews. This very low effort limits the likelihood that the recreational fishery will incidentally capture a smalltooth sawfish.
4. Vessel strikes are also considered unlikely to occur. Smalltooth sawfish are primarily demersal and would rarely be at risk from moving vessels, which need sufficient water to navigate without encountering the bottom. When operating in areas with marginal clearance, vessels generally transit these areas slowly, allowing the species an opportunity to move out of the way. Given the species distribution, there is very limited overlap with vessels participating in the fisheries considered in this Opinion.

Given the habitat preference of smalltooth sawfish for shallow coastal waters and the limited commercial and recreational bluefish effort in Florida, the likelihood of an interaction occurring between commercial bluefish fishing gear and smalltooth sawfish within the range of the DPS is extremely unlikely and discountable.

Designated critical habitat for the smalltooth sawfish DPS, which includes the Charlotte Harbor Estuary Unit and the Ten Thousand Islands/Everglades Unit (74 FR 45353, September 2, 2009), also occurs only in Florida waters west and inshore of the Florida Keys (Figure 26). The features essential to the conservation are red mangroves and shallow euryhaline habitats with depths less than 3 ft (74 FR 45353, September 2, 2009). As it is in state waters, this Opinion is considering vessel transits in this area. Critical habitat occupies a very small fraction of the action area. Given that (1) critical habitat occupies a very small fraction of the action area; (2) the number of participants in the fishery in Florida is small, as described above; and (3) that vessels transiting to the fishing grounds are unlikely to be operating in these shallow waters, any affect to the critical habitat in the action area for the U.S. DPS of smalltooth sawfish will be insignificant and discountable.

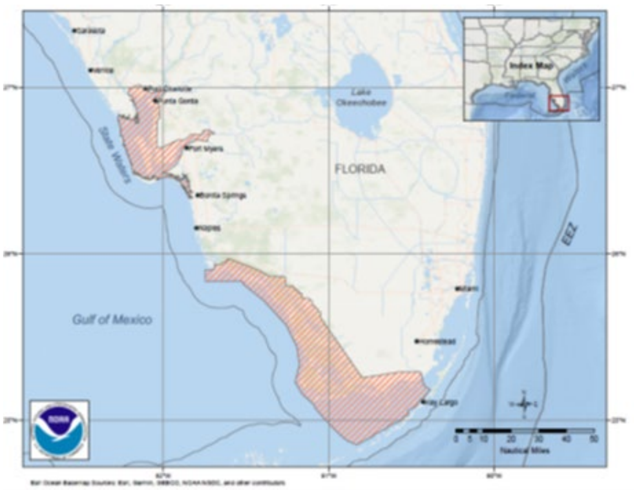

Figure 26: Smalltooth sawfish critical habitat

4.1.8. *Johnson's Sea Grass and Designated Critical Habitat*

Johnson’s seagrass occurs in a variety of habitat types, including intertidal wave-washed sandy shoals, flood deltas near inlets, and near the mouths of canals and rivers, where presumably water quality is sometimes poor and salinity fluctuates widely. It occurs in a patchy, disjunctive distribution from the intertidal zone to depths of approximately 10-13 ft (3-4 m) (NMFS 2007). Johnson’s seagrass is found only in southeast Florida, ranging from Sebastian Inlet to central Biscayne Bay; within this range, 10 areas (Figure 27) are designated as critical habitat (65 FR 17786, April 5, 2000). The general physical and biological features of the critical habitat areas include adequate water quality, salinity levels, water transparency, and stable, unconsolidated sediments that are free from physical disturbance. The specific areas occupied by Johnson’s seagrass are those with one or more of the following criteria: (1) locations with populations that have persisted for 10 years; (2) locations with persistent flowering populations; (3) locations at the northern and southern range limits of the species; (4) locations with unique genetic diversity; and (5) locations with a documented high abundance of Johnson’s seagrass compared to other areas in the species’ range.

The bluefish fishery is the only fishery considered in this Opinion operating in southeastern Florida. The critical habitat is located in state waters. Therefore, effects from vessel transits are assessed here. The number of vessels transiting these areas would be small given that the (1) species and critical habitat occupy a very small fraction of the action area, and (2) the number of vessels participating in the fishery in Florida is small, as described above. Given this, any affect to Johnson's seagrass or critical habitat will be insignificant and discountable.

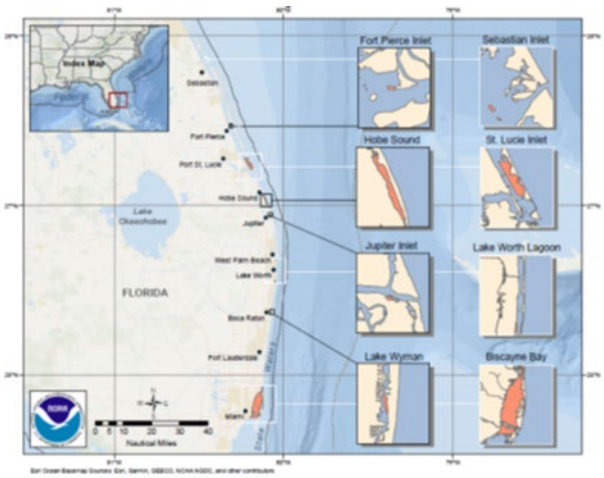

Figure 27: Johnson's seagrass distribution

4.1.9. ***Corals and Designated Critical Habitat***

We evaluated the potential effects of the proposed action on seven ESA-listed corals (elkhorn, staghorn, rough cactus, pillar, lobed star, mountainous star, and boulder star corals) based on the information provided in the species status reviews, recovery plan, and the listing rules (71 FR 26852, May 9, 2006; 79 FR 53852, Sept. 10, 2014).

Coral reefs are formed on solid substrates within a narrow range of environmental conditions. These conditions include relatively narrow temperature, salinity, turbidity, pH, and light ranges (Brainard et al. 2011, Kleypas 1997). All seven species occur in waters off the east coast of Florida. Elkhorn corals commonly grow in turbulent shallow water at depths of 3-16 ft (1-5 m) in depth, but have been found to 98 ft (30 m). Staghorn corals commonly grow in more protected, deeper waters ranging from 16-49 ft (5-15 m) in depth and have been found in rare instances to 197 ft (60 m) (ABRT 2005). Rough cactus coral has been reported to occur in shallow reef environments in water depths of 16-66 ft (5-20 m) (Brainard et al. 2011, Carpenter et al. 2008). It is usually uncommon (Veron 2000 as cited in Brainard et al. 2011) or rare, occurring at densities < 0.8 colonies per 10 m^2 in Florida (Wagner et al. 2010). Monitoring data since 2000 from Florida and elsewhere in the Caribbean show that it the rough cactus coral cover is consistently less than 1 percent, with occasional observations up to 2 percent (Brainard et al. 2011). Pillar, lobed star, mountainous star, and boulder star corals inhabit most reef environments. Pillar star coral occurs in depths from 6.5-82 ft (2-25 m) (Brainard et al. 2011, Carpenter et al. 2008). It is reported to be uncommon (Veron 2000 as cited in Brainard et al. 2011) with isolated colonies across a range of habitats. Overall colony density throughout south Florida was estimated to be ~ 0.6 colonies per 10 m^2 (Wagner et al. 2010). Boulder star corals are reported at depths of 16-164 ft (5-50 m) (Bongaerts et al. 2010, Brainard et al. 2011, Carpenter et al. 2008), lobed corals at depths of 1.6-66 ft (0.5-20 m) (Brainard et al. 2011, Szmant et al. 1997), and mountainous star coral are reported at depths of 1.6-131 ft (0.5-40 m) (Brainard et al. 2011, Carpenter et al. 2008, Weil and Knowton 1994). The environmental conditions of most of the U.S. Atlantic EEZ are not suitable for these seven corals. They are generally found in a small area of the southeast United States.

The bluefish fishery is the only fishery in this Opinion that occurs in southeast waters where the coral species occur. The known routes of effect from fishing on ESA-listed corals are a result of man-made abrasion and breakage resulting from vessel groundings, damaging fishing practices and fishing/marine debris (ABRT 2005). While the bluefish fishery occurs in this area, we consider it extremely unlikely that the fishery will interact with ESA-listed coral species given:

1. The effort in the commercial bluefish fishery is low in Florida. The overlap of the commercial fishery is further limited in that these species occur in only a small area off Florida. In addition, gillnets, the primary gear used in the commercial fishery, are prohibited (Florida Administrative Code 688-4.0081) in Florida state waters where these species overlap.
2. The recreational bluefish fishery does use gear that is known to impact coral reefs. Impacts to corals from hook-and-line fisheries interactions are most common to column and branching coral morphology that are more likely to become entangled by line or broken by gear. The rough cactus, lobed star, mountainous star, and boulder star coral species are characterized as boulder/mound or encrusting corals and area generally flat or round and lack the branching morphology that greatly increases the potential risk of becoming fouled by fishing lines. Pillar coral has protruding columns and the elkhorn and staghorn corals have a branching morphology. Even though these species have a morphology that is potentially susceptible to damage, interactions are extremely unlikely given the low co-occurrence of fishing effort and these corals. As described above, the number of recreational fishermen targeting bluefish is extremely low. Given this low effort, the low density of these listed corals, and the very limited overlap of the gear and species, we expect the probability of interaction to be extremely low.
3. Information in Chiappone et al. (2005) suggests that the level of lost gear from hook-and-line fishing effort needed to impact coral is very high. They report that, while lost hook-and-line fishing gear was ubiquitous in the Florida Keys, it was estimated that < 0.2 percent of the milleporid hydrocorals, stony corals, and gorgonians in the habitats studied showed injury (e.g., colony abrasions and partial mortality) as a result of lost hook-and-line gear interactions (Chiappone et al. 2005). Given that bluefish hook-and-line effort represents only a very small percentage of the overall effort, it is extremely unlikely that gear lost in this fishery would impact corals.
4. Vessel groundings are possible because of the proposed action, but we believe these events are extremely unlikely to occur given the limited effort and available technologies. Over the past 20 years, technological advancements and accessibility to depth gauges and GPS units have also increased vessel operators' ability to detect bottom features and calculate vessel position in relation to mapped coral structures. Experience and the use of technology greatly reduce the likelihood of vessels groundings.
5. Florida Keys National Marine Sanctuary regulations establish specific prohibitions against injuring corals (including *Acropora* species), anchoring on corals, grounding vessels on corals, and discharging fishing/marine debris (15 CFR 922.163).

There are four specific areas designated as critical habitat for elkhorn (*Acropora palmata*) and staghorn (*A. cervicornis*) corals (i.e., Florida, Puerto Rico, St.John/St.Thomas, and St.Croix; 73 FR 72210, November 26, 2008). Of these four areas, only the area designated in Florida occurs in the action area (Figure 28). The physical or biological features of elkhorn and staghorn corals' critical habitat that are essential to their conservation is substrate of suitable quality and availability to support successful larval settlement and recruitment, and reattachment and recruitment of fragments. For purposes of this definition, ''substrate of suitable quality and availability'' means natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover. (73 FR 72210, November 26, 2008).

Figure 28: Elkhorn and staghorn coral critical habitat - Florida Unit

In designating critical habitat for these coral species, the feature essential to the conservation of the species was substrate of suitable quality and availability, in water depths to 98 ft (30 m), to support successful larval settlement, recruitment, and reattachment of fragments (73 FR 72210, November 26, 2008). The only commercial fishing activity identified that may destroy or adversely affect the essential feature involved trap fisheries (NMFS 2008b). Of the fisheries considered in the on-going consultation, only the bluefish fishery overlaps with critical habitat designated for these coral species. Pot/trap is not a gear type used in the commercial or recreational bluefish fishery. Bluefish fishing vessels transiting through critical habitat are also not expected, for reasons described above, to affect substrate of suitable quality and availability.

The low level of fishing effort, low density of ESA-listed corals occurring where fishing is likely to occur, and the measures in place to protect these species make any adverse effects on these species or the critical habitat from the proposed action extremely unlikely to occur. Based on this information, effects of the fisheries on ESA-listed corals and their designated critical habitats is extremely unlikely and, therefore, discountable.

4.1.10. ***North Atlantic Right Whale Critical Habitat***

We have determined that the actions considered in this Opinion are not likely to adversely affect designated critical habitat for North Atlantic right whales (Figure 29). The two areas designated as critical habitat contain approximately 29,763 nmi^2 of marine habitat in the Gulf of Maine and Georges Bank region (Unit 1, Northeastern U.S. Foraging Area) and off the Southeast U.S. coast (Unit 2, Southeastern U.S. Calving Area) (81 FR 4838, January 27, 2016).

Specifically, we considered whether the actions were likely to affect the essential physical or biological features (PBFs) that afford the designated area overall value for the conservation of North Atlantic right whales.

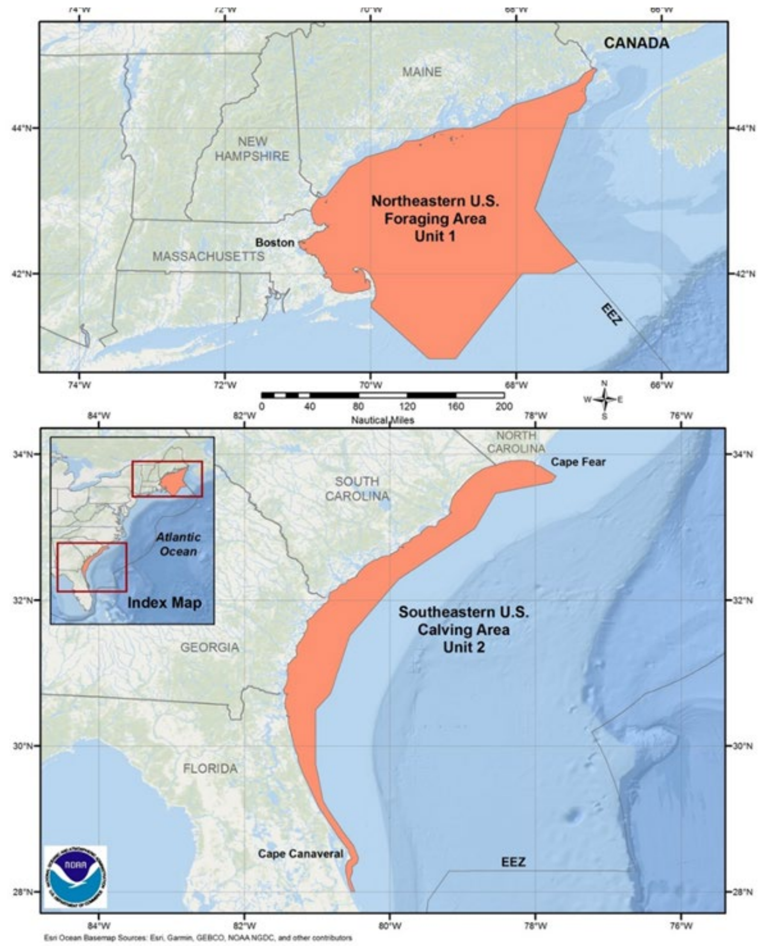

Figure 29: North Atlantic right whale critical habitat

The boundaries of both Unit 1 and Unit 2 are in the action area. The Northeastern U.S. foraging habitat (Unit 1) is defined by the distribution, aggregation and retention of *Calanus finmarchicus*, the primary and preferred prey of North Atlantic right whales (NMFS 2015c). The essential physical features identified in the final rule include prevailing currents, bathymetric features (such as basins, banks, and channels), oceanic fronts, density gradients, and flow velocities. The essential biological features include dense aggregations of copepods, specifically late stage *C. finmarchicus* in the Gulf of Maine and Georges Bank region, as well as aggregations of diapausing (overwintering) populations in the deep basins of the region (i.e., Jordan, George's, and Wilkinson basins) as these populations of *C. finmarchicus* serve as source populations for the overall Gulf of Maine population (Johnson et al. 2006, Lynch et al. 1998, Meise and O'Reilly 1996). It should also be noted that based on changes in right whale and *C. finmarchicus* distributions since 2010, the continental shelf south of New England and the Gulf of Saint Lawrence in Canada (Khan et al. 2018, Record et al. 2019), https://fish.nefsc.noaa.gov/psb/surveys/MapperiframeWithText.html) have become increasingly important foraging habitats for North Atlantic right whales indicating that important prey sources may also be present outside of the designated critical habitat area.

The essential features of right whale calving habitat (Unit 2) are dynamic in their distributions throughout the South Atlantic Bight, varying in time and space. The physical features of right whale calving habitat essential to the conservation of the North Atlantic right whale are calm sea surface conditions of less than or equal to Force 4 on the Beaufort Wind Scale, sea surface temperature greater than 7 °C and less than or equal to 17 °C, and water depths of 20 – 92 ft (6-28 m). These features co-occur over large contiguous areas of ocean waters during the months of November. When these features are available, they are selected by right whale cows and calves in dynamic combinations that are suitable for calving, nursing, and rearing. The optimal combinations of the features vary depending on factors such as weather and the age of the calves. (NMFS 2015c). No essential biological features were identified for Unit 2 (NMFS 2015c).

In designating critical habitat, NMFS evaluated and identified activities that may destroy or adversely modify the essential physical and biological features (NMFS 2015b, c).This analysis evaluated whether fishing activity will adversely affect the late-stage dense *C. finmarchicus* aggregations that trigger right whale foraging behavior, the overwintering populations of *C. finmarchicus* in deep water basins, or the physical and oceanographic features that allow these deep water populations to supply the Gulf of Maine *C. finmarchicus* population. It is extremely unlikely that fishing vessels will have any potential to affect the essential biological and physical oceanographic features (i.e., currents, temperature, bathymetry) of critical habitat. Therefore, the analysis focuses on fishing gears used in the fisheries in this Opinion.

Copepods are extremely small organisms that will pass through or around the fishing gears rather than being captured on or in them. In addition, turbidity created from fishing activities is, as described below, expected to be temporary in nature and will not impact the long-term viability of copepod aggregations. While fishing activity may temporarily disturb localized copepod concentrations, this disturbance is not expected to significantly change the quality or quantity of the aggregations to a degree that will impact the conservation of right whales. In addition, any effects from fishing gear on the environment in areas right whales are present are further limited by the requirement that gear not be set within 500 yards of the sighted right whales (50 CFR 224.103(c)), avoiding localized disturbance of copepod populations on which the whales may be feeding. Haulbacks should also not be initiated if right whales are sighted within, or close to, 500 yards from the vessel.

Bottom-tending mobile gear, such as trawls and dredges, have the potential to temporarily disturb resting copepod populations found in deep water basins as the gear moves through areas where the aggregations occur and temporarily increases turbidity. However the effect of this sediment resuspension is likely minimal for several reasons. First, while fine sediment may take up to 24 hours to resettle, the plumes created by bottom trawling are “laterally advected some distance by tidal currents before settling” (Pilskaln et al. 1998). This dispersal would result in lower concentrations of sediment spread out over a larger area, and the localized turbidity would likely be temporary. Additionally, the Gulf of Maine, particularly Wilkinson and Jordan Basin, already has a pervasive “nepheloid layer” (i.e. a layer of water containing suspended sediment) that can reach between 66-131 ft (20-40 m) in thickness (Pilskaln et al. 1998) so it is expected that any copepod aggregation in those areas is adapted to a highly turbid environment. A recent study of Calanus found that diapausing Calanus can persist across a wide range of conditions in the Northwest Atlantic, and there was little evidence that their vertical distribution was affected by the light conditions (Krumhansl et al. 2018). In addition, laboratory studies have shown that increased sediment loads may affect the feeding efficiency and production of *C. finmarchicus* in

Greenland fjords, however, the results also indicated that copepods can handle very high sediment loads for at least a short period of time (in the study, four days) (Arendt et al. 2011).

As *C. finmarchicus* enter the pre-adult stage, if they build enough lipid stores, they may enter a suspended state of development (diapause) as they retreat to depths where they remain neutrally buoyant. Emergence from diapause is synchronized to allow *C. finmarchicus* or their progeny access to favorable environmental conditions, typically phytoplankton produced during the spring bloom (Baumgartner and Tarrant 2017). The cues triggering the termination of the diapausing state are not clearly understood, however, hypotheses include light and photoperiod cues or the "existence of an endogenous long-range timer that arouses copepods from diapause after some period of time has elapsed" (Campbell et al. 2004, Hirche 1996, Miller et al. 1991). Given that diapause is triggered and maintained; it is unlikely that plumes would disturb Calanus when they are in this state of suspended growth. These plumes are also not expected to have an impact energy stores and temperature/light sensitivity that keep these zooplankton in their overwintering state at depth. This is considered an adaptive strategy that allows the zooplankton to suspend development until conditions are optimal for reproductive success. A portion of the pre-adult population will not retreat to depths and instead will molt into adult stages and reproduce prior to the emergence of the diapausing population if conditions allow (though there is a chance that the conditions will not be favorable) (Baumgartner and Tarrant 2017) These alternative survival strategies allow for maximum productivity and continuous replenishment of the stock with varying environmental conditions.

Bottom-tending mobile gear may also impact resting copepod eggs by increasing mortality of eggs that come into contact with the gear or decreasing the eggs' chances of hatching if it is re-suspended by the plume (Drillet et al. 2014). On the other hand, the resuspension of the eggs into the water column may also potentially increase recruitment back into the water column (Drillet et al. 2014). This trade-off is not well understood, and it is not clear how these impacts differ from natural "bioturbation or storm events under different environmental conditions" (Drillet et al. 2014). There is also no indication of what role sitting eggs play in the overall population recruitment in the Gulf of Maine. A significant supply of the Gulf of Maine's *C. finmarchicus* population found in Jordan and George's Basins are supplied from the Scotian Shelf and Scotian Slope waters (Johnson et al. 2006, Miller et al. 1998) with only Wilkinson's Basin restocking internally (Johnson et al. 2006). Given these multiple sources for Calanus, it is unlikely that any localized decrease in hatching success of resting or re-suspended eggs would affect the Calanus population at-large or to level that would have any detectable effect on North Atlantic right whale foraging,

It should be noted that, when designating critical habitat, NMFS' assessment of activities that may destroy or adversely modify the essential physical and biological features; dredging was also considered. However, in that analysis "dredging" refers to the removal of material from the bottom of water bodies to deepen, widen or maintain navigation corridors, anchorages, or berthing areas, as well as sand mining (NMFS 2015b, c). Dredges typically used for navigational deepening or sand mining operations include hopper and cutterhead dredges. Although dredge size varies by location, hydraulic hopper dredges have draghead widths from a few feet to 12 ft (3.6 m); cutterhead diameters typically range from 16-20 inches (maximum 36 inches) (40-53 cm, maximum 91.5 cm). These dredges disturb the sediment surface (down to 12 inches (30.5 cm) or more) creating turbidity plumes that last up to a few hours. The review found that extracting sediments for navigation or beach nourishment projects would have little to no effect on *C. finmarchicus* and that discharge of dredge material would have ephemeral effects given prevailing currents that would rapidly disperse sediment plumes at depths where essential foraging features are not present (NMFS 2015b). In addition, the ESA 5-year review for right whales concluded that habitat degradation from dredging, among other actions, is not limiting right whale recovery (NMFS 2017e). In contrast to navigational or deepening dredges, scallop dredges ride above the substrate surface, creating turbulence that stirs up the substrate and kicks scallops up and into the bag. The shoes on the dredge are in contact and ride along the surface. Dredges range is width from 5.5 ft (1.7 m) to approximately 15 ft (4.6 m). They are used in high and low energy sand environments and high energy gravel environments (Northeast Region Essential Fish Habitat Steering Committee 2002), Appendix D in NEFMC 2016b). As described above, turbidity from scallop dredges is expected to minimal and overall effects from scallop dredges are expected to be less than those of navigational or deepening dredges. Given that fishing with dredges is localized, that the bottom disturbance is temporary, and that any turbidity created is expected to be minimal and ephemeral, we conclude that dredging will not adversely affect right whale critical habitat.

Other mobile gear, which operate in the water column, such as purse seines and midwater trawls, may temporarily disperse localized Calanus populations. However, given that copepods will easily pass through the mesh of these gear type, they should not interfere with the general aggregations or reproduction. These behaviors are largely dependent on oceanographic processes and currents that the gear types cannot modify.

Fixed fishing gear, such as gillnets and trap/pots, may also temporarily disturb local aggregations of copepods during the setting and hauling of gear due to turbidity caused by the sediment disturbance as the gears are set or dragged over the bottom during retrieval (Northeast Region Essential Fish Habitat Steering Committee 2002). Given the temporary nature of the disturbance, these gears are considered to have low habitat impacts, particularly in areas where sand, mud, and gravel comprise the substrate. Any movement of the gear during active fishing may also cause sediment disturbance, however, this movement would occur in higher energy environments. Local copepod populations would already be adapted to this environment, and the movement would not adversely modify the dense copepod aggregations or the oceanographic features that contribute to their aggregation behaviors. Localized disturbance to dense copepod aggregations by these gear types is further minimized by MMPA gillnet and trap/pot closure areas that exist in temporal and spatial areas where these dense concentrations are expected to trigger foraging behavior (e.g., Massachusetts Bay Restricted Area) (50 CFR 229.23).

Fixed fishing gear also does not block the entire water column or form a wall preventing access. Vertical buoy lines supporting the fixed gear may extend throughout the water column, however, the Gulf of Maine critical habitat feeding area is vast and not constricted by geological or physical barriers, therefore, whales are free to move through and around these gears to reach their feeding resources. The impact of entanglements on individual animals as they access their feeding resources is addressed in section 7.2 of this analysis, but is not considered an impact to whales accessing or moving within critical habitat.

It is extremely unlikely that the fisheries would have any effect on the essential physical oceanographic features (i.e., sea state, temperature, depth) of Unit 2. Fishing gears do not alter sea surface conditions, temperature, or depths. The only fishery considered in this Opinion that overlaps with Unit 2 is the bluefish fishery. This fishery is primarily recreational using hook-and-line gear. There is a small amount of gillnet effort in the bluefish fishery off Florida. Gillnets are not permitted in Florida state water which reduces the overlap of the fishery and critical habitat. In addition, the limited effort that occurs in federal waters is not expected to preclude right whales from accessing these areas.

Based on the above, we have determined that the effects of the fishing gears and vessels used by the fisheries in this Opinion on the availability of copepods for foraging right whales are likely so small that they cannot be meaningfully measured, detected, or evaluated, and, therefore, insignificant. In addition, as noted above, it is extremely unlikely that the operation of fishing gears and vessels will affect the large-scale physical oceanographic conditions in the Gulf of Maine or off the southeast United States. As a result, the effects of the operation of the fisheries on those physical features are discountable. Because the effects of the fisheries on the PBFs that characterize the feeding and calving habitats for North Atlantic right whales are all insignificant and discountable, the fisheries are not likely to adversely affect this critical habitat.

4.1.11. *Northwest Atlantic DPS of Loggerhead Sea Turtle Critical Habitat*

We have determined that the actions in this Opinion are not likely to adversely modify or destroy designated critical habitat for the Northwest Atlantic DPS of loggerhead sea turtles. Critical habitat is designated for nesting beaches and 38 occupied areas within the at-sea range of the Northwest Atlantic DPS (79 FR 29755, July 10, 2014; 79 FR 39856, July 10, 2014). These marine areas in the Atlantic Ocean contain one or a combination of nearshore reproductive habitat, overwintering habitat, breeding habitat, migratory habitat, and *Sargassum* habitat (Figure 30). There is limited overlap of Northwest Atlantic DPS critical habitat and the fisheries considered in this Opinion. Setting and hauling gear and fishing vessel movements are not expected to significantly alter the physical or biological features of the critical habitat areas to levels that would affect life history patterns of individual turtles or the health of prey species found in these habitats.

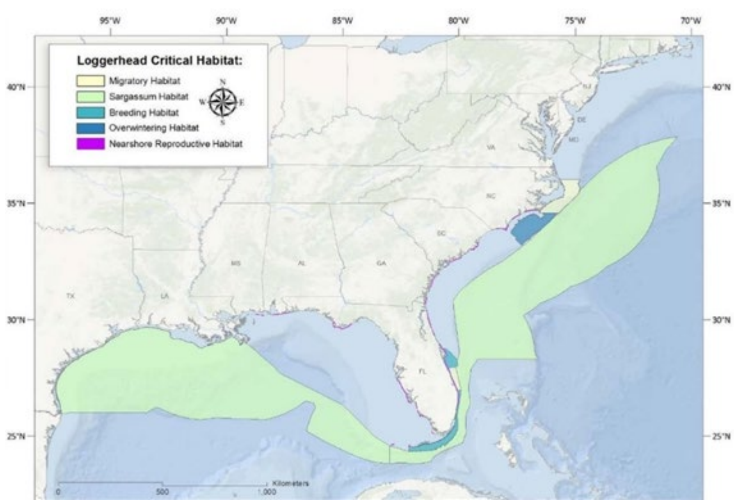

Figure 30: Loggerhead sea turtle critical habitat

The nearshore reproductive habitat within the action area occurs off nesting beaches from North Carolina south through Florida. Primary constituent elements (PCEs) supporting this habitat include waters (1) with direct proximity to nesting beaches that support the highest density nesting aggregations; (2) sufficiently free of obstructions or artificial lighting; and (3) with minimum manmade structures that could promote predators, disrupt wave patterns, and/or create excessive currents (NMFS 2013a). The reproductive habitats identified extend 0.86 nmi (1.6 km) offshore. Gillnets are prohibited in Florida state waters (Florida Administrative Code 688-4.0081). The fisheries will not result in obstructions or manmade structures that will alter the physical environment.

The winter habitat includes warm water habitat south of Cape Hatteras near the western edge of the Gulf Stream. It supports meaningful aggregations of juveniles and adults during the winter months. PCEs that support the habitat are (1) water temperatures above 10 °C from November through April; (2) continental shelf waters in proximity to the western boundary of the Gulf Stream; and (3) water depths between 65-128 ft (20 and 100 m) (NMFS 2013a). The activities considered in this Opinion will not alter water depths, water temperatures, or other physical oceanographic features.

The breeding habitat includes sites along the east coast of Florida that support meaningful aggregations of male and female adult loggerheads during breeding season. The PCEs that support this habitat are: (1) high densities of reproductive loggerheads; (2) proximity to primary Florida migratory corridor; and (3) proximity to Florida nesting grounds (NMFS 2013a). As described elsewhere, the bluefish fishery is the only fishery in this Opinion that extends south to Florida. Fishing activities that disrupt habitat use and, thus, affect concentrations of reproductive loggerheads could affect the breeding habitat. The bluefish fishery has the potential to capture protected loggerhead sea turtles as analyzed later in this Opinion, but we do not believe that this will noticeably affect the density of reproductive males and females in the breeding areas. The fishery overall has limited overlap with these areas and will not alter the physical environment or affect the distance of the breeding habitat in relation to the migratory corridor or nesting grounds. Therefore, any effects on the breeding habitat are insignificant.

The constricted migratory habitat is high use migratory corridors that are limited in width by land on one side and the edge of the continental shelf and Gulf Stream on the other. PCEs that support this habitat are (1) constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory passage and (2) passage conditions that allow for migration to/from nesting, breeding, and/or foraging areas (NMFS 2013a). Migratory habitat in the Atlantic includes areas off North Carolina and Florida. Fisheries using fixed gear (e.g., gillnets and pots/traps) are a concern if the gear is arranged closely together within the designated habitats. These gears could alter the habitat conditions needed for efficient passage of loggerheads through these areas (79 FR 39856, July 10, 2014) when gear is deployed within the critical habitat. However, the operations are wide-ranging, including areas within and outside critical habitat. Any gears deployed in migratory habitat areas fluctuates in time and space and are not permanent obstructions. We do not expect the gears used in the fisheries in this Opinion to meaningfully alter the passage conditions that allow for migration to/from nesting, breeding, and foraging habitats.

Sargassum habitat is important to various life stages, particularly post-hatchlings. Generally, the *Sargassum* habitat included in the designation and occurring in the action area is along the Atlantic coast from the western edge of the Gulf Stream eastward. PCEs that support this habitat include: (1) convergence zones, surface-water downwelling (movement of denser water downward in the water column) areas, major current margins and other locations where there are concentrated components of the *Sargassum* community in suitable water temperatures; (2)

Sargassum in concentrations to support adequate prey abundance; (3) available prey and other material associated with *Sargassum* habitat; and (4) sufficient water depth and proximity to available currents to ensure offshore transport and forage and cover requirements for post-hatchling loggerheads. In designating critical habitat, NMFS identified possible activities that may require special management considerations; commercial fishing activities were not included (79 FR 39856, July 10, 2014). While commercial fishing gear may have some interactions with *Sargassum* during deployment and retrieval, the effects are expected to be temporary and isolated in nature and, because of the fluid nature of the pelagic environment, recovery time is rapid (79 FR 39856, July 10, 2014). The fisheries do not have the capability to affect the location of water depths, currents, convergence zones, downwelling areas, major current margins or other locations with concentrated components of the *Sargassum* community. While vessels may transit the areas, any disruption of Sargassum habitat is not of sufficient magnitude to significantly affect the distribution of Sargassum mats. In addition, the fisheries will not affect the availability of loggerhead prey or other material associated with *Sargassum* because they do not target or harvest smaller prey species or *Sargassum*.

4.2. Species Likely to be Adversely Affected

This section examines the status of each species that are likely to be adversely affected (Table 42) by the proposed action. Under the ESA, species include any subspecies of any species and any distinct population segment of any species of vertebrate fish or wildlife that interbreeds when mature. This section considers the species as listed under the ESA, which may be globally or as a DPS. The status includes the current level of risk that the ESA-listed species face, based on factors considered in documents such as recovery plans, status reviews, and listing decisions. This section helps detail the species' current "reproduction, numbers, or distribution," which is considered in the jeopardy determination as described in 50 CFR. §402.02. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology, is in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on NMFS' website: (https://www.fisheries.noaa.gov/species-directory/threatened-endangered), among others.

4.2.1. *Large Whales*

4.2.1.1. North Atlantic Right Whale (Eubalaena glacialis)

There are three species classified as right whales (genus *Eubalaena*): North Pacific (*E. japonica*), Southern (*E. australis*), and North Atlantic (*E. glacialis*). The North Atlantic right whale is the only species of right whale that occurs in the North Atlantic Ocean (Figure 31) and, therefore, is the only species of right whale that may occur in the action area.

Today, North Atlantic right whales occur primarily in the western North Atlantic Ocean. More recently; however, there have been acoustic detections, reports, and/or sightings of North Atlantic right whales in waters off Greenland (east/southeast), Newfoundland, northern Norway, and Iceland, as well as within Labrador Basin (Hamilton et al. 1998, Jacobsen et al. 2004, Knowlton et al. 1992, Mellinger et al. 2011). These latter sightings/detections are consistent with historic records documenting North Atlantic right whales south of Greenland, in the Denmark straits, and in eastern North Atlantic waters (Kraus et al. 2007). There is also evidence of possible historic North Atlantic right whale calving grounds being located in the Mediterranean

Sea (Rodrigues et al. 2018), an area not currently considered as part of this species historical range.

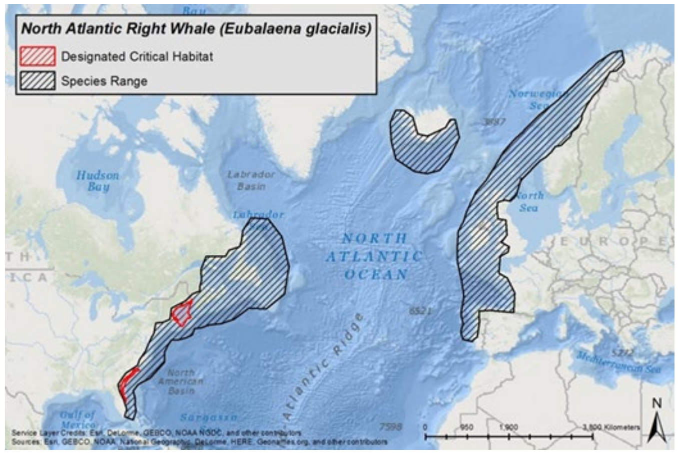

Figure 31: Approximate historic range and currently designated U.S. critical habitat of the North Atlantic right whale.

The North Atlantic right whale is distinguished by its stocky body and lack of a dorsal fin. The species was listed as endangered on December 2, 1970. We used information available in the most recent five-year review for North Atlantic right whales (NMFS 2017e), the most recent stock assessment reports (Hayes 2019, Hayes et al. 2019), and the scientific literature to summarize the species, as follows.

Life history

The maximum lifespan of North Atlantic right whales is unknown, but one individual reached at least 70 years of age (Hamilton et al. 1998, Kenney 2009). Previous modelling efforts suggest that in 1980, females had a life expectancy of approximately 51.8 years of age, which was twice that of males at the time (Fujiwara and Caswell 2001); however, by 1995, female life expectancy was estimated to have declined to approximately 14.5 years (Fujiwara and Caswell 2001). Most recent estimates indicate that North Atlantic right whale females are only living to 45 and males to age 65 (https://www.fisheries.noaa.gov/species/north-atlantic-right-whale). A recent study demonstrates that females, ages 5+, have reduced survival relative to males, ages 5+, resulting in a decrease in female abundance relative to male abundance (Pace et al. 2017). Specifically, state-space mark-recapture model estimates show that from 2010-2015, males declined just under 4.0 percent and females declined approximately 7 percent (Pace et al. 2017).

Gestation is estimated to be between 12 and 14 months, after which calves typically nurse for around one year (Cole et al. 2013, Kenney 2009, Kraus and Hatch 2001, Lockyer 1984). After weaning calves, females typically undergo a ‘resting’ period before becoming pregnant again, presumably because they need time to recover from the energy deficit experienced during lactation (Fortune et al. 2013, Fortune et al. 2012, Pettis et al. 2017). From 1983 to 2005, annual average calving intervals ranged from 3 to 5.8 years (overall average of 4.23 years) (Kraus et al. 2007). Between 2006 and 2015, annual average calving intervals continued to vary within this range, but in 2016 and 2017 longer calving intervals were reported (6.3 to 6.6 years in 2016 and 10.2 years in 2017) (Hayes et al. 2018a, Pettis and Hamilton 2015, Pettis and Hamilton 2016, Pettis et al. 2018a, Pettis et al. 2018b, Pettis et al. 2020). Annual average calving interval was 7 in 2019 and 7.6 in 2020 (Pettis et al. 2020, 2021). The calving index is the annual percentage of reproductive females assumed alive and available to calve that was observed to produce a calf.

This index averaged 47 percent from 2003 to 2010 but has dropped to an average of 17 percent since 2010 (Moore et al. 2021). Females have been known to give birth as young as five years old, but the mean age of a female first giving birth is 10.2 years old (n=76, range 5 to 23, SD 3.3) (Moore et al. 2021). Taken together, changes to inter-birth interval and age to first reproduction suggest that both parous (having given birth) and nulliparous (not having given birth) females are experiencing delays in calving. These calving delays corresponds with the recent distribution shifts. The low reproductive rate or right whales is likely the result of several factors (Moore et al. 2021).

Pregnant North Atlantic right whales migrate south, through the mid-Atlantic region of the United States, to low latitudes during late fall where they overwinter and give birth in shallow, coastal waters (Kenney 2009, Krzystan et al. 2018). During spring, these females and new calves migrate to high latitude foraging grounds where they feed on large concentrations of copepods, primarily *C. finmarchicus* (Mayo et al. 2018, NMFS 2017e). Some non-reproductive North Atlantic right whales (males, juveniles, non-reproducing females) also migrate south, although at more variable times throughout the winter. Others appear to not migrate south and remain in the northern feeding grounds year round or go elsewhere (Bort et al. 2015, Mayo et al. 2018, Morano et al. 2012, NMFS 2017e, Stone et al. 2017). Nonetheless, calving females arrive to the southern calving grounds earlier and stay in the area more than twice as long as other demographics (Krzystan et al. 2018). Little is known about North Atlantic right whale habitat use in the mid-Atlantic, but recent acoustic data indicate near year round presence of at least some whales off the coasts of New Jersey, Virginia, and North Carolina (Davis et al. 2017, Hodge et al. 2015, Salisbury et al. 2016, Whitt et al. 2013). While it is generally not known where North Atlantic right whales mate, some evidence suggests that mating may occur in the northern feeding grounds (Cole et al. 2013, Matthews et al. 2014).

Population dynamics

Today, North Atlantic right whales are primarily found in the western North Atlantic, from their calving grounds in lower latitudes off the coast of the southeastern United States to their feeding grounds in higher latitudes off the coast of New England and Nova Scotia (Hayes et al. 2018a). In recent years, the location of feeding grounds has shifted, with fewer animals being seen in the Great South Channel and the Bay of Fundy and more animals being observed in Cape Cod Bay; the Gulf of Saint Lawrence; the mid-Atlantic; and south of Nantucket, Massachusetts (Daoust et al. 2018, Davis et al. 2017, Hayes et al. 2018a, Hayes et al. 2019, Meyer-Gutbrod et al. 2018, Moore et al. 2021, Pace et al. 2017).

There are currently two recognized populations of North Atlantic right whales, an eastern and a western population. Very few individuals likely make up the population in the eastern Atlantic, which is thought to be functionally extinct (Best et al. 2001). However, in recent years, a few known individuals from the western population have been seen in the eastern Atlantic, suggesting some individuals may have wider ranges than previously thought (Kenney 2009). Specifically, there have been acoustic detections, reports, and/or sightings of North Atlantic right whales in waters off Greenland (east/southeast), Newfoundland, northern Norway, and Iceland, as well as within Labrador Basin (Jacobsen et al. 2004, Knowlton et al. 1992, Mellinger et al. 2011). It is estimated that the North Atlantic historically (i.e., pre-whaling) supported between 9,000 and 21,000 right whales (Monsarrat et al. 2016). The western population may have numbered fewer than 100 individuals by 1935, when international protection for right whales came into effect (Kenney et al. 1995).

Genetic analysis, based upon mitochondrial and nuclear DNA analyses, have consistently revealed an extremely low level of genetic diversity in the North Atlantic right whale population (Hayes et al. 2018a, Malik et al. 2000, McLeod and White 2010, Schaeff et al. 1997). Waldick et al. (2002)concluded that the principal loss of genetic diversity occurred prior to the 18th century, with more recent studies hypothesizing that the loss of genetic diversity may have occurred prior to the onset of Basque whaling during the 16th and 17th century (McLeod et al. 2008, Rastogi et al. 2004, Reeves et al. 2007, Waldick et al. 2002).The persistence of low genetic diversity in the North Atlantic right whale population might indicate inbreeding; however, based on available data, no definitive conclusions can be reached at this time (Hayes et al. 2019, Radvan 2019, Schaeff et al. 1997). By combining 25 years of field data (1980-2005) with high resolution genetic data, Frasier et al. (2013) found that North Atlantic right whale calves born between 1980 and 2005 had higher levels of microsatellite (nuclear) heterozygosity than would be expected from this species gene pool. The authors concluded that this level of heterozygosity is due to postcopulatory selection of genetically dissimilar gametes and that this mechanism is a natural means to mitigate the loss of genetic diversity, over time, in small populations (Frasier et al. 2013).

In the western North Atlantic, North Atlantic right whale abundance was estimated to be 270 animals in 1990 (Pace et al. 2017). Between 1990 to 2011, right whale abundance increased by approximately 2.8 percent per year, despite a decline in 1993 and no growth between 1997 and 2000 (Pace et al. 2017). However, since 2011, when the abundance peaked at 481 animals, the population has been in decline, with a 99.99 percent probability of a decline of just under 1 percent per year (Pace et al. 2017). Between 1990 and 2015, survival rates appeared relatively stable, but differed between the sexes, with males having higher survivorship than females (males: 0.985 ± 0.0038; females: 0.968 ± 0.0073) leading to a male-biased sex ratio (approximately 1.46 males per female) (Pace et al. 2017). Using the methods in Pace et al. (2017), as of January 2017, the median estimate of right whale abundance was 428 animals (95 percent credible intervals (CI) 406-447) and the minimum population estimate (N_{min}) was 418 animals; this estimate did not account for the 17 confirmed mortalities observed in June 2017 (12 in Canada; 5 in the United States) that triggered the designation of a Unusual Mortality Event (UME) for North Atlantic right whales (Hayes 2019). In 2018, there were three confirmed dead stranded right whales in the United States, and, in 2019, 10 confirmed dead stranded right whales (nine in Canada and one in the United States) (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2019-north-atlantic-right-whale-unusual-mortality-event). Each year, NMFS estimates the right whale population abundance and shares that estimate at the North Atlantic Right Whale Consortium's annual meeting. This estimate is considered preliminary and undergoes further review before being finalized in the North Atlantic Right Whale Stock Assessment Report.

Using the methods in Pace et al. (2017), this year's preliminary estimate is 368 (95 percent credible interval range of 356-378) individuals as of January 2019. Prior estimates considered the annual survival rate to be flat across the history of the time series. However, since 2010, annual survival rates have dropped. Therefore, the survival mechanism parameter in the model was adjusted to allow for different rates for different years. Using the original model, the population estimate is 371 (359-381) (Pace 2021). For the purposes of this Opinion, we are using the estimate of 368 individuals.[6] Updated photo-identification data support that the annual mortality rate changed significantly, and the new information reports a faster rate of decline than previously estimated. In these new analyses, the previous estimated of right whales alive as of January 2018 was revised down from 412 to 383. Additionally, the estimated right whale abundance for 2017 was likely lower than the estimated abundance of 428 individuals provided in the 2019 Stock Assessment Report (Hayes 2020).

In addition to finding an overall decline in the North Atlantic right whale population, Pace et al. (2017) also found that between 1990 and 2015, the survival of age 5+ females relative to 5+ males has been reduced; this has resulted in diverging trajectories for male and female abundance. Specifically, there was an estimated 142 males (95% CI=143-152) and 123 females (95% CI=116-128) in 1990; however, by 2015, model estimates show the species was comprised of 272 males (95% CI=261-282) and 186 females (95% CI=174-195; Pace et al. 2017). Calving rates also varied substantially between 1990 and 2015 (i.e., 0.3 percent to 9.5 percent), with low calving rates coinciding with three periods (1993-1995, 1998-2000, and 2012-2015) of decline or no growth (Pace et al. 2017). Using generalized linear models, Corkeron et al. (2018) found that between 1992 and 2016, North Atlantic right whale calf counts increased at a rate of 1.98 percent per year. Relative to three populations of southern right whales that increased 5.34 percent, 6.58 percent, and 7.21 percent per year, this rate of increase for North Atlantic right whales is substantially less (Corkeron et al. 2018). Using the highest annual estimates of survival recorded over the time series from Pace et al. (2017), and an assumed calving interval of approximately four years, Corkeron et al. (2018) suggests that the North Atlantic right whale population could potentially increase at a rate of at least 4 percent per year if there was no anthropogenic mortality.[7] This rate is approximately twice that observed, and the analysis indicates that adult female mortality is the main factor influencing this rate (Corkeron et al. 2018).

Status

The North Atlantic right whale is listed under the ESA as endangered. With anthropogenic mortality limiting the recovery of North Atlantic right whales (Corkeron et al. 2018), currently, none of the species recovery goals (see below) have been met. With whaling now prohibited, the two major known human causes of mortality are vessel strikes and entanglement in fishing gear (Hayes et al. 2018a). Estimates of total annual anthropogenic mortality (i.e., ship strike and entanglement in fishing gear), as well as the number of undetected anthropogenic mortalities for North Atlantic right whales have been provided by Hayes et al. (2020) and Pace et al. (2017); these estimates show that the total annual North Atlantic right whale mortality exceed or equal

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[6] We note that the population estimate of 368 whales is preliminary and has not completed the formal review process that NMFS applies to population estimates of this type. Nevertheless, while recognizing that the number may ultimately change, it was prepared through a transparent process that applied methods that NMFS has used for a number of years to estimate the population. Given that the analysis indicates a further decline in the population and that this consultation is intended to cover fisheries actions that occur after the expected completion of the review process, NMFS is making a conservative assumption that this preliminary number will likely represent the final number and is using it as the basis for this consultation.

[7] Based on information in the North Atlantic Right Whale Catalog, the mean calving interval is 4.69 years (P. Hamilton 2018, unpublished, in Corkeron et al. 2018). Corkeron et al. (2018) assumed a 4 year calving interval as the approximate mid-point between the North Atlantic Right Whale Catalog calving interval and observed calving intervals for southern right whales (i.e., 3.16 years for South Africa, 3.42 years for Argentina, 3.31 years for Auckland Islands, and 3.3 years for Australia).

the number of detected serious injuries and mortalities.[8] These anthropogenic threats appear to be worsening (Hayes et al. 2018a), as evidenced by the North Atlantic right whale UME declared by NMFS on June 7, 2017, as a result of elevated right whale mortalities along the Western North Atlantic Coast. As of April 2021, the confirmed mortalities for the UME are 34 dead stranded right whales (21 in Canada; 13 in the United States) (for more information on UMEs, see https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-unusual-mortality-events). Examinations by necropsy or photo documentation have been conducted on 23 of the 34 whales. Final results from some examinations are pending; however, preliminary findings indicate vessel strikes or rope entanglements as the cause of death. Additionally, since 2017, 15 live-free swimming non-stranded whales have been documented with serious injuries from entanglements (13) or vessel strikes (2). Therefore, the UME has been updated to 49 to include individuals to include both confirmed mortalities and seriously injured free-swimming whales.

The North Atlantic right whale population continues to decline. As provided above, between 1990 to 2011, right whale abundance increased by approximately 2.8 percent per year; however, since 2011 the population has been in decline (Pace et al. 2017). Recent modeling efforts indicate that low female survival, a male biased sex ratio, and low calving success are contributing to the population's current decline (Pace et al. 2017). For instance, five new calves were documented during the 2017 calving season, zero during the 2018 season, and seven during the 2019 season (Pettis et al. 2018a, Pettis et al. 2018b, Pettis et al. 2020), these number of births are well below the number needed to compensate for expected mortalities. More recently, there were 10 calves in the 2020 calving season and 17 calves in 2021, as of March 29. Two of the 2020 calves and one of the 2021 calves died or were seriously injured due to vessel strikes. Two additional calves were reported in the 2021 season, but were not seen as a mother/calf pair. One animal stranded dead with no evidence of human interaction and initial results suggest the calf died during birth or shortly thereafter. The second animal was an anecdotal report of a calf off the Canary Islands.

Long-term photographic identification data also indicate new calves rarely go undetected, so these years likely represent a continuation of low calving rates that began in 2012 (Kraus et al. 2007, Pace et al. 2017). While there are likely a multitude of factors involved, low calving has been linked to poor female health (Rolland et al. 2016) and reduced prey availability (Devine et al. 2017, Johnson et al. 2017, Meyer-Gutbrod and Green 2014, Meyer-Gutbrod and Greene 2018, Meyer-Gutbrod et al. 2018). A recent study comparing North Atlantic right whales to other right whale species found that juvenile, adult and lactating female North Atlantic right whales all had lower body condition scores compared to the southern right whale populations, with lactating females showing the largest difference (Christiansen et al. 2020). North Atlantic right whale calves were in good condition. While some of the difference could be the result of genetic isolation and adaptations to local environmental conditions, the authors suggest that the magnitude indicates that North Atlantic right are in poor condition, which could be suppressing their growth, survival, age of sexual maturation and calving rates. In addition, they conclude that the observed differences are most likely a result of differences in the exposure to anthropogenic factors (Christiansen et al. 2020). Furthermore, entanglement in fishing gear appears to have

[8] Currently, 72 percent of mortalities since 2000 are estimated to have been observed (Hayes et al. 2020).

substantial health and energetic costs that affect both survival and reproduction (Hayes et al. 2018a, Hunt et al. 2016, Lysiak et al. 2018, Pettis et al. 2017, Robbins et al. 2015, Rolland et al. 2017, van der Hoop et al. 2017a).

Kenney (2018) projected that if all other known or suspected impacts (e.g., vessel strikes, calving declines, climate change, resource limitation, sublethal entanglement effects, disease, predation, and ocean noise) on the population remained the same between 1990 and 2016, and none of the observed fishery related M/SI occurred, the projected population in 2016 would be 12.2 percent higher (506 individuals). Furthermore, if the actual mortality resulting from fishing gear is double the observed rate (as estimated in Pace et al. 2017), eliminating all mortalities (observed and unobserved) could have resulted in a 2016 population increase of 24.6 percent (562 individuals) and possibly over 600 in 2018 (Kenney 2018).

Given the above information, the resilience of North Atlantic right whales to future perturbations is expected to be very low (Hayes et al. 2018a). Using a matrix population projection model, it is estimated that by 2029 the population will decline from 160 females to the 1990 estimate of 123 females if the current rate of decline is not altered (Hayes et al. 2018a). Consistent with this, recent modelling efforts indicate that the species may decline towards extinction if prey conditions worsen and anthropogenic mortalities are not reduced (Meyer-Gutbrod et al. 2018). In fact, recent data from the Gulf of Maine and Gulf of St. Lawrence indicate prey densities may already be in decline (Devine et al. 2017, Johnson et al. 2017, Meyer-Gutbrod et al. 2018).

Factors Outside the Action Area Affecting the Status of Right Whale: Fishery Interactions and Vessel Strikes in Canadian Waters

In Canada, right whales are protected under the Species at Risk Act (SARA) and the Fisheries Act. The right whale was considered a single species and designated as endangered in 1980. SARA includes provisions against the killing, harming, harassing, capturing, taking, possessing, collecting, buying, selling, or trading of individuals or its parts (SARA section 32) and damage or destruction of its residence (SARA section 33). In 2003, the species was split to allow separate designation of the North Atlantic right whale, which was listed as endangered under SARA in May 2003. All marine mammals are subject to the provisions of the marine mammal regulations under the Fisheries Act. These include requirements related to approach, disturbance, and reporting. In the St. Lawrence estuary and the Saguenay River, the approach distance for threatened or endangered whales is 1312 ft (400 m).

North Atlantic right whales have died or been seriously injured in Canadian waters by vessel strikes and entanglement in fishing gear (DFO 2014). Serious injury and mortality events are rarely observed where the initial entanglement occurs. After an event, live whales or carcasses may travel hundreds of miles before ever being observed. It is unknown exactly how many serious injuries and mortalities have occurred in Canadian waters historically. However, at least 14 right whale carcasses and 20 injured right whales were sighted in Canadian waters between 1988 and 2014 (Davies and Brillant 2019); 25 right whale carcasses were first sighted in Canadian waters or attributed to Canadian fishing gear from 2015 through 2019. In the sections to follow, information is provided on the fishing and shipping industry in Canadian waters, as well as measures the Canadian government is taking (or will be taking) to reduce the level of serious injuries and mortalities to North Atlantic rights resulting from incidental entanglement in fishing gear or vessel strikes.

Fishery Interactions in Canadian Waters
There are numerous fisheries operating in Canadian waters. Rock and toad crab fisheries, as well as fixed gear fisheries for cod, Atlantic halibut, Greenland halibut, winter flounder and herring have historically had few interactions. While these fisheries deploy gear that pose some risk, this analysis focuses on fisheries that have a demonstrated interactions with ESA-listed species (i.e., lobster, snow crab, mackerel, and whelk). Based on information provided by the Department of Fisheries and Oceans Canada (DFO), a brief summary of these fisheries is provided below.

The American lobster fishery is DFO's largest fishery, by landings. It is managed under regional management plans with 41 Lobster Fisheries Areas (Figure 32), in which 10,000 licensed harvesters across Atlantic Canada and Quebec participate.[9] In addition to the one permanent closure in Lobster Fishery Area 40 (Figure 32) fisheries are generally closed during the summer to protect molts. Lobster fishing is most active in the Gulf of Maine, Bay of Fundy, Southern Gulf of St. Lawrence and coastal Nova Scotia. Most fisheries take place in shallow waters less than 130 ft (40 m) deep and within 8 nmi (15 km) of shore, although some fisheries will fish much farther out and in waters up to 660 ft (200 m) deep. Management measures are tailored to each Area and include limits on the number of licenses issued, limits on the number of traps, limited and staggered fishing seasons, limits on minimum and maximum carapace size (which differs depending on the Area), protection of egg-bearing females (females must be notched and released alive), and ongoing monitoring and enforcement of fishing regulations and license conditions. The Canadian lobster fisheries use trap/pot gear consistent with the gear used in the American lobster fishery. While both Canada and the U.S. lobster fisheries employ similar gears, the two nations employ different management strategies that result in divergent prosecution of the fisheries.

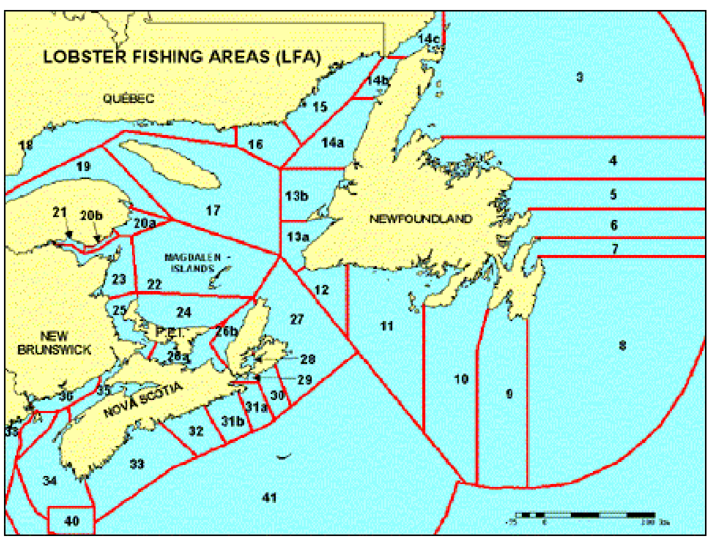

Figure 32: Lobster fishing areas in Atlantic Canada (https://www.dfo-mpo.gc.ca/fisheries-peches/commercial-commerciale/atl-arc/lobster-homard-eng.html)

The snow crab fishery is DFO's second largest fishery, by landings. It is managed under regional management plans with approximately 60 Snow Crab Management Areas in Canada spanning

[9] Of the 41 Lobster Fisheries Areas, one is for the offshore fishery, and one is closed for conservation.

four regions (Scotia-Fundy, Southern Gulf of St. Lawrence, Northern Gulf of St. Lawrence, and Newfoundland and Labrador). In 2010, 4,326 snow crab fishery licenses were issued. The DFO website (http://www.dfo-mpo.gc.ca/stats/commercial/licences-permis/species-especes/se17-eng.htm) indicated that 3,703 permits were issued in 2017. The management of the snow crab fishery is based on annual total allowable catch, individual quotas, trap and mesh restrictions, minimum legal size, mandatory release of female crabs, minimum mesh size of traps, limited seasons and areas. Protocols are in place to close grids when a percentage of soft-shell crabs in catches is reached. Harvesters use baited conical traps and pots set on muddy or sand-mud bottoms usually at depths of 230-460 ft (70-140 m). Annual permit conditions have been used since 2017 to minimize the impacts to North Atlantic right whales, as described below.

DFO manages the Atlantic mackerel fishery under one Atlantic management plan, established in 2007. Management measures include fishing seasons, total allowable catch, gear, Safety at Sea fishing areas, licensing, minimum size, fishing gear restrictions, and monitoring. The plan allows the use of the following gear: gillnet, handline, trap net, seine, and weir. When established, the DFO issued 17,182 licenses across four regions, with over 50 percent of these licenses using gillnet gear. In 2017, DFO issued 7,965 licenses (http://www.dfo-mpo.gc.ca/stats/commercial/licences-permis/species-especes/se17-eng.htm); no gear information was available. Commercial harvest is timed with the migration of mackerel into and out of Canadian waters. In Nova Scotia, the gillnet and trap fisheries for mackerel take place primarily in June and July. Mackerel generally arrive in southwestern Nova Scotia in May, and Cape Breton in June. Migration out of the Gulf of St. Lawrence begins in September, and the fishery can continue into October or early November. They may enter the Gulf of St. Lawrence, depending on temperature conditions. The gillnet fishery in the Gulf of St. Lawrence also occurs in June and July. Most nets are fixed, except for a drift fishery in Chaleurs Bay and the part of the Gulf between New Brunswick, Prince Edward Island, and the Magdalen Islands.

Conservation harvesting plans are used to manage waved whelk in Canadian waters, which are harvested in the Gulf of St. Lawrence, Quebec, Maritimes, and Newfoundland and Labrador regions. The fishery is managed using quotas, fishing gear requirements, dockside monitoring, traps limits, seasons, tagging, and area requirements. In 2017, there were 240 whelk license holders in Quebec; however, only 81 of them were active. Whelk traps are typically weighted at the bottom with cement or other means and a rope or other mechanism is positioned in the center of the trap to secure the bait. Between 50 and 175 traps are authorized per license. The total number of authorized traps for all licenses in each fishing area varies between 550 and 6,400 traps, while the number of used or active traps is lower, with 200 to 1,700 traps per fishing area.

Since 2017, the Government of Canada has implemented measures to protect right whales from entanglement. These measures have included seasonal and dynamic closures for fixed gear fisheries, changes to the fishing season for snow crab, reductions in traps in the mid-shore fishery in Crab Fishing Area 12, and license conditions to reduce the amount of rope in the water. Measures to better track gear, require reporting of gear loss, require reporting of interactions with marine mammals, and increased surveillance for right whales have also been implemented. Measures to reduce interactions with fishing gear are adjusted annually. In 2021, mandatory closures for non-tended fixed gear fisheries, including lobster and crab, will be put in place for 15 days when right whales are sighted. If a whale is detected in days 9-15 of the closure, the closure will be extended. In the Bay of Fundy and the critical habitats in the Roseway and Grand Manan basins, this extension will be for an additional 15 days. If a right whale is detected in the Gulf of St. Lawrence, the closure will be season-long (until November 15, 2021). Outside the dynamic area, closures are considered on a case-by-case basis. There are also gear marking and reporting requirements for all fixed gear fisheries. The Government of Canada will also continue to support industry trials of innovative fishing technologies and methods to prevent and mitigate whale entanglement. This includes authorizing ropeless gear trials in closed areas in 2021. Measures to implement weak rope or weak-breaking points were delayed and will be implemented by the end of 2022. Measures related to maximum rope diameters, sinking rope between traps, and reductions in vertical and floating rope will be implemented after 2022. More information on these measures is available at https://www.dfo-mpo.gc.ca/fisheries-peches/commercial-commerciale/atl-arc/narw-bnan/management-gestion-eng.html.

In August 2016, NMFS published the MMPA Import Provisions Rule (81 FR 54389, August 15, 2016), which established criteria for evaluating a harvesting nation's regulatory program for reducing marine mammal bycatch and the procedures for obtaining authorization to import fish and fish products into the United States. Specifically, to continue in the international trade of seafood products with the United States, other nations must demonstrate that their marine mammal mitigation measure for commercial fisheries are, at a minimum, equivalent to those in place in the United States. A five-year exemption period (beginning January 1, 2017) was created in this process to allow foreign harvesting nations time to develop, as appropriate, regulatory programs comparable in effectiveness to U.S. programs at reducing marine mammal bycatch. To comply with its requirements, it is essential that these interactions are reported, documented and quantified. To guarantee that fish products have access to the U.S. markets, DFO must implement procedures to reliably certify that the level of mortality caused by fisheries does not exceed U.S. standards. DFO must also demonstrate that the regulations in place to reduce accidental death of marine mammals are comparable to those of the United States.

Vessel Strikes in Canadian Waters

Vessel strikes are a threat to right whales throughout their range. In Canadian waters where right whales are present, vessels include recreational and commercial vessels, small and large vessels, and sail, and power vessels. Vessel categories include oil and gas exploration, fishing and aquaculture, cruise ships, offshore excursions (whale and bird watching), tug/tow, dredge, cargo, and military vessels. At the time of development of the Gulf of St. Lawrence management plan, approximately 6400 commercial vessels transited the Cabot Strait and the Strait of Belle Isle annually. This represents a subset of the vessels in this area as it only includes commercial vessels (DFO 2013). To address vessel strikes in Canadian waters, the International Maritime Organization (IMO) amended the Traffic Separation Scheme in the Bay of Fundy to reroute vessels around high use areas. In 2007, IMO adopted and Canada implemented a voluntary seasonal Area to Be Avoided (ATBA) in Roseway Basin to further reduce the risk of vessel strike (DFO 2020). In addition, Canada has implemented seasonal speed restrictions and developed a proposed action plan to identify specific measures needed to address threats and achieve recovery (DFO 2020).

The Government of Canada has also implemented measures to mitigate vessel strikes in Canadian waters. Each year since August 2017, the Government has implemented seasonal speed restrictions (maximum 10 knots) for vessels 20 meters or longer in the western Gulf of St. Lawrence. In 2019, the area was adjusted and the restriction was expanded to apply to vessels greater than 13 m. Smaller vessels are encouraged to respect the limit. Dynamic area management has also been used in recent years. Currently, there are two shipping lanes, south and north of Anticosti Island, where dynamic speed restrictions (mandatory slowdown to 10 knots) can be activated when right whales are present. In 2020 and 2021, the Government of Canada also implemented a trial voluntary speed restriction zone from Cabot Strait to the eastern edge of the dynamic shipping zone at the beginning and end of the season and a mandatory restricted area in or near Shediac Valley mid-season. More information is available at https://www.tc.gc.ca/en/services/marine/navigation-marine-conditions/protecting-north-atlantic-right-whales-collisions-ships-gulf-st-lawrence.html. Modifications to measures in 2021 include refining the size, location, and duration of the mandatory restricted area in and near Shediac Valley and expanding the speed limit exemption in waters less than 20 fathoms to all commercial fishing vessels

Critical Habitat
We have determined that the proposed action is not likely to adversely affect designated critical habitat for North Atlantic right whales (see section 4.1.10; Figure 29).

Recovery Goals
Under the ESA, NOAA publishes recovery plans that outline the path and task required to restore and secure self-sustaining wild population. If successfully implemented, recovery plans result in listed species being reclassified from endangered or threatened or delisting and removal of the species from ESA protection. Recovery plans include objective criteria for measuring recovery. The 2005 Recovery Plan for the North Atlantic right whale (NMFS 2005) does not include delisting criteria. Criteria for downlisting include:

Downlisting
1. Population ecology and vital rates are indicative of an increasing population;
2. Population has increased for 35 years at an average rate of increase equal to or greater than 2 percent per year;
3. None of the known threats are known to limit the population's growth rate; and
4. The population has no more than a 1 percent chance of quasi-extinction in 100 years.

In addition, it includes the following actions/objectives:
1. Significantly reduce sources of human-caused death, injury and disturbance (e.g., vessel collisions, fishery interactions).
2. Develop demographically-based recovery criteria.
3. Identify, characterize, protect and monitor important habitats.
4. Monitor the status and trends of abundance and distribution of the western North Atlantic right whale population.
5. Coordinate federal, state, local, international and private efforts to implement the recovery plan.

4.2.1.2. Fin Whale (Balaenoptera physalus)

Globally there is one species of fin whale, *Balaenoptera physalus*. Fin whales occur in all major oceans of the Northern and Southern Hemispheres (NMFS 2010c) (Figure 33). Within this range, three subspecies of fin whales are recognized: *B. p. physalus* in the Northern Hemisphere, and *B. p. quoyi* and *B. p. patachonica* (a pygmy form) in the Southern Hemisphere (NMFS 2010c). For management purposes in the northern Hemisphere, the United States divides, *B. p. physalus,* into four stocks: Hawaii, California/Oregon/Washington, Alaska (Northeast Pacific), and Western North Atlantic (Hayes et al. 2019, NMFS 2010c).

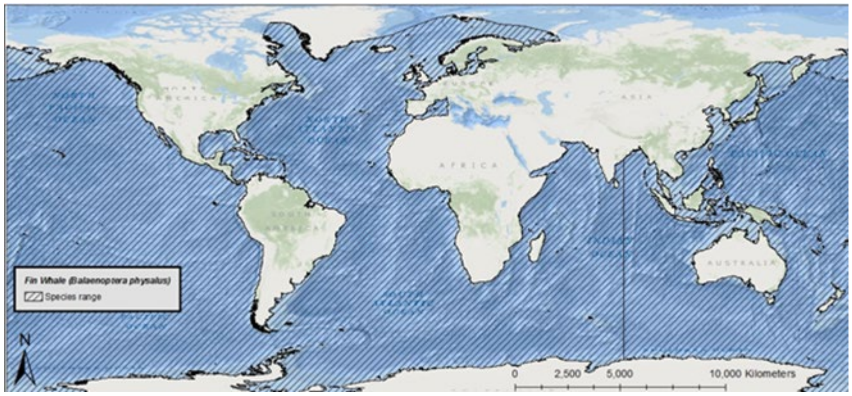

Figure 33: Range of the fin whale

Fin whales are distinguishable from other whales by a sleek, streamlined body, with a V-shaped head, a tall hooked dorsal fin, and a distinctive color pattern of a black or dark brownish-gray body and sides with a white ventral surface. The lower jaw is gray or black on the left side and creamy white on the right side. The fin whale was listed as endangered on December 2, 1970 (35 FR 18319).

Information available from the recovery plan (NMFS 2010c), recent stock assessment reports (Carretta et al. 2019a, Hayes et al. 2019, Muto et al. 2019a), the five-year status review (NMFS 2019d), as well as the recent International Union for the Conservation of Nature's (IUCN) fin whale assessment (Cooke 2018b) were used to summarize the life history, population dynamics and status of the species as follows.

Life History
Fin whales can live, on average, 80 to 90 years. They have a gestation period of less than one year, and calves nurse for six to seven months. Sexual maturity is reached between 6 and 10 years of age with an average calving interval of two to three years. They mostly inhabit deep, offshore waters of all major oceans. They winter at low latitudes, where they calve and nurse, and summer at high latitudes, where they feed, although some fin whales appear to be residential to certain areas.

Population Dynamics
The pre-exploitation estimate for the fin whale population in the entire North Atlantic was approximately 30,000-50,000 animals (NMFS 2010c), and for the entire North Pacific Ocean, approximately 42,000 to 45,000 animals (Ohsumi and Wada 1974). In the Southern Hemisphere, prior to exploitation, the fin whale population was approximately 40,000 whales (Mizroch et al. 1984b). In the North Atlantic Ocean, fin whales were heavily exploited from 1864 to the 1980s; over this timeframe, approximately 98,000 to 115,000 fin whales were killed (IWC 2017). Between 1910-1975, approximately 76,000 fin whales were recorded taken by modern whaling in the North Pacific; this number is likely higher as many whales killed were not identified to species or while killed, where not successfully landed (Allison 2017). Over 725,000 fin whales were killed in the Southern Hemisphere from 1905 to 1976 (Allison 2017).

In the North Atlantic Ocean, the IWC has defined seven management stocks of fin whales: (1) North Norway (2) East Greenland and West Iceland (EGI); (3) West Norway and the Faroes; (4) British Isles, Spain and Portugal; (5) West Greenland and (6) Nova Scotia, (7) Newfoundland and Labrador (Donovan 1991, NMFS 2010c). Based on three decades of survey data in various portions of the North Atlantic, the IWC estimates that there are approximately 79,000 fin whales in this region. Under the present IWC scheme, fin whales off the eastern United States, Nova Scotia and the southeastern coast of Newfoundland are believed to constitute a single stock; in U.S. waters, NMFS classifies these fin whales as the Western North Atlantic stock (Donovan 1991, Hayes et al. 2019, NMFS 2010c). NMFS' best estimate of abundance for the Western North Atlantic Stock of fin whales is 7,418 individuals (N_{min}=6,029); this estimate is the sum of the 2016 NOAA shipboard and aerial surveys and the 2016 Canadian Northwest Atlantic International Sightings Survey (Hayes 2019). Currently, there is no population estimate for the entire fin whale population in the North Pacific (Cooke 2018b). However, abundance estimates for three stocks in U.S. Pacific Ocean waters do exist: Northeast Pacific (N= 3,168; N_{min}=2,554), Hawaii (N=154; N_{min}=75), and California/Oregon/Washington (N= 9,029; N_{min}=8,127) (Nadeem et al. 2016). Abundance data for the Southern Hemisphere stock remain highly uncertain; however, available information suggests a substantial increase in the population has occurred (Thomas et al. 2016).

In the North Atlantic, estimates of annual growth rate for the entire fin whale population in this region is not available (Cooke 2018b). However, in U.S. Atlantic waters NMFS has determined that until additional data is available, the cetacean maximum theoretical net productivity rate of 4.0 percent will be used for the Western North Atlantic stock (Hayes et al. 2019). In the North Pacific, estimates of annual growth rate for the entire fin whale population in this region is not available (Cooke 2018b). However, in U.S. Pacific waters, NMFS has determined that until additional data is available, the cetacean maximum theoretical net productivity rate of 4.0 percent will be used for the Northeast Pacific stock (Muto et al. 2019b, NMFS 2016c). Overall population growth rates and total abundance estimates for the Hawaii stock of fin whales are not available at this time (Carretta et al. 2018). Based on line transect studies between 1991-2014, there was estimated a 7.5 percent increase in mean annual abundance in fin whales occurring in waters off California, Oregon, and Washington; to date, this represents the best available information on the current population trend for the overall California/Oregon/Washington stock of fin whales (Carretta et al. 2019a, Nadeem et al. 2016).[10] For Southern Hemisphere fin whales, as noted above, overall information suggests a substantial increase in the population; however the rate of increase remains poorly quantified (Cooke 2018b).

Archer et al. (2013) examined the genetic structure and diversity of fin whales globally. Full sequencing of the mitochondrial DNA genome for 154 fin whales sampled in the North Atlantic Ocean, North Pacific Ocean, and Southern Hemisphere, resulted in 136 haplotypes, none of which were shared among ocean basins suggesting differentiation at least at this geographic scale. However, North Atlantic fin whales appear to be more closely related to the Southern Hemisphere population, as compared to fin whales in the North Pacific Ocean, which may

[10] Since 2005, the fin whale abundance increase has been driven by increases off northern California, Oregon, and Washington; numbers off Central and Southern California have remained stable (Carretta et al. 2020, Nadeem et al. 2016).

indicate a revision of the subspecies delineations is warranted. Generally, haplotype diversity was found to be high both within and across ocean basins (Archer et al. 2013). Such high genetic diversity and lack of differentiation within ocean basins may indicate that despite some populations having small abundance estimates, the species may persist long-term and be somewhat protected from substantial environmental variance and catastrophes.

Status
The fin whale is endangered as a result of past commercial whaling. Prior to commercial whaling, hundreds of thousands of fin whales existed. Fin whales may be killed under "aboriginal subsistence whaling" in Greenland, under Japan's scientific whaling program, and Iceland's formal objection to the IWC's ban on commercial whaling. Additional threats include vessel strikes, reduced prey availability due to overfishing or climate change, and sound. The species' overall large population size may provide some resilience to current threats, but trends are largely unknown.

Critical Habitat
No critical habitat has been designated for the fin whale.

Recovery Goals
Under the ESA, NOAA publishes recovery plans that outline the path and task required to restore and secure self-sustaining wild population. If successfully implemented, recovery plans result in listed species being reclassified from endangered or threatened or delisting and removal of the species from ESA protection. Recovery plans include objective criteria for measuring recovery. The 2010 Recovery Plan for the fin whale (NMFS 2010c) includes the following criteria for downlisting and delisting:

Downlisting

1. The population in the ocean basin in which it occurs has no more than 1 percent chance of extinction in 100 years and has at least 500 mature reproductive individuals (consisting of at least 250 females and 250 males) in each ocean basin.
2. None of the known threats are known to limit the continued growth of the populations.

Delisting

1. The population in the ocean basin in which it occurs has less than a 10 percent probability of becoming endangered in 20 years.
2. None of the known threats are known to limit the continued growth of the populations.

It also includes the following recovery actions/objectives:

1. Coordinate state, federal, and international actions to implement recovery actions and maintain international regulation of whaling for fin whales.
2. Determine population discreteness and population structure of fin whales.
3. Develop and apply methods to estimate population size and monitor trends in abundance.
4. Conduct risk analysis.
5. Identify, characterize, protect, and monitor habitat important to fin whale populations in U.S. waters and elsewhere.
6. Investigate causes and reduce the frequency and severity of human-caused injury and mortality.
7. Determine and minimize any detrimental effects of anthropogenic noise in the oceans.

8. Maximize efforts to acquire scientific information from dead, stranded, and/or entrapped fin whales.
9. Develop post-delisting monitoring plan.

4.2.1.3. Sei Whale (Balaenoptera borealis)

Globally there is one species of sei whale, *Balaenoptera borealis borealis*. Sei whales occur in subtropical, temperate and subpolar marine waters across the Northern and Southern Hemispheres (Figure 34) (Cooke 2018a, NMFS 2011b). For management purposes, in the Northern Hemisphere, the United States recognizes four sei whale stocks: Hawaii, Eastern North Pacific, and Nova Scotia (NMFS 2011b).

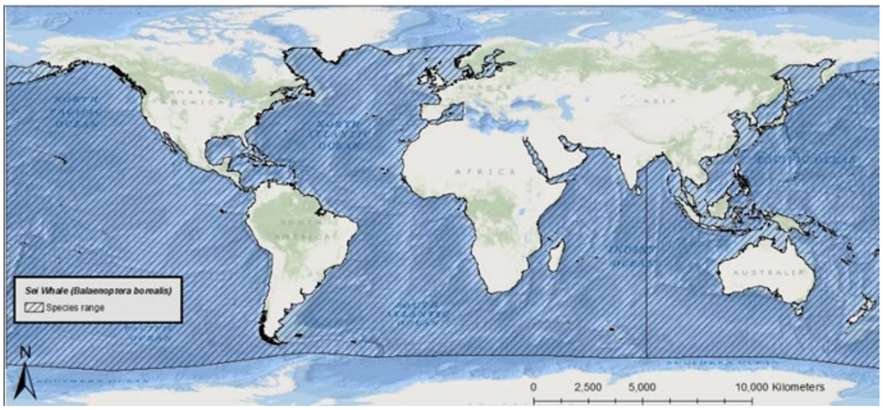

Figure 34: Range of the sei whale

Sei whales are distinguishable from other whales by a long, sleek body that is dark bluish-gray to black in color and pale underneath, and a single ridge located on their rostrum. The sei whale was listed as endangered on December 2, 1970 (35 FR 18319).

Information available from the recovery plan (NMFS 2011b), recent stock assessment reports (Carretta et al. 2019a, Hayes 2019, Hayes et al. 2017a), status review (NMFS 2012f), as well as the recent IUCN sei whale assessment (Cooke 2018a) were used to summarize the life history, population dynamics and status of the species as follows.

Life History
Sei whales can live, on average, between 50 and 70 years. They have a gestation period of 10 to 12 months, and calves nurse for six to nine months. Sexual maturity is reached between 6 and 12 years of age with an average calving interval of two to three years. Sei whales mostly inhabit continental shelf and slope waters far from the coastline. They winter at low latitudes, where they calve and nurse, and summer at high latitudes, where they feed on a range of prey types, including: plankton (copepods and krill), small schooling fishes, and cephalopods.

Population Dynamics
There are no estimates of pre-exploitation sei whale abundance in the entire North Atlantic Ocean; however, approximately 17,000 sei whales were documented caught by modern whaling in the North Atlantic (Allison 2017). In the North Pacific, the pre-whaling sei abundance was estimated to be approximately 42,000 (Tillman 2977 as cited in NMFS 2011b). In the Southern Hemisphere, approximately 63,100 to 65,000 occurred in the Southern Hemisphere prior to exploitation (Mizroch et al. 1984a, NMFS 2011b).

In the North Atlantic, the entire North Atlantic sei whale population, in 1989, was estimated to be 10,300 whales (Cattanach et al. 1993 as cited in NMFS 2011b). While other surveys have been completed in portions of the North Atlantic since 1989, the survey coverage levels in these studies are not as complete as those done in Cattanach et al. (1993) (Cooke 2018a). As a result, to date, updated abundance estimates for the entire North Atlantic population of sei whales are not available. However, in the western North Atlantic, Palka et al. (2017) has provided a recent abundance estimate for the Nova Scotia stock of sei whales. Based on survey data collected from Halifax, Nova Scotia, to Florida between 2010 and 2013, it is estimated that there are approximately 6,292 sei whales (N_{min}=3,098) (Palka et al. 2017); this estimate is considered the best available for the Nova Scotia stock (Hayes 2019). In the North Pacific, an abundance estimate for the entire North Pacific population of sei whales is not available. However, in the western North Pacific, it is estimated that there are 35,000 sei whales (Cooke 2018a). In the eastern North Pacific (considered east of longitude 180º), two stocks of sei whales occur in U.S. waters: Hawaii and Eastern North Pacific. Abundance estimates for the Hawaii stock are 391 sei whales (N_{min}=204), and for Eastern North Pacific stock, 519 sei whales (N_{min}=374) (Carretta et al. 2019a). In the Southern Hemisphere, recent abundance of sei whales is estimated at 9,800 to 12,000 whales. Population growth rates for sei whales are not available at this time as there are little to no systematic survey efforts to study sei whales; however, in U.S. waters, NMFS has determined that until additional data is available, the cetacean maximum theoretical net productivity rate of 4.0 percent will be used for the Hawaii, Eastern North Pacific, and Hawaii stocks of sei whales (Hayes 2019).

Based on genetic analyses, there appears to be some differentiation between sei whale populations in different ocean basins. In an early analysis of genetic variation in sei whales some differences between Southern Ocean and the North Pacific sei whales were detected (Wada and Numachi 1991). However, more recent analyses of mtDNA control region variation show no significant differentiation between Southern Ocean and the North Pacific sei whales, though both appear to be genetically distinct from sei whales in the North Atlantic (Huijser et al. 2018). Within ocean basin, there appears to be intermediate to high genetic diversity and little genetic differentiation despite there being different managed stocks (Danielsdottir et al. 1991, Kanda et al. 2011, Kanda et al. 2006, Kanda et al. 2013, Kanda et al. 2015).

Status

The sei whale is endangered as a result of past commercial whaling. Now, only a few individuals are taken each year by Japan; however, Iceland has expressed an interest in targeting sei whales. Current threats include vessel strikes, fisheries interactions (including entanglement), climate change (habitat loss and reduced prey availability), and anthropogenic sound. Given the species' overall abundance, they may be somewhat resilient to current threats. However, trends are largely unknown, especially for individual stocks, many of which have relatively low abundance estimates.

Critical Habitat

No critical habitat has been designated for the sei whale.

Recovery Goals

Under the ESA, NOAA publishes recovery plans that outline the path and tasks required to restore and secure self-sustaining wild populations. If successfully implemented, recovery plans result in listed species being reclassified from endangered or threatened or delisting and removal of the species from ESA protection. Recovery plans include objective criteria for measuring recovery. The 2011 Recovery Plan for the sei whale (NMFS 2011b) includes the following criteria for downlisting/delisting.

Downlisting

1. The population in the ocean basin in which it occurs has no more than 1 percent chance of extinction in 100 years and the global population has at least 1,500 mature reproductive individuals (consisting of at least 250 mature females and at least 250 mature males in each ocean basin).
2. None of the known threats are known to limit the continued growth of the populations.

Delisting

3. The population in the ocean basin in which it occurs has less than a 10 percent probability of becoming endangered in 20 years.
4. None of the known threats are known to limit the continued growth of the populations.

It also includes the following recovery actions/objectives:

1. Coordinate state, federal, and international actions to maintain international regulation of whaling for sei whales.
2. Develop and apply methods to collect sei whale data.
3. Support existing studies to investigate population discreteness and population structure of sei whales using genetic analyses.
4. Continue to collect data on threats (e.g., fishery interactions, anthropogenic noise, vessel interactions, climate change) and severity of threats to sei whale recovery.
5. Maximize efforts to acquire scientific information from dead, stranded, and entangled sei whales.
6. Estimate population size and monitor trends in abundance.
7. Initiate new studies to determine population discreteness and population structure of sei whales.
8. Conduct risk analyses;
9. Identify, characterize, protect, and monitor habitat important to sei whale populations in U.S. waters and elsewhere.
10. Investigate human-caused threats, and, should they be determined to be medium or high, reduce frequency and severity.
11. Develop post-delisting monitoring plan.

4.2.1.4. Sperm Whale (Physeter microcephalus)

Globally there is one species of sperm whale, *Physeter macrocephalus*. Sperm whales occur in all major oceans of the Northern and Southern Hemispheres (NMFS 2010d) (Figure 35). For management purposes, in the Northern Hemisphere, the United States recognizes six sperm whale stocks: California/Oregon/Washington, Hawaii, North Pacific, North Atlantic, Northern Gulf of Mexico, and Puerto Rico and the U.S. Virgin Islands (NMFS 2010d); see NMFS Marine Mammal Stock Assessment Reports: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessment-reports-species-stock).

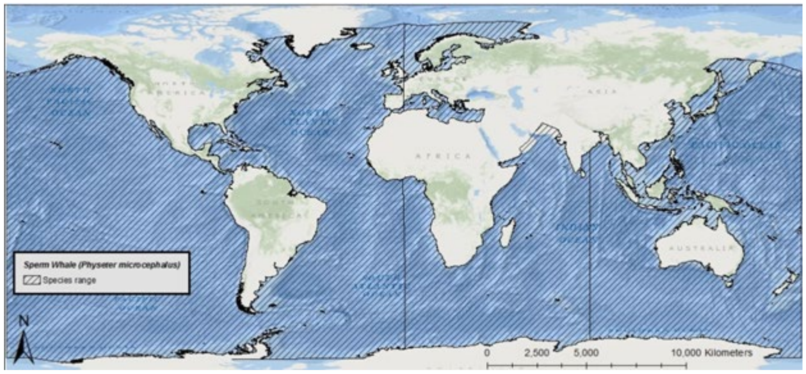

Figure 35: Range of the sperm whale

The sperm whale is the largest toothed whale and distinguishable from other whales by its extremely large head, which takes up 25 to 35 percent of its total body length and a single blowhole asymmetrically situated on the left side of the head near the tip. The sperm whale was originally listed as endangered on December 2, 1970 (35 FR 18319).

Information available from the recovery plan (NMFS 2010d), recent stock assessment reports (Carretta et al. 2018, Hayes et al. 2018b, Muto et al. 2018), status review (NMFS 2015e), as well as the recent IUCN sperm whale assessment (Taylor et al. 2019) were used to summarize the life history, population dynamics and status of the species as follows.

Life History
The average lifespan of sperm whales is estimated to be at least 50 years (Whitehead 2009). They have a gestation period of one to one and a half years, and calves nurse for approximately two years, though they may begin to forage for themselves within the first year of life (Tønnesen et al. 2018). Sexual maturity is reached between 7 and 13 years of age for females with an average calving interval of four to six years. Male sperm whales reach full sexual maturity in their 20s. Sperm whales mostly inhabit areas with a water depth of 1970 ft (600 m) or more, and are uncommon in waters less than 985 ft (300 m) deep. They winter at low latitudes, where they calve and nurse, and summer at high latitudes, where they feed primarily on squid; other prey includes octopus and demersal fish (including teleosts and elasmobranchs).

Population Dynamics
Pre-whaling, the global population of sperm whales was estimated to be approximately 1,100,000 animals (Taylor et al. 2019, Whitehead 2002). By 1880, due to whaling, the population was approximately 71 percent of its original level (Whitehead 2002). In 1999, ten years after the end of large-scale whaling, the population was estimated to be about 32 percent of its original level (Whitehead 2002).

The most recent global sperm whale population estimate is 360,000 whales (Whitehead 2009). There are no reliable estimates for sperm whale abundance across the entire (North and South) Atlantic Ocean. However, estimates are available for two of three U.S. stocks in the western North Atlantic Ocean; the Northern Gulf of Mexico stock is estimated to consist of 763 individuals (N_{min}=560) (Waring et al. 2016) and the North Atlantic stock is estimated to consist of 4,349 individuals (N_{min}=3,451) (Hayes 2019). There are insufficient data to estimate abundance for the Puerto Rico and U.S. Virgin Islands stock. Similar to the Atlantic Ocean, there are no reliable estimates for sperm whale abundance across the entire (North and South) Pacific Ocean. However, estimates are available for two of three U.S. stocks that occur in (Waring et al.

2010) the eastern Pacific; the California/Oregon/ Washington stock is estimated to consist of 1,997 individuals (N_{min}=1,270; Carretta et al. 2019b), and the Hawaii stock is estimated to consist of 4,559 individuals (N_{min}=3,478) (Carretta et al. 2019a). We are aware of no reliable abundance estimates for sperm whales in other major oceans in the Northern and Southern Hemispheres. Although maximum net productivity rates for sperm whales have not been clearly defined, population growth rates for sperm whale populations are expected to be low (i.e., no more than 1.1 percent per year) (Whitehead 2002). In U.S. waters, NMFS determined that, until additional data is available, the cetacean maximum theoretical net productivity rate of 4.0 percent will be used for, among others, the North Atlantic, Northern Gulf of Mexico, and Puerto Rico and the U.S. Virgin Islands stocks of sperm whales (Carretta et al. 2019a, Carretta et al. 2019b, Hayes 2019, Muto et al. 2019a, Muto et al. 2019b, Waring et al. 2010, Waring et al. 2016).

Ocean-wide genetic studies indicate sperm whales have low genetic diversity, suggesting a recent bottleneck, but strong differentiation between matrilineally-related groups (Lyrholm and Gyllensten 1998). Consistent with this, two studies of sperm whales in the Pacific Ocean indicate low genetic diversity (Mesnick et al. 2011, Rendell et al. 2012). Furthermore, sperm whales from the Gulf of Mexico, the western North Atlantic Ocean, the North Sea, and the Mediterranean Sea all have been shown to have low levels of genetic diversity (Engelhaupt et al. 2009). As none of the stocks for which data are available have high levels of genetic diversity, the species may be at some risk to inbreeding and 'allee' effects[11], although the extent to which is currently unknown. Sperm whales have a global distribution and can be found in relatively deep waters in all ocean basins. While both males and females can be found in latitudes less than 40 degrees, only adult males venture into the higher latitudes near the poles.

Status
The sperm whale is endangered as a result of past commercial whaling. Although the aggregate abundance worldwide is probably at least several hundred thousand individuals, the extent of depletion and degree of recovery of populations are uncertain. Commercial whaling is no longer allowed, however, illegal hunting may occur. Continued threats to sperm whale populations include vessel strikes, entanglement in fishing gear, competition for resources due to overfishing, population, loss of prey and habitat due to climate change, and sound. The Deepwater Horizon Natural Resource Damage Assessment Trustees assessed effects of oil exposure on sea turtles and marine mammals. Sperm whales in the Gulf of Mexico were impacted by the oil spill with 3 percent of the stock estimated to have died (DWH NRDA Trustees 2016). The species' large population size shows that it is somewhat resilient to current threats.

Critical Habitat
No critical habitat has been designated for the sperm whale.

Recovery Goals
Under the ESA, NOAA publishes recovery plans that outline the path and tasks required to restore and secure self-sustaining wild populations. If successfully implemented, recovery plans result in listed species being reclassified from endangered or threatened or delisting and removal

[11] Allee effects are broadly characterized as a decline in individual fitness in populations with a small size or density.

of the species from ESA protection. Recovery plans include objective criteria for measuring recovery. The 2010 Recovery Plan for the sperm whale (NMFS 2010d) includes the following downlisting/delisting criteria:

Downlisting

1. The population in the ocean basin in which it occurs has no more than 1 percent chance of extinction in 100 years and has at least 1,500 mature reproductive individuals (consisting of at least 250 females and 250 males) in each ocean basin.
2. None of the known threats are known to limit the continued growth of the populations.

Delisting

1. The population in the ocean basin in which it occurs has less than a 10 percent probability of becoming endangered in 20 years.
2. None of the known threats are known to limit the continued growth of the populations.

It also includes the following recovery actions/objectives:

1. Coordinate state, federal, and international actions to implement recovery action and maintain international regulation of whaling for sperm whales.
2. Develop and apply methods to estimate population size and monitor trends in abundance.
3. Determine population discreteness and population structure of sperm whales.
4. Conduct risk analyses.
5. Identify, characterize, protect, and monitor habitat important to sperm whale populations in U.S. waters and elsewhere.
6. Investigate causes of, and reduce the frequency and severity of, human-caused injury and mortality.
7. Determine and minimize any detrimental effects of anthropogenic noise in the oceans;
8. Maximize efforts to acquire scientific information from dead, stranded, and entangled sperm whales.
9. Develop post-delisting monitoring plan.

4.2.2. ***Sea Turtles***

Kemp's ridley and leatherback sea turtles are currently listed under the ESA at the species level; green and loggerhead sea turtles are listed at the DPS level. Therefore, we include information on the range-wide status of Kemp's ridley and leatherback sea turtles to provide the overall status of each species. Information on the status of loggerhead and green sea turtles is for the DPS affected by this action. Additional background information on the range-wide status of these species can be found in a number of published documents, including sea turtle status reviews and biological reports (Conant et al. 2009, Hirth 1997, NMFS and USFWS 1995, Seminoff et al. 2015, TEWG 1998, 2000, 2007, 2009) and recovery plans and five-year reviews for the loggerhead sea turtle (Bolten et al. 2019, NMFS and USFWS 2008), Kemp's ridley sea turtle (NMFS and USFWS 2015, NMFS et al. 2011), green sea turtle (NMFS and USFWS 1991b), and leatherback sea turtle (NMFS and USFWS 1992, 1998a, 2013).

4.2.2.1. Green Sea Turtle (North Atlantic DPS)

The green sea turtle has a circumglobal distribution, occurring throughout tropical, subtropical and, to a lesser extent, temperate waters. They commonly inhabit nearshore and inshore waters. It is the largest of the hardshell marine turtles, growing to a weight of approximately 350 lbs (159 kg) and a straight carapace length of greater than 3.3 ft (1 m). The species was listed under the ESA on July 28, 1978 (43 FR 32800) as endangered for breeding populations in Florida and the Pacific coast of Mexico and threatened in all other areas throughout its range. On April 6, 2016, NMFS listed 11 DPSs of green sea turtles as threatened or endangered under the ESA (81 FR 20057). The North Atlantic DPS of green turtle is found in the North Atlantic Ocean and Gulf of Mexico (Figure 36) and is listed as threatened. Green turtles from the North Atlantic DPS range from the boundary of South and Central America (7.5° N, 77° W) in the south, throughout the Caribbean, the Gulf of Mexico, and the U.S. Atlantic coast to New Brunswick, Canada (48° N, 77° W) in the north. The range of the DPS then extends due east along latitudes 48° N and 19° N to the western coasts of Europe and Africa.

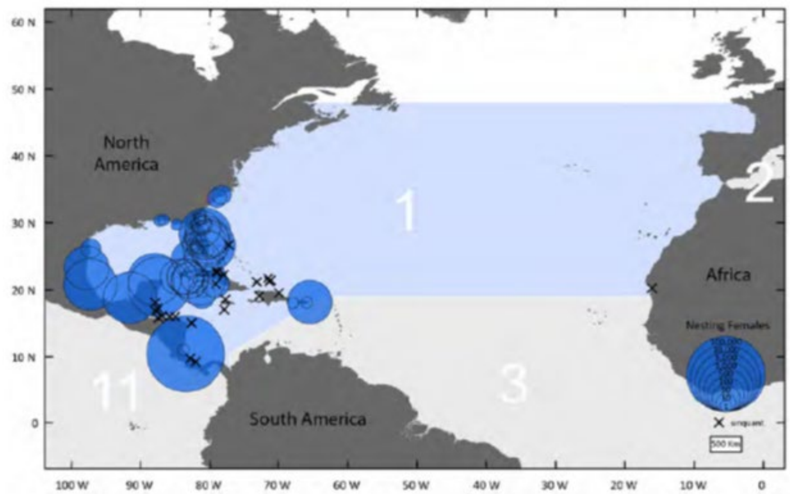

Figure 36: Range of the North Atlantic distinct population segment of green turtle with location and abundance of nesting females (Seminoff et al. 2015).

We used information available in the 2015 Status Review (Seminoff et al. 2015), relevant literature, and recent nesting data from the Florida Fish and Wildlife Conservation Commission's Fish and Wildlife Research Institute (FWRI) to summarize the life history, population dynamics and status of the species, as follows.

Life history

Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, Quintana Roo), United States (Florida) and Cuba (Figure 36) support nesting concentrations of particular interest in the North Atlantic DPS (Seminoff et al. 2015). The largest nesting site in the North Atlantic DPS is in Tortuguero, Costa Rica, which hosts 79 percent of nesting females for the DPS (Seminoff et al. 2015). In the southeastern United States, females generally nest between May and September (Seminoff et al. 2015, Witherington et al. 2006). Green sea turtles lay an average of three nests per season with an average of one hundred eggs per nest (Hirth 1997, Seminoff et al. 2015). The remigration interval (period between nesting seasons) is two to five years (Hirth 1997, Seminoff et al. 2015). Nesting occurs primarily on beaches with intact dune structure, native vegetation and appropriate incubation temperatures during the summer months.

Sea turtles are long-lived animals. Size and age at sexual maturity have been estimated using several methods, including mark-recapture, skeletochronology, and marked, known-aged individuals. Skeletochronology analyzes growth marks in bones to obtain growth rates and age at sexual maturity estimates. Estimates vary widely among studies and populations, and methods continue to be developed and refined (Avens and Snover 2013). Early mark-recapture studies in

Florida estimated the age at sexual maturity 18-30 years (Frazer and Ehrhart 1985, Goshe et al. 2010, Mendonça 1981). More recent estimates of age at sexual maturity are as high as 35–50 years (Avens and Snover 2013, Goshe et al. 2010), with lower ranges reported from known age (15–19 years) turtles from the Cayman Islands (Bell et al. 2005) and Caribbean Mexico (12–20 years) (Zurita et al. 2012). A study of green turtles that use waters of the southeastern United States as developmental habitat found the age at sexual maturity likely ranges from 30 to 44 years (Goshe et al. 2010). Green turtles in the Northwestern Atlantic mature at 2.8-33+ ft (85–100+ cm) straight carapace lengths (SCL) (Avens and Snover 2013).

Adult turtles exhibit site fidelity and migrate hundreds to thousands of kilometers from nesting beaches to foraging areas. Green sea turtles spend the majority of their lives in coastal foraging grounds, which include open coastlines and protected bays and lagoons. Adult green turtles feed primarily on seagrasses and algae, although they also eat other invertebrate prey (Seminoff et al. 2015).

Population dynamics
The North Atlantic DPS has a globally unique haplotype, which was a factor in defining the discreteness of the DPS. Evidence from mitochondrial DNA studies indicates that there are at least four independent nesting subpopulations in Florida, Cuba, Mexico and Costa Rica (Seminoff et al. 2015). More recent genetic analysis indicates that designating a new western Gulf of Mexico management unit might be appropriate (Shamblin et al. 2016).

Compared to other DPSs, the North Atlantic DPS exhibits the highest nester abundance, with approximately 167,424 females at seventy-three nesting sites (using data through 2012), and available data indicated an increasing trend in nesting (Seminoff et al. 2015). Counts of nests and nesting females are commonly used as an index of abundance and population trends, even though there are doubts about the ability to estimate the overall population size.

There are no reliable estimates of population growth rate for the DPS as a whole, but estimates have been developed at a localized level. The status review for green sea turtles assessed population trends for seven nesting sites with more 10 years of data collection in the North Atlantic DPS. The results were variable with some sites showing no trend and others increasing. However, all major nesting populations (using data through 2011-2012) demonstrated increases in abundance (Seminoff et al. 2015).

More recent data is available for the southeastern United States. The FWRI monitors sea turtle nesting through the Statewide Nesting Beach Survey (SNBS) and Index Nesting Beach Survey (INBS). Since 1979, the SNBS had surveyed approximately 215 beaches to collect information on the distribution, seasonality, and abundance of sea turtle nesting in Florida. Since 1989, the INBS has been conducted on a subset of SNBS beaches to monitor trends through consistent effort and specialized training of surveyors. The INBS data uses a standardized data-collection protocol to allow for comparisons between years and is presented for green, loggerhead, and leatherback sea turtles. The index counts represent 27 core index beaches. The index nest counts represent approximately 67 percent of known green turtle nesting in Florida (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/).

Nest counts at Florida's core index beaches have ranged from less than 300 to almost 41,000 in 2019. The nest numbers show a mostly biennial pattern of fluctuation (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/; Figure 37).

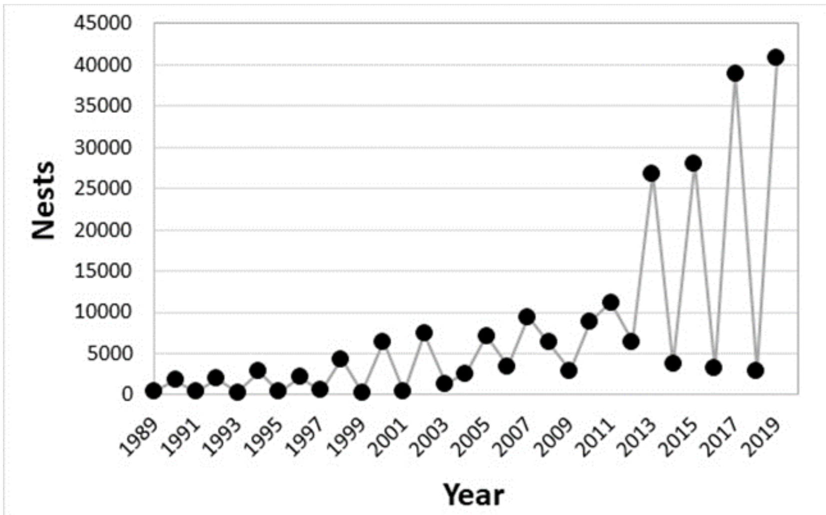

Figure 37: Number of green sea turtle nests counted on core index beaches in Florida from 1989-2019 (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/)

Status

Historically, green sea turtles in the North Atlantic DPS were hunted for food, which was the principle cause of the population's decline. Apparent increases in nester abundance for the North Atlantic DPS in recent years are encouraging but must be viewed cautiously, as the datasets represent a fraction of a green sea turtle generation which is between 30 and 40 years (Seminoff et al. 2015). While the threats of pollution, habitat loss through coastal development, beachfront lighting, and fisheries bycatch continue, the North Atlantic DPS appears to be somewhat resilient to future perturbations.

Critical Habitat

Critical habitat in effect for the North Atlantic DPS of green sea turtles surrounds Culebra Island, Puerto Rico (66 FR 20058, April 6, 2016), which is outside the action area.

Recovery Goals

The recovery plan for green sea turtles has not been recently updated. In the plan, the recovery goal for the U.S. population of green sea turtles is delist the species once the recovery criteria are met (NMFS and USFWS 1991b). The recovery plan includes criteria for delisting related to nesting activity, nesting habitat protection, and reduction in mortality.

Delisting can be considered if, over a period of 25 years,:

1. Florida nesting has increased to an average of 5,000 nests per year for at least six years.
2. At least 25 percent (105 km) of available nesting beaches is in public ownership and encompasses greater than 50 percent of nesting activity.
3. Stage class mortality reduction is reflected in higher abundance counts on foraging grounds.

4. All priority one tasks have been successfully implemented (NMFS and USFWS 1991b).

Major actions needed to help meet the recovery goals include:

1. Providing long-term protection to important nesting beaches.
2. Ensuring at least a 60 percent hatch rate success on major nesting beaches.
3. Implementing effective lighting ordinances/plans on nesting beaches.
4. Determining distribution and seasonal movements of all life stages in the marine environment.
5. Minimizing commercial fishing mortality.
6. Reducing threat to the population and foraging habitat from marine pollution.

4.2.2.2. Kemp's Ridley Sea Turtle

The range of Kemp's ridley sea turtles extends from the Gulf of Mexico to the Atlantic coast (Figure 38). They have occasionally been found in the Mediterranean Sea, which may be due to migration expansion or increased hatchling production (Tomás and Raga 2008). They are the smallest of all sea turtle species, with a nearly circular top shell and a pale yellowish bottom shell. The species was first listed under the Endangered Species Conservation Act (35 FR 18319, December 2, 1970) in 1970. The species has been listed as endangered under the ESA since 1973.

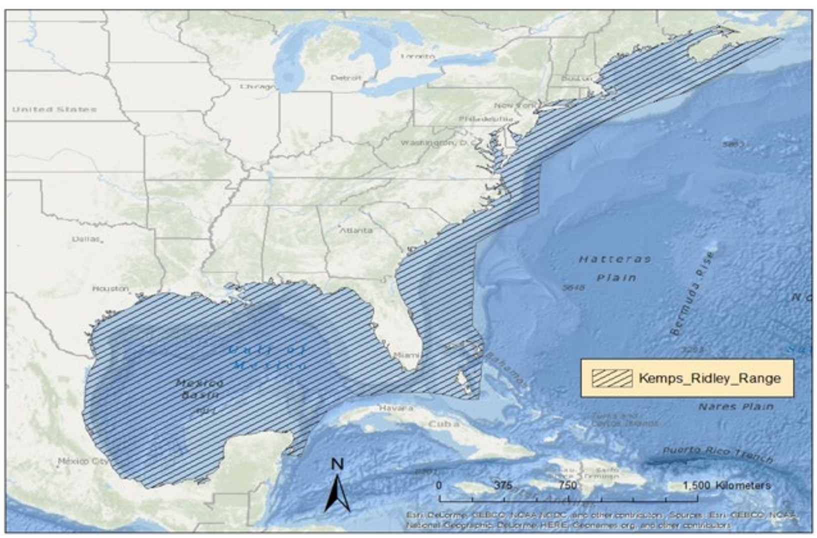

Figure 38: Range of the Kemp's ridley sea turtle

We used information available in the revised recovery plan (NMFS et al. 2011), the five-year review (NMFS and USFWS 2015), and published literature to summarize the life history, population dynamics and status of the species, as follows.

Life History

Kemp's ridley nesting is essentially limited to the western Gulf of Mexico. Approximately 97 percent of the global population's nesting activity occurs on a 90-mile (146-km) stretch of beach that includes Rancho Nuevo in Mexico (Wibbels and Bevan 2019). In the United States, nesting occurs primarily in Texas and occasionally in Florida, Alabama, Georgia, South Carolina, and North Carolina (NMFS and USFWS 2015). Nesting occurs from April to July in large arribadas (synchronized large-scale nesting). The average remigration interval is two years, although intervals of 1 and 3 years are not uncommon (NMFS et al. 2011, TEWG 1998, 2000). Females lay an average of 2.5 clutches per season (NMFS et al. 2011). The annual average clutch size is 95 to 112 eggs per nest (NMFS and USFWS 2015). The nesting location may be particularly important because hatchlings can more easily migrate to foraging grounds in deeper oceanic waters, where they remain for approximately two years before returning to nearshore coastal habitats (Epperly et al. 2013, NMFS and USFWS 2015, Snover et al. 2007). Modeling indicates that oceanic-stage Kemp's ridley turtles are likely distributed throughout the Gulf of Mexico into the northwestern Atlantic (Putman et al. 2013). Kemp's ridley nearing the age when recruitment to nearshore waters occurs are more likely to be distributed in the northern Gulf of Mexico, eastern Gulf of Mexico, and the western Atlantic (Putman et al. 2013).

Several studies, including those of captive turtles, recaptured turtles of known age, mark-recapture data, and skeletochronology, have estimated the average age at sexual maturity for Kemp's ridleys between 5 to 12 years (captive only) (Bjorndal et al. 2014), 10 to 16 years (Chaloupka and Zug 1997, Schmid and Witzell 1997, Schmid and Woodhead 2000, Zug et al. 1997), 9.9 to 16.7 years (Snover et al. 2007), 10 and 18 years (Shaver and Wibbels 2007), 6.8 to 21.8 years (mean 12.9 years) (Avens et al. 2017).

During spring and summer, juvenile Kemp's ridleys generally occur in the shallow coastal waters of the northern Gulf of Mexico from south Texas to north Florida and along the U.S. Atlantic coast from southern Florida to the Mid-Atlantic and New England. In addition, the NEFSC caught a juvenile Kemp's ridley during a recent research project in deep water south of Georges Bank (NEFSC, unpublished data). In the fall, most Kemp's ridleys migrate to deeper or more southern, warmer waters and remain there through the winter. As adults, many turtles remain in the Gulf of Mexico, with only occasional occurrence in the Atlantic Ocean (NMFS et al. 2011). Adult habitat largely consists of sandy and muddy areas in shallow, nearshore waters less than 120 feet (37 meters) deep (Seney and Landry 2008, Shaver et al. 2005, Shaver and Rubio 2008), although they can also be found in deeper offshore waters. As larger juveniles and adults, Kemp's ridleys forage on swimming crabs, fish, mollusks, and tunicates (NMFS et al. 2011).

Population Dynamics

Of the sea turtles species in the world, the Kemp's ridley has declined to the lowest population level. Nesting aggregations at a single location (Rancho Nuevo, Mexico) were estimated at 40,000 females in 1947. By the mid-1980s, the population had declined to an estimated 300 nesting females. From 1980 to 2003, the number of nests at three primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) increased at 15 percent annually (Heppell et al. 2005). However, due to recent declines in nest counts, decreased survival of immature and adult sea turtles, and updated population modeling, this rate is not expected to continue and the overall trend is unclear (Caillouet et al. 2018, NMFS and USFWS 2015). In 2019, there were 11,090 nests, a 37.61 percent decrease from 2018 and a 54.89 percent decrease from 2017, which had the highest number (24,587) of nests (Figure 39; unpublished data). The reason for this recent decline is uncertain.

Using the standard IUCN protocol for sea turtle assessments, the number of mature individuals was recently estimated at 22,341 (Wibbels and Bevan 2019). The calculation took into account the average annual nests from 2016-2018 (21,156), a clutch frequency of 2.5 per year, a remigration interval of 2 years, and a sex ratio of 3.17 females:1 male. Based on the data in their analysis, the assessment concluded the current population trend is unknown (Wibbels and Bevan 2019).

Genetic variability in Kemp's ridley turtles is considered to be high, as measured by nuclear DNA analyses (i.e., microsatellites) (NMFS et al. 2011). If this holds true, rapid increases in population over one or two generations would likely prevent any negative consequences in the genetic variability of the species (NMFS et al. 2011). Additional analysis of the mtDNA taken from samples of Kemp's ridley turtles at Padre Island, Texas, showed six distinct haplotypes, with one found at both Padre Island and Rancho Nuevo (Dutton et al. 2006).

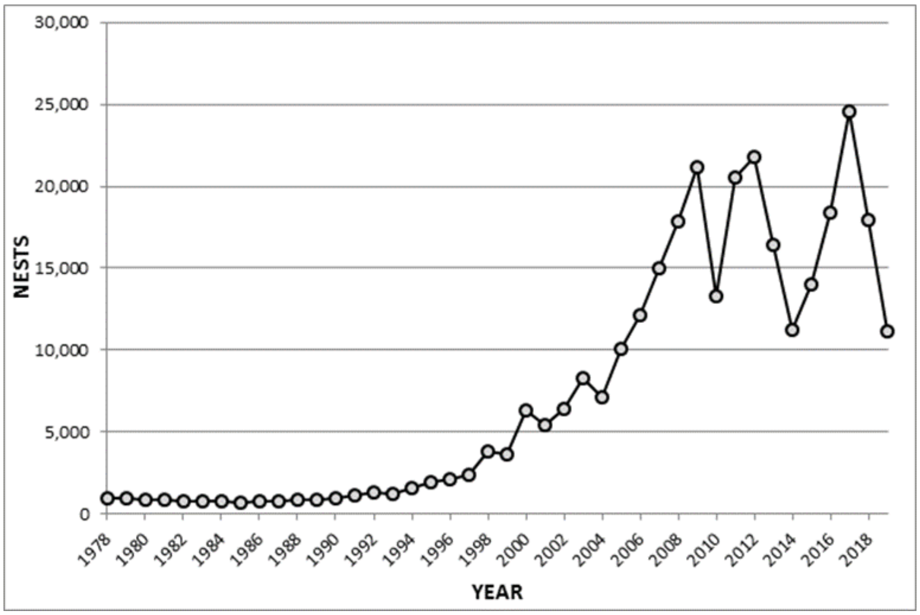

Figure 39: Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2019)

Status

The Kemp's ridley was listed as endangered in response to a severe population decline, primarily the result of egg collection. In 1973, legal ordinances in Mexico prohibited the harvest of sea turtles from May to August, and in 1990, the harvest of all sea turtles was prohibited by presidential decree. In 2002, Rancho Nuevo was declared a Sanctuary. Nesting beaches in Texas have been re-established. Fishery interactions are the main threat to the species. Other threats include habitat destruction, oil spills, dredging, disease, cold stunning, and climate change. The current population trend is uncertain. While the population has increased, recent nesting numbers have been variable. In addition, the species' limited range and low global abundance make it vulnerable to new sources of mortality as well as demographic and environmental randomness, all of which are often difficult to predict with any certainty. Therefore, its resilience to future perturbation is low.

Critical Habitat

Critical habitat has not been designated for Kemp's ridley sea turtles.

Recovery Goals

As with other recovery plans, the goal of the Kemp's ridley recovery plan is to conserve and protect the species so that the listing is no longer necessary. The recovery criteria relate to the number of nesting females, hatchling recruitment, habitat protection, social and/or economic initiatives compatible with conservation, reduction of predation, turtle excluder device (TED) or other protective measures in trawl gear, and improved information available to ensure recovery. The recovery plan includes the complete downlisting/delisting criteria (NMFS et al. 2011). These criteria, which are related to demographic and listing factor criteria, are summarized here.

Downlisting criteria include:

1. A population of at least 10,000 nesting females in a season distributed at primary nesting beaches.
2. Recruitment of at least 300,000 hatchlings to the marine environment per season at the three primary nesting beaches in Mexico.
3. Listing factor criteria related to long-term protection of habitat at two of the primary nesting beaches; initiation of social and/or economic community initiatives; reduction of nest predation; maintenance and enforcement of TED regulations; and identification and review of data on foraging areas, interesting habitats, mating areas, and adult migration routes to provide information to ensure recovery.

Delisting criteria include:

1. Average population of at least 40,000 nesting females per season over a 6-year period distributed among nesting beaches.
2. Average annual recruitment of hatchlings over a 6-year period sufficient to maintain a population of at least 40,000 nesting females per nesting season.
3. Listing factor criteria related to maintaining long-term habitat protection at nesting beaches of Tamaulipas and Texas; maintaining and expanding community socioeconomic programs; reducing nest predation through protective measures; implementing specific, comprehensive legislation/regulations to ensure post-delisting protection, as appropriate; establishing a network on in-water sites to monitor population and implementing surveys; initiating monitoring programs in commercial and recreational fisheries have been initiated and implementing measures to minimize mortality in fisheries; ensuring all other significant anthropogenic mortalities have been sufficiently addressed to ensure recruitment to maintain population level criterion; and continuing STSSN research and data collection to monitor the effectiveness of protection and restoration activities.

Major actions needed to meet the recovery goals include:

1. Protect and manage terrestrial and marine habitats and Kemp's ridley populations.
2. Maintain the STSSN.
3. Manage captive stocks.
4. Develop local, state, national government and community partnerships.
5. Educate the public.
6. Maintain and expand legal protections, promote awareness of these, and increase enforcement.
7. Implement international agreements.

4.2.2.3. *Loggerhead Sea Turtle (Northwest Atlantic Ocean DPS)*

Loggerhead sea turtles are circumglobal and are found in the temperate and tropical regions of the Indian, Pacific and Atlantic Oceans. The loggerhead sea turtle is distinguished from other turtles by its reddish-brown carapace, large head and powerful jaws. The species was first listed as threatened under the Endangered Species Act in 1978 (43 FR 32800, July 28, 1978). On September 22, 2011, the NMFS and USFWS designated nine distinct population segments of loggerhead sea turtles, with the Northwest Atlantic Ocean DPS listed as threatened (76 FR 58868). The Northwest Atlantic Ocean DPS of loggerheads is found along eastern North America, Central America, and northern South America (Figure 40).

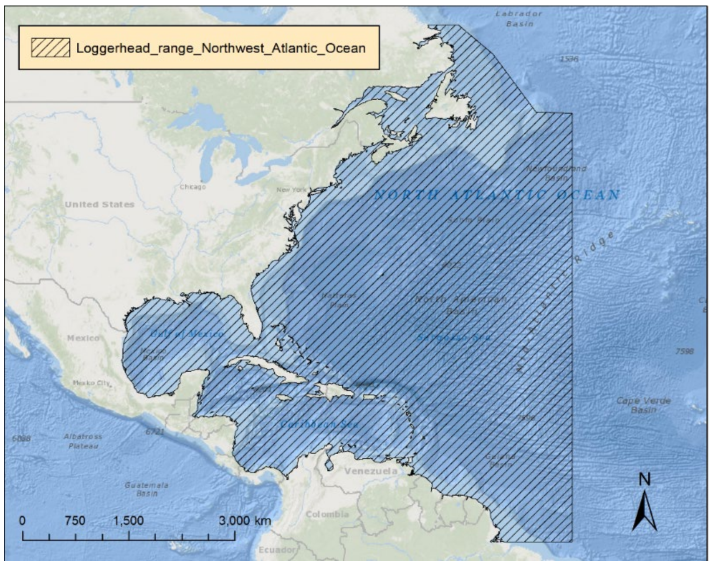

Figure 40: Range of the Northwest Atlantic Ocean DPS of loggerhead sea turtles

We used information available in the 2009 Status Review (Conant et al. 2009), the final listing rule (76 FR 58868, September 22, 2011), the relevant literature, and recent nesting data from the FWRI to summarize the life history, population dynamics and status of the species, as follows.

Life History

Nesting occurs on beaches where warm, humid sand temperatures incubate the eggs. Northwest Atlantic females lay an average of five clutches per year. The annual average clutch size is 115 eggs per nest. Females do not nest every year. The average remigration interval is three years. There is a 54 percent emergence success rate (Conant et al. 2009). As with other sea turtles, temperature determines the sex of the turtle during the middle of the incubation period. Turtles spend the post-hatchling stage in pelagic waters. The juvenile stage is spent first in the oceanic zone and later in coastal waters. Some juveniles may periodically move between the oceanic zone and coastal waters (Bolten 2003, Conant et al. 2009, Mansfield 2006, Morreale and Standora 2005, Witzell 2002). Coastal waters provide important foraging, inter-nesting, and migratory habitats for adult loggerheads. In both the oceanic zone and coastal waters, loggerheads are primarily carnivorous, although they do consume some plant matter as well (Conant et al. 2009). Loggerheads have been documented to feed on crustaceans, mollusks, jellyfish and salps, and algae (Bjorndal 1997, Donaton et al. 2019, Seney and Musick 2007).

Avens et al. (2015) used three approaches to estimate age at maturation. Mean age predictions associated with minimum and mean maturation straight carapace lengths were 22.5-25 and 36-38 years for females and 26-28 and 37-42 years for males. Male and female sea turtles have similar post-maturation longevity, ranging from 4 to 46 (mean 19) years (Avens et al. 2015).

Loggerhead hatchlings from the western Atlantic disperse widely, most likely using the Gulf Stream to drift throughout the Atlantic Ocean. MtDNA evidence demonstrates that juvenile loggerheads from southern Florida nesting beaches comprise the vast majority (71 percent-88 percent) of individuals found in foraging grounds throughout the western and eastern Atlantic: Nicaragua, Panama, Azores and Madeira, Canary Islands and Andalusia, Gulf of Mexico, and Brazil (Masuda 2010). LaCasella et al. (2013) found that loggerheads, primarily juveniles, caught within the Northeast Distant (NED) waters of the North Atlantic mostly originated from nesting populations in the southeast United States and, in particular, Florida. They found that nearly all loggerheads caught in the NED came from the Northwest Atlantic DPS (mean = 99.2 percent), primarily from the large eastern Florida rookeries. There was little evidence of contributions from the South Atlantic, Northeast Atlantic, or Mediterranean DPSs (LaCasella et al. 2013).

A more recent analysis assessed sea turtles captured in fisheries in the Northwest Atlantic and included samples from 850 (including 24 turtles caught during fisheries research) turtles caught from 2000-2013 in coastal and oceanic habitats (Stewart et al. 2019). The turtles were primarily captured in pelagic longline and bottom otter trawls. Other gears included bottom longline, hook and line, gillnet, dredge, and dip net. Turtles were identified from 19 distinct management units; the western Atlantic nesting populations were the main contributors with little representation from the Northeast Atlantic, Mediterranean, or South Atlantic DPSs (Stewart et al. 2019). There was a significant split in the distribution of small (≤ 2 ft (63 cm) SCL) and large (> 2 ft (63 cm) SCL) loggerheads north and south of Cape Hatteras, North Carolina. North of Cape Hatteras, large turtles came mainly from southeast Florida (44 percent±15 percent) and the northern United States management units (33 percent ± 16 percent); small turtles came from central east Florida (64 percent ± 14 percent). South of Cape Hatteras, large turtles came mainly from central east Florida (52 percent ± 20 percent) and southeast Florida (41 percent ± 20 percent); small turtles came from southeast Florida (56 percent ± 25 percent). The authors concluded that bycatch in the western North Atlantic would affect the Northwest Atlantic DPS almost exclusively (Stewart et al. 2019).

Population Dynamics

A number of stock assessments and similar reviews (Conant et al. 2009, Heppell et al. 2005, NMFS SEFSC 2001, 2009, Richards et al. 2011, TEWG 1998, 2000, 2009) have examined the stock status of loggerheads in the Atlantic Ocean, but none have been able to develop a reliable estimate of absolute population size. As with other species, counts of nests and nesting females are commonly used as an index of abundance and population trends, even though there are doubts about the ability to estimate the overall population size.

Based on genetic analysis of nesting subpopulations, the Northwest Atlantic Ocean DPS is divided into five recovery units: Northern, Peninsular Florida, Dry Tortugas, Northern Gulf of Mexico, and Greater Caribbean (Conant et al. 2009). A more recent analysis using expanded mtDNA sequences revealed that rookeries from the Gulf and Atlantic coasts of Florida are genetically distinct (Shamblin et al. 2014). The recent genetic analyses suggest that the

Northwest Atlantic Ocean DPS should be considered as ten management units: (1) South Carolina and Georgia, (2) central eastern Florida, (3) southeastern Florida, (4) Cay Sal, Bahamas, (5) Dry Tortugas, Florida, (6) southwestern Cuba, (7) Quintana Roo, Mexico, (8) southwestern Florida, (9) central western Florida, and (10) northwestern Florida (Shamblin et al. 2012).

The Northwest Atlantic Ocean's loggerhead nesting aggregation is considered the largest in the world (Casale and Tucker 2017). Using data from 2004-2008, the adult female population size of the DPS was estimated at 20,000 to 40,000 females (NMFS SEFSC 2009). More recently, Ceriani and Meylan (2017) reported a 5-year average (2009-2013) of more than 83,717 nests per year in the southeast United States and Mexico (excluding Cancun, Quintana Roo, Mexico; approximately 3.7% of nests in Quintana Roo). These estimates included sites without long-term (≥10 years) datasets. When they used data from 86 index sites (representing 63.4 percent of the estimated nests for the whole DPS with long-term datasets, they reported 53,043 nests per year. Trends at the different index nesting beaches ranged from negative to positive. In a trend analysis of the 86 index sites, the overall trend for the Northwest Atlantic DPS was positive (+2 percent) (Ceriani and Meylan 2017). Uncertainties in this analysis include, among others, using nesting females as proxies for overall population abundance and trends, demographic parameters, monitoring methodologies, and evaluation methods involving simple comparisons of early and later 5-year average annual nest counts. However, the authors concluded that the subpopulation is well monitored and the data evaluated represents 63.4 percent of the total estimated annual nests of the subpopulation and, therefore, are representative of the overall trend (Ceriani and Meylan 2017).

About 80 percent of loggerhead nesting in the southeast United States occurs in six Florida counties (NMFS and USFWS 2008). The Peninsula Florida Recovery Unit and the Northern Recovery Unit represent approximately 87 percent and 10 percent, respectively of all nesting effort in the Northwest Atlantic DPS (Ceriani and Meylan 2017, NMFS and USFWS 2008). As described above, FWRI's INBS collects standardized nesting data. The index nest counts for loggerheads represent approximately 53 percent of known nesting in Florida. There have been three distinct intervals observed: increasing (1989-1998), decreasing (1998-2007), and increasing (2007-2019) (Figure 41). At core index beaches in Florida, nesting totaled a minimum of 28,876 nests in 2007 and a maximum of 65,807 nests in 2016 (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). In 2019, more than 53,000 nests were documented (Figure 41). The nest counts in Figure 42 represent peninsular Florida and do not include an additional set of beaches in the Florida Panhandle and southwest coast that were added to the program in 1997 and more recent years. Nest counts at these Florida Panhandle index beaches have an upward trend since 2010 (Figure 42).

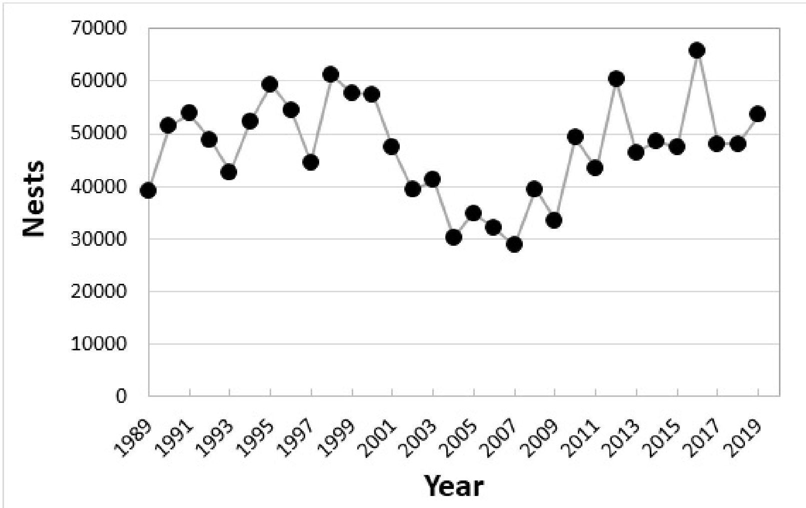

Figure 41: Annual nest counts of loggerhead sea turtles on Florida core index beaches in peninsular Florida, 1989-2019 (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/)

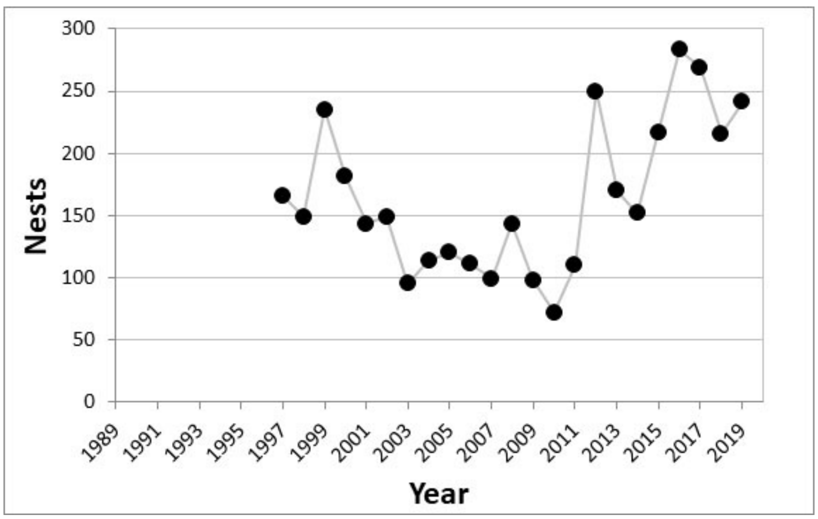

Figure 42: Annual nest counts of loggerhead sea turtles on index beaches in the Florida Panhandle, 1997-2019 (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/)

The annual nest counts on Florida's index beaches fluctuate widely, and we do not fully understand what drives these fluctuations. In assessing the population, Ceriani and Meylan (2017) and Bolten et al. (2019) looked at trends by recovery unit. Trends by recovery unit were variable.

The Peninsular Florida Recovery Unit extends from the Georgia-Florida border south and then north (excluding the islands west of Key West, Florida) through Pinellas County on the west coast of Florida. Annual nest counts from 1989 to 2018 ranged from a low of 28,876 in 2007 to a high of 65,807 in 2017 (Bolten et al. 2019). More recently (2008-2018), counts have ranged from 33,532 in 2009 to 65,807 in 2016 (Bolten et al. 2019). Nest counts taken at index beaches in Peninsular Florida showed a significant decline in loggerhead nesting from 1989 to 2007, most likely attributed to mortality of oceanic-stage loggerheads caused by fisheries bycatch (Witherington et al. 2009). Trend analyses have been completed for various periods. From 2009 through 2013, a 2 percent decrease for this recovery unit was reported (Ceriani and Meylan

2017). Using a longer time series from 1989-2018, there was no significant change in the number of annual nests (Bolten et al. 2019). It is important to recognize that an increase in the number of nests has been observed since 2007. The recovery team cautions that using short term trends in nesting abundance can be misleading and trends should be considered in the context of one generation (50 years for loggerheads) (Bolten et al. 2019).

The Northern Recovery Unit, ranging from the Florida-Georgia border through southern Virginia, is the second largest nesting aggregation in the DPS. Annual nest totals for this recovery unit from 1983 to 2019 have ranged from a low of 520 in 2004 to a high of 5,555 in 2019 (Bolten et al. 2019). From 2008 to 2019, counts have ranged from 1,289 nests in 2014 to 5,555 nests in 2019 (Bolten et al. 2019). Nest counts at loggerhead nesting beaches in North Carolina, South Carolina, and Georgia declined at 1.9 percent annually from 1983 to 2005 (NMFS and USFWS 2008). Recently, the trend has been increasing. Ceriani and Meylan (2017) reported a 35 percent increase for this recovery unit from 2009 through 2013. A longer-term trend analysis based on data from 1983 to 2019 indicates that the annual rate of increase is 1.3 percent (Bolten et al. 2019).

The Dry Tortugas Recovery Unit includes all islands west of Key West, Florida. A census on Key West from 1995 to 2004 (excluding 2002) estimated a mean of 246 nests per year, or about 60 nesting females (NMFS and USFWS 2008). No trend analysis is available because there was not an adequate time series to evaluate the Dry Tortugas recovery unit (Ceriani et al. 2019, Ceriani and Meylan 2017), which accounts for less than 1 percent of the Northwest Atlantic DPS (Ceriani and Meylan 2017).

The Northern Gulf of Mexico Recovery Unit is defined as loggerheads originating from beaches in Franklin County on the northwest Gulf coast of Florida through Texas. From 1995 to 2007, there were an average of 906 nests per year on approximately 300 km of beach in Alabama and Florida, which equates to about 221 females nesting per year (NMFS and USFWS 2008). Annual nest totals for this recovery unit from 1997-2018 have ranged from a low of 72 in 2010 to a high of 283 in 2016 (Bolten et al. 2019). Evaluation of long-term nesting trends for the Northern Gulf of Mexico Recovery Unit is difficult because of changed and expanded beach coverage. However, there are now over 20 years of Florida index nesting beach survey data. A number of trend analyses have been conducted. From 1995 to 2005, the recovery unit exhibited a significant declining trend (Conant et al. 2009, NMFS and USFWS 2008). Nest numbers have increased in recent years (Bolten et al. 2019) (see https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). In the 2009-2013 trend analysis by Ceriani and Meylan (2017), a 1 percent decrease for this recovery unit was reported, likely due to diminished nesting on beaches in Alabama, Mississippi, Louisiana, and Texas. A longer-term analysis from 1997-2018 found that there has been a non-significant increase of 1.7 percent (Bolten et al. 2019).

The Greater Caribbean Recovery Unit encompasses nesting subpopulations in Mexico to French Guiana, the Bahamas, and the Lesser and Greater Antilles. The majority of nesting for this recovery unit occurs on the Yucatán Peninsula, in Quintana Roo, Mexico, with 903 to 2,331 nests annually (Zurita et al. 2003). Other significant nesting sites are found throughout the Caribbean, including Cuba, with approximately 250 to 300 nests annually (Ehrhart et al. 2003), and over 100 nests annually in Cay Sal in the Bahamas (NMFS and USFWS 2008). In the trend analysis by Ceriani and Meylan (2017), a 53 percent increase for this Recovery Unit was reported from 2009 through 2013.

Status
Fisheries bycatch is the highest threat to the Northwest Atlantic DPS of loggerhead sea turtles (Conant et al. 2009). Other threats include boat strikes, marine debris, coastal development, habitat loss, contaminants, disease, and climate change. Nesting trends for each of the loggerhead sea turtle recovery units in the Northwest Atlantic Ocean DPS are variable. Overall, short-term trends have shown increases, however, over the long-term the DPS is considered stable.

Critical Habitat
We have determined that the proposed action is not likely to adversely affect loggerhead critical habitat (see section 4.1.11).

Recovery Goals
The recovery goal for the Northwest Atlantic loggerhead is to ensure that each recovery unit meets its recovery criteria alleviating threats to the species so that protection under the ESA are not needed. The recovery criteria relate to the number of nests and nesting females, trends in abundance on the foraging grounds, and trends in neritic strandings relative to in-water abundance. The 2008 Final Recovery Plan for the Northwest Atlantic Population of Loggerheads includes the complete delisting criteria (NMFS and USFWS 2008).

Delisting criteria include:

1. Each recovery unit has recovered to a viable level and has increased for at least one generation. By recovery unit, over a 50-year period, the annual rate of increase is greater than or equal to 2 percent resulting in at least 14,000 nests annually for the Northern Recovery Unit; greater than or equal to 1 percent resulting in 106,100 nests annually for the Peninsular Florida Recovery Unit; greater than or equal to 3 percent resulting in at least 1,100 nests for the Dry Tortugas Recovery unit; and greater than or equal to 3 percent resulting in at least 4,000 nests annually for the Northern Gulf of Mexico Recovery Unit. For the Greater Caribbean Recovery Unit, the demographic criteria specifies that the total annual number of nests at a minimum of three nesting assemblages, averaging greater than 100 nests annually, has increased over 50 years.
2. The increases in the number of nests for each recovery unit must be a result of corresponding increases in the number of nesting females.
3. A network of in-water sites across the foraging range is established and measure abundance. A composite estimate of relative abundance from these sites is increasing for at least one generation.
4. Stranding trends are not increasing at a rate greater than the in-water relative abundance trends for similar age classes for at least one generation.
5. Listing factor recover criteria include criteria related to maintenance and protection of nesting habitat; development and implementation of a strategy to protect marine habitats important to loggerheads; implementation of nest protection strategies; elimination of legal harvest; reduction of nest predation; implementing legislation to ensure long-term protection of loggerheads and their habitats; implementation of strategies to reduce fisheries bycatch, marine debris ingestion and entanglement, and vessel strikes.

The recovery objectives to meet these goals include:

1. Ensure that the number of nests in each recovery unit is increasing and that this increase corresponds to an increase in the number of nesting females.

2. Ensure the in-water abundance of juveniles in both neritic and oceanic habitats is increasing and is increasing at a greater rate than strandings of similar age classes.
3. Manage sufficient nesting beach habitat to ensure successful nesting.
4. Manage sufficient feeding, migratory and internesting marine habitats to ensure successful growth and reproduction.
5. Eliminate legal harvest.
6. Implement scientifically based nest management plans.
7. Minimize nest predation.
8. Recognize and respond to mass/unusual mortality or disease events appropriately.
9. Develop and implement local, state, federal and international legislation to ensure long-term protection of loggerheads and their terrestrial and marine habitats.
10. Minimize bycatch in domestic and international commercial and artisanal fisheries.
11. Minimize trophic changes from fishery harvest and habitat alteration.
12. Minimize marine debris ingestion and entanglement.
13. Minimize vessel strike mortality.

4.2.2.4. Leatherback Sea Turtle

The leatherback sea turtle is unique among sea turtles for its large size, wide distribution (due to thermoregulatory systems and behavior), and lack of a hard, bony carapace. It ranges from tropical to subpolar latitudes, worldwide (Figure 43).

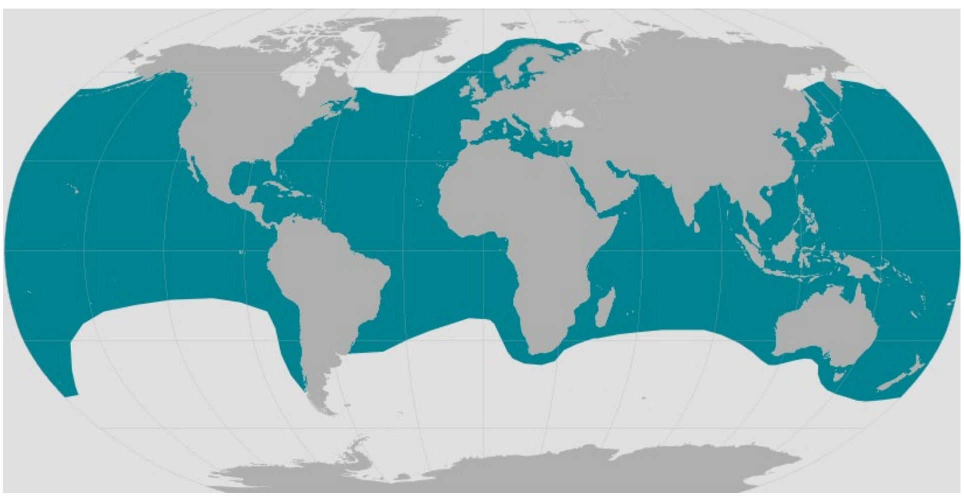

Figure 43: Range of the leatherback sea turtle (https://www.fisheries.noaa.gov/species/leatherback-turtle).

Leatherbacks are the largest living turtle, reaching lengths of six feet long, and weighing up to one ton. Leatherback sea turtles have a distinct black leathery skin covering their carapace with pinkish white skin on their plastron. The species was first listed under the Endangered Species Conservation Act (35 FR 8491, June 2, 1970) and has been listed as endangered under the ESA since 1973. In 2020, seven leatherback populations that met the discreteness and significance criteria of the DPS were identified (NMFS and USFWS 2020). The population found within the action is area is the Northwest Atlantic DPS (Figure 44). NMFS and USFWS concluded that the seven populations, which met the criteria for DPSs, all met the definition of an endangered species. NMFS and USFWS determined that the listing of DPSs was not warranted; leatherbacks continue to be listed at the global level (85 FR 48332, August 10, 2020). Therefore, information is presented on the range-wide status. We used information available in the five-year review

(NMFS and USFWS 2013), the critical habitat designation (44 FR 17710, March 23, 1979), the status review (NMFS and USFWS 2020), relevant literature, and recent nesting data from the Florida FWRI to summarize the life history, population dynamics and status of the species, as follows.

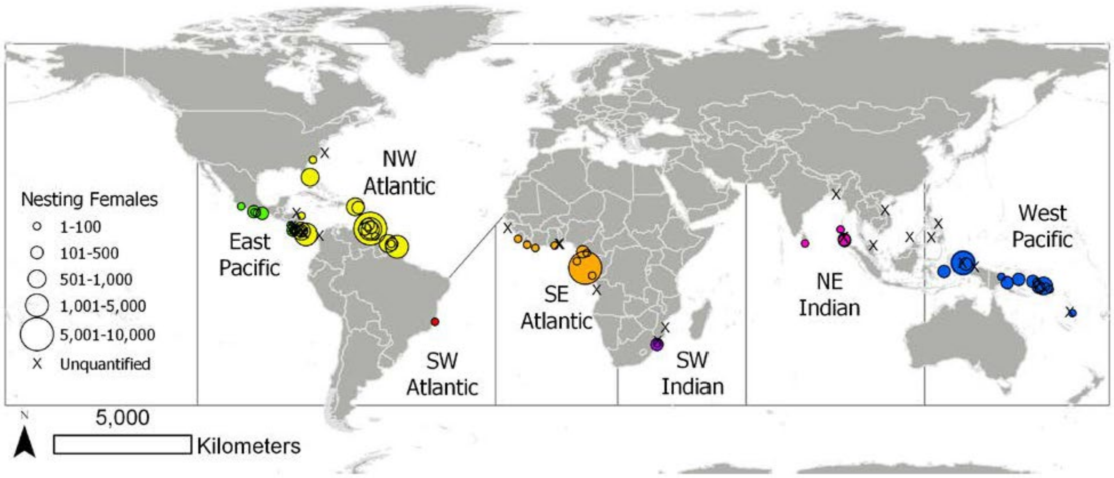

Figure 44: Leatherback sea turtle DPSs and nesting beaches (NMFS and USFWS 2020)

Life History

Leatherbacks are a long-lived species. Preferred nesting grounds are in the tropics; though, nests span latitudes from 34 °S in western Cape, South Africa to 38 °N in Maryland (Eckert et al. 2012, Eckert et al. 2015). Females lay an average of five to seven clutches (range: 1-14 clutches) per season, with 20 to over 100 eggs per clutch (Eckert et al. 2012, Reina et al. 2002, Wallace et al. 2007). The average clutch frequency for the Northwest Atlantic DPS is 5.5 clutches per season (NMFS and USFWS 2020). In the western Atlantic, leatherbacks lay about 82 eggs per clutch (Sotherland et al. 2015). Remigration intervals are 2-4 years for most populations (range 1-11 years) (Eckert et al. 2015, NMFS and USFWS 2020); the remigration interval for the Northwest Atlantic DPS is approximately 3 years (NMFS and USFWS 2020). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergence success) is approximately 50 percent worldwide (Eckert et al. 2012).

Age at sexual maturity has been challenging to obtain given the species physiology and habitat use (Avens et al. 2019). Past estimates ranged from 5-29 years (Avens et al. 2009, Spotila et al. 1996). More recently, Avens et al. (2019) used refined skeletochronology to assess the age at sexual maturity for leatherback sea turtles in the Atlantic and the Pacific. In the Atlantic, the mean age at sexual maturity was 19 years (range 13-28) and the mean size at sexual maturity was 4.2 ft (129.2 cm) CCL (range 3.7-5 ft (112.8-153.8 cm)). In the Pacific, the mean age at sexual maturity was 17 years (range 12-28) and the mean size at sexual maturity was 4.2 ft (129.3 cm) CCL (range 3.6- 5 ft (110.7-152.3 cm)) (Avens et al. 2019).

Leatherbacks have a greater tolerance for colder waters compared to all other sea turtle species due to their thermoregulatory capabilities (Paladino et al. 1990, Shoop and Kenney 1992, Wallace and Jones 2008). Evidence from tag returns, satellite telemetry, and strandings in the western Atlantic suggests that adult leatherback sea turtles engage in routine migrations between temperate/boreal and tropical waters (Bond and James 2017, Dodge et al. 2015, Eckert et al. 2006, Fossette et al. 2014, James et al. 2005a, James et al. 2005b, James et al. 2005c, NMFS and

USFWS 1992). Tagging studies collectively show a clear separation of leatherback movements between the North and South Atlantic Oceans (NMFS and USFWS 2020).

Leatherback sea turtles migrate long, transoceanic distances between their tropical nesting beaches and the highly productive temperate waters where they forage, primarily on jellyfish and tunicates. These gelatinous prey are relatively nutrient-poor, such that leatherbacks must consume large quantities to support their body weight. Leatherbacks weigh about 33 percent more on their foraging grounds than at nesting, indicating that they probably catabolize fat reserves to fuel migration and subsequent reproduction (James et al. 2005c, Wallace et al. 2006). Studies on the foraging ecology of leatherbacks in the North Atlantic show that leatherbacks off Massachusetts primarily consumed lion's mane, sea nettles, and ctenophores (Dodge et al. 2011). Juvenile and small sub-adult leatherbacks may spend more time in oligotrophic (relatively low plant nutrient usually accompanied by high dissolved oxygen) open ocean waters where prey is more difficult to find (Dodge et al. 2011). Sea turtles must meet an energy threshold before returning to nesting beaches. Therefore, their remigration intervals are dependent upon foraging success and duration (Hays 2000, Price et al. 2004).

Population Dynamics
The distribution is global, with nesting beaches in the Pacific, Atlantic, and Indian Oceans. Leatherbacks occur throughout marine waters, from nearshore habitats to oceanic environments (NMFS and USFWS 2020, Shoop and Kenney 1992). Movements are largely dependent upon reproductive and feeding cycles and the oceanographic features that concentrate prey, such as frontal systems, eddy features, current boundaries, and coastal retention areas (Benson et al. 2011).

Analyses of mtDNA from leatherback sea turtles indicates a low level of genetic diversity (Dutton et al. 1999). Further analysis of samples taken from individuals from rookeries in the Atlantic and Indian Oceans suggest that each of the rookeries represent demographically independent populations (NMFS and USFWS 2013). Using genetic data,, combined with nesting, tagging, and tracking data, researchers identified seven global regional management units (RMU) or subpopulations: Northwest Atlantic, Southeast Atlantic, Southwest Atlantic, Northwest Indian, Southwest Indian, East Pacific, and West Pacific (Wallace et al. 2010). The status review concluded that the RMUs identified by Wallace et al. (2010) are discrete populations and, then, evaluated whether any other populations exhibit this level of genetic discontinuity (NMFS and USFWS 2020).

To evaluate the RMUs and fine-scale structure in the Atlantic, Dutton et al. (2013) conducted a comprehensive genetic re-analysis of rookery stock structure. Samples from eight nesting sites in the Atlantic and one in the southwest Indian Ocean identified seven management units in the Atlantic and revealed fine scale genetic differentiation among neighboring populations. The mtDNA analysis failed to find significant differentiation between Florida and Costa Rica or between Trinidad and French Guiana/Suriname (Dutton et al. 2013). While Dutton et al. (2013) identified fine-scale genetic partitioning in the Atlantic Ocean, the differences did not rise to the level of marked separation or discreteness (NMFS and USFWS 2020). Other genetic analyses corroborate the conclusions of Dutton et al. (2013). These studies analyzed nesting sites in French Guiana (Molfetti et al. 2013), nesting and foraging areas in Brazil (Vargas et al. 2019), and nesting beaches in the Caribbean (Carreras et al. 2013). These studies all support three discrete populations in the Atlantic (NMFS and USFWS 2020). While these studies detected fine-scale genetic differentiation in the NW, SW, and SE Atlantic populations, the status review team determined that none indicated that the genetic differences were sufficient to be considered marked separation (NMFS and USFWS 2020).

Population growth rates for leatherback sea turtles vary by ocean basin. An assessment of leatherback populations through 2010 found a global decline overall (Wallace et al. 2013). Using datasets with abundance data series that are 10 years or greater, they estimated that leatherback populations have declined from 90,599 nests per year to 54,262 nests per year over three generations ending in 2010 (Wallace et al. 2013).

Several more recent assessments have been conducted. The Northwest Atlantic Leatherback Working Group was formed to compile nesting abundance data, analyze regional trends, and provide conservation recommendations. The most recent, published IUCN Red List assessment for the NW Atlantic Ocean subpopulation estimated 20,000 mature individuals and approximately 23,000 nests per year (estimate to 2017) (Northwest Atlantic Leatherback Working Group 2019). Annual nest counts show high inter-annual variability within and across nesting sites (Northwest Atlantic Leatherback Working Group 2018). Using data from 24 nesting sites in 10 nations within the Northwest Atlantic DPS, the leatherback status review estimated that the total index of nesting female abundance for the Northwest Atlantic DPS is 20,659 females (NMFS and USFWS 2020). This estimate only includes nesting data from recently and consistently monitored nesting beaches. An index (rather than a census) was developed given that the estimate is based on the number of nests on main nesting beaches with recent and consistent data and assumes a 3-year remigration interval. This index provides a minimum estimate of nesting female abundance (NMFS and USFWS 2020). This index of nesting female abundance is similar to other estimates. The TEWG estimated approximately 18,700 (range 10,000 to 31,000) adult females using nesting data from 2004 and 2005 (TEWG 2007). As described above, the IUCN Red List Assessment estimated 20,000 mature individuals (male and female). The estimate in the status review is higher than the estimate for the IUCN Red List assessment, likely due to a different remigration interval, which has been increasing in recent years (NMFS and USFWS 2020).

Previous assessments of leatherbacks concluded that the Northwest Atlantic population was stable or increasing (TEWG 2007, Tiwari et al. 2013b). However, based on more recent analyses, leatherback nesting in the Northwest Atlantic is showing an overall negative trend, with the most notable decrease occurring during the most recent period of 2008-2017 (Northwest Atlantic Leatherback Working Group 2018). The analyses for the IUCN Red List assessment indicate that the overall regional, abundance-weighted trends are negative (Northwest Atlantic Leatherback Working Group 2018, 2019). The dataset for trend analyses included 23 sites across 14 countries/territories. Three periods were used for the trend analysis: long-term (1990-2017), intermediate (1998-2017), and recent (2008-2017) trends. Overall, regional, abundance-weighted trends were negative across the periods and became more negative as the time-series became shorter. At the stock level, the Working Group evaluated the NW Atlantic – Guianas-Trinidad, Florida, Northern Caribbean, and the Western Caribbean. The NW Atlantic – Guianas-Trinidad stock is the largest stock and declined significantly across all periods, which was attributed to an exponential decline in abundance at Awala-Yalimapo, French Guiana as well as declines in Guyana, Suriname, Cayenne, and Matura. Declines in Awala-Yalimapo were attributed, in part, due to a beach erosion and a loss of nesting habitat (Northwest Atlantic Leatherback Working Group 2018). The Florida stock increased significantly over the long-term, but declined from

2008-2017. The Northern Caribbean and Western Caribbean stocks also declined over all three periods. The Working Group report also includes trends at the site-level, which varied depending on the site and time period, but were generally negative especially in the recent time period. The Working Group identified anthropogenic sources (fishery bycatch, vessel strikes), habitat loss, and changes in life history parameters as possible drivers of nesting abundance declines (Northwest Atlantic Leatherback Working Group 2018). Fisheries bycatch is a well-documented threat to leatherback turtles. The Working Group discussed entanglement in vertical line fisheries off New England and Canada as potentially important mortality sinks. They also noted that vessels strikes result in mortality annually in feeding habitats off New England. Off nesting beaches in Trinidad and the Guianas, net fisheries take leatherbacks in high numbers (~3,000/yr) (Eckert 2013, Lum 2006, Northwest Atlantic Leatherback Working Group 2018).

Similarly, the leatherback status review concluded that the Northwest Atlantic DPS exhibits decreasing nest trends at nesting aggregations with the greatest indices of nesting female abundance. Significant declines have been observed at nesting beaches with the greatest historical or current nesting female abundance, most notably in Trinidad and Tobago, Suriname, and French Guiana. Though some nesting aggregations (see status review document for information on specific nesting aggregations) indicated increasing trends, most of the largest ones are declining. The declining trend is considered to be representative of the DPS (NMFS and USFWS 2020). The status review found that fisheries bycatch is the primary threat to the Northwest Atlantic DPS (NMFS and USFWS 2020).

Within the action area, leatherback sea turtles nest in the southeastern United States. From 1989-2019, leatherback nests at core index beaches in Florida have varied from a minimum of 30 nests in 1990 to a maximum of 657 in 2014 (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). Leatherback nesting declined from 2014 to 2017. Although slight increases were seen in 2018 and 2019, nest counts remain low compared to the numbers documented from 2008-2015 (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/) (Figure 45). The status review found that the median trend for Florida from 2008-2017 was a decrease of 2.1 percent annually (NMFS and USFWS 2020).

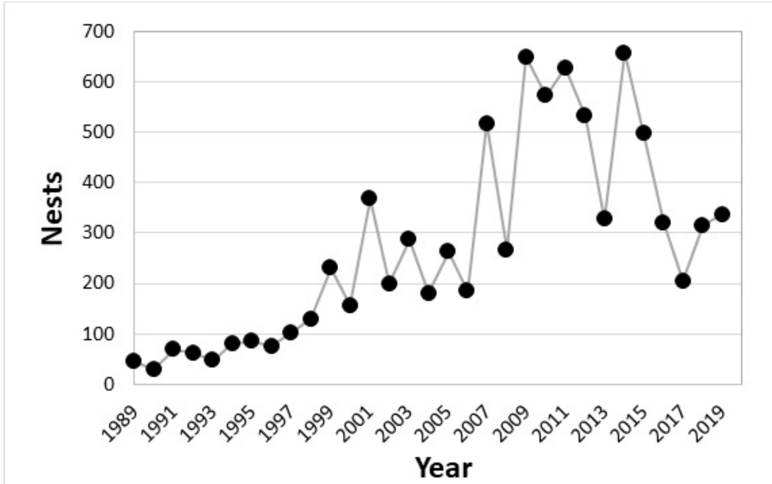

Figure 45: Number of leatherback sea turtle nests on core index beaches in Florida from 1989-2019 (https://myfwc.com/research/wildlife/sea-turtles/nesting/)

For the Southwest Atlantic DPS, the status review estimates the total index of nesting female abundance at approximately 27 females (NMFS and USFWS 2020). This is similar to the IUCN Red List assessment that estimated 35 mature individuals (male and female) using nesting data since 2010. Nesting has increased since 2010 overall, though the 2014-2017 estimates were lower than the previous three years. The trend is increasing, though variable (NMFS and USFWS 2020). The Southeast Atlantic DPS has an index of nesting female abundance of 9,198 females and demonstrates a declining nest trend at the largest nesting aggregation (NMFS and USFWS 2020). The Southeast DPS exhibits a declining nest trend (NMFS and USFWS 2020).

Populations in the Pacific have shown dramatic declines at many nesting sites (Mazaris et al. 2017, Santidrián Tomillo et al. 2017, Santidrián Tomillo et al. 2007, Sarti Martínez et al. 2007, Tapilatu et al. 2013). For an IUCN Red List evaluation, datasets for nesting at all index beaches for the West Pacific population were compiled (Tiwari et al. 2013a). This assessment estimated the number of total mature individuals (males and females) at Jamursba-Medi and Wermon beaches to be 1,438 turtles(Tiwari et al. 2013a). Counts of leatherbacks at nesting beaches in the western Pacific indicate that the subpopulation declined at a rate of almost 6 percent per year from 1984 to 2011 (Tapilatu et al. 2013). More recently, the leatherback status review estimated the total index of nesting female abundance of the West Pacific DPS at 1,277 females, and the DPS exhibits low hatchling success (NMFS and USFWS 2020). The total index of nesting female abundance for the East Pacific DPS is 755 nesting females. It has exhibited a decreasing trend since monitoring began with a 97.4 percent decline since the 1980s or 1990s, depending on nesting beach (Wallace et al. 2013). The low productivity parameters, drastic reductions in nesting female abundance, and current declines in nesting place the DPS at risk (NMFS and USFWS 2020).

Population abundance in the Indian Ocean is difficult to assess due to lack of data and inconsistent reporting. Available data from southern Mozambique show that approximately 10 females nest per year from 1994 to 2004, and about 296 nests per year were counted in South Africa (NMFS and USFWS 2013). A 5-year status review in 2013 found that, in the southwest Indian Ocean, populations in South Africa are stable (NMFS and USFWS 2013). More recently, the 2020 status review estimated that the total index of nesting female abundance for the Southwest Indian DPS is 149 females and that the DPS is exhibiting a slight decreasing nest trend (NMFS and USFWS 2020). While data on nesting in the Northeast Indian Ocean DPS is limited, the DPS is estimated at 109 females. This DPS has exhibited a drastic population decline with extirpation of the largest nesting aggregation in Malaysia (NMFS and USFWS 2020).

Status
The leatherback sea turtle is an endangered species whose once large nesting populations have experienced steep declines in recent decades. There has been a global decline overall. For all DPSs, including the Northwest Atlantic DPS, fisheries bycatch is the primary threat to the species (NMFS and USFWS 2020). Leatherback turtle nesting in the Northwest Atlantic showed an overall negative trend through 2017, with the most notable decrease occurring during the most recent time frame of 2008 to 2017 (Northwest Atlantic Leatherback Working Group 2018). Though some nesting aggregations indicated increasing trends, most of the largest ones are declining. Therefore, the leatherback status review in 2020 concluded that the Northwest Atlantic DPS exhibits an overall decreasing trend in annual nesting activity (NMFS and USFWS 2020). Threats to leatherback sea turtles include loss of nesting habitat, fisheries bycatch, vessel strikes, harvest of eggs, and marine debris, among others (Northwest Atlantic Leatherback Working Group 2018). Because of the threats, once large nesting areas in the Indian and Pacific Oceans are now functionally extinct (Tiwari et al. 2013a) and there have been range-wide reductions in population abundance. The species' resilience to additional perturbation both within the NW Atlantic and worldwide is low.

Critical Habitat
Critical habitat has been designated for leatherback sea turtles in the waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands (44 FR 17710, March 23, 1979) and along the U.S. West Coast (77 FR 4170, January 26, 2012), both of which are outside the action area.

Recovery Goals
There are separate plans for the U.S. Caribbean, Gulf of Mexico, and Atlantic (NMFS and USFWS 1992) and the U.S. Pacific (NMFS and USFWS 1998a) populations of leatherback sea turtles. Neither plan has been recently updated. As with other sea turtle species, the recovery plans for leatherbacks includes criteria for considering delisting. These criteria relate to increases in the populations, nesting trends, nesting beach and habitat protection, and implementation of priority actions. Criteria for delisting in the recovery plan for the U.S. Caribbean, Gulf of Mexico, and Atlantic are described here.

Delisting criteria

1. Adult female population increases for 25 years after publication of the recovery plan, as evidenced by a statistically significant trend in nest numbers at Culebra, Puerto Rico; St. Croix, U.S. Virgin Islands; and the east coast of Florida.
2. Nesting habitat encompassing at least 75 percent of nesting activity in the U.S. Virgin Islands, Puerto Rico, and Florida is in public ownership.
3. All priority one tasks have been successfully implemented (see the recovery plan for a list of priority one tasks).

Major recovery actions in the U.S. Caribbean, Gulf of Mexico and Atlantic include actions to:

1. Protect and manage terrestrial and marine habitats.
2. Protect and manage the population.
3. Inform and educate the public.
4. Develop and implement international agreements.

The Pacific leatherback is a NOAA Species in the Spotlight. The Species in the Spotlight program identifies those species most-at risk of extinction. A 5-year action plan has developed for these species to identify immediate, targeted efforts vital to stabilizing the population and preventing extinction. The following items were the top five recovery actions identified to support in the Leatherback Five Year Action Plan (NMFS 2016d):

1. Reduce fisheries interactions
2. Improve nesting beach protection and increase reproductive output
3. International cooperation
4. Monitoring and research
5. Public engagement

4.2.2.5. Other factors outside the action area affecting the status of sea turtles - 2010 BP Deepwater Horizon Oil Spill

The April 20, 2010, explosion of the Deepwater Horizon oil rig affected sea turtles in the Gulf of Mexico. While the spill occurred outside the action area, it does impact the same sea turtle populations occur in the action area. Therefore, we are considering it in the status of the species. This extensive oiling event contaminated important sea turtle foraging, migratory, and breeding habitats used by different life stages at the surface, in the water column, on the ocean bottom, and on beaches throughout the northern Gulf of Mexico. Sea turtles were exposed to oil when in contaminated water or habitats; by breathing oil droplets, oil vapors, and smoke; by ingesting oil-contaminated water and prey; and potentially by maternal transfer of oil compounds to embryos. Response activities and shoreline oiling also directly injured sea turtles, disrupting and deterring sea turtle nesting in the Gulf (DWH NRDA Trustees 2016).

During direct at-sea capture events, more than 900 turtles were sighted, 574 of which were captured and examined for oiling (Stacy 2012). Of the turtles captured during these operations, greater than 80 percent were visibly oiled (DWH NRDA Trustees 2016). Most of the rescued turtles were taken to rehabilitation facilities; more than 90 percent of the turtles admitted to rehabilitation centers eventually recovered and were released (Stacy 2012, Stacy and Innis 2015). Recovery efforts also included relocating nearly 275 sea turtle nests from the northern Gulf to the Florida Panhandle, with the goal of preventing hatchlings from entering the oiled waters of the northern Gulf. More than 28,000 eggs were moved to an incubation facility in Cape Canaveral, Florida, where they were incubated until emergence and release. Approximately 14,000 hatchlings were released off the Atlantic coast of Florida, 95 percent of which were loggerheads (https://www.fisheries.noaa.gov/national/marine-life-distress/deepwater-horizon-oil-spill-2010-sea-turtles-dolphins-and-whales).

Direct observations of the effects of oil on turtles obtained by at-sea captures, sightings, and strandings represent a fraction of the scope of the injury. As such, the Deep Water Horizon (DWH) National Resource Damage Assessment (NRDA) Trustees used expert opinion, surface oiling maps, and statistical approaches to apply the directly observed adverse effects of oil exposure to turtles in areas and at times that could not be surveyed. The Trustees estimated that between 4,900 and 7,600 large juvenile and adult sea turtles (Kemp's ridleys, loggerheads, and hard-shelled sea turtles not identified to species), and between 55,000 and 160,000 small juvenile sea turtles (Kemp's ridleys, green turtles, loggerheads, hawksbills, and hard-shelled sea turtles not identified to species) died due to the DWH oil spill. Nearly 35,000 hatchling sea turtles (loggerheads, Kemp's ridleys, and green turtles) were also injured by response activities (DWH

NRDA Trustees 2016). Despite uncertainties and some unquantified injuries to sea turtles (e.g., injury to leatherbacks, unrealized reproduction), the Trustees conclude that this assessment adequately quantifies the nature and magnitude of injuries to sea turtles caused by the DWH oil spill and related activities. Other impacts assessed include reproductive failure and adverse health effects. The NRDA report chapter 4 includes details of the assessment and results (DWH NRDA Trustees 2016).

In addition, Wallace et al. (2017) later determined through a modeling approach that the highest probabilities of heavy oil exposure were limited to areas nearest the wellhead and the probability of heavy oiling decreased with increasing distance from the wellhead. They also determined that the estimated distribution of heavily oiled neritic turtles was similar to the estimated distribution of heavily oiled oceanic turtles (Wallace et al. 2017). This modeling approach produced reasonable estimates of heavy oiling probability for both turtles and surface habitats that were not directly observed during the NRDA response and survey efforts. A toxicological estimation of mortality of oceanic sea turtles oiled during the spill concluded that, overall, approximately 30 percent of all oceanic turtles in the region affected by the spill that were not heavily oiled would have died from ingestion of oil (Mitchelmore et al. 2017).

Response methods used to minimize the extent and harm resulting from a spill can also affect sea turtles. These responses may include collection of oil, in situ burning, use of oil booms, and application of dispersants. Incidental entrapment and mortalities can result from oil removal via skimming or burning. The effects of dispersants on sea turtles is poorly understood, and there is a lack of empirical studies and controlled experiments (Stacy et al. 2019). Exposure over the short-term to a dispersant and a mixture of oil/dispersant affected hydration and weight gain in loggerhead hatchlings (Harms et al. 2014). While the effects of dispersants on sea turtles is largely unknown, they remain a concern in sea turtles based on observations in other species (Stacy et al. 2019).

Based on these quantifications of sea turtle injuries and mortalities caused by the DWH oil spill, hard-shelled sea turtles from all life stages and all geographic areas were lost from the northern Gulf of Mexico ecosystem. Injuries to leatherback sea turtles could not be quantified (DWH NRDA Trustees 2016). The DWH NRDA Trustees (2016) conclude that the recovery of sea turtles in the northern Gulf of Mexico from injuries and mortalities caused by the DWH oil spill will require decades of sustained efforts to reduce the most critical threats and enhance survival of turtles at multiple life stages. The ultimate population level effects of the spill and impacts of the associated response activities are likely to remain unknown for some period into the future.

4.2.3. ***ESA-listed Fish***

4.2.3.1. Atlantic Sturgeon

An estuarine-dependent anadromous species, Atlantic sturgeon occupy ocean and estuarine waters, including sounds, bays, and tidal-affected rivers from Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida (ASSRT 2007) (Figure 46). On February 6, 2012, NMFS listed five DPSs of Atlantic sturgeon under the ESA: Gulf of Maine (GOM), New York Bight (NYB), Chesapeake Bay (CB), Carolina, and South Atlantic (77 FR 5880 and 77 FR 5914). The Gulf of Maine DPS is listed as threatened, and the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs are listed as endangered.

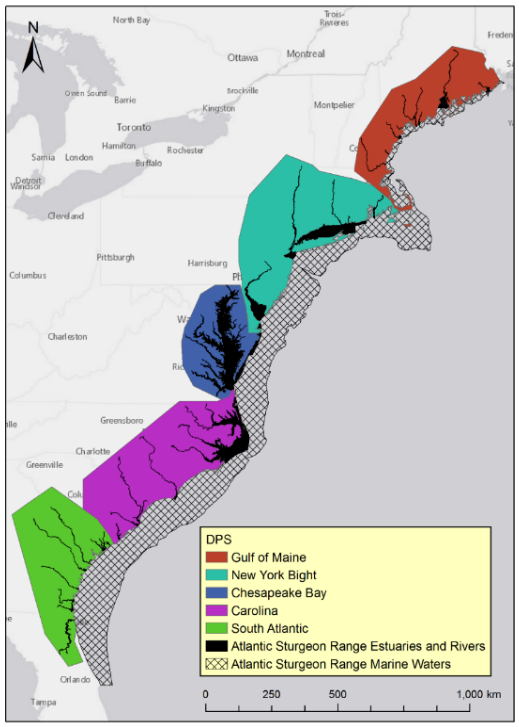

Figure 46: U.S. range of Atlantic sturgeon DPSs

Information available from the 2007 Atlantic sturgeon status review (ASSRT 2007), 2017 ASMFC benchmark stock assessment (ASMFC 2017), final listing rules (77 FR 5880 and 77 FR 5914; February 6, 2012), and material supporting the designation of Atlantic sturgeon critical habitat (NMFS 2017c) were used to summarize the life history, population dynamics, and status of the species.

Life history

Atlantic sturgeon are a late maturing, anadromous species (ASSRT 2007, Balazik et al. 2010, Hilton et al. 2016, Sulak and Randall 2002). Sexual maturity is reached between the ages of 5 to 34 years. Sturgeon originating from rivers in lower latitudes (e.g., South Carolina rivers) mature faster than those originating from rivers located in higher latitudes (e.g., Saint Lawrence River) (NMFS 2017c).

Atlantic sturgeon spawn in freshwater (ASSRT 2007, NMFS 2017d) at sites with flowing water and hard bottom substrate (Bain et al. 2000, Balazik et al. 2012b, Gilbert 1989, Greene et al. 2009, Hatin et al. 2002, Mohler 2003, Smith and Clugston 1997, Vladykov and Greeley 1963).

Water depths of spawning sites are highly variable, but may be up to 88.5 ft (27 m) (Bain et al. 2000, Crance 1987, Leland 1968, Scott and Crossman 1973). Based on tagging records, Atlantic sturgeon return to their natal rivers to spawn (ASSRT 2007), with spawning intervals ranging from one to five years in males (Caron et al. 2002, Collins et al. 2000b, Smith 1985) and two to five years in females (Stevenson and Secor 1999, Van Eenennaam et al. 1996, Vladykov and Greeley 1963). Some Atlantic sturgeon river populations may have up to two spawning seasons comprised of different spawning adults (Balazik and Musick 2015, Collins et al. 2000b), although the majority likely have just one, either in the spring or fall.[12] There is evidence of spring and fall spawning for the South Atlantic DPS (77 FR 5914, February 6, 2012, (Collins et al. 2000b, NMFS and USFWS 1998b), spring spawning for the Gulf of Maine and New York Bight DPSs (NMFS 2017c), and fall spawning for the Chesapeake and Carolina DPSs (Balazik et al. 2012a, Smith et al. 1984). While spawning has not been confirmed in the James River (Chesapeake Bay DPS), telemetry and empirical data suggest that there may be two potential spawning runs: a spring run from late March to early May and a fall run around September after an extended staging period in the lower river (Balazik et al. 2012a, Balazik and Musick 2015).

Following spawning, males move downriver to the lower estuary and remain there until outmigration in the fall (Bain 1997, Bain et al. 2000, Balazik et al. 2012a, Breece et al. 2013, Dovel and Berggren 1983a, Greene et al. 2009, Hatin et al. 2002, Ingram et al. 2019, Smith 1985, Smith et al. 1982). Females move downriver and may leave the estuary and travel to other coastal estuaries until outmigration to marine waters in the fall (Bain 1997, Bain et al. 2000, Balazik et al. 2012a, Breece et al. 2013, Dovel and Berggren 1983a, Greene et al. 2009, Hatin et al. 2002, NMFS 2017c, Smith 1985, Smith et al. 1982). Atlantic sturgeon deposit eggs on hard bottom substrate. They hatch into the yolk sac larval stage approximately 94 to 140 hours after deposition (Mohler 2003, Murawski and Pacheco 1977, Smith et al. 1980, Van Den Avyle 1984, Vladykov and Greeley 1963). Once the yolk sac is absorbed (eight to twelve days post-hatching), sturgeon are larvae. Shortly after, they become young of year and then juveniles. The juvenile stage can last months to years in the brackish waters of the natal estuary (ASSRT 2007, Calvo et al. 2010, Collins et al. 2000a, Dadswell 2006, Dovel and Berggren 1983b, Greene et al. 2009, Hatin et al. 2007, Holland and Yelverton 1973, Kynard and Horgan 2002, Mohler 2003, Schueller and Peterson 2010, Secor et al. 2000, Waldman et al. 1996). Upon reaching the sub-adult phase, individuals enter the marine environment, mixing with adults and sub-adults from other river systems (Bain 1997, Dovel and Berggren 1983a, Hatin et al. 2007, McCord et al. 2007). Once sub-adult Atlantic sturgeon have reached maturity/the adult stage, they will remain in marine or estuarine waters, only returning far upstream to the spawning areas when they are ready to spawn (ASSRT 2007, Bain 1997, Breece et al. 2016, Dunton et al. 2012, Dunton et al. 2015, Savoy and Pacileo 2003).

The life history of Atlantic sturgeon can be divided up into seven general categories as described in Table 43 (adapted from ASSRT 2007).

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[12] Although referred to as spring spawning and fall spawning, the actual time of Atlantic sturgeon spawning may not occur during the astronomical spring or fall season (Balazik and Musick 2015).

Table 43: Descriptions of Atlantic sturgeon life history stages

Age Class	Size	Duration	Description
Egg	~2 mm – 3 mm diameter (Van Eenennaam et al. 1996); p. 773)	Hatching occurs ~3-6 days after egg deposition and fertilization (ASSRT 2007); p. 4)	Fertilized or unfertilized
Yolk-sac larvae (YSL)	~6 mm – 14 mm (Bath et al. 1981); pp. 714-715)	8-12 days post hatch (ASSRT 2007); p. 4)	Negative photo-taxic, nourished by yolk sac
Post yolk-sac larvae (PYSL)	~14mm – 37mm (Bath et al. 1981); pp. 714-715)	12-40 days post hatch	Free swimming; feeding; Silt/sand bottom, deep channel; fresh water
Young of Year (YOY)	0.3 grams <410mm TL	From 40 days to 1 year	Fish that are > 40 days and < 1 year; capable of capturing and consuming live food
Juveniles	>410mm and <760mm TL	1 year to time at which first coastal migration is made	Fish that are at least age 1 and are not sexually mature and do not make coastal migrations.
Subadults	>760 mm and <1500 mm TL	From first coastal migration to sexual maturity	Fish that are not sexually mature but make coastal migrations
Adults	>1500 mm TL	Post-maturation	Sexually mature fish

Population dynamics
A population estimate was derived from the NEAMAP trawl surveys.[13] For this Opinion, as we did in the prior 2013 Opinion, we are relying on the population estimates derived from the NEAMAP swept area biomass assuming a 50 percent catchability (i.e., net efficiency x availability) rate. We consider that the NEAMAP surveys sample an area utilized by Atlantic sturgeon but do not sample all the locations and times where Atlantic sturgeon are present. We also consider that the trawl net captures some, but likely not all, of the Atlantic sturgeon present in the sampling area. Therefore, we assume that net efficiency and the fraction of the population exposed to the NEAMAP surveys in combination result in a 50 percent catchability (NMFS 2013b). The 50 percent catchability assumption reasonably accounts for the robust, yet not complete, sampling of the Atlantic sturgeon oceanic temporal and spatial ranges and the documented high rates of encounter with NEAMAP survey gear. As these estimates are derived directly from empirical data with fewer assumptions than have been required to model Atlantic sturgeon populations to date, we believe these estimates continue to serve as the best available information. Based on the above approach, the overall abundance of Atlantic sturgeon in U.S. Atlantic waters is estimated to be 67,776 fish (see table16 in Kocik et al. 2013). Based on genetic frequencies of occurrence in the sampled area, this overall population estimate was subsequently partitioned by DPS (Table 44). Given the proportion of adults to sub-adults in the NMFS NEFSC observer data (approximate ratio of 1:3), we have also estimated the number of adults and sub-adults originating from each DPS. However, this cannot be considered an estimate of the total number of sub-adults because it only considers those sub-adults that are of a size that are present and vulnerable to capture in commercial trawl and gillnet gear in the marine environment.

It is important to note, the NEAMAP-based estimates do not include young-of-the-year (YOY) fish and juveniles in the rivers; however, those segments of the Atlantic sturgeon populations are at minimal risk from the proposed actions since they are rare to absent within the action area. The NEAMAP surveys are conducted in waters that include the preferred depth ranges of sub-adult and adult Atlantic sturgeon and take place during seasons that coincide with known Atlantic sturgeon coastal migration patterns in the ocean. However, the estimated number of sub-adults in marine waters is a minimum count because it only considers those sub-adults that are captured in a portion of the action area and are present in the marine environment, which is only a fraction of the total number of sub-adults. In regards to adult Atlantic sturgeon, the estimated population in marine waters is also a minimum count as the NEAMAP surveys sample only a portion of the action area, and therefore a portion of the Atlantic sturgeon's range.

[13] Since fall 2007, NEAMAP trawl surveys (spring and fall) have been conducted from Cape Cod, Massachusetts to Cape Hatteras, North Carolina in nearshore waters at depths up to 60 ft (18.3 m). Each survey employs a spatially stratified random design with a total of 35 strata and 150 stations.

Table 44: Calculated population estimates based upon the NEAMAP survey swept area model, assuming 50 percent efficiency

DPS	Estimated Ocean Population Abundance	Estimated Ocean Population of Adults	Estimated Ocean Population of Sub-adults (of size vulnerable to capture in fisheries)
GOM	7,455	1,864	5,591
NYB	34,566	8,642	25,925
CB	8,811	2,203	6,608
Carolina	1,356	339	1,017
SA	14,911	3,728	11,183
Canada	678	170	509

Precise estimates of population growth rate (intrinsic rates) are unknown for the five listed DPSs of Atlantic sturgeon due to a lack of long-term abundance data. The Commission's 2017 stock assessment referenced a population viability assessment (PVA) that was done to determine population growth rates for the five DPSs based on a few long-term survey programs, but most results were statistically insignificant or utilized a model for which the available did not or poorly fit. In any event, the population growth rates reported from that PVA ranged from -1.8 percent to 4.9 percent (ASMFC 2017).

The genetic diversity of Atlantic sturgeon throughout its range has been well-documented (ASSRT 2007, Bowen and Avise 1990, O'Leary et al. 2014, Ong et al. 1996, Waldman et al. 1996, Waldman and Wirgin 1998). Overall, these studies have consistently found populations to be genetically diverse, and the majority can be readily differentiated. Relatively low rates of gene flow reported in population genetic studies (Fritts et al. 2016, Savoy et al. 2017, Wirgin et al. 2002) indicate that Atlantic sturgeon return to their natal river to spawn, despite extensive mixing in coastal waters.

The range of all five listed DPSs extends from Canada through Cape Canaveral, Florida. All five DPSs use the action area. Based on a recent genetic mixed stock analysis (Kazyak et al. 2020, Kazyak et al. 2021)[14], we expect Atlantic sturgeon throughout the action area originate from the five DPSs at the following frequencies: Gulf of Maine 8.7 percent; New York Bight 71.4 percent; Chesapeake Bay 10.7 percent; Carolina 2.6 percent; and South Atlantic 5.6 percent. Approximately 1.0 percent of the Atlantic sturgeon throughout the action area are expected to originate from Canadian rivers or management units. The authors of this recent analysis used 12 microsatellite markers to characterize the stock composition of 1,704 Atlantic sturgeon encountered across the U.S. Atlantic coast dating back to 1980. The primary method to determine the origin of Atlantic sturgeon when they are encountered away from natal habitats is through the use of genetic assignment testing, as was done in Kazyak et al. (2020). However, one

[14] The preliminary analysis (Kazyak 2020) included GARFO and SERO as regions in the output. As the GARFO area most closely aligns with the distribution of fishing effort considered in this Opinion, we used this region in our analysis. See Kazyak 2021 for the published paper based on the same underlying data as the preliminary analysis.

caveat with genetic assignment testing is that not all populations have been discovered and not all discovered populations were used for this assessment. Assignment testing can only assign an individual to a known or defined category. Even if there is very little similarity with the best match, that is where that sample is assigned. Nevertheless, our analyses in this Opinion are done at the DPS level, and we are confident that the five DPSs listed above, in addition to a small percentage of Canadian origin fish which are not differentiated by DPS, represent all the populations of Atlantic sturgeon that occur in the action area and that they occur and may interact with the ten fisheries at the above frequencies.

Depending on life stage, sturgeon may be present in marine and estuarine ecosystems. The action area for this Opinion occurs in marine waters; therefore, this section will focus only on the distribution of Atlantic sturgeon life stages (sub-adult and adult) in marine waters; it will not discuss the distribution of Atlantic sturgeon life stages (eggs, larvae, juvenile, sub-adult, adult) in freshwater ecosystems, specifically, their movements into/out of natal river systems. For more information on Atlantic sturgeon distribution in freshwater ecosystems, refer to ASSRT (2007); 77 FR 5880 (February 6, 2012); 77 FR 5914 (February 6, 2012); NMFS (2017); and ASMFC (2017).

The marine range of U.S. Atlantic sturgeon extends from Labrador, Canada, to Cape Canaveral, Florida. As Atlantic sturgeon travel long distances in these waters, all five DPSs of Atlantic sturgeon have the potential to be anywhere in this marine range. Results from genetic studies show that, regardless of location, multiple DPSs can be found at any one location along the Northwest Atlantic coast, although the Hudson River population from the New York Bight DPS dominates (ASMFC 2017, ASSRT 2007, Dadswell 2006, Dovel and Berggren 1983a, Dunton et al. 2012, Dunton et al. 2015, Dunton et al. 2010, Erickson et al. 2011, Kynard et al. 2000, Laney et al. 2007, O'Leary et al. 2014, Stein et al. 2004b, Waldman et al. 2013, Wirgin et al. 2015a, Wirgin et al. 2015b, Wirgin et al. 2012).

Based on fishery-independent, fishery dependent, tracking, and tagging data, Atlantic sturgeon appear to primarily occur inshore of the 164 ft (50 m) depth contour (Dunton et al. 2012, Dunton et al. 2010, Erickson et al. 2011, Laney et al. 2007, O'Leary et al. 2014, Stein et al. 2004a, b, Waldman et al. 2013, Wirgin et al. 2015a, Wirgin et al. 2015b). However, they are not restricted to these depths and excursions into deeper (e.g., 250 ft (75 m)) continental shelf waters have been documented (Colette and Klein-MacPhee 2002, Collins and Smith 1997, Erickson et al. 2011, Stein et al. 2004b, Timoshkin 1968). Data from fishery-independent surveys and tagging and tracking studies also indicate that some Atlantic sturgeon may undertake seasonal movements along the coast (Dunton et al. 2010, Erickson et al. 2011, Hilton et al. 2016, Oliver et al. 2013, Post et al. 2014, Wippelhauser 2012). For instance, studies found that satellite-tagged adult sturgeon from the Hudson River concentrated in the southern part of the Mid-Atlantic Bight, at depths greater than 66 ft (20 m), during winter and spring; while, in the summer and fall, Atlantic sturgeon concentrations shifted to the northern portion of the Mid-Atlantic Bight at depths less than 66 ft (20 m) (Erickson et al. 2011).

In the marine range, several marine aggregation areas occur adjacent to estuaries and/or coastal features formed by bay mouths and inlets along the U.S. eastern seaboard (i.e., waters off North Carolina, Chesapeake Bay; Delaware Bay; New York Bight; Massachusetts Bay; Long Island Sound; and Connecticut and Kennebec River Estuaries). Depths in these areas are generally no greater than 82 ft (25 m) (Bain et al. 2000, Dunton et al. 2010, Erickson et al. 2011, Laney et al.

2007, O'Leary et al. 2014, Oliver et al. 2013, Savoy and Pacileo 2003, Stein et al. 2004b, Waldman et al. 2013, Wippelhauser 2012, Wippelhauser and Squiers 2015). Although additional studies are still needed to clarify why Atlantic sturgeon aggregate at these sites, there is some indication that they may serve as thermal refugia, wintering sites, or marine foraging areas (Dunton et al. 2010, Erickson et al. 2011, Stein et al. 2004b).

Status

Atlantic sturgeon were once present in 38 river systems and, of these, spawned in 35 (ASSRT 2007). They are currently present in 36 rivers and are probably present in additional rivers that provide sufficient forage base, depth, and access (ASSRT 2007). The benchmark stock assessment evaluated evidence for spawning tributaries and sub-populations of U.S. Atlantic sturgeon in 39 rivers. They confirmed (eggs, embryo, larvae, or YOY observed) spawning in ten rivers, considered spawning highly likely (adults expressing gametes, discrete genetic composition) in nine rivers, and suspected (adults observed in upper reaches of tributaries, historical accounts, presence of resident juveniles) spawning in six rivers. Spawning in the remaining rivers was unknown (ten) or suspected historical (four) (ASMFC 2017). The decline in abundance of Atlantic sturgeon has been attributed primarily to the large U.S. commercial fishery, which existed for the Atlantic sturgeon through the mid-1990s. Based on management recommendations in the ISFMP, adopted by the Commission in 1990, commercial harvest in Atlantic coastal states was severely restricted and ultimately eliminated from most coastal states (ASMFC 1998a). In 1998, the Commission placed a 20-40 year moratorium on all Atlantic sturgeon fisheries until the spawning stocked could be restored to a level where 20 subsequent year classes of adult females were protected (ASMFC 1998a, b). In 1999, NMFS closed the U.S. EEZ to Atlantic sturgeon retention, pursuant to the ACA (64 FR 9449; February 26, 1999). However, many state fisheries for sturgeon were closed prior to this.

The most significant threats to Atlantic sturgeon are incidental catch, dams that block access to spawning habitat in southern rivers, poor water quality, dredging of spawning areas, water withdrawals from rivers, and vessel strikes. Climate change related impacts on water quality (e.g., temperature, salinity, dissolved oxygen, contaminants) also have the potential to affect Atlantic sturgeon populations using impacted river systems.

In support of the above, the Commission released a new benchmark stock assessment for Atlantic sturgeon in October 2017 (ASMFC 2017). Based on historic removals and estimated effective population size, the 2017 stock assessment concluded that all five Atlantic sturgeon DPSs are depleted relative to historical levels (Table 45). However, the 2017 stock assessment does provide some evidence of population recovery at the coastwide scale, and mixed population recovery at the DPS scale (ASMFC 2017). The 2017 stock assessment also concluded that a variety of factors (i.e., bycatch, habitat loss, and ship strikes) continue to impede the recovery rate of Atlantic sturgeon (ASMFC 2017).

Table 45: Stock status determination for the coastwide stock and DPSs (recreated from the Commission's Atlantic Sturgeon Stock Assessment Overview, October 2017)

	Mortality Status	**Biomass/Abundance Status**	
Population	**Probability that $Z>Z_{50\%EPR}$80%***	**Relative to Historical Levels**	**Average probability of terminal year of indices > 1998** value**
Coastwide	7%	Depleted	95%
Gulf of Maine	74%	Depleted	51%
New York Bight	31%	Depleted	75%
Chesapeake Bay	30%	Depleted	36%
Carolina	75%	Depleted	67%
South Atlantic	40%	Depleted	Unknown (no suitable indices)

*EPR= eggs per recruit. The EPR analysis was used to find the value of total mortality (Z) that resulted in an EPR that was 50% of the EPR at the unfished state for ages 4-21 ($Z_{50\%}$).

**For indices that started after 1998, the first year of the index was used as the reference value. The terminal year of a given survey was compared to the fitted abundance index from 1998 (the year the Commission's moratorium for Atlantic sturgeon was implemented).

Despite the depleted status, the Commission's assessment did include signs that the coastwide index is above the 1998 value (95 percent probability). Total mortality from the tagging model was very low at the coastwide level. Small sample sizes made mortality estimates at the DPS level more difficult. By DPS (Table 45), the assessment concluded that there was a 51 percent probability that the Gulf of Maine DPS abundance has increased since 1998 but a 74 percent probability that mortality for this DPS exceeds the mortality threshold used for the assessment. There is a relatively high (75 percent) probability that the New York Bight DPS abundance has increased since 1998, and a 31 percent probability that mortality exceeds the mortality threshold used for the assessment. There is also a relatively high (67 percent) probability that the Carolina DPS abundance has increased since 1998, and a relatively high probability (75 percent) that mortality for this DPS exceeds the mortality threshold used in the assessment. However, the index from the Chesapeake Bay DPS (highlighted red) only had a 36 percent chance of being above the 1998 value and a 30 percent probability that the mortality for this DPS exceeds the mortality threshold for the assessment. There was not enough information available to assess the abundance for the for the South Atlantic DPS relative to the 1998 moratorium, but the assessment did conclude that there was 40 percent probability that the mortality for this DPS exceeds the mortality threshold used in the assessment (ASMFC 2017).

Critical Habitat

Critical habitat has been designated for the five DPSs of Atlantic sturgeon (82 FR 39160, August 17, 2017) in rivers of the eastern United States. These areas are outside the action area.

Recovery Goals

Recovery Plans have not yet been drafted for any of the Atlantic sturgeon DPSs. A recovery outline (see https://www.fisheries.noaa.gov/resource/document/recovery-outline-atlantic-sturgeon-distinct-population-segments) has been developed as interim guidance to direct recovery efforts, including recovery planning, until a full recovery plan is approved.

4.2.3.2. Atlantic Salmon (Gulf of Maine DPS)

The Atlantic salmon is an anadromous fish, migrating up rivers from the ocean to spawn. There are three Atlantic salmon DPSs in the United States: Long Island Sound, Central New England, and the Gulf of Maine DPSs (Fay et al. 2006). The Gulf of Maine (GOM) DPS Atlantic salmon is genetically distinct from other Atlantic salmon. As of 2014, non-native Atlantic salmon were still present in the Central New England and Long Island Sound population segments as an artifact of a reintroduction program that existed in the Connecticut and Merrimack Rivers from 1967 to 2012. In 2013, the USFWS discontinued the federal programs to rebuild these stocks. However, Atlantic salmon persist in some rivers in the Long Island Sound and Central New England DPS because of state efforts. The Atlantic salmon used to support these programs are not part of the listed entity and, therefore, are not protected under the ESA. Only the Gulf of Maine population segment supports native salmon populations (USFWS and NMFS 2019). The GOM DPS, found in watersheds throughout Maine (Figure 47), is the only DPS listed under the ESA. Therefore, this is the only DPS considered in this Opinion.

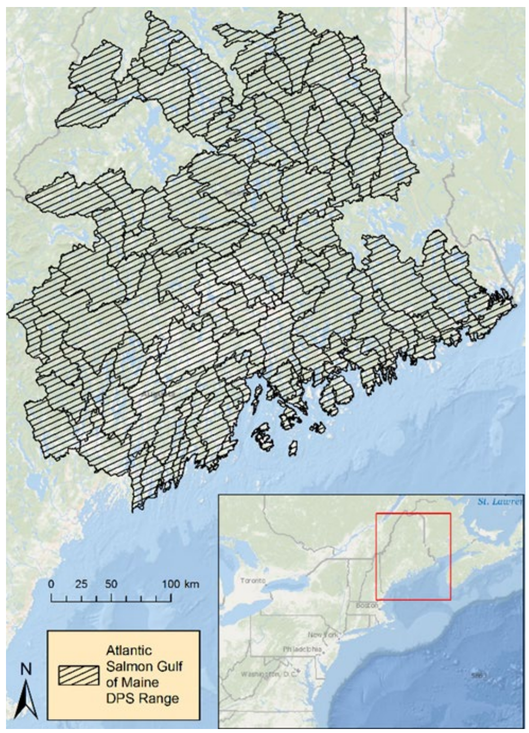

Figure 47: Range of Gulf of Maine DPS of Atlantic salmon

The GOM DPS of Atlantic salmon was initially listed as endangered on November 17, 2000 (65 FR.69459). In 2009, NMFS and USFWS expanded the geographic range for the GOM DPS. The GOM DPS is defined as all anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River, and wherever these fish occur in the estuarine and marine environment. The marine range of the GOM DPS extends from the Gulf of Maine, throughout the Northwest Atlantic Ocean, to the coast of Greenland. Included in the GOM DPS are all associated conservation hatchery populations used to supplement these natural populations. Excluded from the GOM DPS are landlocked Atlantic salmon and those salmon raised in commercial hatcheries for the aquaculture industry (74 FR 29344, June 19, 2009).

In describing the GOM DPS, there are three salmon habitat recovery units (SHRUs). The three SHRUs are the Downeast Coastal SHRU, Penobscot Bay SHRU, and Merrymeeting Bay SHRU. The SHRU delineations were designed to: 1) ensure that a recovered Atlantic salmon population has widespread geographic distribution to help maintain genetic variability; and 2) provide protection from demographic and environmental variation. A widespread distribution of salmon across the three SHRUs will provide a greater probability of population sustainability in the future, which will be needed to achieve recovery of the GOM DPS.

We used information available in the 2006 status review (Fay et al. 2006), the recovery plan (USFWS and NMFS 2019), and recent scientific publications to summarize the life history, population dynamics and status of the species, as follows.

Life History
Atlantic salmon spend most of its adult life in the ocean and return to freshwater to reproduce. Its complex life history includes territorial rearing in rivers to extensive feeding migrations on the high seas. During their life cycle, Atlantic salmon go through several distinct phases that are identified by specific changes in behavior, physiology, morphology, and habitat requirements. They return to rivers in Maine from the Atlantic Ocean primarily between May and early July (Baum and Atlantic Salmon Board 1997), although, they may enter any time from early spring to late summer. Spawning typically occurs in late October through November, and eggs hatch in late March or April (Fay et al. 2006). After spawning, the adults move downstream toward the sea. After reaching the ocean, few survive as indicated by the lack of repeat spawners in the GOM DPS (NMFS and USFWS 2005).

After hatching, Atlantic salmon go through several stages in the river before entering the ocean. Smoltification (the physiological and behavioral changes required for the transition to saltwater) usually occurs at age two for the GOM DPS Atlantic salmon (USASAC 2005). Once entering the marine environment, they travel mainly at the surface of the water column (Renkawitz and Sheehan 2012) and may form shoals, possibly of fish from the same river (Shelton et al. 1997). Atlantic salmon can experience high mortality during the transition to saline environments for reasons that are not well understood (Kocik et al. 2009, Thorstad et al. 2012)

During the late summer and autumn of the first year, North American Atlantic salmon are concentrated in the Labrador Sea and off the west coast of Greenland (Reddin 1985, Reddin and Friedland 1992, Reddin and Short 1991, Renkawitz and Sheehan 2012). The following spring, first year winter and older fish are generally located in the Gulf of St. Lawrence, off the coast of Newfoundland, and on the east coast of the Grand Banks (Dutil and Coutui 1988, Friedland et al. 1999, Reddin 1985, Reddin and Friedland 1992, Ritter 1989).

Population Dynamics
The historic distribution of Atlantic salmon in Maine has been described extensively (Baum 1997). In short, substantial populations of Atlantic salmon existed in nearly every river in Maine that was large enough to maintain a spawning population. The upstream extent of the species' distribution extended far into the headwaters of even the largest rivers. Today, the spatial distribution of Atlantic salmon is limited by obstructions to passage and low abundance levels. Within the range of the GOM DPS, the Kennebec, Androscoggin, Union, Narraguagus, and Penobscot Rivers contain dams that severely limit passage of salmon to significant amounts of spawning and rearing habitat.

Contemporary abundance levels of Atlantic salmon within the GOM DPS are several orders of magnitude lower than historical abundance estimates. For example, Foster and Atkins estimated that roughly 100,000 adult salmon returned to the Penobscot River alone (Foster and Atkins 1869) before the river was dammed, whereas estimates of abundance for the entire GOM DPS have rarely exceeded 5,000 individuals in any given year since 1967 (Fay et al. 2006, USASAC 2013). In the early 1990s, marine survival rates decreased, leading to the declining trend in adult abundance observed throughout the 1990s and early 2000s. Adult returns have fluctuated over the past decade.

Adult returns of Atlantic salmon from 1997 to 2018 ranged from 450 to 4,178. In 2018, there were 869 returns to rivers in the United States. Most (99.2 percent) returns were to the GOM DPS (USASAC 2019). From 2010-2019, the ten year average returns was 1,247 adults, with 120 returns to the Downeast Coastal SHRU, 56 to the Merrymeeting Bay SHRU, and 1,071 to the Penobscot Bay SHRU (Kircheis et al. 2020). The counts include both wild and hatchery-origin fish. The DPS encompasses all anadromous Atlantic salmon in a freshwater range covering the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River and includes all associated conservation hatchery populations used to supplement these natural populations (USFWS and NMFS 2019). Most (88.1 percent in 2018 and 76 percent in 2019) returns were hatchery smolt origin. The remaining returns originated from natural reproduction, 0+ fall stocked parr, hatchery fry, or eggs (USASAC 2019, 2020). Each year, the Penobscot River supported the majority of adult returns (92-98 percent); the Narraguagus River supported between 0.8 to 4.1 percent of adult (Fay et al. 2006). In 2017, over 4 million juvenile salmon (eggs, fry, parr and smolts) and 4,849 adults were stocked in the Connecticut, Merrimack, Saco, Penobscot and five other coastal rivers in Maine. Over 5.5 million juvenile and 5,715 adults were released U.S. rivers in 2018 (USASAC 2019); over 4.7 million juvenile and 5,710 adults were released into U.S. rivers in 2019 (USASAC 2020). Low abundances of both hatchery-origin and naturally reared adult salmon returns to Maine demonstrate continued poor marine survival.

Status

Atlantic salmon face a number of threats to their survival, which are outlined in the recovery plan (USFWS and NMFS 2019). The most significant threats to the GOM DPS of Atlantic salmon include, among others: lack of access to spawning and rearing habitat; reduced habitat complexity; sedimentation of spawning/rearing habitat; degraded water quality; water withdrawal; recreational bycatch; poaching; foreign intercept fishery; competition from introduced species; disease; predation; improper hatchery practices; and climate change.

Genetic diversity is monitored by assessing sea-run adults for the Penobscot River and juvenile fish for other populations. Allelic diversity has remained relatively constant since the mid-1990s; though, slight decreases were detected in the East Machias and Dennys populations (USASAC 2019).The GOM DPS of Atlantic salmon currently exhibits critically low spawner abundance, poor marine survival, and is confronted with a variety of additional threats. The abundance of GOM DPS Atlantic salmon has been low and either stable or declining over the past several decades. The proportion of fish that are of natural origin is small and displays no sign of growth. The spatial distribution of the GOM DPS has been severely reduced relative to historical distribution patterns. The conservation hatchery program assists in slowing the decline and helps stabilize populations at low levels, but has not contributed to an increase in the overall abundance of salmon and has not been able to halt the decline of the naturally reared component of the GOM DPS. Continued reliance on the conservation hatchery program could prevent extinction in the short term, but recovery of the GOM DPS must be accomplished through increases in naturally reared salmon. Based on the information above, the species would likely have a low resilience to additional perturbations.

Critical Habitat

Critical habitat for the GOM DPS of Atlantic salmon has been designated, but is outside the action area of this Opinion.

Recovery Goals

As with other plans, the overall goal of the recovery plan is delisting (USFWS and NMFS 2019). The interim goal is to downlist the DPS from endangered to threatened. Complete down listing/delisting criteria for each SHRU's recovery goals are included in the recovery plan. Reclassification objectives include maintaining sustainable, naturally reared populations with access to suitable habitat in at least two of the SHRUs, ensuring management options for marine survival are better understood, and reducing/eliminating threats that pose an imminent risk of extinction. Delisting criteria include maintaining self-sustaining, wild populations with access to suitable habitat for all SHRUs, ensuring necessary management options for marine survival are in place, and reducing/eliminating threats that pose a risk of endangerment to the DPS (USFWS and NMFS 2019). Recovery actions include:

1. Enhance connectivity between ocean and freshwater habitats important for recovery.
2. Increase adult spawners through the freshwater production of smolts.
3. Increase Atlantic salmon survival through increased ecosystem understanding and identification of spatial and temporal constraints to salmon marine productivity to inform and support management actions that improve survival.
4. Collaborate with partners and engage interested parties in recovery efforts.
5. Ensure federal agencies and associated programs continue to recognize and uphold federal Tribal Trust responsibilities.
6. Provide demographic support and maintain genetic diversity appropriate for recovery through the conservation hatchery program.
7. Maintain the genetic diversity and promote increased fitness of Atlantic salmon populations over time.
8. Identify funding programs that support State, local and NGO conservation efforts.

4.2.3.3. Giant Manta Ray

The giant manta ray (*Manta birostris*) is an elasmobranch that is found worldwide in tropical, subtropical, and temperate oceanic waters and near productive coastlines (Figure 48). The giant manta ray has a diamond-shaped body with wing-like pectoral fins measuring up to 25 feet (8 meters) across. It was listed as threatened on January 22, 2018 (83 FR 2916).

Figure 48: Extent of occurrence (EOO) and area of occurrence (AOO) of giant manta rays (Lawson et al. 2017)

We used information available in the status review (Miller and Klimovich 2017), the final listing (83 FR 2916, January 22, 2018), and recent scientific publications to summarize the life history, population dynamics, and status of the species, as follows.

Life History

Giant manta rays are planktivores, using gill plates (also known as gill rakers) to feed on zooplankton. Reaching sexual maturity at about four to five years old, they give birth to live young, one pup every two to three years. Gestation lasts between 12 to 13 months. Manta rays can live up to 40 years, a female may produce between five to 15 pups in a lifetime. In the western North Atlantic, the maximum age of the giant manta rays is unknown, the age at maturing is 4.3-4.6 for females and >3.5 for males, and the litter size is 1 (Miller and Klimovich 2017).

Giant manta rays are commonly found offshore, in oceanic waters, and near productive coastlines (Kashiwagi et al. 2011, Marshall et al. 2009). In the northwest Atlantic, they have been documented as far north as New Jersey. Additionally, giant manta rays exhibit a high degree of plasticity in terms of their use of depths within their habitat, with tagging studies that show the species conducting night descents of 656-1,312 ft (200-450 m) depths (Rubin and Kumli 2008, Stewart et al. 2016b) and capable of diving to depths exceeding 3,280 ft (1,000 m) (Miller and Klimovich 2017).

The giant manta ray is considered to be a migratory species, with estimated distances travelled of up to 810 nmi (1,500 km) (Miller and Klimovich 2017). However, there is some evidence that *M. birostris* may actually exist as well-structured subpopulations that exhibit a high degree of residency (Stewart et al. 2016a). The species may be capable of occasional long-distance movements; although, these movements may be rare and may not contribute to substantial gene flow or interpopulation mixing of individuals (Stewart et al. 2016a). Additional research is required to better understand the distribution and movement of the species throughout its range.

Population Dynamics

There are no current or historical estimates of range-wide abundance, although there are some rough estimates of subpopulation size based on anecdotal accounts from fishermen and divers. It is difficult to obtain reliable abundance estimates as the species is only sporadically observed. There are about 11 (perhaps more) subpopulations worldwide (Miller and Klimovich 2017). Based on anecdotal diver or fisherman observations, populations potentially range from 100 to

1,500 individuals each (FAO 2012, Miller and Klimovich 2017). While observations of individuals in local aggregations range from around 40 individuals to over 600, estimates of subpopulation size have only been calculated for Mozambique (n=600) and Isla de la Plata, Ecuador (n=1,500) (Miller and Klimovich 2017).

The only abundance data for giant manta rays in the Atlantic are records of more than 70 individuals in the Flower Garden Banks Marine Sanctuary in the Gulf of Mexico, more than 90 individuals off the east coast of Florida, and 60 individuals in the waters off Brazil (Miller and Klimovich 2017). Based on personal observation during aerial surveys conducted off of St. Augustine, Florida, from 2009-2012, F. Young (pers. comm. 2017 as cited in Miller and Klimovich 2017) noted vast schools of giant manta rays, with over 500 manta rays observed per 6-8 hour day of aerial survey. There is no population growth rate available for the giant manta ray. In areas where the species is not subject to fishing, populations may be stable. Population declines in waters where the manta rays are protected have also been observed but attributed to overfishing of the species in adjacent areas within its large home range and its migratory nature.

Status

In areas where the species is not subject to fishing, populations may be stable. However, in regions where giant manta rays are (or were) actively targeted or caught as bycatch populations appear to be decreasing (Miller and Klimovich 2017). Overfishing is the most significant threat to giant manta rays. Giant manta rays are both targeted and caught as bycatch. Manta rays are caught throughout their range in commercial and artisanal fisheries. Fishermen targeting manta rays primarily use harpoons and nets, while significant manta bycatch occurs in purse seine, gillnet, and trawl fisheries targeting other species. The gill plates are highly valued in international trade for use in traditional medicine. Cartilage and skins are also traded internationally while meat is consumed or used for bait locally (Miller and Klimovich 2017).

Due to their association with nearshore habitats, manta rays are at elevated risk for exposure to a variety of contaminants and pollutants, including brevetoxins, heavy metals, polychlorinated biphenyls, and plastics. Many pollutants in the environment have the ability to bioaccumulate in fish species; however, only a few studies have specifically examined the accumulation of heavy metals in the tissues of manta rays (Essumang 2010, Ooi et al. 2015). Plastics within the marine environment may also be a threat to the giant manta ray, as the animals ingest microplastics (through filter feeding) or become entangled in plastic debris, potentially contributing to increased mortality rates. There are few known natural threats to giant manta rays. Disease and shark attacks were ranked as low risk threats, and giant manta rays exhibit high survival rates after maturity (Miller and Klimovich 2017).

Giant manta rays are at risk throughout a significant portion of their range, due in large part to the observed declines in the Indo-Pacific. There have been decreases in landings of up to 95 percent in the Indo-Pacific; such declines have not been observed in other subpopulations such as Mozambique and Ecuador (Miller and Klimovich 2017).

Overall, given the evidence of minimal bycatch of the species in U.S. waters (see Miller and Klimovich (2017) for additional discussion), it is unlikely that overutilization as a result of bycatch mortality is a significant threat to *M. birostris* in the Atlantic Ocean (83 FR 2916; January 22, 2018). As described in section 7.6, between 2010 and 2019, two giant manta rays have been captured in trawl gear and two in gillnet gear in the action area. However, information is severely lacking on both population sizes and distribution of the giant manta ray as well as current catch and fishing effort on the species throughout this portion of its range.

While there is considerable uncertainty regarding the species' current abundance throughout its range, the best available information indicates that the species has experienced population declines of potentially significant magnitude within areas of the Indo-Pacific and eastern Pacific portions of its range, primarily due to fisheries-related mortality (Miller and Klimovich 2017). Yet, larger subpopulations of the species still exist, including off Mozambique, Ecuador, and potentially Thailand. While we assume that declining populations within the Indo-Pacific and eastern Pacific portions of its range will likely translate to overall declines in the species throughout its entire range, there is very little information on the abundance, spatial structure, or extent of fishery-related mortality of the species within the Atlantic portion of its range (Miller and Klimovich 2017).

Critical Habitat
No critical habitat has been designated for the giant manta ray.

Recovery Goals
NMFS has not prepared a recovery plan for the giant manta ray.

5. ENVIRONMENTAL BASELINE

The environmental baseline for this Opinion 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 consultation, 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).

The *Environmental Baseline* includes the effects of several activities that may affect the survival and recovery of right, fin, sei, and sperm whales; loggerhead (Northwest Atlantic DPS), leatherback, Kemp's ridley, and green (North Atlantic DPS) sea turtles; and Atlantic sturgeon, Atlantic salmon, and giant manta rays in the action area. The activities that shape the *Environmental Baseline* of this consultation generally include: federal fisheries management plans; aquaculture; hopper dredging, sand mining and beach nourishment activities; research and other permitted activities; federal vessel operations; military operations; offshore oil and gas; offshore energy development; non-federally regulated fisheries; maritime industry; pollution; coastal development; and recovery activities associated with reducing impacts to listed species.

The overall impacts of each state, federal, and private action or other human activities have on ESA-listed species is not fully known. For actions outside the action area, the impacts of human activities on ESA-listed species are discussed and incorporated into the status of each species (section 4) considered in this Opinion. Section 4 also recognizes the benefits of recovery activities already implemented. In some cases, the benefits of a recovery action may not be evident in the status of the respective population for years, or even decades, given the relatively late age some species (e.g., sea turtles) reach maturity and depending on the age class(es) affected. This section characterizes actions within the action area and their impacts on ESA-listed species.

5.1. Federal Actions with Formal or Early Section 7 Consultations

NMFS has conducted a number of section 7 consultations to address the effects of federal actions on threatened and endangered species in the action area. Each of those consultations sought to develop ways to avoid and reduce impacts of the action on listed species.

As described in section 2.1, we have consulted previously on the operation of the fisheries considered in this Opinion. Gears used in these fisheries (i.e., trap/pot, sink gillnet, bottom trawl, hook and line) are known to have affected ESA-listed species, with some interactions causing injury and death. Therefore, the *Environmental Baseline* for this action includes the effects of the past operation of these fisheries.

5.1.1. *Authorization of Fisheries through Fishery Management Plans*

In the Northwest Atlantic, NMFS GARFO manages federal fisheries from Maine to Cape Hatteras, North Carolina; however, the management areas for some of these fisheries range from Maine through Virginia, while others extend as far south as Key West, Florida. The NMFS Southeast Regional Office (SERO) manages federal fisheries from Cape Hatteras, North Carolina to Texas, including Puerto Rico and the U.S. Virgin Islands. Fisheries managed by NMFS GARFO and SERO overlap in some parts of the action area.

Both regions have conducted ESA section 7 consultation on all federal fisheries authorized under an FMP or ISFMP. NMFS SERO has formally consulted on the following fisheries: (1) coastal migratory pelagics (NMFS 2015d, 2017a); (2) snapper/grouper (NMFS 2015d); (3) dolphin/wahoo (NMFS 2003) (4) southeast shrimp trawl fisheries (NMFS 2021) (5) Atlantic highly migratory species, excluding pelagic longline (NMFS 2020c) and (6) pelagic longline Atlantic highly migratory species (NMFS 2020d). As described in the Consultation History, NMFS GARFO has formally consulted on the American lobster; Northeast multispecies; monkfish; spiny dogfish; Atlantic bluefish; Northeast skate complex; Atlantic mackerel/squid/butterfish; summer flounder/scup/black sea bass; Atlantic deep-sea red crab, and Atlantic sea scallop fisheries.

In these past opinions, only the consultation on the Atlantic highly migratory species, excluding pelagic longline, and the snapper/grouper opinions (NMFS 2020c) concluded that there was a potential for collisions between fishing vessels and an ESA-listed species (specifically, sea turtles). Any effects to their prey and/or habitat were found to be insignificant and discountable. We have also determined that the Atlantic herring, Atlantic surfclam and ocean quahog, and golden and blueline tilefish fisheries are not likely to adversely affect any ESA-listed species or their designated critical habitats (NMFS 2010b, 2017h, 2020e).

Impacts to Large Whales

North Atlantic right, fin, sei, and sperm whales are at risk from entanglement in fishing gear when in the action area. As discussed in the *Status of the Species*, entanglement in the vertical lines of fixed fishing gear is a leading cause of serious injury and mortality to large whales. Past consultations on the fisheries summarized below considered the potential adverse effects to large whales and included triggers for reinitiation if anticipated levels of entanglement are exceeded

(Table 46). It is important to note that whales may not die immediately from an entanglement in fishing gear but may gradually weaken or otherwise be affected so that further injury or death is likely (Hayes et al. 2018a). The sublethal stress of entanglements can have a serious impact on individual health and reproductive rates (Lysiak et al. 2018, Pettis et al. 2017, Robbins et al. 2015). No take of ESA-large whales due to vessel operations is anticipated in the biological opinions in Table 46.

Table 46: Most recent biological opinions prepared by NMFS GARFO and SERO for federally managed fisheries in the action area that result in takes of large whales and their respective reinitiation triggers. Unless noted, reinitiation triggers for take are reviewed on an annual basis.

	Date	**Right whale**	**Fin whale**	**Sei whale**	**Sperm whale**
GARFO FMPs					
American lobster	July 31, 2014	Up to 3.25 M/SI per year over 5 years	Up to 1.7 M/SI per year over 5 years	Up to 0.2 M/SI per year over 5 years	0
Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, Summer Flounder/ Scup/Black Sea Bass (Batched Fisheries)	December 16, 2013 (amended March 10, 2016)	Up to 3 M/SI per year	Up to 3 M/SI per year	Up to 2 M/SI per year	0
Red Crab	February 6, 2002	0	Up to 1 M/SI annually	0	Up to 1 M/SI per year
SERO FMPs					
South Atlantic Snapper-Grouper	December 1, 2016	1 lethal every 25-42 years	0	0	0

Impacts to Sea Turtles

Each of the most recent GARFO and SERO fishery consultations noted above have considered adverse effects to green, Kemp's ridley, loggerhead, and leatherback sea turtles. In each of the fishery opinions, we concluded that the ongoing action was likely to adversely affect but was not likely to jeopardize the continued existence of any sea turtle species. Each of these opinions included an Incidental Take Statement (ITS) exempting a certain amount of lethal or non-lethal take resulting from interactions with the fisheries. These ITSs are summarized below (Table 47). Unless specifically noted, all numbers denote an annual number of captures that may be lethal or non-lethal. The Northeast Fisheries Science Center (NEFSC) has estimated the take of sea turtles in gillnet, dredge, and trawl gear in the Greater Atlantic Region (Table 48). When available, these estimates were considered in developing the ITSs.

Table 47: Most recent biological opinions prepared by NMFS GARFO and SERO for federally managed fisheries in the action area that result in the take of sea turtles and their respective ITSs. Unless noted, levels of incidental take exempted are on an annual basis.

	Date	Loggerhead	Kemp's ridley	Green	Leatherback
GARFO FMPs					
American lobster	July 31, 2014	1 (lethal or non-lethal)	0	0	7 (lethal or non-lethal)
Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, and Summer Flounder/ Scup/Black Sea Bass (Batched Fisheries)	December 16, 2013 (amended March 10, 2016)	1,345 (835 lethal) over 5 years in gillnets; 852 (284 lethal) over a 4 years in bottom trawls; 1 (lethal or non-lethal) in trap/pot gear	4 (3 lethal) in gillnets; 3 (2 lethal) in bottom trawls	4 (3 lethal) in gillnets; 3 (2 lethal) in bottom trawls	4 (3 lethal) in gillnets; 4 (2 lethal) in bottom trawls; 4 (lethal or non-lethal) in trap/pot gear
Atlantic sea scallop	July 12, 2012 (amended November 27, 2018)	322 (92 lethal) over 2 years in dredges; 700 (330 lethal) over 5 years in trawls	3 (2 lethal) in dredges and trawls combined	2 (lethal) in dredges and trawls combined	2 (lethal) in dredges and trawls combined
Red Crab	February 6, 2002	1 (lethal or non-lethal)	0	0	1 (lethal or non-lethal)
SERO FMPs					
Coastal migratory pelagics	June 18, 2015, amended 2017	27 over 3 years (7 lethal)	8 over 3 years (2 lethal)	31 over 3 years (9 lethal)*	1 over 3 years (1 lethal)
South Atlantic snapper-grouper	December 1, 2016	629 (208 lethal) over 3 years	180 (59 lethal) over 3 years	111 (42 lethal) over 3 years	6 (5 lethal) over 3 years
Southeastern U.S. shrimp	April 26, 2021	72,670 (2,150 lethal) over 5 years	84,495 (8,505 lethal) over 5 years	21,214 (1,700 lethal) over 5 years	130 (5 lethal) over 5 years
HMS fisheries, excluding pelagic longline	January 10, 2020	91 (51 lethal) over 3 years	22 (11 lethal) over 3 years	46 (21 lethal) over 3 years	7 (3 lethal) over 3 years

	Date	Loggerhead	Kemp's ridley	Green	Leatherback
HMS, pelagic longline	May 15, 2020	1080 (280 lethal) over 3 years	21 (8 lethal) combined Kemp's ridley, green (includes North Atlantic and South Atlantic DPS), hawksbill, or olive ridley over 3 years		996 (275 lethal) over 3 years
South-Atlantic dolphin-wahoo	August 27, 2003	12 (2 lethal)	3 (1 lethal) combination of Kemp's ridley, green, or hawksbill		12 (1 lethal)

*Coastal migratory pelagic consultation: 31 green sea turtle takes of both DPSs combined is expected, but no more than 30 from the North Atlantic DPS and no more than 2 from the South Atlantic DPS

Table 48 Estimates of average annual turtle interactions in fishing gear. Numbers in parentheses are adult equivalents

Gear	Years	Area	Estimated Interactions (adult equivalents)	Mortalities (adult equivalents)	Source
Sea Scallop Dredge	2009-2014	Mid-Atlantic	Loggerhead: 22 (2)	9-19* (1-2)	Murray (2015a)
Sink Gillnet	2012-2016	Mid-Atlantic	Loggerhead: 141(3.8) Kemp's ridley: 29 Leatherbacks: 5.4 Unid. hardshell: 22.4	Loggerhead: 111.4 Kemp's ridley: 23 Leatherbacks: 4.2 Unid. hardshell: 17.6	Murray (2018)
Bottom Trawl	2014-2018	Mid-Atlantic and Georges Bank	Loggerhead: 116.6 (36.4) Kemp's ridley: 9.2 Green: 3.2 Leatherbacks: 5.2	Loggerhead: 54.4 (17.4) Kemp's ridley: 4.6 Green: 1.6 Leatherbacks: 2.6	Murray (2020)

*Of these interactions, 9-19 would result in mortality depending on whether loggerheads that interacted with chain mats without being captured (the unobservable but quantifiable interactions) survived.

Given that past biological opinions in the Greater Atlantic Region considered the federal fishery to include federally-permitted vessels operating in state waters, the anticipated take of sea turtles in Table 47 includes gear interactions in both state and federal waters by federally-permitted vessels. The distribution and likelihood of sea turtle takes are highly variable such that interactions in nearshore and coastal waters in some years could be higher if greater fishing effort is expended (due to less travel time and ease of access to a wider range of vessels) or sea turtles are present in greater numbers in those waters. The amount of observer coverage allocated to nearshore versus offshore trips may also be a factor in how many sea turtle interactions are documented in certain waters for these fisheries.

Impacts to Atlantic sturgeon

Commercial fisheries that operate in the action area for this consultation capture Atlantic sturgeon originating from each of the five listed DPSs. Given this, consultations on fisheries in the Southeast and Greater Atlantic Regions have considered the take of Atlantic sturgeon (Table 49).

Table 49: Most recent biological opinions prepared by NMFS GARFO and SERO for federally managed fisheries in the action area that result in takes of the five DPSs of Atlantic sturgeon and their respective ITSs. Unless noted, levels of incidental take exempted are on an annual basis.

	Date	Gulf of Maine DPS	New York Bight DPS	Chesapeake Bay DPS	Carolina DPS	South Atlantic DPS
GARFO FMPs						
Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/ Butterfish, and Summer Flounder/ Scup/Black Sea Bass (Batched Fisheries)	December 16, 2013 (amended March 10, 2016)	137 over a 5-year average in gillnet gear (17 adult equivalents lethal); 148 over a 5-year average in bottom trawl gear (5 adult equivalents lethal)	632 over a 5-year average in gillnet gear (79 adult equivalents lethal); 685 over a 5-year average in bottom trawl gear (21 adult equivalents lethal)	162 over a 5-year average in gillnet gear (21 adult equivalents lethal); 175 over a 5-year average in bottom trawl gear (6 adult equivalents lethal)	25 over a 5-year average in gillnet gear (4 adult equivalents lethal); 27 over a 5-year average in bottom trawl gear (1 adult equivalents lethal)	273 over a 5-year average in gillnet gear (34 adult equivalents lethal); 296 over a 5-year average in bottom trawl gear (9 adult equivalents lethal)
Atlantic sea scallop	July 12, 2012 (amended November 27, 2018)	One take annually in scallop trawl gear from any of the five DPSs (one lethal take every 20 years from any of the five DPSs)				
SERO FMPs						
Coastal migratory pelagics	June 18, 2015	2 (12)* every 3 years; 0 lethal	4 (12)* every 3 years; 0 lethal	3 (12)* every 3 years; 0 lethal	4 (12)* every 3 years; 0 lethal	10 (12)* every 3 years; 0 lethal
Southeastern U.S. shrimp	April 26, 2021	2 (0 lethal) every 5 years	7 (2 lethal) every 5 years	19 (4 lethal) every 5 years	66 (15 lethal) every 5 years	103 (24 lethal) every 5 years
HMS fisheries, excluding pelagic longline	January 10, 2020	34 (8 lethal) every 3 years	170 (36 lethal) every 3 years	40 (9 lethal) every 3 years	10 (5 lethal) every 3 years	75 (19 lethal) every 3 years

*The coastal migratory pelagics biological opinion estimates a total take of 12 Atlantic sturgeon. The biological opinion considered the percent each DPS, presented as a range, expected to be in the action area. To be conservative, the biological opinion considered the high end of the range in apportioning take between DPSs. However, in total, no more than 12 Atlantic sturgeon are anticipated to be taken (NMFS 2015d, 2017a).

In a review of bycatch rates on fishing trips from 1989 to 2000, Atlantic sturgeon were recorded in both gillnet and trawl gears, and bycatch rates varied by gear type and target species. Bycatch was highest for sink gillnets in specific areas of the coast. Mortality was higher in sink gillnets than trawls (Stein et al. 2004a). More recent analyses were completed in 2011 and 2016.

In 2011, the NEFSC prepared a bycatch estimate for Atlantic sturgeon captured in federally managed commercial sink gillnet and otter trawl fisheries from Maine through Virginia. This estimate indicated that from 2006-2010, an annual average of 3,118 Atlantic sturgeon were captured in these fisheries with 1,569 in sink gillnet and 1,548 in otter trawls. The mortality rate in sink gillnets was estimated at approximately 20 percent, and the mortality rate in otter trawls was estimated at 5 percent. Based on this estimate, 391 Atlantic sturgeon mortalities were estimated annually in federal fisheries prosecuted in the Greater Atlantic Region (Miller and Shepard 2011).

An updated, although unpublished, Atlantic sturgeon bycatch estimate in Northeast sink gillnet and otter trawl fisheries for 2011-2015 was prepared by the NEFSC in 2016. Using this information, the authors of the recent Atlantic Sturgeon Benchmark Stock Assessment (ASMFC 2017) estimated that 1,139 fish (295 lethal; 25 percent) were caught in gillnet fisheries and 1,062 fish (41 lethal; 4 percent) were caught in otter trawl fisheries each year from 2000-2015. Atlantic sturgeon bycatch estimates for Northeast gillnet and trawl gear from 2011-2015 (approximately 761 fish per year for gillnets, 777 per year for trawls) are substantially lower than those from 2006-2010 (approximately 1,074 fish per year for gillnets, 1,016 per year for trawls) (ASMFC 2017). It should be noted that the models used in 2011 and 2016 differed. While the model framework and selection methodology remained the same, the best performing models changed due to the nature of using a model-based approach in the estimation and the incorporation of additional years of observer data (memorandum from William A. Karp, NEFSC Director, to John Bullard, GARFO Regional Administrator, August 29, 2016).

Impacts to Atlantic Salmon

Atlantic salmon originating from the Gulf of Maine DPS may be captured and die in commercial trawl and gillnet fisheries operating in the action area. Based on observer reports assessed, NMFS, in the 2013 batched fisheries biological opinion, anticipated the observed take of up to five individuals (two lethal) over a five-year average in gillnet gear and up to five individuals (3 lethal) over a five-year average in bottom trawl gear. The Northeast Fisheries Observer Program (NEFOP) and At-Sea Monitor (ASM) observers have not recorded any interactions from 2014 through 2019. The anticipated level of incidental take of Atlantic salmon for the recreational components of the fisheries could not be estimated at the time.

Impacts to Giant Manta Rays

In the Atlantic Ocean, bycatch of giant manta rays has been observed in purse seine, trawl, and longline fisheries, but they do not appear to be a significant component of the bycatch (Miller and Klimovich 2017). The recent consultation on the Atlantic HMS fishery, excluding pelagic longline, anticipates the take of nine (no lethal) giant manta rays over a 3-year period (NMFS 2020c). The most recent consultation on the HMS pelagic longline fishery anticipates the take of 366 giant manta rays over a 3-year period, up to 6 may be lethal (NMFS 2020d)

In the U.S. bottom longline, trawl, and gillnet fisheries operating in the western Atlantic, giant manta rays are a very rare occurrence and available records of observed captures in U.S fisheries indicate that the vast majority of giant manta rays are released alive (C. Horn, pers. comm. December 3, 2018). NEFSC observer data from 2001-2018 confirms that two giant manta rays were captured (both in 2014) in bottom otter trawl gear where the trip was targeting squid or butterfish. Additionally, seven unknown ray species reported captured in trawl gear and four captured in gillnet gear, may have been giant manta rays. In all 13 cases, the animals were released alive.

Hook-and-line gear primary affects giant manta rays through hooking, but also by entanglement and trailing of gear(NMFS 2020c). From 2008 through 2016, Southeast fisheries observers documented three giant manta rays in bottom longline fisheries (one in the Gulf of Mexico reef fish fishery and two in the South Atlantic shark bottom longline research fishery). Two of these giant manta rays are thought to have been released alive, and one was kept. Gillnet gear used the Coastal Migratory Pelagics (CMP) FMP is known to interact with giant manta ray. During 2005-2012, ten giant manta rays caught in CMP gillnet gear were observed to be released alive.

In the Southeast U.S. gillnet fisheries, bycatch of manta rays is low. The NMFS Southeast Gillnet Observer Program covers all anchored (sink and stab), strike, or drift gillnet fishing by vessels operating in waters from Florida to North Carolina and the Gulf of Mexico. From 1998-2015 the number of all mantas observed captured by observers ranged from 0 to 16, with the vast majority (around 89 percent) released alive (see NMFS reports available at http://www.sefsc.noaa.gov/labs/panama/ob/gillnet.htm). Since January 2013, no mantas have been observed caught as bycatch.

5.1.2. ***Aquaculture***

Aquaculture has the potential to impact ESA-listed species through entanglement and/or other interaction with aquaculture gear (e.g., buoys, nets, and vertical lines), introduction or transfer of pathogens, increased vessel traffic and noise, impacts to habitat and benthic organisms, and water quality (Clement 2013, Lloyd 2003, Price and Morris 2013, Price et al. 2017). Current data suggest that documented interactions and entanglements of ESA-listed marine mammals and sea turtles with aquaculture gear are rare (Price et al. 2017). However, this information includes documented interactions only and may not be reflective of actual interaction. There are also concerns about interactions between Atlantic sturgeon and aquaculture with respect to, among others, entanglements, changes to water features related to migration and residency, and habitat conversion. Aquaculture projects have the potential to modify critical habitat through impacts to water quality and habitat conversion. Some components of aquaculture gear and gear used in commercial fisheries are similar; therefore, information on interactions in the similar gear may provide information on the risk aquaculture poses. There are very few reports of ESA-listed marine mammal interactions with aquaculture gear in the U.S. Atlantic, although, it is not always possible to determine whether the gear on animals is from aquaculture or commercial fisheries (Price et al. 2017). There are several reports of sea turtles in the North Atlantic entangled in aquaculture gear (Price et al. 2017), including one entanglement within the action area.

In the United States, marine aquaculture production increased an average of 3.3 percent per year from 2009-2014; however, globally, the United States remains a relatively minor aquaculture producer. Farmed items in the Atlantic include finfish (e.g., Atlantic salmon, steelhead trout), shellfish (e.g., American and European oyster, quahog, blue mussels, softshell clams, sea and bay scallops, and quahogs), and sea vegetables (e.g., sugar kelp). Trials with other species, such as cod and halibut have occurred previously and there is known interest to farm other marine fish species in the future, such as striped bass and black sea bass. Hatchery-raised species are also used to support important commercial and recreational fisheries, as well as for habitat and endangered species restoration. Aquacultured products are grown for medical research, pharmaceuticals, food additives, ornamentals, and aquarium commerce.

The 2018 Census on Aquaculture collected national data about the industry (USDA 2019). In this survey, aquaculture is the farming of aquatic organisms, including baitfish, crustaceans, food fish, mollusks, ornamental fish, sport/game fish, and other products. It includes algae and sea vegetables but does not include other aquatic plants. The 2018 Census reports 774 saltwater farms and 51,674 acres of saltwater aquaculture from Maine through Florida. It should be noted that this includes the west coast of Florida and that, for some states (Georgia, South Carolina, Delaware, New Jersey), the acreage is not reported to preserve confidentiality (USDA 2019). In addition, the farms reported may be in estuaries that are outside the action area.

Currently, marine aquaculture in the action area occurs mainly in state waters and at relatively modest scales; however, many are interested in expanding operations. States have different rules and regulations for permitting or leasing space and for monitoring required by developers. In the southeastern United States, marine aquaculture is dominated by shellfish production, primarily oysters and clams, with soft-shell crab, live rock, and sea vegetables produced at lower levels (Bacheler et al. 2018). Most farms are located in shallow, intertidal areas (Bacheler et al. 2018) where the fisheries in this Opinion do not operate. In the southeastern United States, there are no aquaculture farms in federal waters. In the 2018 Census, 70 percent of the farms and 98 percent of the acreage reported was in the Greater Atlantic Region (USDA 2019). Therefore, the remainder of the section will focus on the Greater Atlantic Region.

Aquaculture in the Greater Atlantic Region is, at present, primarily in state waters. Currently, there is one U.S. Army Corps of Engineers (ACOE) permit for a pilot scale blue mussel aquaculture operation in federal waters of the Atlantic coast. This project is located eight miles off Rockport, MA and has placed three longlines in the water. The permittee submitted an application to ACOE on 12/7/2019 to expand the operation to a total of 20 longlines, but at the time of this consultation has not yet submitted a completed Biological Assessment to initiate the section 7 consultation process.

As provided in Table 50 there are four categories of aquaculture gear used in the Greater Atlantic Region: floating gear, net pen, shell on bottom, and cage on bottom. Based on ESA section 7 consultations conducted in the Greater Atlantic Region between 2015 and January 2019,[15] we compiled a list of states that have aquaculture farms, and, per state, the number and type of aquaculture gear used (Table 51). One case in 2014 was also included due to its offshore location.

The species grown in various gear types include shellfish, finfish, and seaweeds (Table 50). Floating gear includes surface longlines, submerged longlines, and a floating upweller system. Aquaculture longlines are not the same as longline gear used in fisheries. In aquaculture, surface longlines consist of horizontal longline suspended on/near the surface of the water with buoy lines or poles at each end. Various types of cages or flip bags may be used to keep organisms inside an enclosed space. In deeper and higher energy locations, submerged longline are used. Their design consists of horizontal longlines suspended below the surface with moorings/marker buoys at each end. Some may have another mooring in the middle of their run. The longlines are suspended below the water surface and use a series of buoys to maintain the depth. This gear category also includes a floating upweller system (FLUPSY). This system is a dock or pier with

[15] Counts include experimental and/or gear that are no longer deployed.

tanks used to grow shellfish in open water while protecting them from predation. FLUPSY has a motor that pulls water through the bottom of the tanks. As the water moves through the system, it provides a continuous food supply to the shellfish by transporting algae.

Net pens are a type of enclosure culture and involve holding organisms captive within an enclosed space while maintaining a free exchange of water. They are enclosed on the bottom and sides by wooden, mesh or net screens. These types of gear are in direct contact with the surrounding environment. Shell on bottom refers to a technique used to grow shellfish, such as oysters, on the bottom of the ocean floor without cages. Shell on bottom also includes cases used for oyster bed restoration and maintenance, artificial oyster reefs creation, and spat collector installation. Cage on bottom also refers to a technique used to grow shells on the bottom of the ocean floor where cages are used.

Table 50: Examples of organisms grown by aquaculture gear type

Gear type	Examples of grown organisms
Floating gear	Kelp, mussels, oysters, scallops
Net pens	Fish (e.g., Atlantic salmon)
Shell on bottom	Oysters, clams, mussels
Cage on bottom	Oysters, clams

Aquaculture sites may use a combination of gear categories, referred to here as multimode. For instance, both cage on bottom and floating gear were used to grow oysters in the waters near Maryland, so this case was included in this "multimode" category.

Table 51: Aquaculture gear in the Greater Atlantic Region

State	Type of Aquaculture Gear					Total
	Floating Gear	Net Pen	Shell on bottom	Cage on bottom	Multimode	
ME	1	1	2	1	0	5
MA	10	0	1	3	0	15
CT	9	0	3	10	0	22
RI	1	0	0	1	1	3
NY	1	0	3	3	0	7
NJ	3	0	0	8	11	22
MD	7	0	115	33	8	163
VA	2	0	59	1	1	63
Total	34	1	183	60	21	299

5.1.3. *Hopper Dredging, Sand Mining, and Beach Nourishment*

The construction and maintenance of federal navigation channels and sand mining ("borrow") areas may result in take of sea turtles, shortnose sturgeon, and/or Atlantic sturgeon. There are several dredge types used in the action area. A hopper dredge uses pumps to force water and sediment up the dragarm and into the hopper. Hopper dredges may be equipped with screens (UXO screens) for unexploded ordinance on the intake. Cutterhead dredges have a rotating cutter apparatus surrounding the intake of a suction pipe and may be hydraulic or mechanical. Bucket and clamshell dredges are mechanical devices that use buckets to excavate dredge materials (NMFS 2019b). Dredging projects are authorized or carried out by the U.S. ACOE. In the action area, these projects are under the jurisdiction of the districts within the North Atlantic and South Atlantic Divisions.

Hard-shelled sea turtles may be injured by hopper dredges when the draghead is placed, impinged on the screen or entrained in the draghead. It is also possible that sea turtles may become entrained in other intake ports of these dredges. Sturgeon may become entrained during hopper or cutter head dredging or captured by mechanical dredges. Sediment suspension, blasting, and relocation associated with dredging projects may also impact protected species (NMFS 2019b). Relocation trawling may be undertaken to move sea turtles out of the area being dredged and placing them in an area outside of the dredge area.

NMFS has completed ESA section 7 consultations with the U.S. ACOE, NASA, and the U.S. Navy to consider the effects of these dredging, sand mining, and nourishment projects on ESA-listed species in the mid-Atlantic and Northeast (NMFS 2006, 2012a, c, d, 2014c, e, 2018f, 2019b, c, 2020a). Takes of sea turtles, shortnose sturgeon, and Atlantic sturgeon during relocation trawling activities are also included in the consultations and are described below. No takes of Atlantic salmon, giant manta rays, or large whales are anticipated to occur from these project activities.

A regional biological opinion on the U.S. ACOE's hopper dredging in the South Atlantic was completed in 2020. This South Atlantic Regional Biological Opinion (SARBO) (March 27, 2020) concluded that the proposed action would adversely affect, but not likely jeopardize the continued existence of 5 sea turtle species (North Atlantic DPS of green, South Atlantic DPS of green, Kemp's ridley, leatherback, and Northwest Atlantic DPS of loggerhead sea turtles), 6 sturgeon species (Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon, and shortnose sturgeon), giant manta ray, smalltooth sawfish, Johnson's seagrass, and 5 coral species (elkhorn, staghorn, lobed star, mountainous star, and boulder star coral). Anticipated take of sea turtles and sturgeon are included in the table below. In addition, the biological opinion estimates the take of 89 (0 lethal) giant manta rays over a 3-year period (NMFS 2020a).

Aside from commercial fishing and fisheries research activities, these dredging projects represent one of the largest sources of incidental take for sea turtles, shortnose sturgeon, and Atlantic sturgeon in the action area, and, potentially, one of the largest sources of lethal take. Active opinions covering dredging, beach nourishment, and shoreline restoration/ stabilization projects in the action area and the associated ITSs for sea turtles are presented below (Table 52).

Table 52: NMFS' biological opinions for dredging projects in the action area and the anticipated take of sea turtles and sturgeon

Project	Date of Opinion	Loggerhead	Kemp's ridley	Green	Leatherback	Atlantic Sturgeon	Life of Project
ACOE Deepening and Maintenance of the Delaware River Federal Navigation Channel	11/22/2019	37 (37)	3 (3)			1763 non-lethal combination of NYB DPS or shortnose; 116 lethal any DPS or shortnose; 21 lethal NYB DPS or shortnose;	2020-2070
						1.3% of each year class post yolk-sac larvae NYB DPS	
U.S. Navy; ACOE Maintenance Dredging of the Kennebec River FNP	10/25/2019					5 (5) GOM DPS	2019-2029
ACOE Atlantic Coast of Maryland Shoreline Protection Project	11/30/2006	22(22)	2 (2)				2008-2044
U.S. Navy Shoreline Restoration and Protection Project, JEB Little Creek/ Fort Story, VA Beach	7/13/2012	1 (1) loggerhead or Kemp's ridley				2 (2) GOM, NYB, CB, Carolina, or SA DPS	2012-2020
NASA Wallops Island Shoreline Restoration/ Infrastructure Protection Program	8/3/2012	9 (9) of which no more than 1 (1) may be a Kemp's ridley				2 (2) GOM, NYB, CB, Carolina, or SA DPS	2012-2062

Project	Date of Opinion	Loggerhead	Kemp's ridley	Green	Leatherback	Atlantic Sturgeon	Life of Project
ACOE NY and NJ Harbor Deepening	10/25/2012	1 (1) loggerhead or Kemp's ridley				1 (1) GOM, NYB, CB, Carolina, or SA DPS	50 years
ACOE Sea Bright Offshore Borrow Area Beach Nourishment	3/7/2014	8 (8) loggerhead or Kemp's ridley (up to 3 Kemp's total)				2 NYB DPS 1 CB, GOM, CA, SA DPS; 2 Any DPS	50 years
ACOE Sand borrow areas for beach nourishment and hurricane protection, offshore DE and NJ	6/26/2014	29	2	1			50 years
ACOE Dredging of Chesapeake Bay Entrance Channels and Beach Nourishment	10/15/2018	1685 (748 lethal)	341 (66 lethal)	56 (18 lethal)		118 (18 lethal) GOM DPS; 418 (68 lethal) NYB DPS; 179 (29 lethal); 123 (23 lethal) CB DPS; 60 (10); Carolina DPS; SA DPS;	50 years
		Relocation Trawling: up to 938 captures (37 lethal) of loggerheads, 275 captures (11 lethal) of Kemp's ridleys, and 37 captures (2 lethal) of green sea turtles 0				Relocation Trawling: 700 (0 lethal) total; Of these, ≤ 100 GOM, ≤ 350 NYB DPS, ≤ 100 CB DPS; ≤50 Carolina DPS; ≤150 SA DPS	
SARBO	3/27/20	5,484 (214 lethal) and 65 lost egg clutches over 3 years	1,456 (116 lethal) and 1 lost egg clutch over 3 years	860 (118 lethal) and 3 lost egg clutches over 3 years	369 (4 lethal) and 6 lost egg clutches over 3 years	2 (1 lethal) GOM DPS; 39 (5 lethal) NYB DPS; 105 (14 lethal) CB DPS; 366 (47 lethal) Carolina DPS; 572 (73 lethal) SA DPS over 3 years	

5.1.4. ***Research and Other Permitted Activities***

Within the action area, NMFS has completed section 7 consultation on research (either conducted or funded by federal agencies) and other federally-permitted activities that may adversely affect ESA-listed marine mammals, sea turtles, and fish. Below, a description of recently completed section 7 consultations on research and other permitted activities are provided.

NEFSC Fisheries and Ecosystem Research

NEFSC scientists conduct fishery-independent research onboard NOAA-owned and operated vessels or on chartered vessels in coastal, estuarine, and marine waters of the U.S. Atlantic Ocean from Maine to Florida. A number of cooperative research projects also occur within the action area each year. The cooperative research projects are designed to address emerging needs of the fishing industry, for information about particular species, or for modifications to fishing gear to address conservation concerns. Grant programs that fund cooperative research along the U.S. Atlantic coast include the Cooperative Research Partners Program, Northeast Consortium Cooperative Research Program, Commercial Fisheries Research Foundation, and the Research Set-Aside (RSA) Program. A major research initiative is the (NEAMAP nearshore trawl surveys. These fishery surveys are conducted every spring and fall by the Virginia Institute of Marine Science (VIMS) in shallow (up to 120 feet), nearshore waters from Cape Hatteras, North Carolina to Montauk, New York. Those surveys are similar in design and are meant to complement the annual NEFSC spring and fall bottom trawl surveys, which are conducted in deeper waters of the U.S. Atlantic.

NEFSC-conducted or funded fisheries and ecosystem surveys that are known to interact with sea turtles, shortnose and Atlantic sturgeon, and Atlantic salmon include those that utilize bottom trawl, gillnet, and longline gear. Sea turtles have been caught in the following NEFSC survey programs: Cooperative Atlantic States Shark Pupping and Nursery (COASTSPAN) gillnet and longline surveys, Spring and Fall NEFSC Bottom Trawl Surveys, Spring and Fall NEAMAP trawl surveys, and Apex Predators longline surveys. Atlantic sturgeon have been caught during the NEFSC bottom trawl surveys and the spring and fall NEAMAP bottom trawl surveys. A few short-term cooperative research projects have also captured Atlantic sturgeon. All observed catches of Atlantic salmon during NEFSC research activities have occurred in bottom trawls.

In June 2016, NMFS completed a programmatic biological opinion (NMFS 2016b) on all fisheries and ecosystem research activities to be conducted and funded by the NEFSC from June 2016 to June 2021. Based on the information presented in the opinion, we anticipate that these fisheries and ecosystem research projects, over the 5-year period, will result in the capture of:

- up to 85 Northwest Atlantic DPS of loggerhead sea turtles (ten lethal);
- up to 95 Kemp's ridley sea turtles (15 lethal);
- up to 10 North Atlantic DPS of green sea turtles (non-lethal);
- up to 10 leatherback sea turtles (five lethal);
- up to 10 shortnose sturgeon (one lethal);
- up to 595 Atlantic sturgeon (30 lethal)
    - up to 308 from the New York Bight DPS (15 lethal),
    - up to 130 from the South Atlantic DPS (seven lethal),
    - up to 70 from the Chesapeake Bay DPS (four lethal),
    - up to 60 from the Gulf of Maine DPS (three lethal),

    - up to 14 from the Carolina DPS (one lethal),
    - up to 13 Canadian origin (non-listed); and
- up to five Gulf of Maine DPS Atlantic salmon (two lethal).

USFWS Funded State Fisheries Surveys

Under the Dingell-Johnson Sport Fish Restoration Grant program and State Wildlife Grant programs, the USFWS Region 5 provides an annual apportionment of funds to 13 Northeast states and the District of Columbia. Vermont and West Virginia are the only two Northeast states that do not use these funds to conduct surveys in marine, estuarine, or riverine waters where ESA-listed species under NMFS jurisdiction are present. The 11 other states (Maine, New Hampshire, Massachusetts, Connecticut, Rhode Island, New York, New Jersey, Pennsylvania, Delaware, Maryland, Virginia) and the District of Columbia are anticipated to carry out a total of 113 studies, mostly on an annual basis, under these grant programs. There are several broad categories of fisheries surveys including: hook and line; long line; beach seine; haul seine; bottom trawl; surface trawl; fishway trap; fish lift; boat, backpack, and/or barge electrofishing; fyke net; dip net; gill net; push net; hoop net; trap net; cast net; plankton net; pound net; and fish and/or eel trap/pot. These surveys occur in rivers, bays, estuaries, and nearshore ocean waters of those 11 states and the District of Columbia.

We completed a biological opinion on this grant program in October 2018. It bundled together 12 independent actions carried out by the USFWS (i.e., awarding of each grant fund to each state or district is an independent action) and provided an ITS by activity and a summary by state. Overall, we anticipate that the surveys described in the opinion, which will be carried out by the states from 2018 to 2022 will result in the capture of:

- Up to 37 sea turtles;
- Up to 55 shortnose sturgeon (including eight in beach/haul seine studies, one in the Westfield River fish passage facility, ten in bottom trawl studies, two in gill net studies, and 34 interactions during electrofishing activities); and,
- Up to 427 Atlantic sturgeon (including two in beach/haul seine studies, 266 in bottom trawl studies, 158 in gill net studies, and one interaction during electrofishing activities).

The only mortalities that we anticipate to occur are six Atlantic sturgeon (originating from any of the five DPSs) during gillnet surveys carried out by New York, New Jersey, Maryland, and Virginia.

Section 10(a)(1)(A) Permits

NMFS has issued research permits under section 10(a)(1)(A) of the ESA, which authorizes activities for scientific purposes or to enhance the propagation or survival of the affected species. The permitted activities do not operate to the disadvantage of the species and are consistent with the purposes of the ESA, as outlined in section 2 of the Act. Active section 10(a)(1)(A) permits for sea turtles and Atlantic sturgeon are provided in Table 53 and Table 54, respectively. No section 10 permits authorizing serious injury or mortality of marine mammals are currently active.

We searched for research permits on the NMFS' online application system for Authorization and Permits for Protected Species. The search criteria used confined our search to active permits that include take of sea turtles and Atlantic sturgeon within the Atlantic Ocean. Search criteria also limited the search to research states from Florida to Maine. However, many research activities include both the Gulf of Mexico and the Atlantic Ocean, and the requested take did not always specify the waters where take would occur. Thus, some of the requested sea turtle take in Table 53 below include take for activities outside (i.e., in the Gulf of Mexico) the action area.

The requested take reported in Table 53 and Table 54 only includes take authorized under section 10(a)(1)(A) of the ESA. Permits related to stranding and salvage programs are described in that section. In addition, several research projects included take authorized under other authority, e.g., under section 7 of the ESA. These takes are included elsewhere in this Opinion and, therefore, are not included here to avoid double counting of take provided under the ESA.

Table 53: Active section 10(a)(1)(A) permits authorizing take of sea turtles for scientific research

Permittee	File #	Project	Area	Sea Turtle Takes	Research Period
NMFS Southeast Fisheries Center	16733	Demographic and life history studies of sea turtle populations in the Atlantic Ocean, Gulf of Mexico, Caribbean Sea, and tributaries.	Atlantic Ocean DE, MD, NC, NJ, NY, VA	Sample annually 925 loggerheads, 560 greens, 455 Kemp's ridleys, 65 hawksbills, 60 leatherbacks, 10 olive ridleys, and 24 unidentified/hybrid hardshells. In addition, 2620 loggerheads, 565 greens, 615 Kemp's ridleys, 287 hawksbills, 665 leatherbacks, 37 olive ridleys, and 2170 unidentified hardshells observed during aerial, vessel, and acoustic surveys annually	5 years, 08/13/2013 to 08/13/2019
NMFS Northeast Fisheries Science Center	17225	Conservation engineering to reduce sea turtle and Atlantic sturgeon bycatch in fisheries in the Northeast Region	U.S. locations including offshore waters	Over the course of the permit: Northern area (NH to NC): 8 green, 8 Kemp's, 8 leatherbacks, 26 loggerheads; no lethal (capture covered under other authorities) over the course of the permit Southern area (SC to GA): 10 green, 8 hawksbill, 62 Kemp's, 8 leatherback, 148 loggerhead. Incidental mortality: 6 unidentified	5 years, 01/01/2017 to 12/31/2021
Coonamessett Farm Foundation, Inc.	18526	Understanding the impact of the sea scallop fishery on loggerhead sea turtles through satellite tagging	Western Atlantic waters/Mid-Atlantic Bight from Cape Hatteras, North to NY LIS; and from coastal waters to the shelf break	Maximum of 200 loggerhead (20 captured and sonic tagged/80 approached unsuccessfully/100 observed and tracked with ROV). Non-Target species: 2 Kemps ridley, green (captured and sonic tagged); 8 Kemp's ridley, green, leatherback, and/or unidentified (approached unsuccessfully); 20 Kemp's ridley, green, leatherback, and unidentified (observed and tracked with ROV) sea turtles per year.	5 years, 05/27/2015 to 05/31/2020
Atlantic Marine Conservation Society	20294	Marine mammal and sea turtle surveys to assess seasonal abundance and distribution in the Mid-Atlantic region.	Atlantic Ocean Focal area: New York Bight and surrounding waters; Research can occur off MA through N	Aerial Surveys: 125 Kemp's ridley, leatherback 85, 450 loggerhead, 450 unidentified.	5 years, 06/02/2017 to 06/01/2022

Permittee	File #	Project	Area	Sea Turtle Takes	Research Period
NMFS Southeast Fisheries Center (SEFSC)	20339	Application for a scientific research and enhancement permit under the ESA; development and testing of gear aboard commercial fishing vessels.	**Project A**: Turtle Excluder Device (TED) Evaluations in Atlantic and Gulf of Mexico Trawl Fisheries **Project B** research will occur solely within longline commercial fisheries where the incidental capture is already authorized by an existing ESA section 7 biological opinion.	**Project A**: annual take numbers: 220 (70 of these to include capture) loggerheads, 105 (25 captures) Kemp's ridleys, 85 (20 captures) leatherbacks, 50 (15 captures) greens, 30 (10 captures) hawksbills, 30 (10 captures) olive ridleys, and 75 (25 of captures) unidentified/hybrid turtles. A subset of these animals will be captured during trawl research authorized under this permit as noted in the parentheses; the rest of the turtles will be captured within fisheries managed by federal authority. **Project B**, annual take numbers: 30 loggerheads, 10 Kemp's ridleys, 30 leatherbacks, 10 greens, 10 hawksbills, 10 olive ridleys, and 10 unidentified/hybrid turtles. Incidental mortality: 2 green, 1 hawksbill, 2 Kemp’s, 1 leatherback, 3 loggerhead, and 1 olive ridley over the course of the permit.	5 years, 05/23/2017 to 05/31/2022
Virginia Aquarium and Marine Science Center	20561	2018 renewal request for Virginia Aquarium sea turtle research permit	Atlantic Ocean, Long Island Sound, Delaware Bay, Chesapeake Bay, North Carolina Sounds/Estuarine and ocean waters from shore to the continental shelf off of NY through northern NC including inshore brackish waters.	Up to 72 turtles annually (25 green, 22 Kemp's ridley, 25 loggerhead) captured, sampled, and tagged. Up to one leatherback sea turtle may be opportunistically captured, sampled, and tagged. 18 turtles will be captured under other authority annually (5 green, 8 Kemp’s, and 5 loggerhead)	10 years, 08/24/2018 to 09/30/2027

Permittee	File #	Project	Area	Sea Turtle Takes	Research Period
NMFS Northeast Fisheries Science Center (NEFSC)	21233	Demographic and life history studies of sea turtle populations in the Atlantic Ocean, Gulf of Mexico, Caribbean Sea, and tributaries	Project: 1) Cape Lookout Bight, NC 2) Gulf Stream Surveys, NC 3) North Carolina In-water Studies 4) Leatherback Studies, GOM and Atlantic 5) Biscayne National Park and Chassahowitzka National Wildlife Refuge 6) Florida Keys National Marine Sanctuary 7) Trawl captures in Gulf of Mexico 8) Programmatic In-water Studies	**Project 1, 2, and 3**: 555 loggerheads, 390 greens, 18 leatherbacks, 360 Kemp's ridleys, 21 hawksbills, 11 olive ridleys, and 18 unidentified hardshell/hybrids **Project 4**: 50 total leatherbacks captured and satellite tagged per year (25 GOM, 25 Atlantic). Up to 50 leatherbacks observed/pursued during vessel surveys but not captured during unsuccessful capture attempts. Up to 50 leatherbacks observed/pursued during aerial surveys but not captured. Up to 25 leatherbacks captured under other authority (e.g., pelagic longline fishery bycatch) **Project 5**: Up to 140 green turtles, 22 hawksbills, 85 Kemp's ridley and 115 loggerheads captured, processed and released in Biscayne National Park or Chassahowitzka annually. Up to 100 green, 50 loggerhead, and 20 Kemp's ridley turtles pursued without capture during vessel surveys and capture efforts annually. **Project 6:** Up to 60 greens, 35 hawksbills, 15 Kemp's ridleys and 30 loggerheads captured, processed, and released in the Florida Keys annually. Up to 5 hawksbills pursued without capture during survey and capture efforts. **Project 7:** 10 greens, 2 hawksbills, 10 Kemp's ridleys, 10 loggerheads, and 2 leatherbacks captured annually in the Gulf of Mexico **Project 8**: Up to 60 green turtles, 25 hawksbills, 60 Kemp's ridley, and 60 loggerheads captured, processed, and released annually. Up to 25 green turtles, 10 hawksbills, 25 Kemp's ridley, 25 leatherbacks, and 50 loggerheads processed and released after being legally captured under another authority (e.g., commercial fisheries, other section 10 permits) annually. **All:** Incidental mortality over the life of the permit (all capturing and processing) of 2 loggerheads, 2 Kemp's ridleys, 2 greens, 1 leatherback, 1 olive ridley, and 1 hawksbill	10 years, 08/07/2018 to 09/30/2027

Table 54: Active section 10(a)(1)(A) permits authorizing take of Atlantic sturgeon for scientific research

Permittee	File #	Project	Area	Atlantic Sturgeon Takes	Research Period
NMFS Northeast Fisheries Science Center	17225	Conservation engineering to reduce sea turtle and Atlantic sturgeon bycatch in fisheries in the Northeast Region	Western Atlantic waters (Massachusetts through Georgia, including inside COLREGs lines).	Northern area (NH to NC): Non-lethal – 223 sub-adult/adult (capture under other authority) over the course of the permit Southern area (SC to GA): Non-lethal: 204 juvenile/sub-adult/adult over the course of the study Incidental mortality: 6 juvenile/sub-adult/adult over the course of the permit	5 years, 01/01/2017 to 12/31/2021
Connecticut Department of Energy and Environmental Protection, Marine Fisheries	19641	Application to conduct scientific research and monitoring of shortnose sturgeon (*Acipenser brevirostrum*) and Atlantic sturgeon (*A. oxyrinchus oxyrinchus*) in Connecticut Waters and Long Island Sound.	CT waters	Non-lethal - 300 adult, sub-adult, and juvenile annually Incidental mortality: 1 adult/ sub-adult and 1 juvenile annually	10 years, 06/20/2016 to 03/31/2027
University of Maine	20347	Sturgeon of the Gulf of Maine	Gulf of Maine	100 (1 lethal) adults and sub-adults annually 20 (1 lethal) juveniles annually	10 years; 3/31/2017-3/31/2027
Stony Brook University	20351	Atlantic and shortnose sturgeon population dynamics and life history in NY coastal marine and riverine waters	New York (Long Island Sound), New Jersey, Delaware	685 (up to 30 lethal) juveniles, sub-adults, adults annually	10 years; 02/27/2016-03/31/2027
Delaware State University	20548	Reproduction, habitat use, and inter-basin exchange of Atlantic and shortnose sturgeon in the Mid-Atlantic	Coastal New York, New Jersey, Delaware	600 (up to 1 lethal) juvenile, sub-adult, and adult annually	10 years; 03/31/2017-03/31/2027
NMFS, Office of Protected Resources	19642	Characterizing juvenile, sub-adult, and adult life stages of endangered Atlantic and shortnose sturgeon in the York, Rappahannock, Potomac, and Susquehanna Rivers, their tributaries, the Chesapeake Bay, and the Atlantic Coast.	Atlantic Ocean	200 non-lethal; any life stage (capture under other authority over the course of the permit)	5 years; 07/01/2016-06/30/2021

Scientific research on ESA-listed Atlantic salmon has been authorized under the USFWS' endangered species blanket permit (No. 697823) under section 10(a)(1)(A), and covers a number of research projects carried out by NMFS and other research partners contracted by NMFS (e.g., University of Maine). However, the USFWS is anticipating re-structuring their permits. Specifically, the USFWS plans to issue new permits to cover only research directly under the NMFS' direct supervision. The USFWS is also planning to issue separate permits for different research activities conducted through other agencies or partners such as U.S. Geological Survey, Maine Department of Marine Resources (MDMR), and the University of Maine. This will provide a more efficient way of tracking individual take and will allow the USFWS to have a better understanding of ongoing research and level of take associated with these activities through annual reporting requirements.

USFWS is also authorized to conduct the conservation hatchery program at the Craig Brook and Green Lake National Fish Hatcheries. The mission of the hatcheries is to raise Atlantic salmon parr and smolts for stocking into selected Atlantic salmon rivers in Maine. Over 90 percent of adult returns to the GOM DPS are currently provided through production at the hatcheries. Approximately 600,000 smolts are stocked annually in the Penobscot River. The hatcheries provide a significant buffer from extinction for the species.

NMFS currently cooperates in research on Atlantic salmon in the Penobscot River to document changes in fish populations resulting from both the removal of the Veazie and Great Works projects, as well as the construction of the fish bypass at the Howland project. The study uses boat electrofishing techniques to document baseline conditions in the river prior to construction at the dams. Following dam removal and construction of the fish bypass, researchers will re-sample the river. This research will provide a better understanding of how dam removals and fish bypasses benefit Atlantic salmon.

NMFS also is monitoring biomass and species composition in the estuary to look at system-wide effects of dam removal projects. Although these activities will result in some take of Atlantic salmon, these takes are authorized by the existing ESA permit. The information gained from these activities will be used to further salmon conservation actions in the GOM DPS.

Section 10(a)(1)(B) Permits

Section 10(a)(1)(B) of the ESA authorizes NMFS, under some circumstances, to permit non-federal parties to take otherwise prohibited fish and wildlife if such taking is "incidental to, and not the purpose of carrying out otherwise lawful activities" (50 CFR 217-222). As a condition for issuance of a permit, the permit applicant must develop a conservation plan that minimizes negative impacts to the species. There are currently three active section 10(a)(1)(B) permits in the action area (Table 55). Active permits and permit applications are posted online for all species as they become available at https://www.fisheries.noaa.gov/national/endangered-species-conservation/incidental-take-permits.

Table 55: Active section 10(a)(1)(B) permits

Permittee	File #	Project	Area	Annual Endangered Species Takes	Dates
NC Department of Environment and Natural Resources, Division of Marine Fisheries	18102	Inshore anchored gillnet shallow water fishery	NC state waters; Management unit: A - Albemarle, Currituck, Croatan, Roanoke B - Pamlico Sound and the northern portion of Core Sound C - Pamlico, Pungo, Bay, and Neuse river drainages D - southern Core Sound, Back Sound, Bogue Sound, North River, and Newport River E - Atlantic Intracoastal Waterway and adjacent sounds and the New, Cape Fear, Lockwood Folly, White Oak, and Shallotte rivers	Large and small mesh fisheries combined Atlantic sturgeon Carolina DPS Total Lethal: 138 per year Total Non-lethal: 2,124 per year Unit A: 110 lethal and 2,063 non-lethal per year; Unit B: 11 lethal and 27 non-lethal per year; Unit C: 9 lethal and 10 non-lethal per year; Unit D: 4 lethal and 12 non-lethal per year; Unit E: 4 lethal and 12 non-lethal per year Atlantic sturgeon other DPS Total Lethal: 31 per year Total Non-lethal: 634 Unit A: 31 lethal and 618 non-lethal; Unit B: 0 lethal and 12 non-lethal; Unit C: 0 lethal and 4 non-lethal Unit D: no take; Unit E: no take	2014-2024
NC Department of Environment and Natural Resources, Division of Marine Fisheries	16230	Inshore anchored gillnet shallow water fishery	State waters of North Carolina: inshore waters 6 management units	Combined for small and large mesh Green sea turtle Lethal: 165 per year; Non-lethal: 330 per year; Either: 18 per year* Hawksbill sea turtle Lethal: n/a; Non-lethal: n/a; Either: 8 per year* Kemp's ridley sea turtle Lethal: 49 per year; Non-lethal: 98 per year; Either: 12 per year* Leatherback sea turtle Lethal: n/a; Non-lethal: n/a; Either: 8 per year* Loggerhead sea turtle Lethal: n/a; Non-lethal: n/a; Either: 24 per year* Any species Lethal: n/a; Non-lethal: n/a; Either: 8 per year* * Observed take, rest are estimated take based on observed take. N/A if not enough observed take occurred to provide an estimate.	2013-2023
GA Department of Natural Resources	16645	Commercial shad fishery conservation plan	Atlamaha River, Savannah River, Ogeechee River	Atlamaha: 140 Atlantic sturgeon (2.3% mortality) Savannah: 50 Atlantic sturgeon (2.3% mortality) Ogeechee: 10 Atlantic sturgeon (2.3% mortality)	2013-2022

MMPA Incidental Harassment Authorizations and Letters of Authorization
Under section 101(a)(5) of the MMPA, certain incidental taking, via harassment, of a small number of marine mammals during an activity (other than commercial fishing) is allowed through the issuance of Incidental Harassment Authorizations (IHAs) or Letters of Authorization (LOAs).[16] [17] IHAs are issued for actions that have the potential to result in harassment of marine mammals only (i.e., injury or disturbance) and are effective for up to one year. LOAs are issued for actions that have the potential to result in harassment of marine mammals only (i.e., injury or disturbance) and are planned for multiple years, or have the potential to result in serious injury or mortality to the marine mammal species; these authorizations are effective for up to five years.[18]

The types of activities receiving IHAs and LOAs may involve acoustic harassment or habitat disturbance from yacht races, seismic surveys, exploratory drilling surveys, bridge construction, fireworks displays, sonar testing, Navy training and testing programs, and lighthouse restorations, among others. The types of authorized takes include behavioral responses, as well as injuries and mortalities. Currently, no LOAs allow serious injuries and mortalities for ESA-listed cetaceans.

Current and past applications are available for public review at https://www.fisheries.noaa.gov/permit/incidental-take-authorizations-under-marine-mammal-protection-act. Authorizations that are in process for activities in the action area include military readiness (1), other energy (i.e., renewable and liquefied natural gas) (6), construction (1), and fisheries research (1). Active authorizations in the action area include military readiness (3), other energy (9), construction (7), and fisheries and biological research (3). Most of these projects only affect marine mammals that are not listed under the ESA. For those activities that may affect ESA-listed species, NMFS has consulted under section 7 of the ESA on the issuance of the IHAs and LOAs (NMFS 2016b, 2018b, c).

5.1.5. ***Operation of Vessels Carrying out Federal Actions***
Potential sources of adverse effects to whales, sea turtles, and Atlantic sturgeon from federal vessel operations in the action area include operations of the U.S. Navy (USN), U.S. Coast Guard (USCG), Bureau of Ocean Energy Management (BOEM), Maritime Administration (MARAD), Environmental Protection Agency (EPA), NOAA and ACOE vessels. NMFS has previously conducted formal consultations with the Navy and USCG on their vessel-based operations. NMFS has also conducted section 7 consultations with BOEM and MARAD on vessel traffic related to energy projects and has implemented conservation measures. Through the section 7 process, where applicable, NMFS has and will continue to establish conservation measures for federal vessel operations to avoid or minimize adverse effects to listed species.

[16] The MMPA defines harassment as Level A or Level B. Level A harassment has the potential to injure a marine mammal or marine mammal stock. Level B harassment has the potential to disturb a marine mammal or marine mammal stock by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering, but which does not have the potential to injure a marine mammal or marine mammal stock (MMPA section 3(18)(C) and 3(18)(D)).
[17] Note that incidental take of marine mammals during commercial fishing operations is covered separately under the Marine Mammal Authorization Program.
[18] The MMPA defines "serious injury" as any injury that will likely result in mortality (50 CFR § 216.3).

5.1.6. ***Military Operations***

NMFS has completed consultations on individual Navy and USCG activities (see https://www.fisheries.noaa.gov/national/endangered-species-conservation/biological-opinions). In the U.S. Atlantic, the operation of USCG boats and cutters are estimated to take no more than one individual sea turtle, of any species, per year (NMFS 1995, 1998).

In 2018, NMFS issued a biological opinion on the U.S. Navy Atlantic Fleet's military readiness training and testing activities and the promulgation of regulations for incidental take of marine mammals (NMFS 2018b). The action area includes the Gulf of Mexico and the western Atlantic. NMFS concluded that the action is not likely to jeopardize the continued existence of fin, North Atlantic right, sei, or sperm whales, green (North Atlantic DPS), loggerhead (Northwest Atlantic DPS), Kemp's ridley, or leatherback sea turtle ;Atlantic salmon (Gulf of Maine DPS), or Atlantic sturgeon (Gulf of Maine, New York, Chesapeake Bay, Carolina, South Atlantic DPS). NMFS anticipated the following takes from harm due to exposure to impulsive and non-impulsive acoustic stressors annually: 6 fin whales, 6 green (North Atlantic DPS), 5 Kemp's ridley, 97 loggerhead, and 24 leatherback sea turtles. In addition, two lethal takes of loggerhead sea turtles were anticipated. Other marine mammal and sea turtle takes from these stressors are expected to be in the form of harassment. Takes from vessel strikes were anticipated to include the lethal take annually of 1 fin, 1 sei, and 1 sperm whale, 55 green, 20 Kemp's ridley, 75 loggerhead, and 5 leatherback sea turtles. Four green, 4 hawksbill, 5 Kemp's ridley, 11 loggerhead, and 3 leatherbacks were anticipated have non-lethal injuries. For vessel strikes, the opinion also anticipates the take of no more than 6 Atlantic sturgeon (up to 1 from the Gulf of Maine DPS, 1 from the New York Bight DPS, 6 from the Chesapeake Bay DPS, 6 from the Carolina DPS, and 1 from the South Atlantic DPS) combined from all DPSs over a 5-year period. The ITS did not specify the amount or extent of take of ESA-listed fish, but rather used a surrogate expressed as a distance to reach effects in the water column with injury and sub-injury from acoustic stresses. In addition to takes due to acoustic stressors and vessel strikes, take was estimated to occur as a result of small and large ship shock trials. Thirty-six fin, 7 sei, and 6 sperm whales; 2 green (North Atlantic DPS), 1 hawksbill, 4 Kemp's ridley, 41 loggerhead, and 17 leatherback sea turtles are anticipated to be harmed over the course of the action. In addition, 2 lethal takes of loggerheads are estimated. Other takes due to ship shock trials included in the ITS are in the form of harassment. In addition, takes of blue whales, Bryde's whale – Gulf of Mexico subspecies, and Gulf sturgeon were also anticipated.

5.1.7. ***Offshore Oil and Gas***

BOEM oversees leasing of outer continental shelf (OCS) energy and mineral resources; this includes administering the leasing program for OCS oil and gas resources. Currently, BOEM is working under the 2017-2022 National OCS Program, but has initiated a process to develop a program for 2019-2024. No lease sales are scheduled for the Atlantic OCS under the current plan. Under the proposed plan, BOEM has divided the Atlantic OCS into four planning areas: North Atlantic, Mid Atlantic, South Atlantic, and Straits of Florida Planning Areas. The action area overlaps with all four Planning Areas. The draft proposed program for leasing, published in 2018, calls for leasing in the North Atlantic Planning Area in 2021, 2023 and 2025, in the Mid Atlantic Planning Area in 2020, 2022 and 2024, in the South Atlantic Planning Area in 2020, 2022, and 2024; and in the Straits of Florida Planning Area in 2023. At this time, the proposed program has not been approved or finalized.

Geophysical and/or geotechnical surveys to identify hydrocarbon resources would occur if leasing is being pursued in the action area. On November 30, 2018, NMFS issued five IHAs under the MMPA to incidentally harass marine mammals to companies proposing to conduct geophysical surveys, including the use of air guns, in support of hydrocarbon exploration in the Atlantic Ocean (83 FR 63268, December 7, 2018). These were issued to five companies that provide services such as geophysical data acquisition, to the oil and gas industry. No mortality of any individuals is anticipated. Twelve fin whales are expected to experience harm; all other exempted take of marine mammals is in the form of harassment (e.g., ,behavioral disturbance) due to exposure to underwater. NMFS prepared a biological opinion that considered the effects of these activities on ESA-listed species in the action area. In addition to the incidental take of ESA-listed marine mammals, the biological opinion estimated the incidental take of Northwest Atlantic DPS of loggerhead, green, Kemp's ridley, and leatherback sea turtles. This take was in the form of harassment through behavioral responses and temporary hearing threshold shifts. The opinion did not anticipate the death of any individual cetacean or sea turtle exposed to seismic survey activities. The action was also determined not likely to adversely affect any DPS of Atlantic sturgeon, giant manta rays, oceanic white-tip sharks, hawksbill sea turtles (NMFS 2018c). The activities included in the IHA and biological opinion were scheduled to be completed by November 30, 2019.

5.1.8. ***Offshore Renewable Energy***

BOEM is responsible for overseeing offshore renewable energy development in federal waters pursuant to the 2009 final regulations for the OCS Renewable Energy Program, which was authorized by the Energy Policy Act of 2005 (EPAct). These regulations provide a framework for issuing leases, easements, and rights-of-way for OCS activities that support production and transmission of energy from sources other than oil and natural gas (i.e., offshore wind and hydrokinetic projects).

Under the renewable energy regulations (30 CFR § 585), the issuance of leases and subsequent approval of wind energy development on the OCS is a staged decision making process and occurs over several years with each step having varying impacts to marine and/or terrestrial resources. The process follows these general steps: lease issuance, site assessment plan approval, construction and operation plan (COP) review/approval including permitting with cooperating agencies. NMFS has carried out programmatic consultations with BOEM to address the effects of issuance of leases and site assessment activities associated with offshore wind energy. These consultations consider effects from of a suite of activities on listed sturgeon, sea turtles, and marine mammals. The expected effects of the actions considered result from temporary exposure to acoustic sources (e.g., geophysical survey equipment) that may result in behavioral disturbance of individuals. No take in the form of injury or mortality is anticipated.

As of June 2020, BOEM has issued 15 leases for commercial offshore wind energy development along the U.S. Atlantic coast (North Carolina to Massachusetts) and 1 lease for a research site (off the coast of Virginia) where two turbines were installed, the first in federal waters (see https://www.boem.gov/Lease-and-Grant-Information/). A variety of site assessment activities have been completed or are ongoing within the lease blocks, including geophysical and geotechnical surveys and the installation of meteorological buoys or towers at some sites. The effects of these activities on ESA-listed species were considered in the programmatic consultation above. No injury or mortality of any ESA-listed species have been reported to date.

In order for an offshore wind facility on the OCS to be built, BOEM must approve a COP; proposed approval of the COP is the federal action that triggers review under NEPA and ESA section 7 consultation. Generically, effects to be considered include (but are not limited to) noise (pile driving, vessels, surveys), vessel strikes, habitat disturbance/loss, avoidance/displacement from the area, and electromagnetic fields.

In 2014, NMFS conducted a formal consultation on the effects of Deepwater Wind Block Island, LLC's and Deepwater Wind Block Transmission, LLC's proposals to construct and operate the Block Island Wind Farm. No injury of mortality of sea turtles was anticipated. Behavioral disturbance of (harassment) of loggerhead, leatherback, Kemp's ridley, and green sea turtles was anticipated due to exposure to disturbing levels of noise during pile driving. Temporary, short-term behavioral effects due to exposure to underwater noise was also anticipated for Atlantic sturgeon, but NMFS was unable to estimate the number of animals affected. Incidental take of 228 fin and 11 North Atlantic right whales due to harassment was also exempted and an IHA was issued (NMFS 2014a).

In 2020, NMFS concluded a formal consultation on the construction, operation, maintenance, and decommissioning of the Vineyard Wind Offshore Energy Project (NMFS 2020f). Vineyard Wind's proposed activity would occur in the northern portion of the 166,886 acre (675 square km) Vineyard Wind Lease Area, also referred to as the wind development area. Under the maximum impact scenario, pile driving during construction is expected to result in harassment of 20 North Atlantic right, 34 fin, 5 sperm, and 4 sei whales and 3 Northwest Atlantic DPS of loggerhead, 1 North Atlantic DPS of green, 1 Kemp's ridley, and 7 leatherback sea turtles. The pile driving is also expected to result in injury (permanent threshold shift) of 5 fin and 2 sei whales. M/SI of the 17 Northwest Atlantic DPS of loggerhead, 2 North Atlantic DPS of green, 2 Kemp's ridley, and 18 leatherback sea turtles is also anticipated due to vessel strikes. The biological opinion also includes estimated levels of take under other scenarios in which the project installs fewer turbines of larger capacity, if such turbines are available, and fewer electrical service platforms (NMFS 2020f).

5.2. Non-federally Regulated Fisheries

Several fisheries for species not managed by a federal FMP occur in state waters of the action area. In addition, unmanaged fisheries (e.g., hagfish) occur in federal waters. The amount of gear contributed to the environment by these fisheries is currently unknown. In most cases, there is limited observer coverage of these fisheries, and the extent of interactions with ESA-listed species is difficult to estimate. Sea turtles, large whales, Atlantic and shortnose sturgeon, and Atlantic salmon may be vulnerable to capture, injury, and mortality in a number of these fisheries. Captures of loggerhead, Kemp's ridley, green, and leatherback sea turtles (Murray 2007, 2008, 2009a, b, 2013, 2015b, 2018, 2020, Murray and Orphanides 2013, NMFS SEFSC 2001, 2009, Warden 2011a, b) and Atlantic sturgeon (ASSRT 2007, NMFS 2011a) in these fisheries have been reported through state reporting requirements, research studies, VTRs, NMFS NEFSC observer programs, and anecdotal reports.

Interactions with large whales have been documented in fishing gear in state waters. The MMPA List of Fisheries (LOF) evaluates commercial fisheries annually and classifies them by the level of incidental marine mammal death and serious injury. Category I fisheries are those with frequent incidental death or serious injury. Category II are those with occasional incidental death or serious injury. Fisheries in these categories are required to carry observers when requested.

Category III fisheries have a remote likelihood of or no known incidental death or serious injury of marine mammals. Fisheries may be classified in a particular category due to interactions with non-ESA-listed marine mammals.

Large whales are susceptible to entanglement in trap/pot and gillnet gear. Johnson et al. (2005) noted that any part of the gear (buoy line, groundline, floatline, and surface system line) creates a risk for entanglement. As described below, trap/pot and gillnet gear are used in several state and unregulated fisheries. These interactions can occur when and where large whales overlap with commercial or recreational fishing gear, including in state and unregulated fisheries. In the Atlantic, fisheries that have been classified as Category I based on interactions with ESA-listed whales include the Northeast sink gillnet (North Atlantic right whale) and Northeast/mid-Atlantic lobster trap/pot (North Atlantic right whale). Category II fisheries include the Southeastern U.S. Atlantic shark gillnet fishery (North Atlantic right whale) and the Atlantic mixed species trap/pot (fin whale). There are state fishery components of the Northeast sink gillnet, Northeast/mid-Atlantic lobster trap/pot, and the Atlantic mixed species trap/pot. Target species in the Atlantic mixed species trap/pot fishery include both federally regulated and non-federally regulated fisheries. Target species include hagfish, shrimp, conch/whelk, red crab, Jonah crab, rock crab, black sea bass, scup, tautog, cod, haddock, pollock, redfish, white hake, spot, skate, catfish, stone crab, and cunner. In the southeast, gillnet is the primary gear for vessels directing on small coastal sharks. The ALWTRP closes the Southeast U.S. Restricted Area North from November 15-April 15 and the Southeast U.S. Restricted Area South from December 1 – March 31. These areas are off Florida. There is an exemption for shark and Spanish mackerel gillnets in the south area if certain requirements are met and, for mackerel, during certain times. There are also weak link, anchoring, and other gear gillnet requirements in the southeast that apply to both state and federal waters.

Using the analysis presented in section 7.2, we estimate that between 2010 and 2018, an annual average of 7.7 right whales mortalities or serious injury resulted from entanglement in U.S. fishing gear. Additional analysis presented in section 7.2 estimates that an annual average of 4.7 right whale M/SI were the result of entanglement in gear used in the federal component of the U.S. fisheries. By subtracting the estimated M/SI in federal waters (4.7) from the total estimated M/SI in U.S. waters (7.7), we estimate that an annual average of 3 right whale M/SI were the result of entanglement with gear used in state fisheries.

Similarly, sea turtles may interact with fishing gear in state waters. Interactions have been documented with loggerhead, green, Kemp's ridley, and leatherback sea turtles. Gear types used in these fisheries include hook-and-line, gillnet, trawl, pound net and weir, trap/pot, seines, and channel nets. The magnitude and extent of interaction in many of these fisheries is largely unknown. Through the Annual Determination, NMFS identifies U.S. fisheries that are required to take observers upon request. The goals of this coverage is to learn more about interactions in that fishery, evaluate existing measures to prohibit take, and to determine if additional measures may be needed. It is not intended to be a comprehensive list of fisheries with interactions or suspected interactions, but rather those fisheries that NMFS intends to observe over a 5-year period (see Table 56 for current listing).

Table 56: Fisheries currently listed under the Annual Determination

Fishery	Years Eligible to Carry Observers
Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl	2020-2025
Gulf of Mexico mixed species fish trawl	2020-2025
Long Island inshore gillnet	2020-2025
Chesapeake Bay inshore gillnet	2020-2025
Mid-Atlantic gillnet	2018-2022
Gulf of Mexico menhaden purse seine	2018-2022

The available bycatch data for FMP fisheries indicate that sink gillnets and bottom otter trawl gear pose the greatest risk to Atlantic sturgeon (ASMFC 2017); although, Atlantic sturgeon are also caught by hook and line, fyke nets, pound nets, drift gillnets and crab pots (ASMFC 2017). It is likely that this vulnerability to these types of gear is similar to federal fisheries, although there is little data available to support this. Information on the number of Atlantic sturgeon captures and mortalities in non-federal fisheries, which primarily occur in state waters, is extremely limited. An Atlantic sturgeon “reward program” provided commercial fishermen monetary rewards for reporting captures of Atlantic sturgeon in Maryland’s Chesapeake Bay from 1996 to 2012 (Mangold et al. 2007). The data from this program show that Atlantic sturgeon have been caught in a wide variety of gear types, including hook and line, pound nets, gillnets, crab pots, eel pots, hoop nets, trawls, and fyke nets. Pound nets (58.9 percent) and gillnets (40.7 percent) accounted for the vast majority of captures. Of the more than 2,000 Atlantic sturgeon reported in the reward program over 16 years (1996-2012), biologists counted ten individuals that died because of their capture. No information on post-release mortality is available (Mangold et al. 2007).

Efforts are currently underway to obtain more information on the number of Atlantic sturgeon and sea turtle captures and mortalities in fisheries in state waters. Atlantic sturgeon are also vulnerable to capture in fisheries occurring in rivers, such as shad fisheries; however, these riverine areas are outside of the action area considered in this Opinion. Where available, specific information on protected species interactions in non-federal fisheries is provided below.

Atlantic Croaker Fishery

Along the U.S. Atlantic coast, Atlantic croaker are most abundant from the Chesapeake Bay to northern Florida. The Atlantic croaker fishery is managed by the Commission. The fishery is prosecuted with bottom trawl and gillnet gear. In 2018, the majority (97 percent) of commercial landings (in pounds) in came from Virginia (53 percent) and North Carolina (44 percent); the majority of recreational landings (in number of fish) were from Virginia (68 percent) and Florida (13 percent) (ASMFC 2019b). Sea turtle interactions have been documented in this fishery. In previous bycatch estimates where loggerhead bycatch was prorated by managed species landed, croaker was one of the fisheries with a higher number of takes for trawl (Murray 2015a) and gillnet gear (Murray 2018). Atlantic sturgeon interactions have also been observed in the Atlantic croaker fishery, but a quantitative assessment of the number of Atlantic sturgeon captured in the croaker fishery is not available. A mortality rate of Atlantic sturgeon in commercial trawls has been estimated at 5 percent. An earlier review of bycatch rates and landings for the weakfish fishery reported that the weakfish-striped bass fishery had an Atlantic sturgeon bycatch rate of 16 percent from 1989-2000; the weakfish-Atlantic croaker fishery had an Atlantic sturgeon bycatch rate of 0.02 percent, and the weakfish fishery had an Atlantic sturgeon bycatch rate of 1.0 percent. Bycatch rates were the ratio of sturgeon catch weight to the catch weight of all species landed (ASSRT 2007, Stein et al. 2004a). The ASSRT notes that the estimates can be heavily biased and the error rate large as observer coverage was not equal between fisheries or months of sampling and error (ASSRT 2007). In addition, fisheries have changed significantly since these estimates and, therefore, they are likely not applicable to contemporary fisheries.

Weakfish Fishery

Weakfish are found from Nova Scotia to southeastern Florida, but are more common from New York to North Carolina. The weakfish fishery occurs in both state and federal waters. Most commercial landings occur in the fall and winter months (Weakfish Plan Review Team 2019). The dominant commercial gear is gillnets with about 55 percent of commercial landings. There has been a shift in the dominant source of landings from trawls in the 1950s to 1980s to gillnets from the 1990s to present (Weakfish Plan Review Team 2019). Other gears include pound nets, haul seines, and beach seines (ASMFC 2016). North Carolina (34 percent), New York (23 percent), and Virginia (22 percent) had the largest share of the harvest in 2018 (Weakfish Plan Review Team 2019). North Carolina dominates commercial harvest, followed by Virginia and New Jersey. Together, these states have consistently accounted for 70 to 90 percent of the coast-wide commercial harvest since 1950 (ASMFC 2016, Weakfish Plan Review Team 2015, 2016, 2017, 2018, 2019). The recreational fishery catches weakfish using live or cut bait, jigging, trolling, and chumming, and the majority of fish are caught in state waters. The recreational fishery primarily occurs in state waters between New York and North Carolina (Weakfish Plan Review Team 2019).

Sea turtle bycatch in the weakfish fishery has occurred. NMFS originally assessed the impacts of the fishery on sea turtles in a biological opinion issued in 1997 (NMFS 1997). While the most recent gillnet bycatch estimates for 2007-2011 (Murray 2013) and 2012-2016 (Murray 2018) do prorate the bycatch by species landed, they do not include an estimate of loggerhead bycatch in the weakfish gillnet fishery. In an estimate of bycatch from 2002-2006, one loggerhead sea turtle was estimated to have been captured in the weakfish fishery based on a proration by species landed (Murray 2009b). These estimates encompassed both state and federal waters.

A quantitative assessment of the number of Atlantic sturgeon captured in the weakfish fishery is not available. A mortality rate of Atlantic sturgeon in commercial trawls has been estimated at 5 percent. Weakfish has also been identified as the top landed species on observed trips where sturgeon were incidentally captured (NEFSC observer/sea sampling database, unpublished data). In addition, as described above, the weakfish-striped bass fishery was identified as having higher bycatch rates using data from 1989-2000 (ASSRT 2007); however, there are a number of caveats associated with this data.

Whelk/Conch Fishery

A whelk/conch fishery occurs in several parts of the action area, including waters off Maine, Massachusetts, Connecticut, New York, New Jersey, Delaware, Maryland, and Virginia. While pot gear is the predominant gear used, whelk/conch are also harvested by hand and dredge. The fishery is limited entry in Massachusetts, New York, New Jersey and Virginia. Species targeted include waved, Stimpson, channeled, and knobbed whelk. Unlike lobster, there is no uniform, coast-wide management of the whelk fishery. Each state manages the fishery individually. Requirements often include licenses, gear marking, pot limits, and buoy line requirements.

Whelk fisheries overlap in time and space with sea turtles. Loggerhead, leatherback, and green sea turtles are known to become entangled in lines associated with trap/pot gear used in several fisheries including lobster, finfish, whelk, and crab species (Greater Atlantic Region Sea Turtle Disentanglement Network (GAR STDN), unpublished data). Unlike lobster pots, whelk pots in this area are not fully enclosed. This design of whelk pots has been suggested as a potential source of entrapment for loggerhead sea turtles that may be enticed to enter the trap to get the bait or whelks caught in the trap (Mansfield et al. 2001). Whelk fisheries in Massachusetts, New York, Delaware, Maryland, and Virginia were confirmed or probable fisheries involved in 18 sea turtle entanglements from 2010-2019. Thirteen entanglement events involved a leatherback sea turtle and five involved a loggerhead sea turtle. An additional 18 leatherbacks were entangled in either multiple gears (e.g., conch and lobster) or in gear where the fisherman held multiple permits, including conch, and the exact gear could not be identified. Green sea turtles have been documented in whelk/conch gear in previous years (GAR STDN, unpublished data). Atlantic salmon and Atlantic sturgeon interactions with trap/pot gear have never been observed or documented and; therefore, this gear type is not expected to be a source of injury or mortality to these species.

Crab Fisheries

Crab fisheries use a variety of gears including hand, trap/pot, trawl, and dredge. These fisheries occur in federal and state waters and target species such as blue, Jonah, rock and horseshoe crab. While the blue crab fishery occurs throughout the Mid-Atlantic south to the Gulf of Mexico, Maryland, Virginia, and North Carolina harvesters prosecute the majority of the effort. The Chesapeake Bay Program's Blue Crab Management Strategy indicates that there are multiple commercial and recreational gear types, various season lengths and regulations in three management jurisdictions. Fishing practices and the resulting harvest vary because of the complex ways crabs migrate and disperse throughout Chesapeake Bay.

The Jonah and rock crab fisheries may be prosecuted in conjunction with the lobster fishery. In this case, lobster traps are likely to be used. Depending on state regulation, other style traps may be available for use. Jonah crabs are harvested from deeper waters than rock crabs, and presently, are more highly valued. The commercial Jonah crab fishery is centered around Massachusetts and Rhode Island, though landings occur throughout New England and Mid-Atlantic states. The majority of horseshoe crab harvest comes from the Delaware Bay Region, followed by the New York, New England, and the Southeast regions. Trawls, hand harvests, and dredges make up the bulk of commercial horseshoe crab landings.

Sea turtles and large whales can become entangled in the vertical lines of trap/pot gear when they overlap with these fisheries. From 2010-2019, records (confirmed and probable) show 6 leatherbacks and 6 loggerhead sea turtles interacted with the vertical lines of blue crab gear in New Jersey and Virginia (GAR STDN, unpublished data). While these are where takes have been reported, interactions could occur wherever crab gear and sea turtles overlap. Interactions are primarily associated with entanglement in vertical lines, although sea turtles can also become entangled in groundline or surface systems. In 2007, a leatherback sea turtle was entangled in the lines connecting whelk pots (GAR STDN, unpublished data). In 2012, a leatherback was entangled in the surface system of a mooring buoy (GAR STDN, unpublished data), indicating that interactions with surface systems are possible.

Horseshoe crab has also been identified as the top species landed on trips that have incidentally taken sea turtles (NEFSC observer/sea sampling database, unpublished data). These takes were documented in trawl gear. Based on a proration of landings, two loggerheads on average annually were estimated to have been taken in the horseshoe crab trawl fishery from 2009-2013 (Murray 2015b).

The crab fisheries may have other detrimental impacts on sea turtles beyond entanglement in the fishing gear itself. Loggerheads are known to prey on crab species, including horseshoe and blue crabs. In a study of the diet of loggerhead sea turtles in Virginia waters from 1983 to 2002, Seney and Musick (2007) found a shift in the diet of loggerheads in the area from horseshoe and blue crabs to fish, particularly menhaden and Atlantic croaker. The authors suggested that this shift in loggerhead diet may be due to a decline in the crab species (Seney and Musick 2007). The physiological impacts of this shift are uncertain, although, Mansfield (2006) suggested it as a possible explanation for the declines in loggerhead abundance. Maier et al. (2005) detected Seasonal declines in loggerhead abundance coincident with seasonal declines of horseshoe and blue crabs were detected in the same area (Maier et al. 2005). While there is no evidence of a decline in horseshoe crab abundance in the Southeast during the period 1995-2003, declines were evident in some parts of the Mid-Atlantic (ASMFC 2004, Eyler et al. 2007). Given the variety of loggerheads prey items (Bjorndal 1997, Burke et al. 1993, Dodd 1988, Morreale and Standora 1998) and the differences in regional abundance of horseshoe crabs and other prey items (ASMFC 2004, Eyler et al. 2007), a direct correlation between loggerhead sea turtle abundance and horseshoe crab and blue crab availability cannot be made at this time. Nevertheless, the decline in loggerhead abundance in Virginia waters (Mansfield 2006) and possibly Long Island waters (Morreale and Standora 2005) coincident with noted declines in the abundance of horseshoe crab and other crab species raised concerns that crab fisheries may be impacting the forage base for loggerheads in portions of their range.

Atlantic sturgeon are known to be caught in state water horseshoe crab fisheries using trawl gear (Stein et al. 2004a). With the exception of New Jersey state waters, the horseshoe crab fishery operates in all state waters that occur in the action area. Along the U.S. East Coast, hand, bottom trawl, and dredge fisheries account for the majority (86 percent in the 2017 fishery) of commercial horseshoe crab landings in the bait fishery. Other methods used to land horseshoe crab are gillnets, fixed nets, rakes, hoes, and tongs (ASMFC 2019a, Horseshoe Crab Plan Review Team 2019). For most states, the bait fishery is open year round. However, the fishery operates at different times due to movement of the horseshoe crab. New Jersey has prohibited commercial harvest of horseshoe crabs in state waters (N.J.S.A. 23:2B-20-21) since 2006 (Horseshoe Crab Plan Review Team 2019). State waters of Delaware are closed to horseshoe crab harvest and landing from January 1 through June 7 each year (7 Del Admin. C § 3200). Other states also regulate various seasonal and area closures and other state horseshoe crab fisheries are regulated with various seasonal/area closures (Horseshoe Crab Plan Review Team 2019). The majority of horseshoe crab landings from the bait fishery from 2014-2018 came from Maryland, Delaware, New York, Virginia, and Massachusetts. Florida, Georgia, South Carolina had de minimus status in 2018 (Horseshoe Crab Plan Review Team 2019). There is also a smaller fishery for biomedical uses.

An evaluation of bycatch of Atlantic sturgeon using the NEFSC observer/sea sampling database (1989-2000) found that the bycatch rate for horseshoe crabs was low at 0.05 percent (Stein et al. 2004a). An Atlantic sturgeon "reward program," where commercial fishermen were provided monetary rewards for reporting captures of Atlantic sturgeon in the Maryland waters of Chesapeake Bay (Mangold et al. 2007), operated from 1996 to 2012.[19] From 1996-2006, the data showed that one of 1,395 wild Atlantic sturgeon was found caught in a crab pot (Mangold et al. 2007).

Fish Trap, Seine, and Channel Net Fisheries
Incidental captures of sea turtles in fish traps have been reported from several states along the U.S. Atlantic coast (GAR STDN, unpublished data). From 2010-2019, records (confirmed and probable) documented 24 leatherback, 2 Kemp's ridley, 10 loggerheads, and 1 unknown sea turtle in pound nets/weirs from Maine through Virginia. Of the 37 interactions, six animals were documented free swimming (GAR STDN, unpublished data). In this gear, sea turtles may become entangled in the gear or be free swimming in the pound/weir.

The Virginia pound net fishery is contiguous to the action area at the mouth of Chesapeake Bay. Sea turtle interactions with the Virginia pound net fishery have been documented, and interactions reported to the GAR STDN are included above. NMFS has taken regulatory action to address sea turtle bycatch in the Virginia pound net fishery. The most recent biological opinion on the federal rulemaking on Virginia pound nets anticipated the take of up to 805 (1 lethal) loggerhead, 161 Kemp's ridley (1 lethal), 16 green (1 lethal), and 11 Atlantic sturgeon (none lethal) in the pound and heart portions of the gear. The leaders may also capture sea turtles and Atlantic sturgeon. NMFS anticipated that up to 1 (1 lethal), loggerhead, 1 (1 lethal) Kemp's ridley, (up to 1 lethal) green, 8 (4 lethal) leatherback, and 2 Atlantic sturgeon (1 lethal) could occur annually (NMFS 2018e).

Long haul seines, beach seines, purse seines, and channel nets are also known to incidentally capture sea turtles in sounds and other inshore waters along the U.S. Atlantic coast, although no lethal interactions have been reported (NMFS SEFSC 2001). No information on interactions between Atlantic sturgeon and fish traps, long haul seines, or channel nets is currently available; however, depending on where this gear is set and the mesh size, the potential exists for Atlantic sturgeon to be entangled or captured in net gear. Interactions between marine mammals considered in this Opinion and these gears are not known to occur.

American Lobster Trap Fishery
An American lobster trap fishery occurs in state waters of New England and the Mid-Atlantic and is managed under the Commission's Interstate Fishery Management Plan (ISFMP). Like the federal waters component of the fishery, the state waters fishery uses trap/pot gear to land lobster. Trap/pot gear is known to entangle sea turtles and large whales. Often for these entanglements, the gear cannot be documented to a specific fishery. There have been documented takes of North Atlantic right whales in inshore and state lobster gear (Morin et al. 2019) (NMFS, unpublished data; see link to 2000-2018 Marine Animal Incident Data 03/19/2019 at https://archive.fisheries.noaa.gov/garfo/protected/whaletrp/trt/meetings/April%202019/19_april_2019_trt_meeting.html).

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[19] The program was terminated in February 2012, with the listing of Atlantic sturgeon under the ESA.

Leatherback, loggerhead, green and Kemp's ridley sea turtles are known to interact with trap/pot gear. As described above, interactions are primarily associated with entanglement in vertical lines. Records of stranded or entangled sea turtles indicate that fishing gear can wrap around the neck, flipper, or body of the sea turtle and severely restrict swimming or feeding (GAR STDN, unpublished data; NMFS STSSN, unpublished data). As a result, these interactions often result in the injury or mortality to sea turtles.

Using the criteria defined in section 7.3.1.3, there were 81 leatherback entanglements from 2010-2019 in state confirmed to the lobster fishery. Four of the cases were confirmed to recreational pot gear. All entanglements involved the vertical line of the gear. These verified/confirmed entanglements occurred in waters off Maine, Massachusetts, and New York from May through October. The majority were documented in waters off Massachusetts (GAR STDN, unpublished data).

Atlantic salmon and Atlantic sturgeon interactions with trap/pot gear have never been observed (NEFSC observer/sea sampling database, unpublished data) or documented; therefore, this gear type is not expected to be a source of injury or mortality to these species.

American Shad Fishery
An American shad fishery occurs in state waters of New England and the Mid-Atlantic and is managed under the Commission's ISFMP. Amendment 3 to the ISFMP requires states and jurisdictions to develop sustainable FMPs, which are reviewed and approved by the Commission's Technical Committee, in order to maintain recreational and commercial shad fisheries (ASMFC 2010). Eight entities in the action area have developed these FMPs. The fishery occurs in rivers and coastal ocean waters. In 2005, the directed at-sea fishery was closed and subsequent landings from the ocean are only from the bycatch fishery. Given this, the fishery is not expected to interact with Atlantic large whales or sea turtles.

In the past, approximately 40-500 Atlantic sturgeon were reportedly captured in the spring shad fishery in Delaware. In recent years, this fishery has turned more to striped bass. Most of the Atlantic sturgeon were captured in the Delaware Bay, with only 2 percent caught in the Delaware river. The fishery uses five-inch mesh gillnets that are left to soak overnight; based on the available information, there is little bycatch mortality (NMFS 2011a). Recreational hook and line shad fisheries are known to capture Atlantic sturgeon, particularly in southern Maine (NMFS 2011a).

Striped Bass Fishery
Since 1981, the Commission has managed striped bass, from Maine to North Carolina through an ISFMP. The striped bass fishery occurs only in state waters. With the exception of a defined area around Block Island, Rhode Island for possession, federal waters have been closed to the harvest and possession of striped bass since 1990. All states are required to have recreational and commercial size limits, recreational creel limits, and commercial quotas. The commercial striped bass fishery is closed in Maine, New Hampshire, and Connecticut, but open in Massachusetts (hook and line only), Rhode Island, New Jersey (hook and line only), Delaware, Maryland, and Virginia. Recreational striped bass fishing occurs all along the U.S. East Coast.

The striped bass fishery uses gears known to interact with sea turtles, including trap, pound nets, gillnets, trawl, and hook-and-line (ASMFC 2020b) . When prorated by species landed, striped bass was one of the trawl and gillnet fisheries in which sea turtles were estimated (Murray

2015b, 2018). Several states have reported incidental catch of Atlantic sturgeon during striped bass fishing activities (NMFS 2011a). In southern Maine and New Hampshire, the recreational striped bass fishery is known to catch Atlantic sturgeon, although numbers are not available. There are also numerous reports of Atlantic sturgeon bycatch in recreational striped bass fishery along the south shore of Long Island, particularly around Fire Island and Far Rockaway. Unreported mortality is likely occurring.

Data from the Atlantic Coast Sturgeon Tagging Database showed that from 2000-2004, the striped bass fishery accounted for 43 percent of Atlantic sturgeon recaptures (ASSRT 2007). The striped bass-weakfish fishery also had one of the highest bycatch rates of 30 directed fisheries according to NMFS Observer Program data from 1989-2000 (ASSRT 2007).

State gillnet fisheries
State gillnet fisheries occur in many portions of the action area. However, limited information is available on interactions between these fisheries and protected species. Large and small mesh gillnet fisheries occur in state waters. For example, the black drum shark gillnet fisheries in Virginia state waters fisheries uses large mesh (10- to 14-inch) gillnets. Meshes smaller than 10 inches are used in the croaker and dogfish fisheries. Entanglements of sea turtles in large mesh gillnet sets targeting and/or landing black drum have been recorded (NEFSC observer/sea sampling database, unpublished data). Similarly, sea turtles are vulnerable to capture in small mesh gillnet fisheries occurring in state waters. Observer coverage in state gillnet fisheries has been limited. For example, 31 trips were observed in the Long Island Sound gillnet fishery from 2014 through 2018. There has also been limited coverage on coastal gillnet fisheries in the mid-Atlantic on vessels with federal permits and, to a lesser extent, vessels with state only permit. Through this limited coverage, interactions have been recorded with Kemp's ridley, loggerhead, green, and leatherback sea turtles in gillnets operating in state waters (NEFSC observer/sea sampling database, unpublished data). As gillnet gear is known to pose an interaction risk to listed species of sea turtles, sturgeon, and large whales, these fisheries have the potential to interact with these species when the fisheries overlap with them

High levels of strandings in North Carolina in 1999 were determined to likely result from incidental capture in the large mesh gillnet fishery in Pamlico Sound. Since 2000, NMFS has issued five ESA section 10(a)(1)(B) Incidental Take Permits (65 FR 65840, November 2, 2000; 66 FR 51023, October 5, 2001; 67 FR 67150, November 4, 2002; 70 FR 52984, September 6, 2005; 78 FR 57132, September 17, 2013) to the North Carolina Division of Marine Fisheries authorizing the incidental take of sea turtles in certain components of the gillnet fishery. The most recent permit (78 FR 57132, September 17, 2013) authorizes the take through August 2023. Required measures under the permit include restricted soak times, restricted net lengths, attendance requirements, time-area closures, and adaptive management (78 FR 57132, September 17, 2013). North Carolina DMF also has a permit for the incidental take of Atlantic sturgeon DPSs associated with the inshore gillnet fishery. The conservation plan requires specific monitoring for Atlantic sturgeon. If allowable thresholds are approached, North Carolina DMF will place additional restrictions (e.g., closures, attendance requirements) on the fishery. In addition, the observer coverage will identify and adaptively respond to "hotspots" (79 FR 43716, July 28, 2014). The level of take specified in these permits is detailed in section 5.1.4.

The 2017 Benchmark Assessment (ASMFC 2017) used data from the Northeast Fisheries Observer Program, the North Carolina gillnet fisheries, and the South Carolina American shad gillnet fishery to assess Atlantic sturgeon bycatch. For the North Carolina gillnet fisheries predicted bycatch for 2004-2005 ranged from 1,286 Atlantic sturgeon in 2011 to 13,668 Atlantic sturgeon in 2008. The Atlantic sturgeon caught in this fishery were primarily juveniles. The percent observed sturgeon that dies ranged from 0-20 percent with an overall mean of 6 percent. Estimates of dead discards ranged from 0-424 fish (ASMFC 2017).

In 2017, 167 Atlantic sturgeon were reported as bycatch from state water fisheries (0-3 miles offshore, including rivers and estuaries). This included 51 fish in the North Carolina gillnet fishery and 66 fish in the South Carolina American shad fishery. Connecticut (15), Maryland (1), Virginia (11), and Georgia (23) also reported bycatch in 2017 (ASMFC 2019d).

State Trawl Fisheries
Trawl fisheries also occurs in state waters in the action area. Virginia (VA Code Ann. § 28.2-315), New Hampshire (N.H. Stat. Ann. §21149), and Delaware (Del. Code tit. 7, §927) prohibit trawling in state waters. Other states such as Maryland prohibit its use in certain areas.

A Northern shrimp fishery has occurred in waters off Maine, New Hampshire, and Massachusetts, and is managed under the Commission's ISFMP. Due to recruitment failure and a collapsed stock, fishing moratoria were instituted by the Commission for the 2014-2018 fishing seasons. In November 2018, the Commission's Northern Shrimp Section extended the moratorium on commercial fishing through 2021. The majority of northern shrimp are caught with otter trawls, which must be equipped with Nordmore grates (ASMFC 2011). When the fishery is open, it is a winter fishery with the season occurring anytime between December 1 and May 31 (ASMFC 2017).

Bottom otter trawls in the Northern shrimp fishery are known to interact with Atlantic sturgeon, but exact numbers are not available (NMFS 2011a). A majority (84 percent) of Atlantic sturgeon bycatch in otter trawls occurs at depths less than 66 ft (20 m), with 90 percent occurring at depths of less than 98 ft (30 m) (ASMFC 2007). During the NEFSC's spring and fall inshore northern shrimp trawl surveys, northern shrimp are most commonly found in tows with depths of greater than 210 ft (64 m) (ASMFC 2011), which is well below the depths at which most Atlantic sturgeon bycatch occurs.

Given that the Northern shrimp trawl fishery is a winter fishery, it is not expected to overlap with sea turtles in the action area. Given the gear type used in the fishery, it is also not expected to interact with large whales.

Other trawl fisheries occur in state waters, but information is limited. In these fisheries, the gear may operate along or off the bottom. From 2009-2018, observers have documented the take of Kemp's ridley, loggerhead, green, and leatherback sea turtles in state waters (NEFSC observer/sea sampling database, unpublished data). The top landed species on trips that captured turtles included scup, summer flounder, longfin squid, horseshoe crab, and butterfish. Atlantic sturgeon have also been observed captured on state trawl fisheries from 2009-2018. Top landed species on these trips included, among others, summer flounder, little skate, scup, butterfish, longfin squid, spiny dogfish, smooth dogfish, and bluefish. During this period, there were no Atlantic salmon documented captured in state waters. Information available on interactions between ESA-listed species and these fisheries is incomplete.

State Recreational Fisheries
Observations of state hook and line recreational fisheries have shown that loggerhead, Kemp's ridley, leatherback, and green sea turtles can interact with recreational fishing gear. When swimming near rod and reel fishing gear, sea turtles can be "foul-hooked" on the flipper or entangled in the fishing line. Sea turtles are also known to bite the bait and become hooked in the mouth or esophagus, or swallow the hook. Most of the reports of interactions come from fishing piers, but there are also reports of offshore captures (NMFS and USFWS 2008). A summary of known impacts of hook and line captures on loggerhead, Kemp's ridley, and leatherback sea turtles can be found in the TEWG reports (TEWG 1998, 2000, 2007, 2009).

Stranding data also provide evidence of interactions between recreational hook and line gear and sea turtles. While data from stranded animals contain certain biases and cannot be used to quantify the magnitude of a particular threat, it does provide some information on interactions with recreational gear. From Maine through Virginia, there were 186 cases reported from 2016-2018 in the STSSN database in which recreational fishing gear was present (NMFS STSSN, unpublished data). This included 36 loggerhead, 122 Kemp's ridley, 2 green, 1 leatherback, and 25 unknown turtles. NMFS conducts outreach on what to do if you hook or entangle a sea turtle while fishing. In addition, Virginia Aquarium' Stranding Response Program has developed a pier partner program that provides signage for the pier and training to the pier operator on what to do if a sea turtle is hooked. Since the program began in 2014, there have been 253 reports received with 172 animals admitted. In 2018, the Aquarium received a record number of hooked turtle reports. Of the 66 reported cases, they admitted 45 turtles for exam. Almost 87 percent of these turtles were Kemp's ridleys. Turtle captures on recreational hook and line gear are not uncommon, but the overall level of take and post-release mortality are unknown.

Bycatch in recreational fisheries in Maine may result in direct mortality or cause stress, thus reducing reproductive success and survival of Atlantic salmon (USFWS and NMFS 2019). Recreational angling is for freshwater species throughout the range of the GOM DPS. This is outside the area where the fisheries undergoing consultation operate.

Atlantic sturgeon have also been observed captured in hook and line gear, yet the total number of interactions that occur annually is unknown. There have been no post-release survival studies for this species. However, we anticipate that Atlantic sturgeon will likely be released alive, due to the overall hardiness of the species. NMFS also engages in educational outreach efforts on disentanglement, release, and handling and resuscitation of Atlantic sturgeon.

5.3. Other Activities

5.3.1. *Maritime Industry*
Private and commercial vessels, including fishing vessels, operating in the action area of this consultation also have the potential to interact with ESA-listed species. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. Commercial traffic and recreational pursuits can also adversely affect ESA-listed species through propeller- and boat strikes. Vessel interactions have been documented with large whales, sea turtles, Atlantic sturgeon, and giant manta rays. The extent of the problem is difficult to assess because the interactions occur at sea and are often only detected when the animal strands. It is also often not known if the animal was struck pre- or post-mortem. It is important to note that although

minor vessel collisions may not result in a direct mortality, they may weaken or otherwise affect an animal, which may make it more vulnerable to other threats.

5.3.2. ***Pollution***

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local, or private action, may affect ESA-listed species in the action area. Sources of pollutants in the action area include atmospheric loading of pollutants (e.g., PCBs); storm water runoff from coastal towns, cities, and villages; runoff into rivers emptying into bays; groundwater discharges; sewage treatment plant effluents; and oil spills. Oil spills may affect ESA-listed species either directly or through the food chain (see section 5.1.7).

Degraded water quality from point and non-point sources can impact protected species. Run-off can introduce pesticides, herbicides, and other contaminants into the system on which these species depend. Contaminants could degrade habitat if pollution and other factors reduce the food available to marine animals. In 2017, NMFS completed a biological opinion on EPA's registration of certain pesticides. Effects ranged from mortalities to reductions in prey, and impaired growth. Species likely to be affected include, among others, Atlantic salmon, Atlantic sturgeon (all five DPSs), and sea turtles. In specifying the ITS, NMFS identified surrogates for anadromous fish and sea turtles (NMFS 2017).

Oil spills, resulting from anthropogenic activities (e.g., commercial vessel traffic/shipping), directly and indirectly affect all components of the marine ecosystem. Larger oil spills may result from severe accidents, although these events would be rare. The pathological effects of oil spills on sea turtles specifically have been documented in several laboratory studies (Vargo et al. 1986). There have been a number of documented smaller oil spills in the northeastern United States.

As many ESA-listed species ranges extend beyond that of the action area, oil spills that occur outside the action area, but within the range of the species, also have the potential to affect ESA-listed species that occur within the action area. For instance, on April 20, 2010, the Deepwater Horizon oil spill occurred off the coast of Louisiana, in the Gulf of Mexico. The effects of this spill on ESA-listed species is discussed in the *Status of the Species* section.

Marine debris (e.g., discarded fishing line, boat lines, plastics) can directly or indirectly affect listed species. Discarded line (fishing or boat) can entangle large whales, Atlantic sturgeon, or sea turtles, causing injury or mortality. Large whales and sea turtles ingest plastic. In the case of sea turtles, they may mistake debris for food. For instance, jellyfish are a preferred prey for leatherbacks, and plastic bags, which may look like jellyfish to the turtles, are often found in the turtles' stomach contents (Mrosovsky et al. 2009, Nelms et al. 2015, NRC 1990, Schuyler et al. 2014). While marine debris is known to affect these species, the effects have not been quantified and impacts at the population level are not well understood.

5.3.3. ***Coastal development***

Beachfront development, lighting, and beach erosion control are ongoing activities along the coastlines of the United States and within the action area. In the southeast and mid-Atlantic, these activities potentially reduce or degrade sea turtle nesting habitats or interfere with hatchling movement to sea. Human activities along nesting beaches at night may also discourage sea turtles from nesting sites. The extent to which these activities reduce sea turtle nesting and hatchling production is unknown. However, more and more coastal counties are adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting.

Coastal development may also impact Atlantic sturgeon if it disturbs or degrades foraging habitats or otherwise affects the ability of sturgeon to use coastal habitats.

5.4. Reducing Threats to ESA-listed Species

5.4.1. *Education and Outreach Activities*

Education and outreach activities are some of the primary tools to effectively reduce the threats to all protected species. For example, NMFS has been active in public outreach to educate fishermen about sea turtle handling and resuscitation techniques and educate recreational fishermen and boaters on how to avoid interactions with marine mammals, sea turtles, and sturgeon. NMFS is engaged in a number of education and outreach activities aimed specifically at increasing mariner awareness of the threat of ship strikes to protected species. NMFS also offers educational programs to students. One such program is "SCUTES" (Student Collaborating to Undertake Tracking Efforts for Sturgeon), which offers educational programs and activities about the movements, behaviors, and threats to Atlantic sturgeon. While the effects of these efforts at reducing impacts to protected species cannot be quantified, they are anticipated to reduce impacts through education and promoting stewardship. Outreach occurs through websites, NMFS presence at industry meetings, outreach events and trade shows, publications in industry trade journals and news outlets, and dockside interactions between staff and industry. NMFS intends to continue these outreach efforts in an attempt to reduce interactions and the likelihood of injury to protected species and to potentially improve the condition of the ESA-listed species or its designated critical habitat in the action area.

5.4.2. *Stranding and Salvage Programs*

The STSSN does not directly reduce the threats to sea turtles. However, the extensive network of STSSN participants along the Atlantic and Gulf of Mexico coasts not only collects data on dead sea turtles, but also rescues and rehabilitates live stranded turtles, reducing mortality of injured or sick animals. NMFS manages the activities of the STSSN. Data collected by the STSSN are used to monitor stranding levels, to identify areas where unusual or elevated mortality is occurring, and to identify sources of mortality. The data are also used to monitor incidence of disease, study toxicology and contaminants, and conduct genetic studies to determine population structure. All of the states that participate in the STSSN tag live turtles when encountered (either via the stranding network, through incidental takes, or permitted in-water studies). Tagging studies help improve our understanding of sea turtle movements, longevity, and reproductive patterns, all of which contribute to our ability to reach recovery goals for sea turtle species.

NMFS was designated the lead agency to coordinate the Marine Mammal Health and Stranding Response Program (MMHSRP), which was formalized by the 1992 Amendments to the MMPA. The program consists of state volunteer stranding networks, biomonitoring, Analytical Quality Assurance for marine mammal tissue samples, a Working Group on Marine Mammal Unusual Mortality Events (UME) and a National Marine Mammal Tissue Bank. Additionally, a serum bank and long-term storage of histopathology tissue are being developed. The MMHSRP's permit (permit #18786) includes the incidental take of unidentified sea turtles (10), leatherback sea turtles (2), and Atlantic sturgeon (3).

A salvage program operating under an ESA section 10(a)(a)(A) permit is in place for Atlantic sturgeon. Atlantic sturgeon carcasses can provide pertinent life history data and information on new or evolving threats to Atlantic sturgeon. Their use in scientific research studies can reduce the need to collect live Atlantic sturgeon. The NMFS Sturgeon Salvage Program is a network of individuals qualified to retrieve and/or use Atlantic and shortnose sturgeon carcasses and parts for scientific research and education. All carcasses and parts are retrieved opportunistically and participation in the network is voluntary.

5.4.3. *Disentanglement Networks*

In 2002, in response to the high number of leatherback sea turtles found entangled in pot gear along the U.S. Northeast Atlantic coast, NMFS Northeast Region (now GARFO) established the NMFS Greater Atlantic Region Sea Turtle Disentanglement Network (GAR STDN). The GAR STDN is a component of the larger STSSN program, and operates in all states in the region. The GAR STDN responds to entangled sea turtles, disentangling and releasing live animals, thereby reducing injury and mortality. In addition, the GAR STDN collects data on sea turtle entanglement events, providing valuable information for management purposes. GARFO oversees the GAR STDN program and manages the GAR STDN database.

Any agent/employee of NMFS, the USFWS, the USCG, any other federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his/her official duties, is allowed to take endangered sea turtles encountered in the marine environment if such taking is necessary to: (1) aid a sick, injured, or entangled endangered sea turtle; (2) dispose of a dead endangered sea turtle; or (3) salvage a dead endangered sea turtle for scientific or educational purposes (70 FR 42508, July 25, 2005. NMFS affords the same protection to sea turtles listed as threatened under the ESA (50 CFR 223.206(b)).

In 1984, the Center for Coastal Studies (CCS) in partnership with NMFS, developed techniques for disentangling free-swimming large whales from life threatening entanglements. Over the next decade, CCS and NMFS continued working on the development of the technique to safely disentangle both anchored and free swimming large whales. In 1995, NMFS issued an official permit to CCS to disentangle large whales. This initial partnership led to the establishment of the Atlantic Large Whale Disentanglement Network (ALWDN). This network represents a coordinated effort between NMFS, large whale researchers, state agencies, and other federal partners, to implement and monitor efforts to remove and recover entangling fishing gear from large whales along the entire Atlantic coast. The ALWDN is managed by NMFS and is implemented through a series of permits and Memorandums of Agreement. Due to the success of the disentanglement networks, NMFS believes protected species that may otherwise have succumbed to complications from entangling gear have been freed and have survived.

5.4.4. *Regulatory Measures for Sea Turtles*

Large-Mesh Gillnet Requirements in the Mid-Atlantic

Since 2002, NMFS has regulated the use of large mesh gillnets in federal waters off North Carolina and Virginia (67 FR 13098, March 21, 2002) to reduce the impact of these fisheries on ESA-listed sea turtles. These restrictions were revised in 2006 (73 FR 24776, April 26, 2006). Currently, gillnets with stretched mesh size of 7 inches (17.8 cm) or larger are prohibited in the Exclusive Economic Zone during the following times and in the following areas:

(1) north of the North Carolina/South Carolina border to Oregon Inlet, North Carolina at all times,
(2) north of Oregon Inlet, North Carolina to Currituck Beach Light, North Carolina from March 16 through January 14,
(3) north of Currituck Beach Light, North Carolina to Wachapreague Inlet, Virginia from April 1 through January 14, and
(4) north of Wachapreague Inlet, Virginia to Chincoteague, Virginia from April 16 through January 14.

NMFS has also issued regulations to address the interaction of sea turtles in gillnet gear fished in Pamlico Sound, North Carolina. Waters of Pamlico Sound are closed to fishing with gillnets with a stretched mesh size larger than 4 ¼ inches (10.8 cm) from September 1 through December 15 each year to protect sea turtles. The closed area includes all inshore waters of Pamlico Sound, and all contiguous tidal waters, south of 35° 46.3' N, north of 35° 00' N, and east of 76° 30' W (50 CFR 223.206). As described above, NMFS has also issued incidental take permits for Atlantic sturgeon and sea turtles in Pamlico Sound gillnet fisheries. The permit includes mandatory measures to reduce take, and impacts from take, in this fishery.

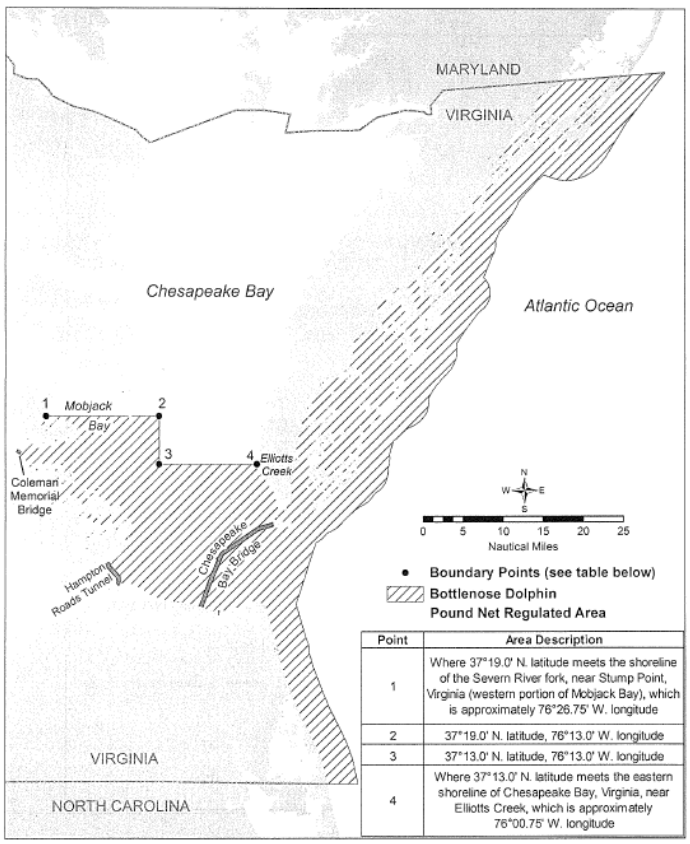
TED Requirements in Trawl Fisheries
Turtle excluder devices (TEDs) are required in the summer flounder and southeast shrimp fisheries. TEDs allow sea turtles to escape the trawl net, reducing injury and mortality resulting from capture in the net. Approved TEDs are required in the shrimp trawl fishery operating in the Atlantic and Gulf Areas (50 CFR 222.102) unless the trawler is fishing under one of the exemptions (e.g., bait shrimper, pusher-head trawl,) and all requirements of the exemption are met (50 CFR 223.206). On February 21, 2003, NMFS issued a final rule to amend the TED regulations to enhance their effectiveness in the Atlantic and Gulf Areas of the southeastern United States by requiring an escape opening designed to exclude leatherbacks as well as large loggerhead and green turtles (68 FR 8456). NMFS published a final rule, effective April 1, 2021, that requires TEDs to exclude small turtles on skimmer trawls vessels 40 ft (12 m) or greater in length (84 FR 70048, December 20, 2019). On March 31, 2021, NMFS delayed the effective date until August 1, 2021 (86 FR 16677).

TEDs are also required for summer flounder trawlers in the summer flounder fishery-sea turtle protection area. This area is bounded on the north by a line extending along 37° 05’ N (Cape Charles, VA) and on the south by a line extending out from the North Carolina-South Carolina border. Vessels north of Oregon Inlet, NC are exempt from the TED requirement from January 15 through March 15 each year (50 CFR 223.206). The TED requirements for the summer flounder trawl fishery do not require the use of the larger escape opening.

Pound net requirements in Virginia
NMFS has issued several regulations to help protect sea turtles from entanglement in and impingement on Virginia pound net gear (66 FR 33489, June 22 2001; 67 FR 41196; June 17, 2002; 68 FR 41942, July 16, 2003; 69 FR 24997, May 5, 2004; 71 FR 36024; June 23, 2006; 73 FR 68348, November 18, 2008; 80 FR 6925, February 9, 2015). All offshore pound leaders in Pound Net Regulated Area I (Figure 49) must meet the definition of a modified pound net leader from May 6 through July 15. The modified leader has been found to be effective in reducing sea turtle interactions as compared to the unmodified leader. Under the ESA regulations, nearshore pound net leaders in Pound Net Regulated Area I and all pound net leaders in Pound Net

Regulated Area II must have mesh size less than 12 inches (30.5 cm) stretched mesh and may not employ stringers (50 CFR 223.206) from May 6 through July 15 each year. A pound net leader is exempt from these measures only if it meets the definition of a modified pound net leader. The 2015 regulation (80 FR 6925) modified the definitions of offshore and inshore pound net leaders under the ESA. In addition, there are compliance training, monitoring and reporting requirements in this fishery (50 CFR 223.206).

Under the Bottlenose Dolphin Take Reduction Plan (section 5.4.9), fishermen with offshore pound nets must use a modified pound net leader year-round within the Bottlenose Dolphin Pound Net Regulated Area (Figure 49). Pound nets fished in offshore and inshore areas must be fished with all three continuous sections (i.e., pound, heart, and leader) in the Bottlenose Dolphin Pound Net Regulated Area or Regulated Areas I and II under the ESA sea turtle conservation requirements. An exception is that one or more sections may be missing for up to 10 days for setting, removing, and/or repairing the gear.



Point	Area Description
1	Where 37°19.0' N. latitude meets the shoreline of the Severn River fork, near Stump Point, Virginia (western portion of Mobjack Bay), which is approximately 76°26.75' W. longitude
2	37°19.0' N. latitude, 76°13.0' W. longitude
3	37°13.0' N. latitude, 76°13.0' W. longitude
4	Where 37°13.0' N. latitude meets the eastern shoreline of Chesapeake Bay, Virginia, near Elliotts Creek, which is approximately 76°00.75' W. longitude

Figure 49: Bottlenose dolphin pound net regulated area

Longline requirements in the HMS fishery

In 2020, NMFS SERO completed two biological opinions on the FMP for the Atlantic HMS fisheries for swordfish, tunas, and sharks (NMFS 2020c, d). These opinions concluded that the actions are not likely to jeopardize the continued existence of any hard-shell or leatherback sea turtle. Sea turtle conservation requirements in the HMS fishery are related to the fishing gear, bait, and disentanglement gear and training (50 CFR 648.21). NMFS requires the use of specific gears and release equipment in the pelagic longline component of the HMS fishery in order to minimize lethal impacts to sea turtles. NMFS has developed sea turtle handling and release protocols for the HMS fishery (NMFS 2010a). Sea turtle handling and release placards are required to be posted in the wheelhouse of certain commercial fishing vessels. NMFS has also initiated an extensive outreach and education program for commercial fishermen that engage in these fisheries in order to minimize the impacts of this fishery on sea turtles. As part of the program, NMFS has distributed sea turtle identification and resuscitation guidelines to HMS fishermen who may incidentally hook, entangle, or capture sea turtles during their fishing activities and has also conducted hands-on workshops on safe handling, release, and identification of sea turtles.

Modified Dredge Requirements in the Atlantic Sea Scallop Fishery
In response to the observed capture of sea turtles in scallop dredge gear, including injuries and mortality as a result of capture, NMFS required federally-permitted scallop vessels fishing with dredge gear to modify their gear by adding an arrangement of horizontal and vertical chains (hereafter referred to as a “chain mat”) between the sweep and the cutting bar. This modification was required when fishing in Mid-Atlantic waters south of 41° 9’ N from the shoreline to the outer boundary of the EEZ during the period of May 1-November 30 each year (70 FR 30660, May 27, 2005). The requirement was subsequently modified by emergency rule on November 15, 2006 (71 FR 66466) and by final rules published on April 8, 2008 (73 FR 18984) and May 5, 2009 (74 FR 20667). In 2015, NMFS aligned the requirements with the turtle deflector dredge (TDD) requirements as described below. Since 2006, the chain mat modifications have reduced the severity of most sea turtle interactions with scallop dredge gear (Murray 2011, 2015a). However, these modifications are not expected to reduce the overall number of sea turtle interactions with scallop dredge gear.

Beginning May 1, 2013, all limited access scallop vessels, as well as Limited Access General Category vessels with a dredge width of 10.5 ft (3.2 m) or greater, were required to use a TDD in the Mid-Atlantic (west of 71° W) from May 1 through October 31 each year (77 FR 20728, April 6, 2012). The purpose of the TDD requirement is to deflect sea turtles over the dredge frame and bag rather than under the cutting bar, so as to reduce sea turtle injuries due to contact with the dredge frame on the ocean bottom (including being crushed under the dredge frame). When combined with the effects of chain mats, which decrease captures in the dredge bag, the TDD should provide greater sea turtle benefits by reducing injury and mortality due to interactions with the dredge frame, compared to a standard New Bedford dredge.

In 2015, NMFS aligned the TDD and chain mat requirements (80 FR 22119, April 21, 2015). Currently, chain mats are required on any vessel with a sea scallop dredge and required to have a federal Atlantic sea scallop fishery permit, regardless of dredge size or vessel permit category, entering waters west of 71° W from May 1 through November 30. Similarly, any limited access scallop vessel and limited access general category vessel with a dredge width of 10.5 ft (3.2 m) or greater is required to use a TDD west of 71° W from May 1 through November 30.

Handling and Resuscitation Requirements
NMFS has developed and published sea turtle handling and resuscitation techniques for sea turtles that are incidentally caught during scientific research or fishing activities (66 FR 67495, December 31, 2001). Persons participating in fishing activities or scientific research are required to handle and resuscitate (as necessary) sea turtles as prescribed in the final rule. These measures help to prevent mortality of hard-shelled turtles caught in fishing or scientific research gear.

NMFS has conducted outreach to fishermen participating in fisheries in the Greater Atlantic Region, providing wheelhouse cards detailing the requirements.

5.4.5. *Atlantic Large Whale Take Reduction Plan*

The Plan reduces the risk of serious injury or mortality of North Atlantic right, fin, and humpback whales due to incidental entanglement in U.S. commercial trap/pot and gillnet fishing gear. The Plan is required by the Marine Mammal Protection Act (MMPA) and has been developed through a collaborative Take Reduction Team process implemented by NMFS. The Plan covers the U.S. Atlantic EEZ from Maine through Florida (26°46.5' N). Requirements are year-round in the Northeast, and seasonal in the Mid and South Atlantic. Fisheries in this Opinion that are regulated under the ALWTRP include Northeast multispecies, monkfish, spiny dogfish, skate, bluefish, American lobster, Jonah crab, and red crab. This section describes measures currently in effect (see section 3.2.1 for proposed measures for the lobster and Jonah crab fisheries).

The Atlantic Large Whale Take Reduction Team (ALWTRT) consists of fishing industry representatives, environmentalists, state and federal officials, and other interested parties. The Take Reduction Plan is an evolving plan that changes as NMFS and the Team learn more about why whales become entangled and how fishing practices might be modified to reduce the risk of entanglement. Regulatory actions are directed at reducing serious entanglement injuries and mortalities of right, humpback, and fin whales from fixed gear fisheries (i.e., trap/pot and gillnet fisheries). The non-regulatory component of the Plan is composed of four principal parts: (1) gear research and development, (2) disentanglement, (3) the Right Whale Sighting Advisory System (RWSAS), and (4) education/outreach. These components will be discussed in more detail below. The first regulations stemming from the Plan went into effect in 1997.

Regulatory Measures to Reduce the Threat of Entanglement on Whales

The regulatory component of the Plan includes a combination of broad fishing gear modifications and time-area restrictions supplemented by progressive gear research to reduce the chance that entanglements will occur, or that whales will be seriously injured or die because of an entanglement. The long-term goal, established by the 1994 Amendments to the MMPA, is to reduce entanglement related serious injuries and mortalities of right, humpback and fin whales to insignificant levels approaching zero within five years of Plan implementation. Despite measures of the Plan, entanglements, including serious injuries or mortalities, continued to occur. Data on whale distribution, gear distribution and configuration, and all gear observed on or taken off whales is examined. Revisions are made to the Plan by implementing regulations as new information and technology becomes available.

The Team initially concluded that all parts of gillnet and trap/pot gear can and have caused entanglements. Research and testing has been ongoing to identify risk reduction measures that are feasible. Initial measures included, among others, seasonal closures, gear marking and the introduction of weak links to facilitate a whale's ability to break through the gear (62 FR 39157, July 22, 1997). In 2000, additional or new gear modifications were implemented in the specific areas (65 FR 80368, December 21, 2000). In 2002, further gear modifications (67 FR 1300, January 10, 2002), as well as regulations requiring dynamic (67 FR 1133, January 9, 2002) and seasonal management (67 FR 1142, January 9, 2002) in response to whale occurrence were required. The regulations implemented in 2009, among others, removed the dynamic management requirement due to difficulty effectively implementing rapid requirements and needing further protection outside these areas, and in place focused broad-based sinking horizontal ground line (line between traps) requirements to remove line from the water column, required expanded gear modifications (e.g., gear marking), and regulated additional trap/pot and gillnet fisheries. The 2014 regulations focused on reducing the number of and associated risk posed by vertical buoy lines, increased the size and the frequency of the required gear marks, and expanded one of the seasonal trap/pot closures (79 FR 36586, June 27, 2014 and 79 FR 73848, December 12, 2014). In 2015, the Team determined that additional unique gear marking was needed in certain areas to better determine where entanglements are actually occurring. Therefore, the newly expanded gear marking scheme was modified further (80 FR 30367, May 28, 2015).

The 2014 regulations implementing the "vertical line strategy" prioritized risk reduction in areas where there is the greatest co-occurrence of vertical lines and large whales using whale and vertical line distribution. These data were overlaid to demonstrate the combined densities by area. A model was developed and was constructed to allow gear configuration alternatives to be manipulated to determine what relative co-occurrence reductions (as a proxy for risk) could be achieved by gear configuration changes and/or effort reductions by area. This co-occurrence analysis was an integral component of the vertical line strategy.

Plan requirements may vary by gear type (gillnet or trap/pot) and area. The major requirements include:

- No buoy line floating at the surface.
- No wet storage of gear (all gear must be hauled out of the water at least once every 30 days. In federal waters in the Southeast trap/pots must be returned to shore at the end of every trip).
- In most waters, surface buoys and buoy line need to be marked to identify the vessel or fishery.
- All buoys, flotation devices, and/or weights must be attached to the buoy line with a weak link. Specific breaking strengths may vary by area. This measure is designed so that if a large whale does become entangled, it should be able to exert enough force to break the weak link and break free of the buoy (lobster gear) or net panels (gillnet), increasing the chance of releasing the gear and reducing the risk of injury or mortality.
- In most waters, groundline must be made of sinking line.
- All buoy lines need to be marked three times (top, middle, bottom) with three marks along a 12 inch (30.5 cm) area. This measure is intended to help managers learn more about where and when entanglements occur.
- Minimum trap per trawl requirements based on area fished and miles from shore.

There are also two seasonal trap/pot closures (Figure 50) under the Plan: the Great South Channel Trap/Pot Closure (50 CFR 229.32(c)4) and the Massachusetts Restricted Area (50 CFR 229.32(c)3). Great South Channel Trap/Pot Closure prohibits fishing with, setting, or possessing trap/pot gear in this area unless stowed in accordance with §229.2 from April 1 through June 30. Cape Cod Bay is also closed to gillnet fishing from January 1 to May 15. These periods coincide with the presence of right whales in these areas. The Massachusetts Restricted Area prohibits fishing with, setting, or possessing trap/pot gear in this area unless stowed in accordance with §229.2 from February 1 to April 30. As described in the *Cumulative Effects*, the Commonwealth of Massachusetts has implemented additional measures. The current measures include a February 1st through May 15th seasonal closure of all waters under the jurisdiction of the Commonwealth to trap gear fishing. This closure does not apply to waters under the jurisdiction of the Commonwealth within Lobster Management Area 2. There is also a January 1st through May 15th closure of Cape Cod Bay and certain adjacent waters to gillnet gear. These closures can be extended beyond the end date in response to the continued presence of right whales in the waters under the jurisdiction of the Commonwealth (322 CMR 12).

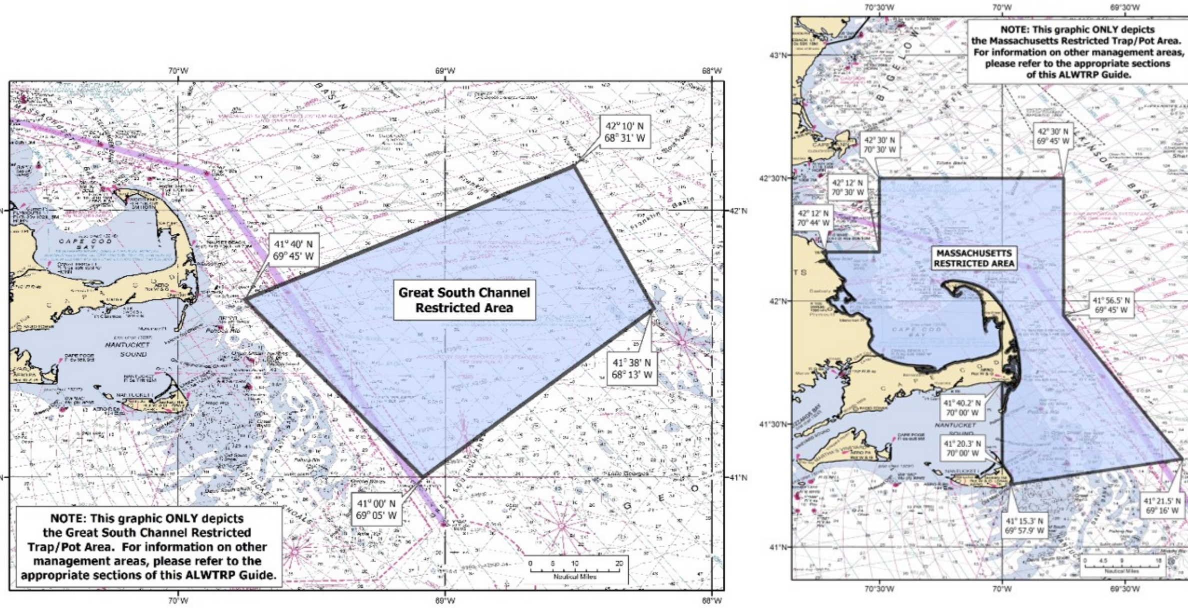


Figure 50: Great South Channel trap/pot closure and the Massachusetts Restricted Area under the ALWTRP

Non-regulatory Components of the ALWTRP

Gear research and development is a critical component of the Plan, with the aim of finding new ways of reducing the number and severity of large-gear interactions while still allowing for fishing activities. Development of gear modifications, including development of a roadmap to a future with ropeless fishing, are ongoing and are primarily used to minimize risk of large whale entanglement.

Outreach activities and products are considered important tools needed to reduce the threats to all protected species from human activities, including fishing activities. Outreach efforts to fishermen under the Plan aim to foster a more cooperative relationship between all parties interested in the conservation of threatened and endangered species. Outreach methods and tools include, but are not limited to, informative websites, NMFS presence at industry meetings, outreach events and trade shows, publications in industry trade journals and news outlets, and dockside interactions between port agents, regional gear liaisons, and industry members. Outreach guides and fact sheets have also been produced to help consolidate Plan requirements for easier access and understanding. Outreach and Plan information is also provided to the NEFSC Observer Program, USCG, and state/federal enforcement agents.

5.4.6. ***Measures to Reduce Vessel Strikes to Large Whales***

The Ship Strike Reduction Program is currently focused on protecting the North Atlantic right whale, but the operational measures are expected to reduce the incidence of ship strike on other large whales to some degree. The program consists of five basic elements and includes both regulatory and non-regulatory components. Elements of the program include:

1. operational measures for the shipping industry, including speed restrictions and routing measures.
2. section 7 consultations with federal agencies that maintain vessel fleets.
3. education and outreach programs.
4. a bilateral conservation agreement with Canada.
5. ongoing measures to reduce ship strikes of right whales (e.g., RWSAS, ongoing research into the factors that contribute to ship strikes, and research to identify new technologies that can help mariners and whales avoid each other).

Restricting Vessel Approach to Right Whales

In a right whale recovery action aimed at reducing vessel-related impacts, including disturbance, NMFS published an interim final rule in 1997 restricting vessel approach to right whales to a distance of 500 yards (62 FR 6729, February 13, 1997). The Recovery Plan for the North Atlantic right whale identified anthropogenic disturbance as one of many factors that had some potential to impede right whale recovery (NMFS 2005). With certain exceptions, the rule prohibits both boats and aircraft from approaching any right whale closer than 500 yards. Exceptions for closer approach are provided for the following situations, when:

1. compliance would create an imminent and serious threat to a person, vessel, or aircraft.
2. a vessel is restricted in its ability to maneuver around the 500-yard perimeter of a whale.
3. a vessel is investigating or involved in the rescue of an entangled or injured right whale.
4. the vessel or aircraft is participating in a permitted activity, such as a research project.

If a vessel operator has unknowingly approached closer than 500 yards, the rule requires that a course be steered away from the whale at slow, safe speed. In addition, all aircraft, except those involved in whale watching activities, are exempted from these approach regulations (50 CFR 224.103). This rule is expected to reduce the potential for vessel collisions and other adverse vessel-related effects in the *Environmental Baseline*.

Mandatory Ship Reporting System (MSR)

In April 1998, the USCG submitted, on behalf of the United States, a proposal to the International Maritime Organization (IMO) requesting approval of a mandatory ship reporting system (MSR) in two areas off the east coast of the United States: the right whale feeding grounds in the Northeast and the right whale calving grounds in the Southeast (Figure 51). The package was submitted to the IMO's Subcommittee on Safety and Navigation, as well as IMO's Marine Safety Committee; the package was approved in December 1998. On June 1, 1999, the USCG published an interim rule implementing the two MSRs (64 FR 29229); on November 20, 2001, the interim rule became final (69 FR 58066). The USCG and NOAA play important roles in helping to operate the MSR system. Ships entering the northeast and southeast MSR boundaries are required to report the vessel identity, date, time, course, speed, destination, and other relevant information. In return, the vessel receives an automated reply with the most recent right whale sightings, management areas, and information on precautionary measures to take while near right whales.

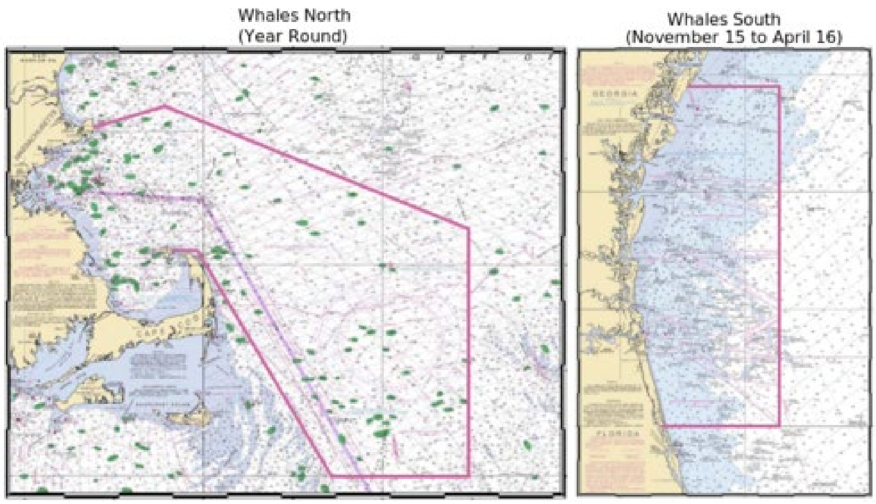


Figure 51: Mandatory ship reporting areas for North Atlantic right whales

Vessel Speed Restrictions

A key component of NOAA's right whale ship strike reduction program is the implementation of speed restrictions for vessels transiting the U.S. Atlantic in areas and seasons where right whales predictably occur in high concentrations. The Northeast Implementation Team (NEIT)-funded report "Recommended Measures to Reduce Ship Strikes of North Atlantic Right Whales" found that seasonal speed and routing measures could be an effective means of reducing the risk of ship strike along the U.S. East Coast (Russell 2001). Based on these recommendations, NMFS published regulations on October 10, 2008 to implement a 10-knot speed restriction for all vessels 65 feet (19.8 meters) or longer in Seasonal Management Areas (SMAs) (Figure 52) along the East Coast of the U.S. Atlantic seaboard at certain times of the year (73 FR 60173; October 10, 2008); on December 9, 2013, NMFS issued a final rule (78 FR 73726) eliminating the sunset provision on speed restrictions outlined in the 2008 rule. The SMAs encompass areas of high risk for whale-vessel collision along the U.S. Atlantic seaboard where right whale sightings predictably and consistently occur each year.

SMAs are supplemented by Dynamic Management Areas (DMAs) that are implemented for 15 days in areas in which right whales are sighted outside of SMA boundaries. When NOAA aerial surveys or other reliable sources report aggregations of three or more right whales in a density that indicates the whales are likely to persist in the area, NOAA calculates a buffer zone around the aggregation (Clapham and Pace 2001). The size of the DMA depends on the number of right whales sighted in the area. Mariners are requested to either avoid the area or travel through it at 10 knots (18.5 km/hr) or less. Compliance with the DMA zones is voluntary. NMFS announces the boundaries of the DMA via various mariner communication outlets, including NOAA

Weather Radio, USCG Broadcast Notice to Mariners, an interactive Google Map website, the Whale Alert app, MSR return messages, and email distribution lists, and the RWSAS.

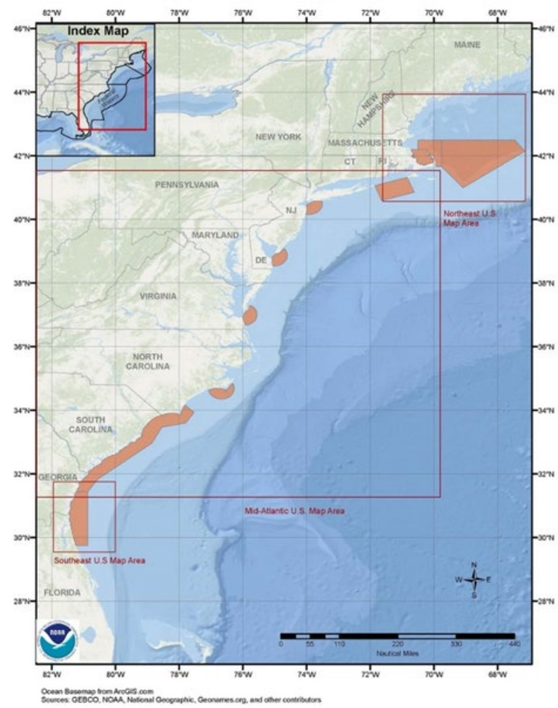


Figure 52: Seasonal Management Areas (SMAs) to protect North Atlantic right whales

NMFS's 2008 10-knot vessel speed restrictions reduced the risk of lethal strikes of right whales by 38.5 percent in waters off the southeast United States (Lagueux et al. 2011)and 56.7 percent in waters off New England (Wiley et al. 2011). A subsequent analysis using data from 1990-2012 found that, though not directly coincident with SMA implementation, North Atlantic right whale vessel-strike mortalities significantly declined from 2.0 (2000-2006) to 0.33 per year (2017-2012). Large whale vessel-strike moralities decreased inside active SMAs and increased outside active SMAs (van der Hoop et al. 2015). The research used to initiate vessel speed restrictions and studies subsequent to implementation of the regulations support continued use of the restrictions (Silber and Bettridge 2012).

NOAA recently conducted a review of the rule and assessed its effectiveness, as it pertains to right whale management (NMFS 2020g). The review found that vessel compliance varied by vessel type in active SMAs during 2018-2019. Fishing vessels showed the highest level of compliant transit (93%) while other cargo (44%) and pleasure vessels (31%) had particularly low levels of compliance (NMFS 2020g). This review concluded that since the speed rule was implemented, there has been a decline in the total number of documented right whale mortalities due to vessel strike, but an increase in serious[20] and non-serious injuries. The increase in seriously or non-seriously injured right whales by vessel interactions may be due to right whales being better able to avoid fatal vessel collisions due to slower vessel speeds, however, additional

[20] Under the MMPA, NMFS defines a serious injury as "any injury that will likely result in mortality (50 CFR 229.2) and further interprets the word "likely" as presenting a "greater than 50 percent chance of death."

analysis is required to fully evaluate this likelihood. Although the review lacked sufficient data to quantitatively assess a connection between the speed rule and the decline in observed right whale mortality due to vessel strikes, the assessment showed the speed rule had a positive effect in contributing to this change. Additionally, the review found that the speed restrictions put in place for right whales are not providing additional protection for other large whale species. The report also concluded that continued speed restrictions are warranted in light of the positive effect at reducing mortalities and also recommended that the regulations be strengthened (NMFS 2020g).

In addition to these federal regulations, the Massachusetts Division of Marine Fisheries has established similar measures to prevent strikes of whales by vessels smaller than 65 ft (20 m) (322 CMR 12.05(2) and (3)). During the period of March 1 through April 30, all vessels measuring less than 65 ft (20 m) overall length and operating within the Cape Cod Bay Restricted Speed Area[21] shall travel at a speed of 10 knots (18.5 km/hr) or less. Exemptions include inshore areas (waters within Plymouth, Kingston and Duxbury Harbors, Barnstable Harbor and Wellfleet Harbor), as well as enforcement and emergency personnel.

Vessel Routing Measures to Reduce the Co-occurrence of Ships and Whales
Another critical, non-regulatory component of NOAA's right whale ship strike reduction program involves the development and implementation of routing measures that reduce the co-occurrence of vessels and right whales, thus reducing the risk of vessel collisions. Recommended routes were developed for the Cape Cod Bay feeding grounds and Southeast calving grounds by overlaying right whale sightings data on existing vessel tracks, and plotting alternative routes where vessels could expect to encounter fewer right whales. Details of these routes were completed at the end of November 2006. The routes are charted on all NOAA electronic and printed charts, published in U.S. Coast Pilots, and sent to mariners through USCG Notices to Mariners.

Through a joint effort between NOAA and the USCG, the United States also submitted a proposal to the IMO to shift the northern leg of the existing Boston Traffic Separation Scheme (TSS) 12 degrees to the north. Overlaying sightings of right whales and all baleen whales on the existing TSS revealed that the existing TSS directly overlaps with areas of high whale densities, while an area slightly to the north showed a considerable decrease in sightings. Analysis indicated the proposed TSS would overlap with 58 percent fewer right whale sightings and 81 percent fewer sightings of all large whales (Merrick 2005, Silber and Bettridge 2012), thus considerably reducing the risk of collisions between ships and whales. The proposal was submitted to the IMO in April 2006, and was adopted by the Maritime Safety Committee in December 2006 (Silber and Bettridge 2012). The shift took effect on July 1, 2007. In 2009, this TSS was modified by narrowing the width of the north-south portion by one mile to reduce the threat of ship collisions with endangered right whales and other whale species (Silber and Bettridge 2012).

[21] The Cape Cod bay Restricted Speed Area is defined as all waters of Cape Cod Bay south of 42° 08' north latitude and those waters north and east of Cape Cod west of 70° 10' west longitude.

In 2009, NOAA and the USCG established the Great South Channel as an Area to be Avoided (ATBA). This is a voluntary seasonal ATBA for ships weighing 300 gross tons or more. The ATBA is in effect each year from April 1 to July 31 (Silber and Bettridge 2012), when right whales are known to congregate around the Great South Channel. The ATBA coupled with narrowing the TSS by one nautical mile will reduce the relative risk of right whale ship strikes by an estimated 74 percent during April-July (63 percent from the ATBA and 11 percent from the narrowing of the TSS (Merrick and Cole 2007)).

Right Whale Sighting Advisory System (RWSAS)
The Right Whale Sighting Advisory System (RWSAS) was initiated in early 1997 as a partnership among several federal and state agencies and other organizations to conduct aerial and ship board surveys to locate right whales and to alert mariners to right whale sighting locations in near real time. These right whale sighting reports are obtained from a variety of sources including the USCG, aerial surveys, shipboard surveys, whale watch vessels, and other sources (commercial ships, fishing vessels, and the general public). Right whale sightings Virginia to Maine can also be reported from by calling the NOAA hotline. In order to increase public awareness about the presence of right whales and the need to report sightings to the NOAA, right whale signs have been distributed throughout this region at boat ramps and marinas. In 2009, the Right Whale Sighting Advisory System was reengineered to support the SMA and DMA regulations to reduce the probability of lethal injury to right whales from collisions with ships (73 FR 60173, December 9, 2008).

Fishermen and other vessel operators can obtain RWSAS sighting reports, and make necessary adjustments in operations to decrease the potential for interactions with right whales. The RWSAS has also served as the only form of active entanglement monitoring in the Cape Cod Bay and Great South Channel feeding areas. Some of these sighting efforts have resulted in successful disentanglement of right whales. RWSAS flights have also contributed sightings of dead floating animals that can occasionally be retrieved, and necropsied to increase our knowledge of the biology of the species and effects of human impacts.

WhaleALERT
WhaleALERT is an application designed to augment existing ship navigation tools informing mariners of the safest and most current information to reduce the risk of ship and right whale collisions. Mariners along the U.S. east coast can now download an application that warns them when they enter areas of high risk of collision with North Atlantic right whales. The WhaleALERT app provides one source for information about right whale management measures and the latest data about right whale detections, all overlaid on NOAA digital charts.

5.4.7. ***North Atlantic Right Whale Recovery Plan Implementation Teams***
Two multi-disciplinary teams advise NMFS on issues related to the status of right whales from Maine through Virginia (Northeast Implementation Team (NEIT)) and south of Virginia (Southeast Implementation Team). These teams work collaboratively on coastwide issues and independently on regional issues. They work to coordinate and effect recovery plan implementation through involving stakeholders, promoting creative solutions, monitoring effectiveness of implementation, and identifying and prioritizing information needs. The Recovery Plan Implementation Teams include a population evaluation tool subgroup which is working to develop a population viability analysis or other assessment tool that will help us

better characterize extinction risk, taking into account current and future threats. This analysis is still under development.

5.4.8. ***Harbor Porpoise Take Reduction Plan (HPTRP)***

NMFS has implemented the HPTRP to decrease interactions between harbor porpoises and commercial gillnet gear in waters off New England and the Mid-Atlantic. Fisheries in this Opinion that are regulated under the HPTRP include Northeast multispecies, monkfish, spiny dogfish, bluefish, and skate. The HPTRP includes time and area closures and gear modification requirements. Time and area closures implemented by the HPTRP may decrease the chance of interactions between ESA-listed species that are present in the area at the time of the closure and gillnet gear. NMFS published a final rule amending the original plan on February 19, 2010 (75 FR 7383). In New England, amendments included the expansion of seasonal and temporal requirements within some existing HPTRP management areas, incorporation of additional management areas, and establishment of a consequence closure area strategy as an incentive to increase compliance and reduce bycatch levels in areas with historically high levels of harbor porpoise bycatch. In the Mid-Atlantic, amendments included the establishment of an additional management area, and modification to tie-down requirements for large mesh gillnet gear. The final rule also incorporated a research provision and amended some existing regulatory text for minor corrections and clarifications. For more information on the HPTRP including time and area closures, visit: https://www.fisheries.noaa.gov/action/harbor-porpoise-take-reduction-plan-regulations.

5.4.9. ***Bottlenose Dolphin Take Reduction Plan (BDTRP)***

Under the BDTRP, NMFS has implemented restrictions for small, medium, and large-mesh gillnets along the Atlantic coast from New Jersey to Florida and in the Virginia pound net fishery. Fisheries in this Opinion that are regulated under the BDTRP include monkfish, spiny dogfish, skate, Northeast multispecies, and bluefish. The regulatory requirements seek to reduce soak times and modify fishing practices to limit bycatch of bottlenose dolphins. These regulations may also benefit ESA-listed species that are present in the area during BDTRP regulatory measures. The take reduction team meets periodically to monitor implementation and effectiveness of the plan. For more information on the BDTRP, visit: https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan.

5.4.10. ***Magnuson-Stevens Fishery Conservation and Management Act and Atlantic Coastal Fisheries Cooperative Management Act***

There are numerous regulations mandated by the MSA and ACA that may benefit ESA-listed species. Many fisheries are subject to different time and area closures. These area closures can be seasonal, year-round, and/or gear based. Closure areas may benefit ESA-listed species due to elimination of active gear in areas where ESA-listed species are present. However, if closures shift effort to areas with a comparable or higher density of ESA-listed marine mammals, sea turtles, or fish, and/or the shift in effort results in increases in gear soak or tow time and/or quantity of fishing gear set/towed in the affected area, then risk of interaction could increase. Fishing effort reduction (i.e., landing/possession limits or trap allocations) measures may also benefit ESA-listed species by limiting the amount of time that gear is present in the species environment. Additionally, gear restrictions and modifications required for fishing regulations may also decrease the risk of capture or entanglement with endangered species. National Standard 9 of the MSA specifies conservation and management measures shall, to the extent practicable, (a) minimize bycatch and (b) to the extent bycatch cannot be avoided, minimize the mortality of such bycatch. This includes bycatch of sea turtles and ESA-listed fish. For a complete listing of fishery regulations in the action area visit: https://www.fisheries.noaa.gov/content/greater-atlantic-region-regulations.

5.4.11. ***Reducing Threats to Atlantic Sturgeon***

Several conservation actions aimed at reducing threats to Atlantic sturgeon are currently ongoing, including dam removals, moratoria on commercial and recreational fishing, and the implementation of a Sturgeon Salvage Network and educational programs for sturgeon throughout the U.S. Atlantic (e.g., SCUTES). In the near future, NMFS will be convening a recovery team, and drafting a recovery plan that will outline recovery goals, criteria, and steps necessary to recover all Atlantic sturgeon DPSs. To develop population estimates for each DPS, numerous research activities are underway, involving NMFS and other federal, state and academic partners, to obtain more information on the distribution and abundance of Atlantic sturgeon throughout their range. Guidelines developed by sturgeon researchers in cooperation with NMFS staff (Damon-Randall et al. 2010, Kahn and Mohead 2010, Moser et al. 2000) provide standardized research protocols that minimize the risk to sturgeon species from capture, handling and sampling. Efforts are also underway to better understand threats, such as poor water quality and bycatch, faced by the populations and ways to minimize these threats. Gear research is underway to design fishing gear that minimizes interactions with Atlantic sturgeon while maximizing retention of targeted fish species. Several states are in the process of preparing ESA section 10 Habitat Conservation Plans aimed at minimizing the effects of state fisheries on Atlantic sturgeon.

5.4.12. ***Reducing Threats to Atlantic Salmon***

NMFS has worked with the Maine Department of Marine Resources (MDMR), USFWS, the Penobscot Indian Nation (PIN), and other partners to pursue a range of management and research activities to mitigate and reduce the most severe threats to Atlantic salmon. We have also worked with such partners to improve our understanding of salmon abundance and population health.

Since Atlantic salmon were listed, NMFS, MDMR, USFWS, PIN, and other partners have taken a number of steps to restore ecosystem function. Numerous dams have been removed and many new fishways have been constructed in Maine (USFWS and NMFS 2019). Among these are dam removals, including the recent removal of the Great Works and Veazie Dams located on the Penobscot River. Removal of these two dams allows Atlantic salmon and other diadromous fish unimpeded access to sections of the Penobscot River that they have not had in 200 years. Several small projects, such as bypasses, fishways, culvert replacements, and barrier (including dams) removal, helped restore physical and biological features necessary to further salmon recovery in the GOM DPS. There have also been efforts on the Kennebec River and the Seasticook River (USFWS and NMFS 2019). Comprehensive efforts encompassed the work of the Penobscot River Restoration Project and NMFS' designation of the Penobscot Habitat Focus Area. In addition, active stocking and fisheries management is supporting recovery of other diadromous species. In addition, the overall threat from aquaculture to the GOM DPS has also decreased substantially over the past decade (USFWS and NMFS 2019). Other recovery activities include: (1) conducting reviews of Species Protection Plans for FERC-licensed hydroelectric projects in the GOM DPS; (2) developing fish passage guidelines; and (3) consulting with federal partners to assure that federal actions minimize harm to Atlantic salmon.

NMFS also supports several annual assessment and monitoring efforts to gain greater understanding of Atlantic salmon movement patterns and community. This information will help inform future management decisions. Among these efforts are: (1) a satellite-tagging project of adult Atlantic salmon off the coast of West Greenland to track ocean movements; (2) a fish community study in the Penobscot River estuary; and (3) telemetry studies measuring Atlantic salmon smolt survival form the Penobscot River to the Gulf of Maine and monitoring fish at Halifax, Nova Scotia.

NMFS participates in the North Atlantic Salmon Conservation Organization (NASCO), the international governing body that jointly manages Atlantic salmon. Participation in NASCO has led to the development of multi-year regulatory measures for high-seas Atlantic salmon fisheries, international guidelines for salmon stocking, mitigation of threats from aquaculture practices, and country specific Action Plans that outline the implementation of all the NASCO guidelines.

We work with international partners to conduct annual sampling of the Atlantic salmon fishery in West Greenland. From this sampling, biological information is used to confirm catch, support international Atlantic salmon stock assessments, and determine salmon continent-of-origin, while providing a platform for research evaluating the ecological health of Atlantic salmon at Greenland.

5.4.13. *Reducing Threats under CITES*

North Atlantic right, fin, sei, and sperm whales and green, Kemp's ridley, loggerhead, and leatherback sea turtles are included on Appendix I of CITES. Appendix I lists species that are the most endangered among CITES-listed species. CITES prohibits international trade in specimens of these species except when the purpose of the import is not commercial, for instances for scientific research. In these cases, import and export permits are required to authorize trade to take place.

Atlantic sturgeon and giant manta rays are listed under Appendix II. This Appendix lists species that are not necessarily now threatened with extinction but may become so unless trade is tightly controlled. It also includes species whose specimens in trade look like those species listed for conservation reasons ("look-alike species"). International trade may be authorized through permits that ensure that the products were legally acquired and that the Scientific Authority of the State of export has advised that such export will not be detrimental to the survival of that species (after taking into account factors such as its population status and trends, distribution, harvest, and other biological and ecological elements). Restrictions under CITES may help address the threat of overutilization for these species by ensuring that international trade is sustainable.

5.5. Status of the Species within the Action Area

5.5.1. *North Atlantic Right Whale*

North Atlantic right whales occur along the U.S. eastern seaboard of the Northwest Atlantic Ocean. Specifically, North Atlantic right whales occur from calving grounds in the southeastern United States (waters off of Georgia and Florida, with some suggestion that calving grounds may extend as far north as Cape Fear, North Carolina) to feeding grounds in New England waters into Canadian waters (Hayes et al. 2020; W.A. McLellan, Univ. of North Carolina Wilmington, pers. comm. as cited in Hayes et al. 2020). There is also at least one recent case of a calf apparently being born in the Gulf of Maine (Patrician et al. 2009), and another newborn was detected in Cape Cod Bay in 2013 (CCS, unpublished data, as cited in Hayes et al. 2020).

Right whales predominantly occupy waters of the continental shelf, but are also known to make lengthy excursions into deep waters off the shelf (Baumgartner and Mate 2005, Davis et al. 2017, Hayes et al. 2020, Mate et al. 1997). Offshore of the Maine coast, the likelihood of a North Atlantic right whale being present increases with distance from shore (Roberts et al. 2016). Surveys have demonstrated the existence of several areas where North Atlantic right whales congregate seasonally, including areas in the action area such as the coastal waters of the southeastern U.S.; the Great South Channel; Jordan Basin; Georges Basin along the northeastern edge of Georges Bank; Cape Cod; Massachusetts Bay; and the continental shelf south of New England (Brown et al. 2002, Cole et al. 2013, Hayes et al. 2020, Leiter et al. 2017).

In the late fall months (e.g., October), pregnant female right whales move south to their calving grounds off Georgia and Florida, while the majority of the population likely remains on the feeding grounds or disperses along the eastern seaboard. Recent research indicates our understanding of their movement patterns remains incomplete (Davis et al. 2017). A review of visual and passive acoustic monitoring data in the western North Atlantic demonstrated nearly continuous year-round presence across their entire habitat range (for at least some individuals), including in locations previously thought of as migratory corridors (e.g., waters off New Jersey and Virginia). This suggests that not all of the population undergoes a consistent annual migration (Bort et al. 2015, Cole et al. 2013, Davis et al. 2017, Hayes et al. 2020, Leiter et al. 2017, Morano et al. 2012, Whitt et al. 2013).

New England waters are important feeding habitats for right whales, where they feed primarily on copepods (Hayes et al. 2020). The distribution of right whales is linked to the distribution of their principal zooplankton prey, calanoid copepods (Baumgartner and Mate 2005, NMFS 2005, Waring et al. 2012, Winn et al. 1986). Right whale calls have been detected by autonomous passive acoustic sensors deployed between 2005 and 2010 at three sites (Massachusetts Bay, Stellwagen Bank, and Jeffreys Ledge) in the southern Gulf of Maine (Morano et al. 2012, Mussoline et al. 2012). Comparisons between detections from passive acoustic recorders and observations from aerial surveys in Cape Cod Bay between 2001 and 2005 demonstrated that aerial surveys found whales on approximately two-thirds of the days during which acoustic monitoring detected whales (Clark et al. 2010).

Recent changes in right whale distribution (Kraus et al. 2016) are driven by warming deep waters in the Gulf of Maine (Record et al. 2019). Prior to 2010, right whale movements followed the seasonal occurrence of the late stage, lipid-rich copepod *C. finmarchicus* from the western Gulf of Maine in winter and spring to the eastern Gulf of Maine and Scotian Shelf in the summer and autumn (Beardsley et al. 1996, Mayo and Marx 1990, Murison and Gaskin 1989, Pendleton et al. 2009, Pendleton et al. 2012). Warming in the Gulf of Maine has resulted in changes in the seasonal abundance of late-stage *C. finmarchicus*, with record high abundances in the western Gulf of Maine in spring and significantly lower abundances in the eastern Gulf of Maine in late summer and fall (Record et al. 2019). One of the consequences of this has been a shift of right whales out of habitats such as the Great South Channel and the Bay of Fundy, and into areas such as the Gulf of St. Lawrence in the summer and south of New England and Long Island in the fall and winter (NMFS NEFSC, unpublished data).

In summary, we anticipate individual right whales to occur year round in the action area in both coastal, shallower waters as well as offshore, deeper waters. We expect these individuals to be moving throughout the action area, making seasonal migrations, foraging in northern parts of the action area when copepod patches of sufficient density are present, and calving during the winter months in southern waters of the action area.

5.5.2. *Fin Whale*

Fin whales occurring in the North Atlantic belong to the western North Atlantic stock (Hayes et al. 2020). They are typically found along the 328-ft (100-m) isobath but also in shallower and deeper water, including submarine canyons along the shelf break (Kenney and Winn 1987). Fin whales are migratory, moving seasonally into and out of feeding areas, but the overall migration pattern is complex and specific routes are unknown (Hayes et al. 2018b). The species occur year-round in a wide range of latitudes and longitudes, but the density of individuals in any one area changes seasonally. Thus, their movements overall are patterned and consistent, but distribution of individuals in a given year may vary according to their energetic and reproductive condition, and climatic factors (NMFS 2010c).

The northern Mid-Atlantic Bight represents a major feeding ground for fin whales as the physical and biological oceanographic structure of the area aggregates prey. Fin whales in this area feed on krill (*Meganyctiphanes norvegica* and *Thysanoessa inermis*) and schooling fish such as capelin (*Mallotus villosus*), herring (*Clupea harengus*), and sand lance (*Ammodytes* spp.) (Borobia et al. 1995) by skimming the water or lunge feeding. Several studies suggest that distribution and movements of fin whales along the east coast of the United States is influenced by the availability of sand lance (Kenney and Winn 1986, Payne 1990).

Fin whales are believed to use the North Atlantic water primarily for feeding and more southern waters for calving. Movement of fin whales from the Labrador/Newfoundland region south into the West Indies during the fall have been reported (Clark 1995). However, neonate strandings along the U.S. Mid-Atlantic coast from October through January indicate a possible offshore calving area (Hain et al. 1992).

In summary, we anticipate individual fin whales to occur in the action area year-round, with the highest numbers in the spring and summer. We expect these individuals to be making seasonal coastal migrations, and to be foraging during spring and summer months.

5.5.3. *Sei Whale*

Sei whales occurring in the North Atlantic belong to the Nova Scotia stock (Hayes et al. 2020). They can be found in deeper waters of the continental shelf edge waters of the northeastern United States and northeastward to south of Newfoundland (Hain et al. 1985), and NMFS aerial surveys found substantial numbers of sei whales in this region, in particular south of Nantucket, in the spring of 2001. Sei whales often occur along the shelf edge to feed, but also may come up to shallower shelf waters. Although known to eat fish in other oceans, sei whales off the northeastern U.S. are largely planktivorous, feeding primarily on euphausiids and copepods (Flinn et al. 2002, Hayes et al. 2017a). These aggregations of prey are largely influenced by the dynamic oceanographic processes in the region. The southern portion of the species' range during spring and summer includes the northern portions of the U.S. EEZ; the Gulf of Maine and Georges Bank (Hayes et al. 2017a). Spring is the period of greatest sei whale abundance in New England waters, with sightings concentrated along the eastern margin of Georges Bank and into the Northeast Channel area, and along the southwestern edge of Georges Bank in the area of

Hydrographer Canyon (CETAP 1982). NMFS aerial surveys in 1999, 2000 and 2001 found concentrations of sei and right whales along the northern edge of Georges Bank in the spring. In years of greater abundance of copepod prey sources, sei whales are reported in more inshore locations, such as the Great South Channel (in 1987 and 1989) and Stellwagen Bank (in 1986) (Waring et al. 2014).

During seasonal aerial surveys conducted from 2011-2015 in waters off Massachusetts and Rhode Island, sei whales were observed between March and June every year, with the greatest number of sightings in May (n = 8) and June (n = 13) (Kraus et al. 2016). From 1981 to 2018, sightings data indicate that sei whales may occur in the area in relatively moderate numbers during the spring and in low numbers in the summer (North Atlantic Right Whale Consortium 2018).

In summary, we anticipate individual sei whales to occur in the action area primarily in the spring and summer months. We expect these individuals to be making seasonal migrations, and to be foraging when krill are present. Foraging adult sei whales are most common in the action area but adult sei whales with calves have been observed during spring and summer months (Kraus et al. 2016).

5.5.4. *Sperm Whale*

Sperm whales from the North Atlantic Stock regularly occur in waters of the U.S. EEZ in the Atlantic Ocean. However, the sperm whales that occur in the eastern U.S. Atlantic EEZ likely represent only a fraction of the total stock (Waring et al. 2015). Sperm whales are generally found on the continental shelf edge, over the continental slope, and into mid-ocean regions (Waring et al. 2015). This offshore distribution is more commonly associated with the Gulf Stream edge and other features (Waring et al. 1993, Waring et al. 2001). Based on reviews of many types of stock studies, (i.e., tagging, genetics, catch data, mark-recapture, biochemical markers, etc.), researchers suggested that sperm whale populations have no clear geographic structure (Dufault et al. 1999, Reeves and Whitehead 1997). In the U.S. Atlantic EEZ waters, there appears to be a distinct seasonal cycle (CETAP 1982, Scott and Sadove 1997). In winter, sperm whales are concentrated east and northeast of Cape Hatteras. In spring, the center of distribution shifts northward to east of Delaware and Virginia and is widespread throughout the central portion of the Mid-Atlantic Bight and the southern portion of Georges Bank. In summer, the distribution is similar but now also includes the area east and north of Georges Bank and into the Northeast Channel region, as well as the continental shelf (inshore of the 328-ft (100-m) isobath) south of New England. In the fall, sperm whale occurrence south of New England on the continental shelf is at its highest level, and there remains a continental shelf edge occurrence in the Mid-Atlantic Bight.

The average depth of sperm whale sightings observed during the CeTAP surveys was 5,880 ft (1,792 m) (CETAP 1982). Female sperm whales and young males usually inhabit waters deeper than 3,280 ft (1,000 m) and at latitudes less than 40° N (Whitehead 2002). Sperm whales feed on larger organisms that inhabit the deeper ocean regions (Whitehead 2002). Calving for the species occurs in low latitude waters outside of the action area.

In summary, we anticipate adult individual sperm whales to occur infrequently in deeper, offshore waters of the action area primarily in summer and fall months. We expect these individuals to be moving through the project area as they make seasonal migrations, and to be foraging along the shelf break.

5.5.5. ***Sea Turtles***

The fishing considered in this Opinion and habitat use by sea turtles overlap in the action area. Adult and/or juvenile loggerhead, leatherback, green, and Kemp's ridley sea turtles may be migrating or foraging in the areas where the fisheries will occur. As described in the *Status of the Species*, the occurrence of loggerhead, Kemp's ridley, green, and leatherback sea turtles along the U.S. Atlantic coast is primarily temperature dependent. In general, sea turtles move up the U.S. Atlantic coast from southern wintering areas to foraging grounds as water temperatures warm in the spring. The trend is reversed in the fall as water temperatures cool. By December, sea turtles have passed Cape Hatteras, returning to more southern waters for the winter (Braun-McNeill and Epperly 2002, Ceriani et al. 2012, Griffin et al. 2013, James et al. 2005b, Mansfield et al. 2009, Morreale and Standora 1998, Morreale and Standora 2005, NEFSC and SEFSC 2011, Shoop and Kenney 1992, TEWG 2009, Winton et al. 2018).

Within the action area, sea turtles are found as far north as the Gulf of Maine seasonally. They occur throughout the bays and estuaries of nearly all southeast and mid-Atlantic states and some Northeast ones as well (e.g., Cape Cod Bay, Massachusetts), from shallow waters along the shoreline and near river mouths to deeper waters of the Atlantic Ocean. They are present in Greater Atlantic Region waters from May to November each year, with the highest number of individuals present from June to October. Sea turtles arrive in waters off Virginia in late April/early May and in the Gulf of Maine in June. (Braun-McNeill and Epperly 2002, Ceriani et al. 2012, Griffin et al. 2013, Morreale and Standora 2005, Palka et al. 2017, Winton et al. 2018). Leatherback sea turtles have a similar seasonal distribution but have a more extensive range compared to the hard-shelled species (Archibald and James 2016, Dodge et al. 2014, James et al. 2005a, James et al. 2005b, Mitchell et al. 2002, Shoop and Kenney 1992).

Sea turtles have been documented in the action area through aerial and vessel surveys and satellite tracking programs and fisheries observers (Archibald and James 2016, Barco et al. 2018, James et al. 2005a, James et al. 2005b, James et al. 2005c, Kraus et al. 2016, NMFS 2015a, 2016a, 2018a, 2019a, Patel et al. 2018, Winton et al. 2018). The Atlantic Marine Assessment Program for Protected Species (AMAPPS) is a comprehensive program to assess abundance, distribution, and ecology of marine mammals, sea turtles, and seabirds throughout the U.S. Atlantic. From 2010-2018, aerial and shipboard surveys (approximately 103,132 nmi (191,000 km) of trackline) from Nova Scotia, Canada through Florida detected more than 8,000 turtles including green, Kemp's ridley, loggerhead, and leatherback turtles (Palka et al. 2017) . These sightings occurred throughout most of the action area (see AMAPPS sightings at http://seamap.env.duke.edu/). From 2010-2018, the NEFSC and Coonamessett Farm Foundation deployed 180 satellite tags on loggerhead sea turtles. Data from these satellite tags was used to assess the relative density of sea turtles (Palka et al. 2017, Winton et al. 2018). Researchers also continue to tag loggerhead and leatherback sea turtles though this program (NMFS 2015a, 2016a, 2018a, 2019a). The satellite tracks of loggerheads studied as part of the AMAPPS program are at http://www.seaturtle.org/tracking/?project_id=537&dyn=1324309895. Other studies have focused on exploring species distribution relative to prey and physical oceanography.

In the summer of 2010, as part of the AMAPPS project, the NEFSC and SEFSC estimated the abundance of juvenile and adult loggerhead sea turtles in the portion of the northwestern Atlantic continental shelf between Cape Canaveral, Florida and the mouth of the Gulf of St. Lawrence, Canada (NMFS 2011c). The abundance estimates were based on data collected from an aerial line-transect sighting survey as well as satellite tagged loggerheads. The preliminary regional abundance estimate was about 588,000 individuals (approximate inter-quartile range of 382,000-817,000) based on only the positively identified loggerhead sightings, and about 801,000 individuals (approximate inter-quartile range of 521,000-1,111,000) when based on the positively identified loggerheads and a portion of the unidentified sea turtle sightings (NMFS 2011c).

Barco et al. (2018) estimated loggerhead sea turtle abundance and density in the southern portion of the Mid-Atlantic Bight and Chesapeake Bay using data from 2011-2012. As Chesapeake Bay falls outside the action area, the focus here is on the results for the ocean waters off Virginia and Maryland. During aerial surveys, loggerhead sea turtles were the most common turtle species detected, followed by greens and leatherbacks, with few Kemp's ridleys documented. Density varied both spatially and temporally. Loggerhead abundance and density estimates in the ocean were higher in the spring (May-June) than the summer (July-August) or fall (September-October). Ocean abundance estimates of loggerheads ranged from highs of 27,508-80,503 in the spring months of May-June to lows of 3,005-17,962 in the fall months of September-October (Barco et al. 2018).

AMAPPS data, along with other sources, have been used in recent modelling studies. Winton et al. (2018) modelled the spatial distribution of satellite-tagged loggerhead sea turtles in the Western North Atlantic. The Mid-Atlantic Bight was identified as an important summer foraging area and the results suggest that the area may support a larger proportion of the population, over 50 percent of the predicted relative density of loggerheads north of Cape Hatteras from June to October (NMFS 2019a, Winton et al. 2018). Using satellite telemetry observations from 271 large juvenile and adult sea turtles collected from 2004 to 2016, the models predicted that overall densities were greatest in the shelf waters of the U.S. Atlantic coast from Florida to North Carolina (Figure 53, left side). Tagged loggerheads primarily occupied the continental shelf from Long Island, New York to Florida, with some moving offshore. Monthly variation in the Mid-Atlantic Bight (Figure 53, right side) indicated migration north to the foraging grounds from March to May and migration south from November to December. In late spring and summer, predicted densities were highest in the shelf waters from Maryland to New Jersey. In the cooler months, the predicted densities in the Mid-Atlantic Bight were higher offshore (Winton et al. 2018). South of Cape Hatteras, there was less seasonal variability and predicted densities were high in all months. Many of the individuals tagged in this area remained in the general vicinity of the tagging location. The authors did caution that the model was driven, at least in part, by the weighting scheme chosen, is reflective only of the tagged population, and has biases associated with the non-random tag deployment. Most loggerheads tagged in the Mid-Atlantic Bight were tagged in offshore shelf waters north of Chesapeake Bay in the spring. Thus, loggerheads in the nearshore areas of the Mid-Atlantic Bight may have been under-represented (Winton et al. 2018). Despite these caveats, this data is the best scientific and commercial data on loggerhead density in the action area.

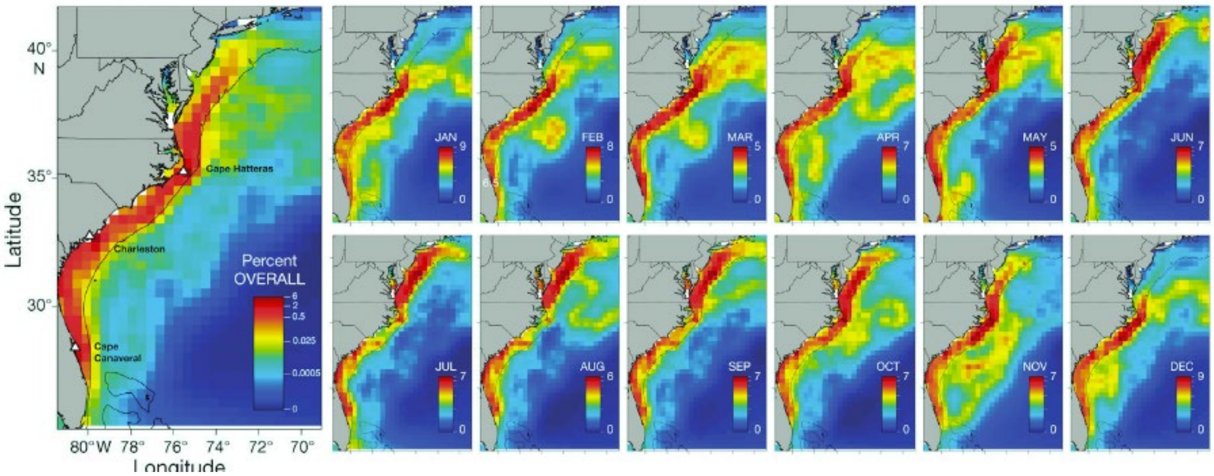


Figure 53: Overall and monthly log density of tagged loggerhead sea turtles predicted from a space-time geostatistical mixed effects model. The proportion of the predicted density in each cell is indicated by the key (Winton et al. 2018).

One of the main factors influencing sea turtle presence in mid-Atlantic waters and north is seasonal temperature patterns (Ruben and Morreale 1999). The distribution of sea turtles is limited geographically and temporally by water temperatures (Braun-McNeill et al. 2008, Epperly 1995, James et al. 2006b, Mansfield et al. 2009), with warmer waters in the late spring, summer, and early fall being the most suitable. Water temperatures too low or too high may affect feeding rates and physiological functioning (Milton and Lutz 2003); metabolic rates may be suppressed when a sea turtle is exposed for a prolonged period to temperatures below 8-10° C (George 1997, Milton and Lutz 2003, Morreale et al. 1992). That said, loggerhead sea turtles have been found in waters as low as 7.1-8 ° C (Braun-McNeill et al. 2008, Smolowitz et al. 2015, Weeks et al. 2010). However, in assessing critical habitat for loggerhead sea turtles, the review team considered the water-temperature habitat range for loggerheads to be above 10° C (NMFS 2013a). Sea turtles are most likely to occur in the action area when water temperatures are above this temperature, although depending on seasonal weather patterns and prey availability, they could be also present in months when water temperatures are cooler (as evidenced by fall and winter cold stunning records as well as year round stranding records).

To better understand loggerhead behavior on the Mid-Atlantic foraging grounds, Patel et al. (2016) used a remotely operated vehicle (ROV) to document the feeding habitats (and prey availability), buoyancy control, and water column use of 73 loggerheads recorded from 2008-2014. When the mouth and face were in view, loggerheads spent 13 percent of the time feeding on non-gelatinous prey and 2 percent feeding on gelatinous prey. Feeding on gelatinous prey occurred near the surface to depths of 52.5 ft (16 m). Non-gelatinous prey were consumed on the bottom. Turtles spent approximately 7 percent of their time on the surface (associated with breathing), 42 percent in the near surface region, 44 percent in the water column, 0.4 percent near bottom, and 6 percent on bottom. When diving to depth, turtles displayed negative buoyancy, making staying at the bottom easier (Patel et al. 2016).

Patel et al. (2018) evaluated temperature-depth data from 162 satellite tags deployed on loggerhead sea turtles from 2009 to 2017 when the water column is highly stratified (June 1 – October 4). Turtles arrived in the Mid-Atlantic Bight in late May as the Cold Pool formed and departed in early October when the Cold Pool started to dissipate. The Cold Pool is an oceanographic feature that forms annually in late May. During the highly stratified season, tagged turtles were documented throughout the water column from June through September. Fewer bottom dives occurred north of Hudson Canyon early (June) and late (September) in the foraging season (Patel et al. 2018).

Satellite tagging studies have also been used to understand leatherback sea turtle behavior and movement in the action area (Dodge et al. 2014, Dodge et al. 2015, Eckert et al. 2006, James et al. 2005a, James et al. 2005b, James et al. 2006a). These studies show that leatherback sea turtles move throughout most of the North Atlantic from the equator to high latitudes. Key foraging destinations include, among others, the eastern coast of United States (Eckert et al. 2006). Telemetry studies provide information on the use of the water column by leatherback sea turtles. Based on telemetry data for leatherbacks (n=15) off Cape Cod, Massachusetts, leatherback turtles spent over 60 percent of their time in the top 33 ft (10 m) of the water column and over 70 percent in the top 49 ft (15 m) (Dodge et al. 2014). Leatherbacks on the foraging grounds moved with slow, sinuous area-restricted search behaviors. Shorter, shallower dives were taken in productive, shallow waters with strong sea surface temperature gradients. They were highly aggregated in shelf and slope waters in the summer, early fall, and late spring. During the late fall, winter, and early spring, they were more widely dispersed in more southern waters and neritic habitats (Dodge et al. 2014). Leatherbacks (n=24) tagged in Canadian waters primarily used the upper 98 ft (30 m) of the water column and had shallow dives (Wallace et al. 2015).

Dodge et al. (2018) used an autonomous underwater vehicle (AUV) to remotely monitor fine-scale movements and behaviors of nine leatherbacks off Cape Cod, Massachusetts. The "TurtleCam" collected video of tagged leatherback sea turtles and simultaneously sampled the habitat (e.g., chlorophyll, temperature, salinity). Representative data from one turtle was reported in Dodge et al. (2018). During the 5.5 hours of tracking, the turtle dove continuously from the surface to the seafloor (0-66 ft (0-20 m)). Over a two-hour period, the turtle spent 68 percent of its time diving, 16 percent swimming just above the seafloor, 15 percent at the surface and 17 percent just below the surface. The animal frequently surfaced (>100 times in ~2 hours). The turtle used the entire water column, feeding on jellyfish from the seafloor to the surface. The turtle silhouetted prey 36 percent of the time, diving to near/at bottom and looking up to locate prey. The authors note that silhouetting prey may increase entanglement in fixed gear if a buoy of float is mistaken for jellyfish (Dodge et al. 2018).

5.5.6. *Atlantic Sturgeon*

The marine and estuarine range of all five Atlantic sturgeon DPSs overlaps and extends from Canada through Cape Canaveral, Florida. Based on the best available scientific and commercial data, Atlantic sturgeon originating from any of five DPSs could occur in the waters of the action area (Damon-Randall et al. 2013, Wirgin et al. 2015b). Eggs, early life stages, and juveniles (as used here referring to Atlantic sturgeon offspring that have not emigrated from the natal river) are not present in the action area. Sub-adult and adult Atlantic sturgeon occur in waters off the Northeast and Mid-Atlantic year round. Atlantic sturgeon are known to use the action area for migration and foraging. Foraging behaviors typically occur in areas where suitable forage and appropriate habitat conditions are present. These areas include tidally influenced flats and mud, sand, and mixed cobble substrates (Stein et al. 2004b). Within the marine range of Atlantic sturgeon, several marine aggregation areas have been identified adjacent to estuaries and/or coastal features formed by bay mouths and inlets along the U.S. eastern seaboard. Depths in these areas are generally no greater than 82 ft (25 m) (Dunton et al. 2010, Erickson et al. 2011, Laney et al. 2007, Stein et al. 2004b). Given the depth range, it is expected that these identified aggregations are primarily in state waters. The fisheries under the ten FMPs and Atlantic sturgeon do overlap, suggesting that if suitable forage and/or habitat features are present, adults and sub-adults from any of the five listed DPSs may be foraging or undertaking migrations in the areas where fishing activities will occur.

5.5.7. *Atlantic Salmon*

Atlantic salmon also use the action area as a migratory route and for foraging. Upon completion of the physiological transition to salt water, the post-smolt Atlantic salmon grows rapidly and has been documented to move in small schools loosely aggregated close to the surface (Dutil and Coutui 1988). After entering into the nearshore waters of Canada, the U.S. post-smolts become part of a mixture of stocks of Atlantic salmon from various North American streams. Their diet includes invertebrates, amphipods, euphausiids, and fish (Fraser 1987, Hislop and Shelton 1993, Hislop and Youngson 1984, Jutila and Toivonen 1985). Results from a 2001-2005 post-smolt trawl survey in Penobscot Bay and the nearshore waters of the Gulf of Maine indicate that Atlantic salmon post-smolts are prevalent in the upper water column (Sheehan et al. 2005).

Most of the GOM DPS-origin salmon spend two winters in the ocean before returning to streams for spawning. Aggregations of Atlantic salmon may still occur after the first winter at sea, but most evidence indicates that they travel individually (Reddin 1985). At this stage, Atlantic salmon primarily eat fish, feeding upon capelin, herring, and sand lance (Hansen and Pethon 1985, Hislop and Shelton 1993, Reddin 1985).

5.5.8. *Giant Manta Ray*

In the Atlantic Ocean, it is unlikely that overutilization as a result of bycatch mortality is a significant threat to giant manta rays (Miller and Klimovich 2017). However, information is severely lacking on both population sizes and distribution of the giant manta ray as well as current catch and fishing effort on the species throughout this portion of its range (Figure 48).

Based on the giant manta ray’s distribution, the species may occur in coastal, nearshore, and pelagic waters off the U.S. east coast. Along the U.S. East Coast, giant manta rays are usually found in water temperatures between 19 and 22 °C (Miller and Klimovich 2017) and have been observed as far north as New Jersey. In the action area, very little information on *M. birostris* populations is available. Based on personal observation during aerial surveys conducted off of St. Augustine, Florida, from 2009-2012, F. Young (pers. comm. 2017) noted vast schools of giant manta rays, with over 500 manta rays observed per 6-8 hour day of aerial survey. Given that the species is rarely identified in the fisheries data in the Atlantic, it may be assumed that populations within the Atlantic are small and sparsely distributed (Miller and Klimovich 2017).

5.6. Impact of the Environmental Baseline on ESA-Listed Species

Collectively, the stressors described above have had, and likely continue to have, lasting impacts on the ESA-listed species considered in this consultation. Some of these stressors (e.g., vessel strike, entanglement) result in mortality or serious injury to individual animals, whereas others (e.g., a fishery that impacts prey availability) result in more indirect or non-lethal impacts. Assessing the aggregate impacts of these stressors on species is difficult, especially since many of the species in this Opinion are wide ranging and subject to stressors in locations throughout the action area and outside the action area.

We consider the best indicator of the aggregate impact of the *Environmental Baseline* on ESA-listed resources to be the status and trends of those species. As noted in the *Status of the Species*, some of the species considered in this consultation are experiencing increases in population abundance, some are declining, and for others, their status remains unknown. In considering these trends, we must also consider that some are based on a proxy for the overall population. For example, sea turtle trends are primarily based on nesting data that assesses a subset of the population. The trends must be considered in this context. Taken together, this indicates that the *Environmental Baseline* is impacting species in different ways. The species experiencing increasing population abundances are doing so despite the potential negative impacts of the *Environmental Baseline*. Therefore, while the *Environmental Baseline* may slow their recovery, recovery is not being prevented. For the species that may be declining in abundance, it is possible that the suite of conditions described in the *Environmental Baseline* is preventing their recovery. However, it is also possible that their populations are at such low levels (e.g., due to historic commercial whaling) that even when the species' primary threats are removed, the species may not be able to achieve recovery. At small population sizes, species may experience phenomena such as demographic stochasticity, inbreeding depression, and allee effects [22], among others, that cause their limited population size to become a threat in and of itself. A thorough review of the status and trends of each species is discussed in the *Status of Species* section of this Opinion.

6. CLIMATE CHANGE

The discussion below presents background information on global climate change, as well as information on past and predicted future effects of global climate change throughout the range of the listed species considered here. Additionally, we present the available information on predicted effects of climate change in the action area and how those predicted environmental changes may affect listed species. Climate change is relevant to the *Status of the Species*, *Environmental Baseline*, and *Cumulative Effects* sections of this Opinion. Therefore, rather than include partial discussions in several sections of this Opinion, we are synthesizing this information into one discussion. Consideration of the effects of the proposed action in light of predicted changes in environmental conditions due to anticipated climate change are included in the *Effects of the Action* below (see section 7).

6.1. Background Information on Global Climate Change

In a special report "Global Warming of 1.5 °C", the Intergovernmental Panel on Climate Change (IPCC) found that human activities are estimated to have caused approximately 1 °C (likely range 0.8 °C to 1.2 °C) of global warming over pre-industrial levels. It is likely to reach 1.5 °C between 2030 and 2050 under current conditions (high confidence) (IPCC 2018). Reflecting this trend, observed global mean sea surface temperature (SST) for 2006-2015 was 0.87 °C likely between 0.1 °C and 0.3 °C higher than the average from 1850-1900 (very high confidence)

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[22] Demographic stochasticity is caused by random independent events of individual mortality and reproduction which cause random fluctuations in population growth rate. It is most strong in small populations. Inbreeding depression is the reduced biological fitness of a population from breeding of related individuals, inbreeding. As described earlier, allee effects are broadly characterized as a decline in individual fitness in populations with a small size or density.

(IPCC 2018). Ocean temperatures on the Northeast Continental Shelf have risen an average of 0.03 °C each year between 1982–2013, which is three times faster than the global average observed during this time period (Pershing et al. 2015). From 2007-2016, the regional trend increased 0.14 °C per year, four times faster than the national average (Dupigny-Giroux et al. 2018). In the Northwest Atlantic, 2012 was a particularly anomalous year rising a full 2 °C above the average temperature between 1982 and 2011 (Dupigny-Giroux et al. 2018).

Model projections of global mean sea level rise (relative to 1986-2005) suggest an indicative range of 0.85-2.5 ft (0.260 0.77 m) by 2100 for 1.5 °C of global warming which is 0.32 ft (0.1 m) less than for global warming of 2 °C (medium confidence). Sea level rise is expected to continue well beyond 2100 (high confidence) and the magnitude and rate of rise depend on future emission pathways (IPCC 2018). Temperature increase will very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. Climate warming has also resulted in increased river discharge and glacial and sea-ice melting (Greene et al. 2008).

Ocean temperature in the U.S. Northeast Shelf and surrounding Northwest Atlantic waters have warmed faster than the global average over the last decade (Pershing et al. 2015). New projections for the U.S. Northeast Shelf and Northwest Atlantic Ocean suggest that this region will warm two to three times faster than the global average; given this, existing projections from the IPCC may be too conservative (Saba et al. 2015).

The past few decades have witnessed major changes in ocean circulation patterns in the Arctic, and these were accompanied by climate associated changes as well (Greene et al. 2008). Shifts in atmospheric conditions have altered Arctic Ocean circulation patterns and the export of freshwater to the North Atlantic . Large discharges of freshwater into the North Atlantic subarctic seas can lead to intense stratification of the upper water column and a disruption of North Atlantic Deep Water (NADW) formation (Greene et al. 2008, IPCC 2007). There is evidence that the NADW has already freshened significantly (IPCC 2007). This in turn can lead to a slowing down of the global ocean thermohaline (large-scale circulation in the ocean that transforms low-density upper ocean waters to higher density intermediate and deep waters and returns those waters back to the upper ocean), which can have climatic ramifications for the entire world (Greene et al. 2008). Changes in salinity and temperature are also thought to be the result of changes in the Earth's atmosphere caused by anthropogenic forces (IPCC 2007). Specifically, recent research on the North Atlantic Oscillation (NAO), which impacts climate variability throughout the Northern Hemisphere, has found potential changes in NAO characteristics under future climate change until 2100 (Hanna and Cropper 2017).

Global warming of 1.5 °C is projected to shift the ranges of many marine species to higher latitudes and drive the loss of coastal resources. The risk of irreversible loss of many marine and coastal ecosystems increases with global warming, especially at 2 °C or higher (high confidence) (IPCC 2018). There is a high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as changes in ice cover, salinity, oxygen levels, and circulation. Changes to the marine ecosystem due to climate change may result in changes in the distribution and abundance of the prey for protected species.

While predictions are available regarding potential effects of climate change globally, it is more difficult to assess the potential effects of climate change on smaller geographic scales, such as the action area. The effects of future change will vary greatly among coastal regions of the United States. For example, sea level rise is projected to be worse in low-lying coastal areas where land is sinking (e.g., the Gulf of Mexico) than in areas with higher, rising coastlines (e.g., Alaska) (Jay et al. 2018). Climate change can cause or exacerbate direct stress on ecosystems through high temperatures, a reduction in water availability, and altered frequency of extreme events and severe storms. As climate warms, water temperatures in streams and rivers are likely to increase; this will likely result wide-ranging effects to aquatic ecosystems. Changes in temperature will be most evident during low flow periods when the water column in waterways are more likely to warm beyond the physiological tolerance of resident species (NAST 2000). Low flow can impede fish entry into waterways and low flow combined with high temperatures can reduce survival and recruitment in anadromous fishes like Atlantic salmon (Jonsson and Jonsson 2009).

Expected consequences of climate change for river systems are wide ranging. Rivers are already under a great deal of stress due to excessive water withdrawal or land development, and this stress may be exacerbated by changes in climate (Hulme 2005). Rivers could experience a decrease in the amount of dissolved oxygen in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch et al. 2000). Increased water volume in a warmer-wetter climate could ameliorate poor water quality conditions in places where human-caused concentrations of nutrients and pollutants currently degrade water quality (Murdoch et al. 2000). Increases in water temperature and changes in seasonal patterns of runoff will very likely disturb fish habitat and affect recreational uses of lakes, streams, and wetlands. Surface water resources along the U.S. Atlantic coast are intensively managed with dams and channels and almost all are affected by human activities; in some systems water quality is either below recommended levels or nearly so. Within 50 years, river basins that are impacted by dams or by extensive development will experience greater changes in discharge and water stress than unimpacted, free-flowing rivers (Palmer et al. 2008). Given this, a global analysis of the potential effects of climate change on river basins indicates that large river basins impacted by dams will need a higher level of reactive or proactive management interventions in response to climate change than basins with free-flowing rivers (Palmer et al. 2008). Human-induced disturbances also influence coastal and marine systems, often reducing the ability of the systems to respond and/or adapt to change. Given the above, under a continually changing environment, maintaining healthy riverine ecosystems will likely require adaptive management strategies (Hulme 2005).

6.2. Species Specific Information on Climate Change Effects

6.2.1. *Right, Fin, Sei, and Sperm Whales*

The impact of climate change on cetaceans is likely to be related to habitat degradation and changes in prey availability caused by changes in sea temperatures and circulation, decreased salinity, sea level rise, and ocean acidification (Davis et al. 2020, Simmonds and Eliott 2009). Additional indirect physiological effects include changes in reproductive capacity and susceptibility to diseases (Simmonds and Eliott 2009). Of the main factors affecting distribution of cetaceans, water temperature can have a large influence on geographic ranges of cetacean species (MacLeod 2009). Changes in water temperature due to climate change may affect the distribution of cetaceans (Becker et al. 2018). In general, baleen whales have experienced a northward shift since 2010, matching the shifts in their prey distribution (Davis et al. 2020). MacLeod (2009) found that range changes due increases in water temperature may have a favorable impact on or may not change the conservation status of 69 percent of baleen whales and three species of sperm whales. However, the findings differ by species. For species considered in this Opinion, MacLeod (2009) found that the conservation implication for sei whales was favorable, while it would be unchanged for fin and sperm whales. For right whales, increases in water temperature may result in longer migration routes, resulting in unfavorable implications for their conservation status. The difference between how climate change affects these species could be due to the difference in the way these species feed, as right whales filter feed through dense patches of prey, other species can gulp feed and target additional prey species (Davis et al. 2020). Macleod (2009) recommended further research to assess whether these predictions are correct. A more recent study evaluated a broader range of factors, applying a qualitative framework to assess the vulnerability of seven cetacean species in the Northeast Atlantic to climate change. Factors considered included population size, geographic distribution, diet diversity, migration, human activities, genetic variability, and IUCN status. Fin and sperm whales were the only large whales evaluated, and they were found to have a higher vulnerability to climate change than the other species in the study. While geographic distribution did not contribute to the vulnerability score for these species, all other factors contributed to varying degrees to the higher risk score (Sousa et al. 2019). A framework for assessing climate change impacts (Lettrich et al. 2019) is currently being applied to assess the vulnerability of marine mammal stocks to climate change in the Northwest Atlantic, Gulf of Mexico, and Caribbean.

In regards to marine mammal prey species, there are many potential effects that global climate change may have on prey abundance and distribution. These effects on prey availability, in turn, pose potential behavioral and physiological effects to marine mammals. Species with more dietary flexibility are expected to be more resilient to fluctuations in prey resources (Gavrilchuk et al. 2014) than specialists and therefore, are less likely to be impacted by changes in prey resources. For example, ocean warming has had a significant impact on the plankton ecology of the Gulf of Maine, including effects on *Calanus finmarchicus*, a primary prey species for North Atlantic right whales (Greene 2016, Record et al. 2019). Decreases in zooplankton (*C. finmarchicus*) prey abundance have been correlated with reduced productivity of North Atlantic right whales (Meyer-Gutbrod et al. 2015). This is an important consideration for right whale recovery. Although warming in the Gulf of Maine has resulted in changes in the seasonal abundance of late-stage *C. finmarchicus*, with record high abundances in the western Gulf of Maine in spring and significantly lower abundances in the eastern Gulf of Maine in late summer and fall (Record et al. 2019), it is projected that *C. finmarchicus* concentrations will decrease within the U.S. Northeast Continental Shelf by the end of the century (Grieve et al. 2017). However, there are still many complex processes that need to be investigated when considering fluctuations of *C. finmarchicus* in the Northwest Atlantic (Runge et al. 2014). Cephalopods such as squid dominate the diet of sperm whales, who would likely re-distribute following changes in the distribution and abundance of their prey. If, however, cephalopod populations collapse or decline dramatically, sperm whales would likely decline as well. Long-term shifts of sperm whale prey in the California Current have been attributed to the redistribution of their prey resulting from climate-based shifts in oceanographic variables (Salvadeo et al. 2011).

More information is needed to determine the potential impacts of global climate change on the timing and extent of population movements, abundance, recruitment, distribution, and species composition of prey (Learmonth et al. 2006). Changes in climate patterns, ocean currents, storm frequency, rainfall, salinity, melting ice, and an increase in river inputs/runoff (nutrients and pollutants) will all directly affect the distribution, abundance and migration of prey species (Learmonth et al. 2006, Tynan and DeMaster 1997, Waluda et al. 2001). These changes will likely have several indirect effects on marine mammals, which may include changes in distribution; displacement from ideal habitats; decline in individual and population fitness; increased susceptibility to disease and contaminants; and changes in abundance, migration patterns, community structure, and reproductive success (Jenssen 2006, MacLeod 2009, Simmonds and Eliott 2009). Global climate change may also indirectly affect marine mammals via changes to the range and abundance of competitors and predators (Learmonth et al. 2006).

Recent reviews have continued to recommend additional research on the impacts of climate change on protected species (Hare et al. 2016a, NMFS 2012f, 2017e, 2019d). For example, research is needed on understanding climate change effects on North Atlantic right whale foraging, migration, habitat use, reproduction, and distribution (NMFS 2017e). The effects of a changing climate on right whale life history and distribution is one of a number of complex factors limiting their recovery (NMFS 2017e). The future NMFS marine mammal climate vulnerability assessment, expected to publish in 2021, will help provide additional information on which species are most vulnerable to help guide management.

6.2.2. ***Sea Turtles***

Sea turtle species have persisted for millions of years. They are ectotherms, meaning that their body temperatures depends on ambient temperatures. Throughout this time they have experienced wide variations in global climate conditions and are thought to have previously adapted to these changes through changes in nesting phenology and behavior (Poloczanska et al. 2009). Given this, climate change at normal rates (thousands of years) is not thought to have historically been a problem for sea turtle species. However, at the current rate of global climate change, future effects to sea turtles are probable. Climate change has been identified as a threat to all species of sea turtles found in the action area (Conant et al. 2009, NMFS and USFWS 2013, NMFS et al. 2011, Seminoff et al. 2015). However, trying to assess the likely effects of climate change on sea turtles is extremely difficult given the uncertainty in all climate change models, the difficulty in determining the likely rate of temperature increases, and the scope and scale of any accompanying habitat or behavior effects. In the Northwest Atlantic, specifically, loggerhead, green, and leatherback sea turtles are predicted to be among the more resilient species to climate change, while Kemp's ridley turtles are among the least resilient (Fuentes et al. 2013). Leatherbacks may be more resilient to climate change in the Northwest Atlantic because of their wide geographic distribution, low nest-site fidelity, and gigantothermy (Dutton et al. 1999, Fuentes et al. 2013, Robinson et al. 2009). Gigantothermy refers to the leatherbacks ability to use their large body size, peripheral tissues as insulation, and circulatory changes in thermoregulation (Paladino et al. 1990). Leatherbacks achieve and maintain substantial differentials between body and ambient temperatures through adaptations for heat production, including adjustments of the metabolic rate, and retention (Wallace and Jones 2008). However, modeling results show that global warming poses a "slight risk" to females nesting in French Guiana and Suriname relative to those in Gabon/Congo and West Papua, Indonesia (Dudley et al. 2016).

Sea turtles are most likely to be affected by climate change due to:

1. changing air/land temperatures and rainfall at nesting beaches that could affect reproductive output including hatching success, hatchling emergence rate, and hatchling sex ratios.

2. sea level rise, which could result in a reduction or shift in available nesting beach habitat, an increased risk of erosion and nest inundation, and reduced nest success.
3. changes in the abundance and distribution of forage species, which could result in changes in the foraging behavior and distribution of sea turtle species as well as changes in sea turtle fitness and growth.
4. changes in water temperature, which could possibly lead to a shift in their range, changes in phenology (timing of nesting seasons, timing of migrations) and different threat exposure.
5. increased frequency and severity of storm events, which could impact nests and nesting habitat, thus reducing nesting and hatching success.

Current approaches have limited power to predict the magnitude of future climate change, associated impacts, whether and to what extent some impacts will offset others, or the adaptive capacity of this species. Within the next decade, sea surface temperatures are expected to rise less than 1 °C. It is unknown if that is enough of a change to contribute to shifts in the range, distribution and recruitment of sea turtles or their prey. Theoretically, we expect that as waters in the action area warm, more sea turtles could be present or present for longer periods.

As climate continues to warm, feminization of sea turtle populations is a concern for many sea turtle species, which undergo temperature-dependent sex determinations. Rapidly increasing global temperatures may result in warmer incubation temperatures and higher female-biased sex ratios (Glen and Mrosovsky 2004, Hawkes et al. 2009). Increases in precipitation might cool beaches (Houghton et al. 2007); thereby, mitigating some impacts relative to increasing sand temperature. Feminization occurs over a small temperature range (1-4 °C) (Wibbels 2003) and several populations in the action area already are female biased (Gledhill 2007, Laloë et al. 2016, Patino-Martinez et al. 2012, Witt et al. 2010). The existing female bias among juvenile loggerhead sea turtles is estimated at approximately three to two females per males (Witt et al. 2010). Feminization is a particular concern in tropical nesting areas where over 95 percent female biased nests are already suspected for green turtles, and leatherbacks are expected to cross this threshold within a decade (Laloë et al. 2014, Laloë et al. 2016, Patino-Martinez et al. 2012). It is possible for populations to persist, and potentially increase with increased egg production, with strong female biases (Broderick et al. 2000, Coyne and Landry 2007, Godfrey et al. 1999, Hays et al. 2003), but population productivity could decline if access to males becomes scarce (Coyne 2000). Low numbers of males could also result in the loss of genetic diversity within a population. Behavioral changes could help mitigate the impacts of climate change, including shifting breeding season and location to avoid warmer temperatures. For example, the start of the nesting season for loggerheads has already shifted as the climate has warmed (Weishampel et al. 2004). Nesting selectivity could also help mitigate the impacts of climate on sex ratios as well (Kamel and Mrosovsky 2004).

At St. Eustatius in the Caribbean, there is an increasing female biased sex ratio of green turtle hatchlings (Laloë et al. 2016). While this is partly attributable to imperfect egg hatchery practices, global climate change is also implicated as a likely cause as warmer sand temperatures at nesting beaches can result in the production of more female embryos. At this time, we do not know how much of this bias is also due to hatchery practices as opposed to temperature. Global warming may exacerbate this female skew. An increase in female bias is predicted in St. Eustatius, with only 2.4 percent male hatchlings expected to be produced by 2030 (Laloë et al. 2016). The study also evaluated leatherback sea turtles on St. Eustatius. The authors found that the model results project the entire feminization of the green and leatherback sea turtles due to increased air temperature within the next century (Laloë et al. 2016). The extent to which sea turtles may be able to cope with this change, by selecting cooler areas of the beach or shifting their nesting distribution to other beaches with smaller increases in sand temperature, is currently unknown.

Several leatherback nesting areas are already predominantly female, a trend that is expected to continue with some areas expecting at least 95 percent female nests by 2028 (Gledhill 2007, Laloë et al. 2016, Patino-Martinez et al. 2012). Hatchling success has declined in St. Croix (Garner et al. 2017), though there is some evidence that the overall trend is not climate or precipitation related (Rafferty et al. 2017). Excess precipitation is known to negatively impact hatchling success in wet areas but can have a positive effect in dry climates (Santidrián Tomillo et al. 2015). In Grenada, increased rainfall (another effect of climate change) was found to have a cooling influence on leatherback nests, so that more male producing temperatures (less than 29.75 °C) were found within the clutches (Houghton et al. 2007). There is also evidence for very wet conditions inundating nests or increasing fungal and mold growth, reducing hatching success (Patino-Martinez et al. 2014). Very dry conditions may also affect embryonic development and decrease hatchling output. Leatherbacks have a tendency towards individual nest placement preferences, with some clutches deposited in the cooler tide zone of beaches and have relatively weak nesting site fidelity; this may mitigate the effects of long-term changes in climate on sex ratios (Fuentes et al. 2013, Kamel and Mrosovsky 2004).

If nesting can shift over time or space towards cooler sand temperatures, these effects may be partially offset. A shift towards earlier onset of loggerhead nesting was associated with an average warming of 0.8 °C in Florida (Weishampel et al. 2004). Early nesting could also help mitigate some effects of warming, but has also been linked to shorter nesting seasons in this population (Pike et al. 2006), which could have negative effects on hatchling output. Nesting beach characteristics, such as the amount of precipitation and degree of shading, can effectively cool nest temperatures (Lolavar and Wyneken 2015). However, current evidence suggests that the degree of cooling resulting from precipitation and/or shading effects is relatively small and therefore, even under these conditions, the production of predominantly female nests is still possible (Lolavar and Wyneken 2015). However, the impact of precipitation, as well as humidity and air temperature, on loggerhead nests is site specific and data suggest temperate sites may see improvements in hatchling success with predicted increases in precipitation and temperature (Montero et al. 2018, Montero et al. 2019). Conversely, tropical areas already produce 30 percent less output than temperate regions and reproductive output is expected to decline in these regions (Pike 2014).

Warming sea temperatures are likely to result in a shift in the seasonal distribution of sea turtles in the action area. In the northern part of the action area, sea turtles may be present earlier in the year if northward migrations from their southern overwintering grounds begin earlier in the spring. Likewise, if water temperatures are warmer in the fall, sea turtles could remain in the more northern areas later in the year. Potential effects of climate change include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson et al. 2009). McMahon and Hays (2006) reported that warming has caused a generally northerly migration of the 15 °C SST isotherm from 1983 to 2006. In response to this, leatherbacks have expanded their range in the Atlantic north by 330 km (McMahon and Hays 2006). An increase in cold stunning of Kemp's ridley sea turtles in New England has also been linked to climate change and could pose an additional threat to population resilience (Griffin et al. 2019).

In addition, although nesting occurs in the south and mid-Atlantic (i.e., North Carolina and into Virginia), recent observations have caused some speculation that the nesting range of some sea turtle species may shift northward as the climate warms and that nest crowding may increase as sea level rises and available nesting habitat shrinks (Reece et al. 2013). Recent instances include a Kemp's ridley nesting in New York in July 2018 (96 hatchlings), a loggerhead nesting in Delaware in July 2018 (48 hatchlings), and a loggerhead nesting in Maryland in September 2017 (7 live hatchlings). The ability to shift nesting in time and space towards cooler areas could reduce some of the temperature-induced impacts of climate change (e.g., female biased sex ratio). Fuentes et al. (2020) modelled the geographic distribution of climatically suitable nesting habitat for sea turtles in the U.S. Atlantic under future climate scenarios, identified potential range shifts by 2050, determined sea-level rise impacts, and explored changes in exposure to coastal development as a result of range shifts. Overall, the researchers found that, with the exception of the northern nesting boundaries for loggerhead sea turtles, the nesting ranges were not predicted to change. Fuentes et al. (2020) noted that range shifts may be hindered by expanding development. They also found that loggerhead sea turtles would experience a decrease (10 percent) in suitable nesting habitat followed by declines in nesting habitat for green turtles. No significant changes was predicted in the distribution of climatically suitable nesting area for leatherbacks by 2050. Sea level rise is projected to inundate current habitats; however, new beaches will also be formed and suitable habitats could be gained, with leatherback sea turtles potentially experience the biggest gain in suitable habitat (Fuentes et al. 2020).

Despite site-specific vulnerabilities of the Northwest Atlantic Ocean loggerhead DPS, this DPS may be more resilient to changing climate than other management units (Fuentes et al. 2013). Van Houtan and Halley (2011) recently developed climate based models to investigate loggerhead nesting (considering juvenile recruitment and breeding remigration) in the Northwest Atlantic and North Pacific. These models found that climatic conditions and oceanographic influences explain loggerhead nesting variability. Specifically, the climate models alone explained an average 60 percent (range 18 percent-88 percent) of the observed nesting changes in the Northwest Atlantic and North Pacific over the past several decades. In terms of future nesting projections, modeled climate data predict a positive trend for Florida nesting (the Northwest Atlantic Ocean DPS), with increases through 2040 as a result of the Atlantic Multidecadal Oscillation (Van Houtan and Halley 2011). In a separate model, Arendt et al (2013) suggested that the variability represents a lagged perturbation response to historical anthropogenic impacts. The nest count increases since 2008 may reflect a potential recovery response (Arendt et al. 2013).

Climate change may also increase hurricane activity, leading to an increase in debris in nearshore and offshore environments. This, in turn, could increase the occurrence of entanglements, ingestion of pollutants, or drowning. In addition, increased hurricane activity may damage nesting beaches or inundate nests with seawater. Increasing temperatures are expected to result in increased polar melting and changes in precipitation that may lead to rising sea levels (Titus and Narayanan 1995).

Hurricanes and tropical storms occur frequently in the southeastern United States. They impact nesting beaches by increasing erosion and sand loss and depositing large amounts of debris on the beach. A lower level of leatherback nesting attempts occurred on sites more likely to be impacted by hurricanes (Dewald and Pike 2014). These storm events may ultimately affect the amount of suitable nesting beach habitat, potentially resulting in reduced productivity (TEWG 2007). These storms may also result in egg loss through nest destruction or inundation. Climate change may be increasing the frequency and patterns of hurricanes (IPCC 2014) which may result in more frequent impacts.

These environmental/climatic changes could result in increased erosion rates along nesting beaches, increased inundation of nesting sites, a decrease in available nesting habitat, and an increase in nest crowding (Baker et al. 2006, Daniels et al. 1993, Fish et al. 2005, Reece et al. 2013). Changes in environmental and oceanographic conditions (e.g., increases in the frequency of storms, changes in prevailing currents), as a result of climate change, could accelerate the loss of sea turtle nesting habitat, and thus, loss of eggs (Antonelis et al. 2006, Baker et al. 2006, Conant et al. 2009, Ehrhart et al. 2014).

Tidal inundation and excess precipitation can contribute to reduce hatchling output, particularly in wetter climates (Pike 2014, Pike et al. 2015, Santidrián Tomillo et al. 2015). This is especially problematic in areas with storm events and in highly-developed areas where the beach has nowhere to migrate. Females may deposit eggs seaward of erosion control structures, potentially subjecting nests to repeated tidal inundation. A recent study by the U.S. Geological Survey found that sea levels in a 620-mile (998-km) “hot spot” along the east coast are rising three to four times faster than the global average (Sallenger et al. 2012). In the next 100 years, the study predicted that sea levels will rise an additional 7.9-10.6 inches (20-27 cm) along the Atlantic coast “hot spot” (Sallenger et al. 2012). The disproportionate sea level rise is due to the slowing of Atlantic currents caused by fresh water from the melting of the Greenland Ice Sheet. Sharp rises in sea levels from North Carolina to Massachusetts could threaten wetland and beach habitats, and negatively affect sea turtle nesting along the North Carolina coast. If warming temperatures moved favorable nesting sites northward, it is possible that rises in sea level could constrain the availability of nesting sites on existing beaches (Reece et al. 2013). There is limited evidence of a potential northward range shift of nesting loggerheads in Florida, and it is predicted that this shift, along with sea level rise, could result in more crowded nesting beaches (Reece et al. 2013).

In the case of the Kemp’s ridley, most of their critical nesting beaches are undeveloped and may still be available for nesting despite shifting landward. Unlike much of the Texas coast, the Padre Island National Seashore (PAIS) shoreline in Texas, where increasing numbers of Kemp’s ridley are nesting, is accreting. Given the increase in nesting at the PAIS, as well as increasing and slightly cooler sand temperatures than at other primary nesting sites, PAIS could become an increasingly important source of males for a species, which already has one of the most restricted nesting ranges of all sea turtles. Nesting activity of Kemp’s ridleys in Florida has also increased over the past decade, suggesting the population may have some behavioral flexibility to adapt to a changing climate (Pike 2013). Still, current models predict long-term reductions in sea turtle fertility as a result of climate change; however, these effects may not be seen for 30 to 50 years because of the longevity of sea turtles (Davenport 1997, Hawkes et al. 2007, Hulin and Guillon 2007).

Changes in water temperature may also alter the forage base and, thus, foraging behavior of sea turtles (Conant et al. 2009). Likewise, if changes in water temperature affected the prey base for green, loggerhead, Kemp's ridley, or leatherback sea turtles, there may be changes in the abundance and distribution of these species in the action area. Depending on whether there was an increase or decrease in the forage base and/or a seasonal shift in water temperature, there could be an increase or decrease in the number of sea turtles in the action area. Seagrass habitats may suffer from decreased productivity and/or increased stress due to sea level rise, as well as changes in salinity, light levels, and temperature (Duarte 2002, Saunders et al. 2013, Short and Neckles 1999). If seagrasses in the action area decline, it is reasonable to expect that the number of foraging green sea turtles would also decline as well. Rising water temperatures, and associated changes in marine physical oceanographic systems (e.g., salinity, oxygen levels, and circulation), may also impact the distribution/abundance of leatherback prey (i.e., jellyfish) and in turn, impact the distribution and foraging behavior of leatherbacks (Attrill et al. 2007, Brodeur et al. 1999, NMFS and USFWS 2013, Purcell 2005, Richardson et al. 2009). Loggerhead sea turtles are thought to be generalists (NMFS and USFWS 2008), and, therefore, may be more resilient to changes in prey availability. As noted above, because we do not know the adaptive capacity of these individuals, or what level of temperature change would cause a shift in distribution, it is not possible to predict changes to the foraging behavior of sea turtles over the next ten years. If sea turtle distribution shifted along with prey distribution, it is likely that there would be minimal, if any, impact to sea turtles due to the availability of food. Similarly, if sea turtles shifted to areas where different forage was available, and sea turtles were able to obtain sufficient nutrition from that new source of forage, any effect would be minimal. However, should climatic changes cause sea turtles to shift to an area or time where insufficient forage is available, impacts to these species would be greater.

Kemp's ridley sea turtles are also the most commonly documented species during cold stun events in the Greater Atlantic Region. With prolonged exposure to low water temperatures, sea turtles become hypothermic and can experience debilitating lethargic conditions. These events occur in the fall at higher latitudes when sea turtles do not migrate south before water temperatures decline. Griffin et al. (2019) suggest that warming sea surface temperatures in the Gulf of Maine are associated with increased strandings of Kemp's ridley sea turtles in Massachusetts. The warmer temperatures may be allowing Kemp's ridley distribution to expand and may act as an ecological bridge between the Gulf Stream and nearshore waters (Griffin et al. 2019).

6.2.3. ***Atlantic Sturgeon***

Hare et al. (2016) assessed the vulnerability to climate change of a number of species that occur along the U.S. Atlantic coast. The authors define vulnerability as "the extent to which abundance or productivity of a species in the region could be impacted by climate change and decadal variability." Atlantic sturgeon were given a vulnerability rank of very high (99 percent certainty from bootstrap analysis) and a climate exposure rank of very high. Three exposure factors contributed to this score: sea surface temperature (4.0), ocean acidification (4.0) and air temperature (4.0). The authors concluded that Atlantic sturgeon are relatively invulnerable to distribution shifts. Climate factors such as sea level rise, reduced dissolved oxygen and increased temperatures have the potential to decrease productivity, but the magnitude and interaction of effects is difficult to assess (Hare et al. 2016b). Increasing hypoxia, in combination with increasing temperature, impacts juvenile Atlantic sturgeon metabolism and survival (Secor and Gunderson 1998). A multivariable bioenergetics and survival model predicted that within the Chesapeake Bay, a 1 ºC increase in Bay-wide temperature reduced suitable habitat for juvenile

Atlantic sturgeon by 65 percent (Niklitschek and Secor 2005). These studies highlight the importance of the availability of water with suitable temperature, salinity and dissolved oxygen; climate conditions that reduce the amount of available habitat with these conditions would reduce the productivity of Atlantic sturgeon.

Changes in water availability may also impact the productivity of populations of Atlantic sturgeon. In rivers with dams, or other barriers that limit access to upstream freshwater reaches, spawning and rearing habitat may be restricted by increased saltwater intrusion; however, no estimates of the impacts of such change are currently available.

6.2.4. *Atlantic Salmon*

Hare et al. (2016) gave Atlantic salmon a vulnerability rank of very high (100 percent certainty from bootstrap analysis) as well as a climate exposure rank of very high and a distributional vulnerability rank of moderate (87 percent certainty from bootstrap analysis). Due to the effects of warming on freshwater and marine habitats, and the potential to affect the phenology of Atlantic salmon migration, the effect of climate change on Atlantic salmon in the Northeast U.S. Shelf Ecosystem is very likely to be negative (>95 percent certainty in expert scores) (Hare et al. 2016b). Ocean acidification could also affect olfaction, which Atlantic salmon use for natal homing.

As described in Hare et al. (2016), several studies have examined the effects of climate on the abundance and distribution of Atlantic salmon. A review of the likely effects of climate change found that the thermal niche of Atlantic salmon will likely shift northward causing decreased production and possibly extinction at the southern end of the species range (Jonsson and Jonsson 2009). The GOM DPS is the southernmost populations of Atlantic salmon in the Northwest Atlantic Ocean. Declines in post-smolt survival were associated with ocean warming (Friedland et al. 2014). The authors hypothesized that in the Northwest Atlantic, the decline in survival was due to early ocean migration by post-smolts (Friedland et al. 2014). Results of a recent study suggest that poor trophic conditions, likely due to climate-driven environmental factors, and warmer ocean temperatures are constraining the productivity and recovery of Atlantic salmon in the Northwest Atlantic (Mills et al. 2013). Available evidence suggests that climate change and long-term climate variability will reduce the productivity of the GOM DPS of Atlantic salmon.

6.2.5. *Giant Manta Ray*

Given that giant manta rays are migratory and considered ecologically flexible (e.g., low habitat specificity), they may be less vulnerable to the impacts of climate change compared to other sharks and rays (Chin et al. 2010). However, giant manta rays frequently rely on coral reef habitat for important life history functions (e.g., feeding, cleaning), and depend on planktonic food resources for nourishment. As coral reef habitat and planktonic organisms (e.g., zooplankton) are both highly sensitive to environmental changes (Brainard et al. 2011, Guinder and Molinero 2013), climate change is likely to have an impact on the distribution and behavior of the giant manta ray. Coral reef degradation from anthropogenic causes, particularly climate change, is projected to increase through the future (Miller and Klimovich 2017). There is insufficient information to indicate how, and to what extent, changes in the reef community structure will affect the status of the giant manta ray. The projected increase in coral habitat degradation may potentially lead to a decrease in the abundance of manta ray cleaning fish (e.g., *Labroides* spp., *Thalassoma* spp., and *Chaetodon* spp.), as well as an overall reduction in the number of cleaning stations available to manta rays within these habitats. Decreased access to cleaning stations may negatively impact the fitness of the mantas by hindering their ability to reduce parasitic loads and dead tissue, which in turn, could lead to an increase in diseases and a decline in reproductive fitness and survival.

Changes in climate and oceanographic conditions, such as acidification, are also known to affect zooplankton structure (size, composition, diversity), phenology, and distribution (Guinder and Molinero 2013). As such, the migration paths and locations of both resident and seasonal aggregations of manta rays, which depend on these animals for food, may similarly be altered (Couturier et al. 2012, Government of Australia 2012). It is likely that those manta ray populations that exhibit site-fidelity behavior will be most affected by these changes. As research to understand the exact impacts of climate change on marine phytoplankton and zooplankton communities is still ongoing, the severity of this threat to manta rays has yet to be fully determined.

7. EFFECTS OF THE PROPOSED ACTION

In *Effects of the Action* section, we present the results of our assessment of the probable effects of federal actions that are the subject of this consultation on threatened and endangered species and designated critical habitat. Effects of the action are defined as 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. 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. (50 CFR 402.02)

The analysis in this section forms the foundation for our jeopardy analysis in section 9. The quantitative and qualitative analyses in this section are based upon the best available commercial and scientific data on species biology and the effects of the action. Data are limited, so we are often forced to make assumptions to overcome the limits in our knowledge. Sometimes, the best available information may include a range of values for a particular aspect under consideration or different analytical approaches may be applied to the same data set. When appropriate in those cases, the uncertainty is resolved in favor of the species (see House of Representatives Conference Report No. 697, pg. 1442, 96th Congress, Second Session, 12 (1979)). We generally select the value that would lead to conclusions of higher, rather than lower, risk to endangered or threatened species. This approach provides the “benefit of the doubt” to threatened and endangered species.

7.1. Approach to the Assessment

We began our analysis of the effects of the action by first reviewing what activities (e.g., gear types and techniques, vessel transits) associated with the proposed action are likely to adversely affect whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays in the action area (i.e., the proposed action stressors). We next reviewed the range of responses to an individual’s exposure to that stressor and the factors affecting the likelihood, frequency, and severity of exposure. Afterwards, our focus shifted to evaluating and quantifying exposure. We estimated the number of individuals of each species likely to be exposed and the likely fate of those animals. As described in the *Description of the Proposed Action*, we are also consulting on the implementation of the Habitat Amendment. After assessing the effects of the operations of the fisheries on ESA-listed species, we consider how the Habitat Amendment modifies the operation of fisheries, and, if changes occur, whether this exposes the species to additional or increased stressors.

The *Integration and Synthesis* section of this Opinion follows the *Effects of the Action*, and integrates information we presented in the *Status of the Species* and *Environmental Baseline* with the results of our exposure and response analyses to estimate the probable risks the proposed action poses to endangered and threatened species. Because we previously concluded that the proposed action is not likely to adversely affect several listed species and areas designated as critical habitat for listed species (section 4.1), these listed species and critical habitat are not considered in the analyses that follow.

To identify, describe, and assess the effects to listed species considered in this Opinion, we reviewed information on: (1) entanglements of right, fin, sei, and sperm whales; and sea turtles documented in the GAR Marine Animal Incident and STDN databases and from published literature (Hayes 2019, Hayes et al. 2019, Johnson et al. 2005, Waring et al. 2015); (2) bycatch of loggerhead, leatherback, Kemp's ridley, and green sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays in these fisheries from the NEFSC observer/sea sampling database and the published literature (ASMFC 2017, Murray 2018, 2020); (3) vessel interactions with sea turtles documented in the STSSN database; (4) life history of large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays, and (5) the effects of fishing gear interactions on large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays that have been published in a number of documents. These sources include status reviews, stock assessments, and biological reports, recovery plans, and numerous other sources of information from the published literature as cited in this Opinion.

Potential Stressors

We consider all stressors from the proposed action that may adversely affect endangered or threatened species, their ecological interactions, or critical habitat designated for the listed species. At any point in time, a single vessel may be the source of one or more of these potential stressors, and listed individuals may be exposed to one or more of these stressors.

Potential stressors from the proposed action include capture or entanglement in fishing gear and vessel strikes. Effects caused by the authorization of the fisheries on threatened and endangered species stem primarily from interactions with the fishing gear that results in the capture, injury, or death of an individual, listed species. Our analysis, therefore, assesses adverse effects from physical contact with fishing gear. We also assume the potential effects of each gear type are proportional to the number of interactions between the gear and each species. Four basic types of fishing gear are primarily used in the fisheries: sink gillnets, bottom trawls, trap/pots, and hook and line gear. Other potential effects of the proposed action on listed species occur vessel interactions, resulting in injury and/or death of an individual.

Additional consequences caused by or resulting from the proposed action, may occur later in time and are reasonably certain to occur. These may include such effects as habitat degradation and reduction of prey/foraging base. Of all the gears used in the fisheries, bottom trawl is the only gear type that has the potential to adversely affect bottom habitat in the action area. Effects of otter trawl gear may include: (1) scraping or plowing of the doors on the bottom, sometimes creating furrows along their path; (2) sediment suspension resulting from the turbulence caused by the doors and the ground gear on the bottom; (3) removal or damage to benthic or demersal species; and (4) removal or damage to structure forming biota. An assessment of fishing gear impacts found that, for trawl, mud, sand, and cobble features are more susceptible, while granule-pebble and scattered boulder features are less susceptible. Geological structures generally recovered more quickly from trawling on mud an sand substrates than on cobble and boulder substrates; while biological structures recovered at similar rates across substrates. Susceptibility was defined as the percentage of habitat features encountered by the gear during a hypothetical single pass event that had their functional value reduced, and recovery was defined as the time required for the functional value to be restored (see Appendix D in NEFMC 2016b, NEFMC 2020b). We do not consider gillnet, trap/pot, or hook and line gear to be gear types that would affect either bottom or pelagic habitats in the action area and, therefore, our effects to habitat analysis only considers impacts from trawl gear. As described below, there are no indirect effects associated with the proposed action that are likely to adversely affect large whales, sea turtles, ESA-listed fish, or critical habitat. Therefore, the analyses will focus on direct effects. As described in the *Status of the Species*, prey for ESA-listed species within the action areas varies greatly depending on the predator species. Given this range, impacts from changes in the prey/foraging base are considered under the subsections below.

7.2. Effects to Large Whales

Impacts of entanglement events attributed to gear used in Canadian fisheries are considered in the *Status of the Species*. Impacts of entanglement events attributed to gear used in state fisheries are considered in the *Environmental Baseline* and *Cumulative Effects* sections. State and Canadian entanglement events are not included as part of the effects analysis because they are not the result of the action under consultation. The effects of the proposed action in the context of information presented in the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* will be examined in the *Integration and Synthesis of Effects.*

7.2.1. *Gear Interactions*

Certain fishing gears may directly affect whales. These include gillnet, hook and line gear, and the lines of trap/pot gear. There are few records of interactions between trawl gear and large cetaceans, and the interactions that have occurred involved species that are not ESA-listed. From 2010-2019, there was one interaction between a large whale and trawl gear. Prior to this, minke whales have been observed or reported entangled in bottom and mid-water trawl gear, but interactions are rare (since 2000, two interactions in bottom trawl gear and one in midwater trawl gear) (Hayes et al. 2020, Waring et al. 2015, Waring et al. 2013). In 2020, a live, anchored humpback whale was disentangled by trained responders from the ALWDN. The gear was recovered and identified as a trawl net. Due to the weight of the gear, fresh wounds on the animal, and the fact that the animal was a juvenile, it was determined the trawl net was of U.S. origin. There have been no observed or reported interactions of right, fin, sei or sperm whales with bottom otter trawl gear (NEFSC observer/sea sampling database, unpublished data; GAR Marine Animal Incident database, unpublished data). Bottom otter trawl gear is not expected to directly affect right, fin, sei, and sperm whales given that these large cetaceans have the speed and maneuverability to get out of the way of oncoming mobile gear, including bottom trawl gear used in these fisheries. Given that this action does not change the fishing practices of the trawl fisheries and the information above, it is extremely unlikely that ESA-listed large whales will interact with bottom otter trawl gear in these fisheries and, therefore, the effects are discountable. In addition, due to their size, right, fin, sei, and sperm whales cannot get caught in the trap/pot itself since the opening is far smaller than any of these species. We focus the remainder of this section on gears that may directly impact large whales.

7.2.1.1. Factors Affecting Whale Interactions

Any line in the water column, including line resting on or floating above, the seafloor has the potential to entangle a whale (Hamilton et al. 2018, Hamilton et al. 2019, Johnson et al. 2005). Entanglements may involve the head, flippers, or fluke; effects range from no apparent injury to death. Of the fisheries in this Opinion, large whales are vulnerable to entanglement in vertical or ground lines associated with fisheries using sink gillnet and trap/pot gear as well as the net panels of gillnet gear. No interactions with hook and line gear have been documented for right, sei, or sperm whales. However, one fin whale interaction with hook and line gear was documented in 2016 that resulted in serious injury. While whale interactions with hook and line gear are expected to be rare, we anticipate that any whale species may become entangled in hook and line gear used in these fisheries.

The general scenario that leads to a whale becoming entangled in gear begins with a whale encountering gear. It may move along the line until it comes up against something such as a buoy or knot. When the animal feels the resistance of the gear, it is likely to thrash, which may cause it to become further entangled in the lines associated with gear. The buoy may become caught in the whale's baleen, against a pectoral fin, or on some other body part.

The probability that a marine mammal will initially survive an entanglement in fishing gear depends on the characteristics of the gear, the species, and the health and age of the marine mammal involved. If the gear attached to the line is too heavy and prevents the whale from surfacing, drowning may result immediately. However, many whales have been observed swimming with portions of the line, with or without additional fishing gear, wrapped around a pectoral fin, the tail stock, the neck or the mouth. Documented cases show that entangled animals may travel for extended periods of time and over long distances before freeing themselves, being disentangled by humans, or dying as a result of the entanglement (Angliss and DeMaster 1998).

Determining which part of gear creates the most entanglement risk for ESA-listed species is difficult due to uncertainties surrounding the nature of the entanglement event, as well as unknown biases associated with reporting effort and the lack of information about the types and amounts of gear being used (Johnson et al. 2005). The vertical and ground lines of several different fisheries have been found to entangle large whales. Netting is also known to pose a risk of entanglement to whales. In many events, the animal was entangled in more than one set of gear. The animal may be entangled in the line of one set, which then becomes tangled with the bottom gear or vertical line of a second or third set of gear.

There are generally three initial attachment points for gear to attach to large whales: (1) the gape of the mouth, (2) around the flippers, and (3) around the tail stock. Knots in the line hinder the ability of the line to pass through the baleen. Anchors on the gear or the weight of the gear itself offers resistance against which the whale may struggle and result in further entanglement of the fishing gear across the mouth and/or body of the whale. Conversely, the extra resistance could increase the effectiveness of weak links to assist in shedding gear from entangled whales. Weak links are breakable components of the gear that will part when subject to a certain tension load.

The overlap of the fisheries and large whales in space and time also influences the likelihood that gear entanglement will occur. Atlantic large whales are at risk of becoming entangled in fishing gear as they feed and travel in the action area. North Atlantic right, fin, and sei whales follow a similar, general pattern of foraging at high latitudes (e.g., southern New England and Canadian waters) in the spring and summer months and calving in lower latitudes (i.e., off of Florida for right whales) in the winter months (CETAP 1982, Clark 1995, Hain et al. 1992, Horwood 2002, Kenney 2009, Perry et al. 1999). The highest abundances of North Atlantic right, fin, and sei whale populations occur from March through November in New England waters, which is also the peak fishing period for the fisheries in these waters. Sperm whales have a different seasonal distribution as described in more detail below.

For many years, right whales aggregated seasonally in seven known areas: the coastal waters of the southeastern United States; the Great South Channel; Jordan Basin; Georges Basin along the northeastern edge of Georges Bank; Cape Cod and Massachusetts Bays; the Bay of Fundy; and the Roseway Basin on the Scotian Shelf. Since 2010, fewer whales have been using some of these established habitats such as the Great South Channel and the Bay of Fundy (Davis et al. 2017, Hayes 2019). Meanwhile, the use of Cape Cod and Massachusetts Bay seems to have increased and a large portion of the right whale population is now using an area south of Nantucket and Martha's Vineyard from winter through early spring (Davis et al. 2017, Hayes 2019). This area is also used in other seasons (Oleson et al. 2020). In addition, right whales also use more of the U.S. eastern seaboard than previously believed and can be present in the Mid-Atlantic year round (Davis et al. 2017, Hodge et al. 2015, Salisbury et al. 2016, Whitt et al. 2013). The frequency with which right whales occur in offshore waters in the southeastern United States remains unclear (Hayes 2019).

New England waters are an important feeding ground for fin whales. They are common year round in the EEZ from Cape Hatteras, North Carolina northward; their density varies seasonally (Hayes 2019). Sei whales are present in the action area during spring and summer. The southern portion of the species' range includes the Gulf of Maine and Georges Bank. Spring is the period of greatest abundance of sei whales in U.S. waters (Hayes et al. 2017b).

Sperm whales appear to have a distinct seasonal cycle (CETAP 1982, Scott and Sadove 1997). They are concentrated east and northeast of Cape Hatteras, North Carolina in winter and shift north to east of Delaware and Virginia in spring when they are widespread through the central portion of the Mid-Atlantic Bight and the southern portion of Georges Bank. In summer, the distribution is similar but also includes the area east and north of Georges Bank and into the Northeast Channel region and the continental shelf south of New England. In the fall, sperm whale occurrence south of New England on the continental shelf is at its highest level, and there remains a continental shelf edge occurrence in the Mid-Atlantic Bight. In the U.S. EEZ, they occur on the continental shelf edge, over the continental slope and into mid-ocean regions (Waring et al. 2015).

Because of substantial interannual and geographic variation in whale occurrence and lack of complete data for seasonal distributions, the potential exists for whales to interact with gear used in the fisheries year-round throughout the entire action area.

7.2.1.2. Existing Information on Interactions with Whales

Information available on interactions with large whales comes from reports documented in the GAR Marine Animal Incident Database. Cases in the database include observed cases resulting from entanglement, vessel strike, or unknown cause. We obtain these reports from aerial surveys and vessels on the water that call into the reporting system. The level of information collected for each case varies, but may include details on the animal, gear, and any other information about the interaction (e.g., location, description, etc.). Each case is evaluated using defined criteria to assign the case to an injury/information category using all available information and scientific judgement. In this way, the injury severity and cause of injury/death for the event is evaluated (Henry et al. 2019, NMFS 2012e). When a case lacks the details necessary for a more specific category, the injury determination is prorated, resulting in a fractional value. For example, cases with evidence of entanglement but with insufficient information to assign it to one of the other categories with a high degree of confidence are prorated at 0.75 (Henry et al. 2019, NMFS 2012e). Serious injury determinations for large whales commonly include animals carrying gear when these entanglements are constricting or appear to interfere with foraging (Hayes et al. 2019). For the purposes of this analysis, we assume that serious injuries result in mortalities. Additionally, depending on the level of information available for each case, the interaction may be attributed to a particular country or unassigned.

When gear is identified, the identification is based on an analysis of recovered gear, fishermen interviews, or information (e.g., gear markings) documented in the field. Frequently, entangled whales have evidence on other body parts of entanglement trauma without gear present, or subsequent sightings may show that the configuration of entangling gear has changed since the initial observation. In these cases, the physical struggle during the initial entanglement may have broken free portions of the gear, including weak links. For example, if an entanglement case had recovered sinking groundline, it is possible the animal had been entangled in other parts of the gear and shed a significant portion of it, with the sinking line being the only part remaining to be recovered.

Evaluating all of the information available for each interaction to make a final determination takes time and the validated data for each case are often not available for one to two years following an interaction. Available preliminary entanglement reports are considered in estimating interaction rates for each species in the sections below. Preliminary reports often change when cases are evaluated more thoroughly. Changes may include adding/deleting cases and changing the determination or status of a case. Preliminary entanglement reports for each year should be considered a minimum number and not comprehensive.

Other information available to assess interactions with fishing gear include scarring estimates. Because whales often free themselves of gear following an entanglement event, scarring is a useful indicator in monitoring fisheries interactions with large whales. Scarring information is available for right whale interactions using research conducted annually using the North Atlantic Right Whale Catalogue (Hamilton et al. 2019). Scar coding and analyses in Hamilton et al. 2019 used the details and methodology presented in papers by Knowlton et al. (2012, 2016), which describes that for a scar to be attributed to entanglement, it must show evidence of the rope having “wrapped” on a given body part. Scars with any uncertainty of source or cause were not included. Data from the catalogue are analyzed to produce annual entanglement estimates using scarring data. In our assessment, we will evaluate the annual entanglement estimates presented in Hamilton et al. (2019) to estimate the rate of right whale entanglements. This rate of entanglement was obtained by accessing a subset of animals adequately photographed in back-to-back years. The information presented in Hamilton et al. (2019) represents the best available information to estimate an annual right whale entanglement rate. There is currently no dedicated research to analyze the scarring rates of the other ESA-listed large whale species. Additionally, the scarring estimate for right whales is not an acceptable surrogate for the other large whale species given the differences in species behavior, distribution, co-occurrence, and body type that likely contribute to the susceptibility to entanglement. Therefore, at this time there are not enough scarification data available for the other large whale species to calculate a reasonable and scientifically supportable estimate.

Some whale mortalities may never be observed; thus, the annual observed entanglement-related mortalities are likely less than the actual number of entanglement-related mortalities occurring. The methods of Pace et al. (2017) can be used to produce estimates of annual mortality, including both observed and cryptic[23] (unobserved) mortality (Hayes et al. 2019). Subtracting the observed mortality from the estimated annual mortality, we calculated the cryptic mortality. In our assessment, we will evaluate cryptic mortality as well as observed mortality.

A decision support tool (DST) was developed by the NEFSC to aid in the comparison of spatial management measures toward the development of the ALWTRP proposed rule to reach a 60 percent risk reduction target. This model calculates North Atlantic right whale entanglement risk based on three components: (1) line density, (2) whale density, and (3) gear threat per line. The line density component of the DST is based on the peer-reviewed NMFS Vertical Line Model and co-occurrence model developed by IEC. The distribution of whales is from a habitat density model analyzing right whale distribution from 2010-2018.[24] The gear threat model was used to determine the potential severity of entanglements of different lines (NMFS 2020b). Together, these components roughly estimate the approximate risk of an entanglement that will result in serious injury or mortality, where a higher density of lines or predicted whales, and/or certain gear characteristics (e.g., high line strength, longer trawls) increase risk. This enables a semi-quantitative comparison of how different management scenarios and gear modifications are predicted to change entanglements that result in serious injury or mortality. For the purpose of this Opinion, the DST was used to assess how much of the risk to right whales is occurring in state vs federal waters. The results show that 60.4 percent of the risk of M/SI to right whales occurs in federal waters. More thorough documentation of the model and its components are available in appendix 3.1 (NMFS 2020b).

To summarize, we are using data from several sources in order to assess the effects of the proposed action on North Atlantic right whales. To evaluate the total number of entanglements, we use the scarification data. To evaluate M/SI, we use the GAR Marine Animal Incident Database to assess observed interactions resulting in M/SI. Annual counts of right whale carcasses do a poor job of indicating the total mortality for that year, and carcass detection rates seem to vary with effective survey effort (Pace et al. 2021). Therefore, we use the total mortality estimate (using methods from Pace et al. 2017) to assess the total mortality. We then subtract the observed interactions (derived from GAR Marine Animal Incident Database) from the total mortality estimates (derived from methods used by Pace et al. (2017) to assess the level of cryptic mortality. The observed M/SI plus cryptic mortality are then apportioned across vessel strikes and entanglements and to the United States and Canada, using the methods described below. This gives an estimate of entanglements resulting in M/SI occurring in U.S. waters. We

[23] Cryptic mortality refers to the death of an animal without resulting in an observed carcass.

[24] In the draft Biological Opinion, the distribution of whales was from either a habitat density model analyzing right whale distribution through 2017 or, in Southern New England where distribution has recently shifted, the North Atlantic Right Whale Consortium's Sighting per Unit Effort data from 2014-2018. A new habitat density model is now available and, therefore, we were able to use a single, updated model in the final Biological Opinion..

then use the DST to apportion the entanglements occurring in U.S. waters between state and federal fisheries.

Sublethal Effects of Entanglement
Entanglements may cause sublethal effects that impact an animal's health and reproductive rates (Lysiak et al. 2018, Pettis et al. 2017, Robbins et al. 2015). Sublethal effects of entanglement are difficult to observe or quantify. Therefore, we qualitatively assess the sublethal effects entanglements may have on whales.

When a large whale becomes entangled in fishing gear, they may be partially disentangled or break free of anchor points and carry a portion of the gear with them. Whales carrying gear have a higher energy expenditure due to the increased drag and increased thrust power required to swim (van der Hoop et al. 2017b). Increased drag from carrying gear for long periods can be energetically more costly for a female than the migratory and developmental costs of a pregnancy (van der Hoop et al. 2017a, van der Hoop et al. 2017b, van der Hoop et al. 2017c). Non-lethal entanglements may also reduce or prevent births in the population (Hayes et al. 2018a), cause systemic infection or debilitating tissue damage (Cassoff et al. 2011), or damage the baleen plates preventing efficient filter feeding (Hayes et al. 2018a). Any injury or entanglement that restricts a whale from rotating its jaw while feeding, prevents it from forming a hydrostatic oral seal, compromises the integrity of its baleen, or prevents it from swimming at speeds necessary to capture prey will reduce its foraging capabilities and may lead to starvation (Cassoff et al. 2011, van Der Hoop et al. 2013). As baleen grows slowly, damage to the plates can prevent efficient feeding for years (Hayes et al. 2018a).

While non-lethal entanglement in fishing gear may negatively affect the health or body condition of a whale, multiple stressors (e.g., prey abundance, climate variation, reproductive state, exposure to harmful algal blooms, vessel collisions) co-occur and, individually and cumulatively, can affect the health of animals, and, subsequently, the calving rate of the population. Limited foraging conditions, increases in energetic costs, and stress of entanglements are all likely having an influence on the health and calving rates of whales, however, it is difficult to distinguish the relative effects of each (Hayes et al. 2018a). It is the interplay of these multiple stressors that contributes to the overall health of the animal rather than a single co-variate such as entanglement (Rolland et al. 2016, Schick et al. 2013, van der Hoop et al. 2016). Further, recent literature addressing individual and population level health of right whales are model based, and therefore, the health of whales is based on postulations of the possible impact of multiple stressors, including anthropogenic stressors (e.g., vessel strikes, fishing gear entanglement) (Rolland et al. 2016, Schick et al. 2013). Based on these studies, it is likely that entanglement is a co-variate in the long-term health of right whales. While entanglement results in sublethal effects (van der Hoop et al. 2017a), these effects are often confounded by other factors. Based on the best available scientific and commercial data, we believe at least some of the observed variability in right whale calving rates is due to the sublethal effects of entanglements in U.S. federal fishing gear but cannot quantify the degree to which entanglements are affecting calving rates at this time.

7.2.1.3. Estimating Interactions with and Serious Injury/Mortality of North Atlantic Right Whales

Since June 7, 2017, elevated North Atlantic right whale mortalities have been documented, and were declared an Unusual Mortality Event[25] (UME). As part of the UME investigation process, additional funding and resources were made available to review the data collected. Therefore, more comprehensive and recent data exists for right whales than the other large whale species. Right whale preliminary entanglement reports are presented below (Table 57). Given that there is comprehensive, validated data available for right whales, preliminary entanglement reports will not be used in the analysis to estimate interactions with right whales. We have reviewed these preliminary reports, and they are consistent with what has been reported in the past so their exclusion from this analysis is not expected to alter any conclusions. For the purpose of this assessment, we are relying on data (e.g., serious injury, country of origin) that have undergone review through the determination process (Henry et al. 2019, NMFS 2012e) described in section 7.2.1.2.

Table 57: Entanglement reports for right whales from January 2019 through March 2021

Date	First seen	Status
4/25/2019	East of Orleans, MA	Partially disentangled; gear shed
6/29/2019	East of Miscou Island, NB	Partially disentangled; gear shed
7/4/2019	East of Perce, QC	Partially disentangled
8/6/2019	Northeast of Iles de la Madeleine, Quebec	Deceased
12/21/2019	South of Nantucket	Entangled
2/24/2020*	South of Nantucket	Entangled
3/16/2020*	Georges Bank	Entangled
10/11/2020*	East of Sea Bright, NJ	Entangled
10/19/2020*	South of Nantucket	Mortality (preliminary cause of death: entanglement)
1/11/2021*	East of Fernandina Beach, FL	Entangled
3/10/2021*	North of Sandwich, MA	Entangled (partially disentangled)

*2020 and 2021 data are preliminary

[25] For more information on the UME, see https://www.fisheries.noaa.gov/marine-life-distress/frequent-questions-2017-north-atlantic-right-whale-unusual-mortality-event

To estimate future entanglement of right whales resulting in M/SI due to the operation of the federal fisheries, we use data from the GAR Marine Animal Incident Database for the years 2010-2018[26]. Additionally, to estimate the annual total entanglements resulting from the federal fisheries, we use the most recent scarring estimates for 2010 – 2017 presented in Hamilton et al. (2019). Annual total entanglements includes animals with non-serious entanglements as well as those resulting in serious injury and mortality.

We chose 2010 as the earliest year as this year coincided with a zooplankton regime shift in the Gulf of Maine. The Gulf of Maine has been markedly different in the past decade than in the 2000s. Small bodied zooplankton are now more abundant than large zooplankton such as *C. finmarchicus* (NMFS 2020h). Regime shifts in zooplankton community composition are abrupt changes between contrasting states of a system that persist through time (deYoung et al. 2008). Changes in zooplankton productivity may be one of the most important pathways for climate to impact higher trophic levels of the Northeast continental shelf (Morse et al. 2017). Regime shifts have occurred periodically in the Gulf of Maine (for spring early 1980s, 2002, 2007), on Georges Bank (for spring, early 1990s; for fall, late 1980s), and the Mid-Atlantic Bight (for spring, 1990s; for fall, early 2000s). A regime shift in seasonal sea surface temperature occurred in 2010 in the spring and fall in the Gulf of Maine and Georges Bank, which was followed in 2012 by a shift in the Gulf of Maine zooplankton community (Morse et al. 2017).

The regime shift in 2010 coincided with a noticeable shift in right whale distribution and habitat use (Davies et al. 2019, Davis et al. 2017, Pettis et al. 2018b, Record et al. 2019). Since 2010, right whale habitat use patterns in areas where most of the population has been observed in previous years seems to have changed considerably (Hayes et al. 2019). Climate-driven changes in the Gulf of Maine have shifted the seasonal patterns for essential right whale prey (*C. finmarchicus*), which likely caused right whales to shift their distribution in search of adequate sources of prey (Davis et al. 2017, Morse et al. 2017, Record et al. 2019). There is no information available to suggest when a shift in the zooplankton community may occur again. Given this information, the 2010-2018 data represent the best available information to estimate future right whale interactions with the fisheries.

Between 2010 and 2018, there were 107 confirmed right whale entanglements with 48.5 resulting in M/SI (Table 58). Of these, 16 cases (7.75 M/SI) were confirmed to be entanglement with Canadian fishing gear, and 8 cases (2 M/SI) were confirmed to be entanglement in U.S. fishing gear. The remaining 83 (38.75 M/SI) cases were the result of entanglement with gear from an unknown country of origin. While sightings and acoustic records indicate an extended range (i.e., outside of U.S. or Canadian waters) for at least some individuals (Hayes et al. 2020), all interactions with fishing gear attributed to country were from U.S. or Canadian gear. Therefore, this analysis attributes entanglements to either U.S. or Canadian fishing gear. As

[26] Note that the draft Biological Opinion used data from 2010-2019. In developing the final Biological Opinion, we discovered a misalignment in the data. This has been corrected in this final Biological Opinion. Although the 2019 right whale observed interactions data is available, the total mortality estimate for 2019 is not available. Given that we need the total mortality estimate to calculate the cryptic mortality in 2019, we do not include the 2019 data in our average annual estimates below.

described above, injury cases may be prorated resulting in fractional values in the data presented below.

Table 58: Observed entanglements of North Atlantic right whales from 2010 through 2018 by country of origin. Entanglements resulting in M/SI are presented in the parentheses. Source: GAR Marine Animal Incident Database

	Number of Entanglements	**Confirmed Canada**	**Confirmed U.S.**	**Unknown Country of Origin**
2010	6 (4)	0	1	5 (4)
2011	14 (5.5)	0	2	12 (5.5)
2012	12 (4)	0	1 (1)	11 (3)
2013	5 (0.75)	0	0	5 (0.75)
2014	17 (8)	1	1 (1)	15 (7)
2015	9 (3.5)	1	0	8 (3.5)
2016	15 (9.5)	3 (3)	1	11 (6.5)
2017	15 (6)	8 (3)	1	6 (3)
2018	14 (7.25)	3 (1.75)	1	10 (5.5)
Total	**107 (48.5)**	**16 (7.75)**	**8 (2)**	**83 (38.75)**

Although the observed entanglement data include non-M/SI events, these observed events are considered a minimum estimate, and the actual entanglement rate is likely higher. To account for this underrepresentation of non-M/SI events in the observed entanglement data, our annual entanglement estimate for this Opinion is based on the scarring analysis presented in Hamilton et al. (2019). This approach provides the benefit of the doubt to the species and a more conservative estimate of total right whale entanglements. Therefore, non-M/SI entanglements documented in the GAR Marine Animal Incident Database will not be considered further in this Opinion.

Assignment of an observed entanglement event to a specific fishery or country of origin is rarely possible. Gear is often not retrieved. In cases where gear is retrieved, identification may not be possible because the same gear (e.g., lines and webbing) is used in multiple fisheries. Therefore, we must make assumptions on the origin of the gear for cases where that information is not available. As described below, we use different sources of data to partition M/SI between countries, gear type, cause of M/SI, and state and federal fisheries. Different data are available for each of these categories. In each case, we determined which data source represented the best available data and used that to partition the take. The rationale behind these assumptions is described below and summarized in Table 60.

Unknown Country Apportionment

The estimated M/SI with unknown country of origin was partitioned between the United States and Canada following the approach used by the ALWTRT for their April 2019 meeting; a 50/50 split, to apportion take between countries. In developing the transboundary approach for the ALWTRT, NMFS considered whether there was sufficient information to follow the guidance related to transboundary stocks provided in the NMFS Guidelines for Assessing Marine Mammal Stocks (GAMMS). Under this guidance, NMFS would assign serious injuries and mortalities that could not be identified to a country of origin based on the percentage of time right whales occur in each country's waters. This likely would have assigned a higher than 50 percent portion of unknown sources to U.S. fisheries based on historical distribution of the animals. However, given the stock's recent distribution shift, the large portion of the population increasingly occurring in Canadian waters, and the lack of rangewide survey coverage, it was determined that there was insufficient information to quantify the fraction of time right whales now spend on each side of the border. In developing the target for the ALWTRP, NMFS also considered apportioning based on known risk. It is clear from recent documented M/SI incidents where gear has been present that heavier snow crab gear poses a greater mortality risk than buoy lines associated with most nearshore lobster fisheries. However, given the large number of lines in U.S. waters including larger diameter lines and long traps/trawl configurations in offshore U.S. fisheries, significant risk occurs on both sides of the border. Seasonal and dynamic measures have been implemented in Canada, but broadscale measures have been in place in U.S. waters for many years. Given limited distribution information and transboundary fishery attributes, NMFS assessed an equal division of the unassigned serious injuries and mortalities between the United States and Canada. These methods were peer reviewed by the Center for Independent Experts, and while the reviewers did not come to consensus on accuracy, they considered the approach reasonable.[27] Therefore, for the purpose of this Opinion, we determined that this 50/50 split represented the best available information.

Unknown Gear Type Apportionment

For entanglements with no gear type identified, we assessed a variety of information to apportion these interactions to trap/pot or gillnet gear. Entanglements are categorized as unknown when there is no gear present or the gear that is present cannot be identified to a particular gear type. When there is net gear present, but it is unknown whether the net gear is gillnet or another net, the gear is categorized generally as net. There were 82 entanglements categorized as unknown and 2 as net (Table 59). Of the 56 non-serious injury entanglement cases from 2010-2018, 43 are categorized as unknown gear, with 23 percent of these unknown gear cases (10 of 43) reported to have gear present. Of the unknown gear M/SI cases (39), 56 percent (22) of the cases were reported with gear present. With the exception of one case, when gear was present and the entanglement case was classified as unknown gear, it was described as lines, sometimes with associated buoys or polyballs. Without identifying marks, we cannot know whether the line is from gillnet gear, trap/pot gear, or another source. In one case, the gear was described as a bridle, configuration unclear. Bridles are used in trap/pot fisheries.

[27] See peer review reports for North Atlantic right whale DST review report (2019-2) at https://www.st.nmfs.noaa.gov/science-quality-assurance/cie-peer-reviews/cie-review-2019.

Table 59: Number of entanglement interactions and M/SI by gear type from 2010-2018 Source: GAR Marine Animal Incident Database

	Unknown	Gillnet	Net	Pot/Trap	Total
Entanglements	82	6	2	17	107
M/SI	37	1.75	0.75	9	48.5

Based on the information provided in Table 59, M/SI appears to vary by gear type with 53 percent of known trap/pot, 29 percent of gillnet, and 38 percent of net interactions resulting in M/SI. For the cases in unknown gear, 45 percent of the interactions resulted in M/SI. However, this data must be interpreted cautiously given the small number of cases. Net interactions are associated with unidentified nets, which may include gillnets, cast nets, and weirs, among others. For gillnets and vertical lines, there are requirements under the ALWTRP (e.g., weak links in the buoy lines, weak links on the head rope) to reduce the severity of whale interactions with gillnet gear. Similar measures for other net types are not in place. It is possible that the differences in the M/SI between trap/pot and gillnet gear are due to the ability of the animal to break free from gillnet gear.

Interactions that involve gillnet net panels may be more easily detected and identified on large whales; resulting in a higher percentage of these interactions being identified in the database. In addition, 71 percent[28] of the interactions in confirmed gillnet gear were categorized as non-serious; while only 47 percent of interactions in confirmed trap/pot gear were categorized as non-serious. As noted above, large whales interacting with gillnet net panels may be able to break free of the gear, and the data indicate that large whales may be able to shed net panel gear.

The other component of the gillnet gear that entangles large whales are the vertical lines. As described above, the records indicate that gear was documented in approximately half of the interactions with unknown gear (which account for 77 percent of all interactions) that resulted in M/SI. With the exception of one, these cases all involved some type of line. Entanglement records in which no gear was present are determined based on wounds and scars. In these cases, it is possible that the interaction was with a different portion of the gear; however, given the data, a portion of these cases likely also involve vertical line gear.

The interaction rate with the gear is based, at least in part, on the co-occurrence of species and the gear. There is no data to indicate that the interaction rate with gillnet vertical lines would differ from the interaction rate with vertical lines associated with trap/pot gear. However, 99.7

[28] There were six gillnet entanglement cases, of which 1.75 resulted in M/SI. Although 4 of 6 (67 percent) of the cases were categorized as non-serious, the prorated value results in 71 percent of gillnet entanglements being categorized as non-serious.

percent of vertical lines in the action area are from trap/pot lines[29] (2016 IEC, unpublished data). Therefore, we anticipate that the interactions with unknown line gear are much more likely to be from trap/pot gear than gillnet gear.

Based on this information, we determined that it is more reasonable to apportion unknown M/SI to trap/pot gear, rather than simply using the ratio of confirmed gillnet to trap/pot gear to assign unknown M/SI cases given that (1) the records indicate line was the predominant gear involved in cases with unknown gear and the majority of the cases involved unknown gear; (2) interactions with net panels may result in less severe injuries as the animal may be able to break free from the gear; and (3) interactions with vertical lines are more likely to be trap/pot gear given the co-occurrence of the right whales and trap/pot gear. If these assumptions are not correct and gillnet gear is involved in a larger portion of the unknown entanglements, this may be compensated as interactions with unknown net gear were assumed to be from gillnet gear.

U.S. State vs Federal Apportionment

The NEFSC DST was used in this Opinion to compare right whale serious injury and mortality risk in U.S. waters from trap/pot gear in state and federal waters as equitably as possible. To capture how much risk from trap/pot gear is occurring in state versus federal waters, the DST assessed the risk reduction that would occur if all federal waters were closed to all trap/pot fishing gear. The DST showed that showed 60.4 percent of the risk from trap/pot gear to right whales is occurring in federal waters. Given this information, we apportion 60.4 percent of the estimated right whale entanglements occurring in U.S. trap/pot gear to the federal fisheries. The DST was not used to apportion the presumed entanglements in gillnet gear. As described above, all of the presumed gillnet entanglements are attributed to U.S. federal fisheries.

Entanglement vs Vessel Strike Apportionment

For apportioning takes with unknown cause between entanglement and vesssel strikes, we used observed cases from the most recent decade (2010-2019) for which information is available.

Other approaches were considered in apportioning take between entanglement and vessel strike. From 2003-2018, a review of 70 mortalities found that of the examined (56) and necropsied (44) carcasses, entanglements accounted for 58 percent and vessel strikes accounted for 43 percent of the mortalities (Sharp et al. 2019). In a separate study, from 1990 to 2017, NMFS reported 62 live right whales as having serious injuries that were defined as life-threatening and subsequently disappeared from the population (Pace et al. 2021). Entanglement accounted for the vast majority (54 of 62, or 87 percent) of these cases (Pace et al. 2021). During this same time period, 41 carcasses were examined, and only 49 percent of the deaths were determined to be entanglement related (Pace et al. 2021). These data suggest that cryptic deaths due to entanglements significantly outnumber cryptic deaths from vessel strikes (Pace et al. 2021).

For this Opinion, we used the observed data from 2010-2019[30]. Between 2010 and 2019, 15.04 M/SI were confirmed to be due to vessel strikes, and 50.5 M/SI were due to entanglement (GAR

[29] This percentage is an average across all months and includes all waters non-exempt and exempted from the ALWTRP.
[30] The 2019 observed data was used as it provides the most recent sightings data relevant to this apportionment.

Marine Animal Incident Database, unpublished data). Based on this ratio, we assume 23 percent of the cases in the United States where the cause was unknown were due to vessel strike and 77 percent due to entanglement. While this estimate of mortalities due to entanglement is slightly lower than the estimate included in (Pace et al. 2021) and the authors caution about using detected carcasses when determining cause of death, we believe it represents the best available information as it accounts for serious injury and mortality after the shift in right whale distribution and the exposure to different threats due to this shift. It is also consistent with the findings of Pace et al. (2021) that entanglements significantly outnumber vessel strikes in cryptic mortalities.

There are six categories of entanglement cases considered for attribution to U.S. state and federal fisheries: (1) observed entanglement, confirmed United States; (2) observed entanglement, country unknown; (3) observed, unknown cause, confirmed United States; (4) observed, unknown cause, unknown country; (5) unobserved (cryptic), unknown cause, unknown country; and (6) animals documented with entanglement scars.

Observed Entanglement, Confirmed United States: Entanglements confirmed to be from U.S. trap/pot or gillnet gear were assigned to the U.S. fisheries. Between 2010 and 2018, there were two observed M/SI entanglement cases involving confirmed U.S. fishing gear. One of these was in unknown trap/pot gear, and the other was in unknown gear.

Observed Entanglement, Country Unknown:

There were 38.75 M/SI observed entanglement cases involving gear from an undetermined country of origin from 2010 through 2018 (Table 58). Of these, 1.0 M/SI was identified as trap/pot, 1.75 M/SI as gillnet, 0.75 M/SI as net, and 35.25 M/SI as unknown gear type. As described above, although the net gear entanglements may not have been a result of gillnet gear, to ensure our estimate is conservative to the species, the 0.75 M/SI entanglements involving net gear were assumed to be gillnet gear resulting in a total of 2.5 M/SI entanglements in gillnet gear.

We applied the 50 percent U.S./Canada split to the totals, and presumed 19.375 M/SI entanglements were the result of the U.S. fisheries. Further, 0.5 M/SI entanglements were presumed to be in U.S. trap/pot gear, 1.25 M/SI in U.S. gillnet gear, and 17.625 in U.S. unknown gear.

Phase 1 of the Framework (ALWTRP proposed rule) will reduce risk to large whales from entanglement in lobster and Jonah trap/pot gear. Therefore, in order to properly evaluate the proposed measures, we apportioned the 19.375 M/SIs in unknown U.S. fisheries between gillnet and trap/pot. For entanglements with a gear type identified, the entanglement was assigned to that gear type. As described above, for entanglements with no gear type identified, we are assuming that all of the presumed U.S. entanglements in unknown gear were from trap/pot gear. Therefore, of the 19.375 M/SI entanglements presumed to be a result of the U.S. fisheries, 18.125 M/SI are presumed to be from U.S. trap/pot gear and 1.25 M/SI from gillnet gear.

Observed, Unknown Cause, Confirmed United States. For these cases, we applied the ratio described above (23 percent vessel strike, 77 percent entanglement) of all observed M/SI resulting from entanglement and vessel strikes from 2010 - 2019[31]. There was one mortality confirmed to the United States with an unknown cause. Applying the apportionment, 0.77 right whales were presumed to be entangled and died in U.S. fishing gear.

Observed, Unknown Cause, Unknown Country: There are also observed cases that are unknown cause and not assigned to country. As above, we assumed that 50 percent of these occurred in the United States, and, of those, 77 percent were assumed to be due to entanglement. When both of these assumptions are applied, it is equivalent to 38.5 percent.

There were 12 M/SI cases observed from 2010 to 2018 where the country and cause were unknown. Of these, 6 M/SI (50 percent of 12) are presumed to have occurred in the United States. Of these entanglements, 4.62 M/SI (77 percent of 6) cases are assumed to be a result of entanglement.

Unobserved (Cryptic), Unknown Cause, Unknown Country: We also used the vessel strike/entanglement ratio and the U.S./Canada split to apportion cryptic mortality. Natural mortality is not included in the apportionment as there is little evidence showing this to be a cause of right whale mortality except at the calf stage (Corkeron et al. 2018).

We estimate cryptic mortality by subtracting the observed mortality in U.S. and Canadian waters from the total mortality. The total right whale mortality estimate generated by the NEFSC from the state space model under status quo for 2010-2018 is 190 (using methods from Pace et al. 2017). From the total mortality estimate (190), we subtracted the observed M/SI due to entanglement (48.5), vessel strike (11.04), and unknown cause (20). We determined that 110.46 mortalities occurred that were not observed and, therefore, considered cryptic mortalities. Of these mortalities, 55.23 (50 percent of 110.46) are presumed to have occurred in the United States. Of these, 42.53 M/SI (77 percent of 55.23) cases are assumed to be a result of entanglement.

Scarring Rates: Research using the North Atlantic Right Whale Catalogue has indicated that between 2010-2017, on average, 30.25 percent of right whales acquired new wounds or scars from fishing gear annually (Hamilton et al. 2019). We used the U.S./Canada split to apportion the total annual entanglement rate. Assuming 50 percent of the 30.25 percent are the result of entanglement with U.S. fishing gear, we determined that an average of 15.125 percent of the right whale population becomes entangled annually in U.S. fishing gear (Table 61).

[31] The 2019 observed data was used as it provides the most recent sightings data relevant to this apportionment.

Table 60: Summary of the apportionment of M/SI cases between 2010 and 2018 that we assume to be from entanglement in U.S. gear.

Added to the Total	Assumption	Rationale	M/SI Entanglements Attributed to U.S. Fisheries 2010-2018 (Annual Average)
All confirmed U.S. gear entanglement cases	NA	Observed numbers being used	2 (0.22)
50 percent of confirmed entanglement cases from unknown country	Assumes half of cases belong to United States and half to Canada Assumes all entanglement cases not identified to gear type are trap/pot	Observed numbers being split between countries following apportionment from TRT process 99.7 percent of vertical line gear (2016 IEC data) is trap/pot gear[32]	19.375 (2.15)
77 percent of cases with unknown cause confirmed in U.S.	Assumes that ratio of confirmed entanglement and vessel strike case applies to cases with unknown cause. Assumes all entanglements are trap/pot	Observed numbers apportioned based on confirmed cases ratio. Scientific opinion differs on whether an observer bias may occur for these two types of risk; going with observed apportionment is best available.	0.77 (0.086)
38.5 percent of the cases with an unknown cause/unknown country	Assumes 50 percent of cases belong to United States and half to Canada Assumes that ratio of confirmed entanglement and vessel strike case applies to cases with unknown cause (77 percent). Assumes all entanglements are trap/pot	Carrying forward the same assumptions and ratios from observed/confirmed cases, U.S./Canada split, and assigning unknown entanglements to trap/pot.	4.62 (0.513)
38.5 percent of cryptic mortality	Assumes 50 percent of cases belong to United States and half to Canada Assumes that ratio of confirmed entanglement and vessel strike case applies to cryptic mortality (77 percent). Assumes all entanglements are trap/pot	Carrying forward the same assumptions and ratios from observed/confirmed cases, U.S./Canada split, and assigning unknown entanglements to trap/pot.	42.53 (4.725)
Total			69.29 (7.7)

[32] When considering U.S. waters that are non-exempt under the ALWTRT, the percentage decreases to 99.4.

Table 61: Summary of the apportionment of the estimated annual entanglement rate based on scarring analysis in Hamilton et al. 2019 between 2010 and 2017 that we assume to be from entanglement in U.S. gear.

50 percent of the annual average entanglement rate based on scarring analysis (Hamilton et al. 2019).	Assumes half of cases belong to United States and half to Canada	Scarring rate split between countries following apportionment from TRT process	15.125 percent of population

Total Entanglements in the U.S. Federal Fisheries

To estimate how much risk from trap/pot gear is occurring in U.S. federal fisheries, we used the results of the DST that showed 60.4 percent of the risk from trap/pot gear to right whales is occurring in federal waters. Given this information, we apportion 60.4 percent of the estimated right whale entanglements occurring in U.S. trap/pot gear to the federal fisheries.

As described above, to estimate the annual rate of total entanglements from the U.S. fisheries, we use the data presented in Hamilton et al. (2019) based on analysis of scarring rates between 2010 and 2017. Based on this information, we determined that the U.S. fisheries will entangle an average of 15.125 percent of the total right whale population annually (Table 61). Of the 15.125 percent of the population entangled annually, we apportioned 60.4 percent of the entanglements to the U.S. federal fisheries analyzed in this Opinion. Given this information, we determined that an average of 9.14 percent of the right whale population is entangled annually in U.S. federal fishing gear.

Total Entanglements in U.S. Federal Fisheries Resulting in M/SI

As described above, our analysis includes data from 2010 to 2018 for observed entanglements, presumed entanglements, and cryptic mortality. The final estimate of right whale M/SI as a result of entanglement in U.S. fishing gear between 2010 and 2018 is 69.29 (annual average 7.7) (Table 60).

The DST showed 60.4 percent of the risk from trap/pot gear to right whales in U.S. waters is occurring in federal waters. All entanglements presumed to be from U.S. gillnet gear (1.25 (annual average 0.125)) were assigned to the federal fisheries in this Opinion. While these entanglements may have occurred in gillnet gear used in fisheries not included in this Opinion, we are giving the "benefit of the doubt" to the species and assuming the interactions were caused by gear used in the federal fisheries in this Opinion.

To estimate the annual average number of entanglements that occurred in U.S. federal fishing gear, we first subtracted the estimated annual average of 0.125 M/SI that occurred in gillnet gear[33] from the total annual average estimated entanglements in U.S. fishing gear (7.7), which

[33] The gillnet estimate is based on the observed data from 2010-2019. The most recent 10-year period represents the best available information for apportioning interactions by gear type. The total mortality estimate is not available for

results in an annual average of 7.57 entanglements in trap/pot gear. We then apportioned 60.4 percent of the estimated entanglements in trap/pot gear (annual average 4.57) to the U.S. federal fisheries and then added back the estimated annual average of 0.125 entanglements in gillnet gear. This results in an estimated annual average of 4.7 M/SI. Given this information, we determined that an annual average of 4.7 right whale M/SI are the result of entanglement from gear used in the U.S. federal fisheries.

Applying the Framework Risk Reduction Percentage

As described in section 3.23.2.1, the Framework will implement four phases designed to reduce entanglement risk to right whales in federal waters. Here, we assess how M/SI in the federal fisheries will be reduced over the implementation of the Framework. The Framework includes:

- Phase 1: The implementation of the 2021 ALWTRP proposed rule measures to reduce the risk of entanglement resulting in M/SI by 58.1 percent[34] to right whales in trap/pot gear used in lobster and Jonah crab fisheries.
- Phase 2: In 2023, NMFS implements measures to reduce M/SI by 60 percent to right whales in federal gillnet and other trap/pot (e.g., finfish, red crab) fisheries.
- In 2023-2024, NMFS evaluates measures implemented in the 2023 action as well as new data on the right whale population and threats to assess progress towards achieving the conservation goals of the Framework. At this time, we also expect to be able to assess and incorporate effectively the risk reduction measures taken by Canada to address M/SI in their fisheries.
- Phase 3: In 2025, NMFS implements rulemaking to further reduce the remaining M/SI in federal fixed gear fisheries by 60 percent risk.
- In 2025-2026, NMFS evaluates measures implemented in 2025 action as well as new data on the right whale population and threats to assess progress towards achieving the conservation goals of this Framework. Based on the results of this evaluation, NMFS will determine the degree to which additional measures are needed to ensure the fisheries are not appreciably reducing the likelihood of survival and recovery.
- Phase 4: In 2030, further reduction will be implemented. These reductions will achieve the additional level of risk reduction (expected to be up to an 87% additional reduction) in M/SI of right whales as a result of entanglement with gear used in the federal fisheries that is needed at the time to ensure the likelihood of survival and recovery of the species.

The implementation of the Framework will change how the fisheries considered in this Opinion operate, and as such, these changes are considered in our analysis. While the Framework takes an adaptive management approach, NMFS is committed to achieving the level of reduction needed and specified. Therefore, this analysis considers the implementation of the Framework at the maximum levels specified.

2019, therefore we cannot calculate cryptic mortality and the 2019 data was not considered in estimating M/SI in pot/trap gear.

[34] The ALWTRP proposed rule has a risk reduction greater than 60 percent as it accounts for risk reduction previously achieved in Massachusetts state waters. In this Opinion, we analyze only the future risk reduction measures which will achieve a 58.1 percent reduction in risk to right whales.

Phase 1

Phase 1 on the Framework (ALWTRP proposed rule) will incorporate several provisions to reduce the risk of entanglements resulting in M/SI to right whales in the lobster and Jonah crab trap/pot fisheries by at least 58.1 percent. The gear modification requirements include reducing the number of lines in the water (e.g., via increasing the number of traps per trawl, time/area closures to persistent buoy lines) and reducing serious injury and mortality in the remaining lobster and Jonah crab buoy lines by specifying a low (no greater than 1700 lb) maximum breaking strength or weak inserts for vertical line to be used in certain areas depending on gear configurations. Additionally, measures will include gear marking requirements. More detailed information on the measures can be found in the ALWTRP proposed rule and draft Environmental Impact Statement (EIS). The discussion below describes the impact of each of the measures on whale entanglement risks.

- **Line Reduction Requirements:** Measures to reduce the number of vertical lines fished benefit large whales by reducing co-occurrence and associated opportunity for entanglement in buoy lines and associated gear. Measures include requirements to increase the minimum number of traps per trawl in the Northeast to reduce the number of vertical buoy lines in the water. These measures are expected to reduce the total number of entanglements.
- **Seasonal Buoy Line Closures:** Closures protect areas of predictable seasonal aggregations of right whales. The regulatory changes include several restrictions on when and where trap/pot gear can be set with persistent buoy lines. Two existing closures to trap/pot fishing would instead be closed to fishing trap/pot gear with persistent buoy lines; allowing “ropeless” fishing. Ropeless fishing is usually done by storing buoy lines on the bottom and remotely releasing the buoy to retrieve the line when fishermen are on site to haul in their trawl of traps, or other bottom gear. Proposed measures include two new seasonal closure areas. These measures are expected to reduce the total number of entanglements.
- **Weak Line/Insert Requirements:** The regulatory changes include provisions such as requiring that lobster and crab trap/pot gear modify buoy lines to using rope that breaks at 1700 lb. for substantial lengths of the buoy line or for some weak insertions. The specified strength rope or weak inserts is based on a study that indicated that, if a large whale does become entangled, it is more likely to exert enough force to break the rope before a severe entanglement occurs, reducing risk of serious injury or mortality. These measures are expected to reduce the severity of entanglements.
- **Gear marking requirements:** Gear marking requirements will not reduce impacts to large whales but will provide information that will enable us to better understand what fisheries are interacting with these species.

The 2021 ALWTRP proposed rule will reduce the risk of M/SI to right whales from U.S. lobster and Jonah crab trap/pot gear only. However, we are unable to partition the entanglement data above between the different trap/pot fisheries (e.g., lobster pots, fish pots, crab pots). Therefore, our analysis includes all entanglements that are presumed to be in U.S federal gear and will apply the federal portion of the 58.1 percent risk reduction to the estimated number of entanglements resulting in M/SI. Given that some of the risk reduction measures are designed to reduce the severity of entanglements and not the likelihood, as a conservative approach, the risk reduction was not applied to the estimated total number of right whale entanglements (i.e., non-serious entanglements).

Using results from the DST, we determined the proportion of the 58.1 percent reduction in M/SI due to the ALTWRP measures that would occur in state fisheries vs. federal fisheries. The results showed that 45.78 percent of the total ALWTRP proposed 58.1 percent risk reduction is from federal trap/pot fisheries. This indicates that of the 58.1 percent risk reduction in M/SI due to measures implemented under the ALWTRP, a 26.6 percent (45.78 percent of 58.1 percent) risk reduction of the total U.S. trap/pot entanglements will occur in federal waters. A 26.6 percent reduction of the annual estimate of total U.S. entanglements in trap/pot gear (7.57) results in a reduction of 2.01 M/SI entanglements. We subtract this reduction (2.01) from the total estimated M/SI entanglements in trap/pot gear used in the federal fisheries (4.57) resulting in 2.56 M/SI entanglements in the U.S. federal trap/pot fisheries after the implementation of the ALWTRP rulemaking. We then added in the estimated gillnet M/SI entanglements (0.125) and determined that with the measures implemented in the ALWTRP proposed rule, an annual average of 2.69 M/SI right whale entanglements will occur as a result of gear used in the U.S. federal fisheries (Table 62).

Table 62: Annual average number of right whale M/SI entanglements with measures implemented under the ALWTRP proposed rule, based on apportionments calculated using the DST

	Estimated M/SI in trap/pot gear (60.4% Fed)	**Reduction of M/SI with measures implemented under the proposed rule (26.6% Fed)**	**Remaining M/SI in trap/pot gear with measures implemented under the proposed rule**	**% Risk reduction in trap/pot gear**	**Remaining M/SI with measures implemented under the proposed rule (including gillnet)**
State	3	2.39	0.61	79.7%	0.61
Federal	**4.57**	**2.01**	**2.56**	**44%**	**2.69**
Total	7.57	4.4	3.17	58.1%	3.3

We have determined that, after the implementation of the 2021 ALWTRP proposed rule, the operation of the U.S. federal fisheries will entangle an annual average of 9.14 percent of the right whale population resulting in an annual average of 2.69 M/SI.

Phase 2

Phase 2 of the Framework will implement measures to reduce right whale M/SI in other federal trap/pot fisheries (i.e., red crab, scup, black sea bass) and federal gillnet fisheries by 60 percent. The gear modification requirements for Phase 2 are expected to be similar to those implemented in Phase 1. This includes, but is not limited to, reducing the number of lines in the water (e.g., via increasing the number of traps per trawl, time/area closures to persistent buoy lines) and reducing serious injury and mortality in the remaining buoy lines by specifying a low maximum breaking strength or weak inserts for vertical line to be used in certain areas depending on gear configurations.

As previously described, the 2021 ALWTRP proposed rule will reduce risk in American lobster and Jonah crab fisheries. Although most often we do not know which fishery's gear was involved in these entanglements, the lobster and Jonah crab fisheries represent the vast majority of the gear in U.S. federal waters, and we expect entanglements in other federal trap/pot gear to be rare. Therefore, we assumed that the 58.1 percent reduction from the ALWTRP proposed rule applied to the total M/SIs in trap/pot fisheries. Although it is possible that the risk reduction we assumed in Phase 1 may apply to entanglements that may be from federal trap/pot gear used in trap/pot fisheries not regulated in Phase 1 (e.g., red crab or finfish fisheries), there is no way to separate these out. If a rare entanglement occurs in federal gear that is not regulated under Phase 1, it is addressed in Phase 2 , which will reduce M/SIs occurring in federal gillnet and other trap/pot fisheries (i.e., red crab, scup, black sea bass) by 60 percent. Therefore, as the risk reduction in all trap/pot gear entanglements resulting in M/SI has been analyzed in Phase 1, Phase 2 will analyze a 60 percent risk reduction of right whale M/SI entanglements in gillnet gear.

Phase 2 is expected to reduce right whale M/SI entanglements in gillnet gear in federal waters by 60 percent. As previously described, we anticipate an annual average of 0.125 right whale M/SI as a result of entanglement in gillnet gear. A 60 percent risk reduction is expected to reduce the annual average of right whale M/SI entanglements in gillnet gear by 0.075 to 0.05. This reduction is subtracted from the estimated M/SI occurring after Phase 1 (annual average 2.69), which results in an estimated annual average of 2.61 right whale M/SI entanglements expected to occur in federal waters.

Given that some of the risk reduction measures will be designed to reduce the severity of entanglements and not the likelihood, the risk reduction was not applied to the estimated total number of right whale entanglements (i.e., non-serious entanglements). Therefore, we have determined that after the implementation of Phase 2 of the Framework in 2023, the U.S. federal fisheries will entangle an annual average of 9.14 percent of the right whale population and an annual average of 2.61 right whale M/SI entanglements in federal waters (Table 63).

Phase 3

For the federal fisheries to implement Phases 3 and 4 of the Framework, additional management measures will need to be put in place. Here, the Framework described general measures that could be implemented (independently or in conjunction) to achieve the required risk reduction of these phases, but NMFS will work with its partners to develop the specific measures to reduce M/SI. The measures listed here are not intended to be exhaustive, and any additional strategies to reduce risk to right whales that achieve the conservation goal will be acceptable towards achieving the risk reduction target of Phases 3 and 4 of the Framework.

Buoy Line Closures: Temporary or permanent closures of areas to fishing fixed gear with persistent buoy lines. Vessels would be allowed to fish with fixed gears in these areas without persistent buoy lines. These closures would protect areas of predictable seasonal aggregations of right whales.

Line Reduction Requirements: Reduce the number of vertical lines in the water. This can be achieved by increasing the minimum number of traps per trawl, trap reductions, or implementing line cap allocations of buoy lines in federal waters. Measures to reduce the number of vertical lines fished benefit right whales by reducing co-occurrence and opportunities for entanglement in buoy lines and associated gear.

“Ropeless Fishing”: Ropeless fishing technologies generally store buoy lines on the bottom and remotely releasing the buoy to retrieve the line when fishermen are on site to haul in their gear. Requiring ropeless fishing would remove buoy lines from the water and would benefit right whales by reducing opportunities for entanglement in buoy lines. A number of systems are currently being tested for operational effectiveness in trap/pot fisheries in the Greater Atlantic Region and are expected to be ready for implementation during the implementation of the Framework.

Phase 3 of the Framework will implement measures to reduce right whale M/SI in all federal fisheries by an additional 60 percent. A 60 percent risk reduction will reduce the annual average of 2.61 right whale M/SI entanglements in federal waters by 1.57, resulting in an annual average of 1.04 M/SI in the federal fisheries after the implementation of Phase 3. Although final measures for Phase 3 have not been determined, in order to achieve the risk reduction requirements in Phase 3, the federal fisheries will need to implement measures that will substantially reduce the co-occurrence of right whales and vertical lines. This reduction in co-occurrence is expected to reduce the number of total entanglements of right whales in federal waters. Although we cannot quantify the reduction in the number of entanglements, we expect it to be lower than 9.14 percent of the right whale population. Therefore, we have determined that after the implementation of Phase 3 of the Framework in 2025, the U.S. federal fisheries will entangle an annual average of less than 9.14 percent of the right whale population, and an annual average of 1.04 entanglements are anticipated to result in M/SI (Table 63).

<u>Phase 4</u>

Phase 4 of the Framework will implement measures to reduce right whale M/SI in all federal fisheries by an additional 87 percent. An 87 percent risk reduction will reduce the annual average of 1.04 right whale M/SI entanglements in federal waters by 0.91, resulting in an annual average of 0.136 M/SI in the federal fisheries after the implementation of Phase 4. Although final measures for Phase 4 have not been determined, in order to achieve the risk reduction requirements in Phase 4, the federal fisheries will need to implement measures that will substantially reduce the co-occurrence of right whales and vertical lines. This reduction in co-occurrence is expected to reduce the number of total entanglements of right whales in federal waters. Although we cannot quantify the reduction in the number of entanglements, we expect it to be lower than 9.14 percent of the right whale population. Therefore, we have determined that after the implementation of Phase 4 in 2030, the U.S. federal fisheries will entangle substantially less than 9.14 percent of the right whale population on average annually and will entangle an annual average of 0.136 that are anticipated to result in M/SI (Table 63).

Table 63: Annual average number of right whale M/SI entanglements in federal fisheries before and after measures implemented under the Framework

	Estimated M/SI in federal trap/pot gear	**Estimated M/SI in federal gillnet gear**	**Reduction of M/SI**	**Remaining M/SI with measures implemented**
Phase 1	4.57	0.125	2.01 (in trap/pot)	2.69
Phase 2	2.56	0.125	0.075 (in gillnet)	2.61
Phase 3	2.56	0.05	1.57 (any gear type)	1.04
Phase 4	1.04 (trap/pot and gillnet combined)		0.91 (any gear type)	0.136

As described in the Framework, NMFS will evaluate whether the requirement to reduce M/SI by 87 percent can be lowered, depending on any changes to the population status and environmental baseline (i.e., increased calving rates, risk reductions in vessel strikes and fisheries in Canada, risk reductions in U.S. state fisheries, and/or vessel-strike reductions in U.S. waters). The M/SI reduction may be reduced from the 87 percent target if an action outside the federal fisheries reduces risk to right whales by 0.5 M/SI on average annually (1 whale every two years). Should this occur, the M/SI reduction requirement will be reduced from 87 percent to 39 percent. It is possible that population-wide risk reduction measures will reach a level at which further action in the federal fisheries is not needed. If M/SI from other sources is reduced by greater than one M/SI on average annually, we will evaluate whether further action in the federal fisheries is needed and, if so, at what level.

We have determined that after the implementation of the Framework, the U.S. federal fisheries will entangle substantially less than 9.14 percent of the right whale population on average annually. Depending on if the criterion is met to reduce the risk requirement of the fourth and final phase, entanglements will result in the M/SI specified below (Table 64).

Table 64: Criteria for reductions in the final action of the Framework

Criteria	Reduction Required	Remaining M/SI in federal fisheries with measures implemented
No change in the status	87%	0.136
M/SIs outside the action reduced by 0.5 animals	39%	0.64

Summary of Estimating Right Whale Entanglements
Based on the apportionment information described above, we estimated that, on average, 7.57 North Atlantic right whales were seriously injured or died as a result of entanglement in U.S. trap/pot fishing gear annually from 2010 through 2018. Based on the results from the DST, 60.4 percent (4.57 M/SIs) of these M/SI entanglements occurred in federal trap/pot fisheries. The implementation of the 2020 ALWTRP proposed rule (Phase 1) will reduce the risk of M/SI due to entanglement in U.S. state and federal trap/pot fisheries by 58.1 percent; 26.6 percent of that reduction is expected to occur in federal waters. Therefore, after the implementation of the ALWTRP rule, we anticipate that an annual average of 2.56 right whale entanglements in trap/pot gear used in the U.S. federal fisheries will result in M/SI. In addition, we anticipate the M/SI of 0.125 (annual average) right whales in federal gillnet fisheries. The implementation of the ALWTRP rule will not reduce M/SI due to gillnet entanglements. Therefore, after the implementation of the ALWTRP proposed rule, the U.S. federal fisheries will entangle an annual average of 9.14 percent of the right whale population and an annual average of 2.69 M/SIs.

The implementation of Phase 2 of the Framework will reduce right whale M/SI entanglements in federal gillnet gear by 60 percent. Reducing takes in federal gillnet fisheries (annual average reduction of 0.075 M/SIs) will further reduce the estimated annual average M/SIs expected to occur in federal fixed gear fisheries to 2.61 right whales. Therefore, after the implementation of Phase 2 of the Framework in 2023, the U.S. federal fisheries will entangle an annual average of 9.14 percent of the right whale population and an annual average of 2.61 M/SI.

Phase 3 of the Framework will be implemented within 5 years to reduce the annual average of 2.61 right whale M/SI entanglements in federal waters by an additional 60 percent. Reducing takes in federal fisheries (annual average reduction of 1.56 M/SIs) will further reduce the estimated annual average M/SIs expected to occur in federal fixed gear fisheries to 1.04 right whales. The reduction in co-occurrence from Phase 3 is expected to reduce the number of total entanglements of right whales in federal waters and after the implementation of Phase 3 in 5 years, we expect less than an annual average of 9.14 percent of the right whale population to become entangled in federal waters and an annual average of 1.04 M/SI in federal fixed gear fisheries considered in this Opinion.

Phase 4 of the Framework will be implemented within 10 years to reduce the annual average of 1.04 right whale M/SI entanglements in federal waters by an additional 87 percent. Reducing takes in federal fisheries (annual average reduction of 0.9 M/SIs) will further reduce the estimated annual average M/SIs expected to occur in federal fixed gear fisheries to 0.136 right whales. The reduction in co-occurrence from Phase 4 is expected to further reduce the number of total entanglements of right whales in federal waters. After the implementation of Phase 4 in 10 years, we expect substantially less than 9.14 percent of the right whale population to become entangled in federal waters on average annually and an annual average of 0.136 M/SI in federal fixed gear fisheries considered in this Opinion.

7.2.1.4. Estimating Interactions with and Serious Injury/Mortality of Fin Whales

There have been frequent reports of fin whale entanglements over the years that provide adequate information to estimate the number of entanglements and M/SI rates. There have been no preliminary reports of fin whale entanglements from 2018-March 2021. For the purpose of this assessment, we are relying on data that has undergone review through the determination process (e.g., serious injury, country of origin).

To estimate future entanglement of fin whales due to the operation of the federal fisheries, we used the most recent validated data from the GAR Marine Animal Incident Database for the years 2009-2017. These years differ from the right whale assessment because any 2018 fin whale entanglement data has not been reviewed through the determination process (e.g., serious injury, country of origin). Additionally, there is no indication that an ecosystem regime shift caused a distribution shift of fin whales.

Between 2009 and 2017, there were 23 (14.75 M/SI) confirmed fin whale entanglements. Of those, 6 cases (5 M/SI) were confirmed to be the result of entanglement with Canadian fishing gear, 3 cases (1 M/SI) were confirmed to be the result of entanglement with U.S. fishing gear and 14 cases (8.75 M/SI) were the result of entanglement with gear from an unknown country of origin.

For the purpose of this Opinion we focus on entanglement events that are of undetermined origin or confirmed U.S. origin. The 6 Canadian cases (5 M/SI) will not be included in the analysis below. Although it is possible that gear of unknown origin may be from fisheries not considered in this Opinion, we are taking the most conservative approach and assuming the 14 cases (8.75 M/SI) with an unknown country of origin were the result of entanglement with U.S. federal fishing gear. Therefore, between 2009 and 2017, 17 fin whale entanglements (9.75 M/SI) are presumed to have occurred in U.S. federal fishing gear.

Given the small data set of observed interactions for fin whales, our analytical approach is conservative to account for uncertainty in the data and ensure where appropriate we provide the benefit of the doubt to the ESA-listed species. Of the 17 fin whale entanglements (9.75 M/SI), 14 (9 M/SI) involved an unknown gear type, 2 (0 M/SI) involved trap/pot gear, and 1 (0.75 M/SI) involved hook/monofilament gear. Given the limited information available on gear types involved in fin whale entanglements, we assume entanglements could occur as a result of any gear type used in the fisheries.

Based on the observed range of reported entanglements between 2009 and 2017, we anticipate an annual average of 1.89 fin whale entanglements resulting in an annual average of 1.08 M/SI due to the operation of the federal fisheries.

The implementation of the Conservation Framework as part of the proposed action will implement several measures that may reduce the risk of M/SI to fin whales. However, the measures being implemented are focused on reducing risk to right whales from entanglements in trap/pot gear. Given the uncertainty of the benefits that these measures will provide to fin whales, we are taking a conservative approach by not considering any reduction in the risk to the fin whale entanglement estimate. Given this information, we have determined that the operation of the U.S. federal fisheries with the Conservation Framework measures implemented will entangle an annual average of 1.89 fin whales resulting in an annual average of 1.08 M/SI. Although fin whale entanglements could occur as a result of any U.S. fishery (state and/or federal), given that recovered gear is rarely identified to a specific fishery, for the purpose of this Opinion we assume that the fin whale entanglements will occur as a result of the federal fisheries analyzed in this Opinion.

7.2.1.5. Estimating Interactions with and Serious Injury/Mortality of Sei Whales

Between 2009-2017, there were two (one M/SI) documented interactions with sei whales in fishing gear from unknown country of origin. Records of observed sei whale entanglements are so limited that a more general approach to estimating the number of entanglements and M/SI is required. For sei whales, we also use preliminary data to assert an assumption on future effects due to entanglement. Sei whale preliminary entanglement reports for 2018-2020 are presented below. These reports include one sei whale entanglement with unknown gear in both 2018 and 2019 (Table 65).

Table 65: Preliminary entanglement reports for sei whales (January 2018-March 2021)

Date	First seen	Status
3/12/18	Marathon Key, FL	Deceased (line documented at initial sighting but not present at necropsy)
4/26/19	Southwest of Cape Sable Island, Nova Scotia	Entangled

The 2018 case that resulted in mortality was found to have been the result of entanglement. The low level of reports may be a reflection of their generally offshore distribution where entanglements are less likely to be observed. Based on these documented entanglements and that the distribution of these species will overlap with the distribution of gear used in the fisheries known to be an entanglement risk, it is likely that sei whales may become entangled in gear used by the fisheries. Although sei whale entanglements are not documented each year, the observed entanglements are likely an underestimate of the actual level of entanglement occurring.

Therefore, we have determined that the federal fisheries will entangle one sei whale on average annually. The low occurrence of these observations does not allow us to make a valid determination on the anticipated levels of M/SI for these entanglements; therefore, we assume that the entanglements will result in M/SI.

The implementation of the Conservation Framework as part of the proposed action will implement several measures that may reduce the risk of M/SI to sei whales. However, the measures being implemented are focused on reducing risk to right whales from entanglements in trap/pot gear. Given the uncertainty of the benefits that these measures will provide to sei whales, we are taking a conservative approach by not considering any reduction in the risk to the sei whale entanglement estimate. Given this information, we have determined that the U.S. federal fisheries with the Conservation Framework measures implemented will entangle an annual average of one sei whale resulting in M/SI.

Although sei whale entanglements could occur as a result of any U.S. fishery (state and/or federal), given that recovered gear is rarely identified to a specific fishery, for the purpose of this Opinion, we assume that the sei whale entanglements will occur as a result of the federal fisheries analyzed in this Opinion.

7.2.1.6. Estimating Interactions with and Serious Injury/Mortality of Sperm Whales

There is very limited information on entanglement of sperm whales in fishing gear. Between 2009 and 2017, there have been no documented cases of sperm whales entangled in any gear used in the U.S. fisheries. Additionally, no preliminary sperm whales entanglements were reported in 2018 - March 2021. The most recent SAR confirms that there was one mortality as a result of entanglement in Canadian trap/pot gear in 2009, and one entanglement which resulted in mortality in Canadian pelagic longline in both 2009 and 2010 (Waring et al. 2015). A sperm whale was reported entangled in monkfish net on the Canadian Grand Banks in 2011, but was released alive and gear free (Waring et al. 2015).The lack of observed sperm whale entanglements may be a reflection of their generally offshore distribution where entanglements are less likely to be observed. The distribution of sperm whales overlaps the distribution of gear used in the red crab and lobster fisheries. The gear used in these fisheries is consistent with gear known to be an entanglement risk to large whales causing serious injury and mortality. Therefore, we assume sperm whales entanglements in gear used by the federal fisheries are possible. Given this information, we have determined that the federal fisheries will result in an annual average of one sperm whale entanglement. The lack of observations does not allow us to make a precise determination on the anticipated levels of M/SI for this entanglement; therefore, we assume that it will result in M/SI.

The implementation of the Conservation Framework as part of the proposed action will implement several measures that may reduce the risk of M/SI to sperm whales. However, the measures being implemented are focused on reducing risk to right whale entanglements in trap/pot gear. Given the uncertainty of the level of benefit that these measures will provide to sperm whales, we are taking a conservative approach by not considering any reduction in the risk to the sperm whale entanglement estimate.

7.2.2. ***Vessel Strikes***

The proposed action would expose all ESA-listed whale species under NMFS' jurisdiction to the risk of collision with vessels. Vessel collisions with marine mammals can result in death by massive trauma, hemorrhaging, broken bones, and propeller wounds (Campbell-Malone 2007, Knowlton and Kraus 2001). If relatively superficial, some individuals can recover from seemingly serious collisions, as evidenced by photographic time series of deep lacerations healing on individual animals (Silber et al. 2009).

Injuries and mortalities from vessel strikes are a threat to North Atlantic right, fin, sei, and sperm whales. Reports from 2009 to 2018 indicate that right whales experienced four vessel strike mortalities and five serious injuries, two of which were prorated serious injuries, in the United States or in an unknown country of origin. The annual average of vessel strikes between 2012 and 2016 in U.S. waters was 1.4 and 0.8 for fin and sei whale respectively (Hayes et al. 2019). The 2014 SAR for sperm whales indicates one vessel strike resulting in mortality in 2012 (Waring et al. 2015). From 2013-2020, no strandings of sperm whales were classified as human interactions (Hayes et al. 2020). While vessel collisions with marine mammals have been documented, there are few records of interactions between commercial fishing vessels and large cetaceans, and the interactions that have occurred involved species that are not ESA-listed. From 2010-2019, there was only one interaction reported between a large whale and a fishing vessel. In 2015, a self-report from a fishing vessel indicated the vessel interacted with a live, humpback whale. There have been no observed or reported interactions of right, fin, sei or sperm whales with federal fishing vessels. This analysis focuses on whether interactions with fishing vessels participating in the fisheries considered in this Opinion are likely to impact ESA-listed large whales.

Fishing vessels actively fishing either operate at relatively slow speeds, drift, or remain idle, when setting, soaking and hauling gear. Thus, any listed species in the path of a fishing vessel would be more likely to have time to move away before being struck. Fishing vessels transiting to and from port or between fishing areas can travel at greater speeds, particularly recreational vessels, and thus do have more potential to strike a vulnerable species than during active fishing. However, larger vessels are required to comply with seasonal management areas that have speed restrictions to help protect large whales.

Several large scale management efforts to mitigate vessel strikes have proven to be successful (Laist et al. 2014), including the ship speed restriction rule implemented in 2008 (50 CFR 225.105), shifts in traffic separation schemes in the Bay of Fundy and Boston, and the designation of the Roseway Basin and Great South Channel as Areas to be Avoided. In the United States, the Seasonal and Dynamic Management Areas will continue to reduce vessel traffic around aggregations of right whales and lower the risk of any vessel striking a right whale. As described in Section 5.4.6, since the implementation of the speed restriction, there has been a decline in large whale mortalities resulting from vessel strikes.

Given the rarity of vessel strikes when considering (1) the large amount of vessel traffic in the action area, (2) that all fishing vessels (state, federal, and unregulated) represent only a portion of marine vessel activity[35], (3) that fishing vessels considered in this Opinion represent an even smaller portion of marine activity; and (4) regulations in place to reduce the risk of vessel strike to whales, it seems extremely unlikely and discountable that a fishing vessel would strike a whale, even during transiting. Based on this information, we have determined that all listed marine mammals in the action area are not likely to be adversely affected by fishing vessels operating under the proposed action.

7.2.3. *Prey*

We have determined that the operation of the fisheries will not have any adverse effects on the availability of prey for right, fin, sei, and sperm whales. Right whales and sei whales feed on copepods (Perry et al. 1999). Dense aggregations of late stage and diapausing *C. finmarchicus* in the action area will not be affected by the fisheries. As described in section 4.1.10, the fisheries will not affect the availability of copepods for foraging right and sei whales. Copepods are very small organisms that will pass through fishing gear rather than being captured in it. In addition, copepods will not be affected by turbidity created by the gear moving through the water. Fin whales feed on krill and small schooling fish (e.g., sand lance, herring, mackerel) (Aguilar 2002). The fisheries' fishing gear operates on or very near the bottom, while schooling fish such as herring and mackerel occur higher in the water column. Therefore, with the exception of the mackerel/squid/butterfish fishery, the fisheries target and bycatch species are not foraged on or a primary prey species (Perry et al. 1999). Although small schooling fish species (including mackerel) may be caught in net gear targeting mackerel/squid/butterfish, it is one of many prey species rather than a dominant prey species on which they depend. Therefore, because fin whales are less prey selective than some other large whale species and are not expected to be food limited, the fisheries will not affect the availability of prey for fin whales (Perry et al. 1999). Sperm whales feed during summer at high latitudes, where they feed primarily on squid; other prey includes octopus and demersal fish. Sperm whales primarily only overlap with the red crab and lobster fishery, and these fishery are not expected to capture any prey species of sperm whales. We have determined that the operation of the fisheries will not affect the availability of prey for foraging whales.

7.2.4. *Habitat*

As described above, bottom trawl is the only gear type used by the ten fishery management plans considered in this Opinion that has the potential to adversely affect bottom habitat in the action area. Although trawl gear may interact with bottom habitat in the action area, any alterations of bottom habitat are not expected to affect right, fin, sei, and sperm whale's ability to forage, migrate, or breed (Baumgartner et al. 2003, IWC 1992, Pace and Merrick 2008, Perry et al.

[35] Fishing vessels accounted for less than 2 percent of the trips made by commercial fishing (all vessels reporting into the VTR system) and recreational vessels in the Greater Atlantic Region vessels in 2012 and 2013. This percentage is likely even lower as it does not include all vessel types (e.g., ferries, tankers). We also reviewed the AIS data from 2016 and 2017 (Memorandum from the Jennifer Anderson to the Record, December 23,2020 for the caveats associated with this data). AIS is required on vessels that are 65 ft or greater in length. It is not possible to separate fishing vessels participating in the fisheries in this Opinion from other fishing vessels captured in the AIS data. Approximately 15 percent of the vessels permitted in the ten FMPs are greater than or equal to 65 ft in length. A large number of vessels (e.g., recreational and fishing vessels <65 ft) operating in the Greater Atlantic Region are not required to use AIS. Therefore, we do not believe that the AIS provides the most accurate picture of the percentage of trips within the region that are taken by the fisheries considered in this Opinion.

1999). As described above, any disturbance to bottom habitat from trawl gear is expected to be temporary and localized. For these reasons, and the lack of any evidence that fishing practices affect habitats in degrees that harm or harass ESA-listed whales, we find that while the fisheries may potentially interact with benthic habitats, the effects of the habitat interactions will be too small to be meaningfully measured or detected and will therefore will have an insignificant effect on ESA-listed whales.

7.3. Effects to Sea Turtles

7.3.1. *Gear Interactions*

Certain fishing gears may directly affect sea turtles. Of the gears used by fisheries in this Opinion, sea turtles are known to interact with bottom otter trawls, gillnet, the lines of trap/pot gear, and hook and line gear.

7.3.1.1. Factors Affecting Sea Turtle Interactions

The primary factors affecting sea turtle interactions with the ten fisheries assessed in this Opinion are: (1) overlap in time and space, (2) method of fishing, (3) the behavior of sea turtles in the presence of gear, and (4) oceanographic features. As described in the *Status of the Species*, the occurrence of loggerhead, leatherback, Kemp's ridley, and green sea turtles in Northwest Atlantic waters is primarily temperature dependent. In general, sea turtles move up the U.S. Atlantic coast from southern wintering areas as water temperatures warm in the spring (Braun-McNeill and Epperly 2002, Braun-McNeill et al. 2008, James et al. 2005a, James et al. 2005b, James et al. 2005c, Keinath et al. 1987, Morreale and Standora 1998, Morreale and Standora 2005, Musick and Limpus 1997, Shoop and Kenney 1992). Recreational anglers have reported sightings of sea turtles in inshore waters (bays, inlets, rivers, or sounds) as far north as New York as early as March-April, but in relatively low numbers (Braun-McNeill and Epperly 2002). The trend is reversed in the fall as water temperatures cool. Near Cape Hatteras during late fall and early winter, the narrowness of the continental shelf and influence of the Gulf Stream helps to concentrate sea turtles, making them more susceptible to fishery interactions (Epperly 1995). Greater numbers of loggerheads, Kemp's ridleys, and greens are found in inshore, nearshore, and offshore waters of the southern Mid-Atlantic (Virginia and North Carolina) from May-November (Mansfield et al. 2009) and in inshore, nearshore, and offshore waters of the northern Mid-Atlantic (New York and New Jersey) from June-October (Braun-McNeill and Epperly 2002, Keinath and Musick 1993, Morreale and Standora 1994). Hard-shelled sea turtles are more commonly found in waters south of Cape Cod and Georges Bank, but may also occur in waters farther north (Morreale and Standora 1994). Leatherback sea turtles have a similar seasonal distribution, but have a more extensive range into the Gulf of Maine compared to the hard-shelled sea turtle species (James et al. 2005a, James et al. 2005b, Mitchell et al. 2002, Shoop and Kenney 1992).

Extensive survey effort of the continental shelf from Cape Hatteras to Nova Scotia, Canada in the 1980s revealed that loggerheads were observed at the surface from the beach to bottom depths up to 14,700 ft (4,481 m) (CETAP 1982). However, they were generally found in waters where bottom depths ranged from 72-161 ft (median 120 ft) (22-49 m (median 36.6 m)) (Shoop and Kenney 1992). Leatherbacks were sighted at the surface in waters with bottom depths ranging from 3.3 - 13,620 ft (1-4,151 m) (Shoop and Kenney 1992). However, 84.4 percent of leatherback sightings occurred in waters where the bottom depth was less than 590 ft (180 m); whereas 84.5 percent of loggerhead sightings occurred in waters where the bottom depth was less than 262 ft (80 m) (Shoop and Kenney 1992). Neither species was commonly found in waters over Georges Bank, regardless of season (Shoop and Kenney 1992). The CeTAP study did not include Kemp's ridley and green sea turtle sightings, given the difficulty of sighting and identifying these sea turtle species (CETAP 1982).

More recently as part of an AMAPPS survey, the NEFSC and SEFSC conducted two shipboard and two aerial line transect surveys covering U.S. Atlantic waters from Florida to Maine, from the coastline to the U.S. EEZ and slightly beyond from June 27 to September 28, 2016 (NMFS 2016a). The aerial abundance surveys targeted sea turtles in Atlantic continental shelf waters from the shore to about the 328-ft (100 m) or 656-ft (200 m) depth contour, depending on the location. The shipboard abundance surveys targeted sea turtles in waters at the shelf break, starting from the offshore edge of the plane's survey area to waters farther offshore to the U.S. EEZ and slightly beyond. The surveys completed about 18,338 nmi (33,963 km) of track lines: 5,796 nmi (10,735 km) from ships and 12,542 nmi (23,228 km) from planes. The most frequently detected sea turtles were loggerheads, with about 1,000 individuals that ranged from 26-41° N, mostly in waters on the continental shelf. Studies conducted in 2016 also investigated methods to estimate spatial and temporal distributions of tagged loggerhead sea turtle densities (NMFS 2016a).

Researchers also conducted aerial surveys in coastal ocean waters of Maryland and Virginia from spring through fall in 2011 and 2012 (Barco et al. 2018). Ocean abundance estimates of loggerheads were highest in the spring months of May-June and lower in the fall months of September-October. Ocean abundance estimates for loggerheads during the summer months of July-August were in between the spring and fall ranges, while no surveys were flown in the winter months from November-March (Barco et al. 2018).

Sea turtle interactions with trawl and gillnet gear used in the ten fisheries can result in entanglements of the head, limbs, or carapace or captures of the animal. Captures of sea turtles in gillnets are a severe type of interaction as they often result in injury and death. Gillnets are so effective at catching sea turtles they were commonly used in the historical sea turtle fishery (Witzell 1994). Drowning may occur due to forced submergence from the weight of the gear or, at a later time, if trailing gear becomes lodged between rocks and ledges below the surface. Although drowning due to forced submergence is the most serious risk to sea turtles in gillnet gear, constriction of a sea turtle's flippers can lead to infection or amputation of limbs, which may result in mortality or impaired foraging or swimming ability. Sea turtles that do escape often retain pieces of gear that can inhibit their foraging or survival. If the turtle is released or escapes with line attached, the flipper may eventually become occluded, infected, and necrotic.

Sea turtles may also drown due to forced submergence in bottom trawl gear. Recent studies have also shown that capture in fishing gear followed by rapid decompression may result in gas bubble formation in the blood stream (embolism) and tissues. This can lead to organ injury, impairment, and mortality in some animals (Crespo-Picazo et al. 2020, Fahlman et al. 2017, García-Párraga et al. 2014, Parga et al. 2020). Gas embolism has been documented in green, loggerhead, olive ridley, and leatherback sea turtles (Crespo-Picazo et al. 2020, García-Párraga et al. 2014, Parga et al. 2020) and in turtles caught in gillnet and trawl gear (Fahlman et al. 2017). The likelihood of fatal decompression increases with increasing depth fished (Fahlman et al. 2017). Size of the gear (e.g., mesh size), duration of sets/tows, and effectiveness of gear modifications (TEDs in trawls) will influence the likelihood of injury and mortality to sea turtles that are incidentally caught (Epperly et al. 2002, Murray 2007, 2008, 2009a, 2018, 2020, Stacy et al. 2015, Warden 2011a).

Available entanglement data for sea turtles indicate they are also vulnerable to entanglement in trap/pot gear. Aside from hatchling life stages, sea turtles cannot be caught in the pots or traps themselves since the vents/openings leading inside are far smaller than any of these species. Since hatchling sea turtles are pelagic rather than benthic once entering the water from the nesting beach, they are also not susceptible to entering the pot or trap on the ocean bottom. The most commonly documented turtle entanglements are with the vertical lines of fishing gear. However, sea turtles also entangle in groundlines or surface system lines of trap/pot gear. Given data documented in the GAR STDN database, leatherback sea turtles seem to be the most vulnerable turtle to entanglement in vertical lines of fixed fishing gear in the action area. Long pectoral flippers may make leatherback sea turtles more vulnerable to entanglement. Leatherbacks entangled in fixed gear are often restricted with the line wrapped tightly around the flippers multiple times suggesting entangled leatherbacks are typically unable to free themselves from the gear (Hamelin et al. 2017). Hamelin et al. (2017) observed leatherbacks that encountered a vertical line attempt to push the buoy and line away with one or both front flippers or attempt to swim away with broad, raised flipper strokes followed by a struggle with flailing the flippers, diving, and rolling. This behavior repeatedly encircles the flippers and neck, which may result in injuries or suffocation. This behavior can also cause the vertical line of fixed fishing gear to shorten, which can lead to the leatherback being forcibly submerged and can result in drowning (Hamelin et al. 2017). The leatherback's diet is composed predominantly of jellyfish species. Leatherbacks may mistake a surface buoy or submerged float for their jellyfish prey, potentially resulting in an increased risk of entanglement in vertical lines (Dodge et al. 2018). Leatherback entanglements in trap/pot gear may be more prevalent at certain times of the year when they are feeding on jellyfish in nearshore waters (i.e., Cape Cod Bay) where trap/pot fishing gear is concentrated. Hard-shelled turtles also entangle in vertical lines of trap/pot gear. Due to leatherback sea turtles large size, they have the strength to wrap fixed fishing gear lines around themselves, whereas small turtles such as Kemp's ridley or smaller juvenile hard-shelled turtles likely do not (Sampson, pers comm January 16, 2020). The factors influencing entanglements of larger hard-shelled sea turtle in trap/pot fishing gear used by the fisheries in this Opinion are unclear.

Records of stranded or entangled sea turtles show entanglement of trap/pot lines around the neck, flipper, or body of the sea turtle; these entanglements can severely restrict swimming or feeding (Balazs 1985). Drowning may occur quickly if the weight of the gear prevents the turtle from reaching the surface to breathe or, at a later time, if trailing gear becomes lodged between rocks and ledges below the surface. If a sea turtle is entangled when young, the line could become tighter and more constricting as the sea turtle grows, cutting off blood flow and causing deep gashes, some severe enough to remove an appendage (Balazs 1985). A sustained stress response, such as repeated or prolonged entanglement in gear makes these species less able to fight infection or disease, and may make them more prone to boat/ship strikes and predation (Lutcavage and Lutz 1997). Of the entangled sea turtles that do not die from their wounds, some may suffer impaired swimming or foraging abilities, altered migratory behavior, or altered breeding or reproductive patterns due to injuries resulting from the entanglement (Balazs 1985).

Leatherbacks may be more susceptible to drowning compared to other sea turtles due to their unusual physiology and metabolic processes (Lutcavage and Lutz 1997). Leatherbacks lack calcium, which aids in the neutralizing of lactic acid that builds up by increasing bicarbonate levels. The dive behavior of leatherbacks consists of continuous aerobic activity. When an entanglement occurs, available oxygen decreases allowing anaerobic glycolysis to take over producing high levels of lactic acid in the blood (Lutcavage and Lutz 1997). Therefore, especially when bycaught, the stored oxygen is likely to be quickly used. The softer epidermal tissue of leatherbacks may also make them more susceptible to severe injuries from entangling gear. As with gillnet gear, constriction of a sea turtle's neck or flippers can lead to severe injury or mortality. While drowning is the most serious consequence of entanglement, constriction of a sea turtle's flippers can amputate limbs, also leading to death by infection or to impaired foraging or swimming ability. If the turtle escapes or is released from the gear with line attached, the flipper may eventually become occluded, infected, and necrotic. Entangled sea turtles can also be more vulnerable to collision with boats, particularly if the entanglement occurs at or near the surface (Lutcavage et al. 1997).

In regards to the recreational component of the ten fisheries, stranding data provide some evidence of interactions between recreational hook and line gear and ESA-listed species. Sea turtles are known to ingest baited hooks or have their appendages snagged by hooks, both of which have been recorded in the STSSN database. Loggerheads and Kemp's ridleys are the species caught most often in the Greater Atlantic Region; these turtles frequently ingest the hooks. Deceased sea turtles with hooks in their digestive tract have been reported, although, it is assumed that most sea turtles hooked by recreational fishermen are released alive. Some turtles will break free on their own and escape with embedded/ingested hooks and/or trailing line. Others may be cut free by fishermen and intentionally released. These sea turtles will escape with embedded or swallowed hooks or trailing varying amounts of monofilament fishing line, which may cause post-release injury or death. The ingested hook and/or the trailing, monofilament fishing line may ultimately be swallowed and ingested by the animal, potentially leading to constriction and strangulation of the sea turtle's internal digestive organs; or the line may become entangled around the animal's limbs (which may lead to limb amputations) or around seafloor obstructions, preventing the animal from surfacing (leading to drowning). Thus, some of these hooking/entanglement interactions may eventually be lethal.

NMFS has considered other factors that might affect the likelihood that sea turtles will become entangled in fishing gear. As described above, these other factors include the behavior of sea turtles around fishing gear. Sea turtles have been observed to remain at the bottom or dive to the bottom and hunker down when alarmed by loud noise or gear (Memorandum to the File, L. Lankshear, December 4, 2007, DeAlteris et al. 2010), which could place them in the path of a trawl. Video footage recorded by NMFS SEFSC's Pascagoula Laboratory showed that loggerhead sea turtles will keep swimming in front of an advancing shrimp trawl, rather than deviating to the side, until the turtles become fatigued and are caught by the trawl or the trawl is hauled up (NMFS 2002b). At a workshop on mitigating interactions in trawl fisheries, it was noted that sometimes sea turtles remained on the bottom with bottom disturbance from trawl gear, while others shot to the top (DeAlteris 2010). There was also additional discussion about whether sea turtle behavior in front of approaching trawl gear might be indicative of how long it had been since the turtle had last surfaced for air (DeAlteris 2010). The information on behavior in front of and around trawl gear is inconclusive (DeAlteris 2010).

Benthic immature and adult loggerhead and Kemp's ridley sea turtles feed on benthic organisms such as crabs, whelks, and other invertebrates including bivalves (Burke et al. 1994, Burke et al.

1993, Dodd 1988, Keinath et al. 1987, Lutcavage and Musick 1985, Morreale and Standora 2005, Seney and Musick 2005, Seney and Musick 2007). Green sea turtles also feed on the ocean bottom. Therefore, if loggerhead, Kemp's ridley, and green sea turtles are foraging on the bottom or swimming through the water column in areas where the fisheries operate, they would be at risk.

Research conducted on the use of the water column by sea turtles provides additional information on the co-occurrence of sea turtles and fisheries. Starting in 2007, Coonamessett Farm began a series of research projects to assess and implement the use of an ROV to observe sea turtle behavior in the water column and on the sea floor in the Mid-Atlantic. The ROV studies focused on Atlantic sea scallop fishing grounds with water depths of 131 – 262 ft (40-80 m) during the months of June (2008, 2009), July (2009), August, (2008) and September (2007, 2009) (Smolowitz and Weeks 2010, Weeks et al. 2010). During these studies, over 50 sea turtles were tracked by ROV for periods ranging from two minutes to over eight hours (Smolowitz and Weeks 2010, Weeks et al. 2010). In addition to footage collected from the ROV, visual observations and recordings from the masthead were obtained. A range of loggerhead behaviors were observed, including feeding, diving, swimming, surface, and social behaviors. Loggerheads were observed feeding on jellyfish within the top 33 ft (10 m) of the surface and on crabs and scallops on the ocean bottom (Smolowitz and Weeks 2010, Weeks et al. 2010). A number of sea turtles were recorded on the ocean bottom at depths of 161-230 ft (49-70 m), and water temperatures of 7.5 °C-11.5 °C (Smolowitz and Weeks 2010, Weeks et al. 2010). Bottom times in excess of 30 minutes were recorded (Weeks et al. 2010).

The effect of certain oceanographic features may affect the likelihood of an interaction with sea turtles. A review of the data associated with 11 sea turtles captured by the scallop dredge fishery in 2001 concluded that the captured sea turtles appeared to have been near the shelf/slope front (D. Mountain, pers. comm.). Intensity of biological activity in the Northwest Atlantic has been associated with oceanographic fronts, including nutrient fluxes and biological productivity. Particular oceanographic features and processes that influence biological activity include vertical mixing by tides; seasonal heating and cooling that leads to winter convection and vertical stratification in summer; pressure gradients from density contrasts set up by deep water inflows and lower salinity waters; and influxes of the cold, fresher waters associated with Scotian Shelf Water (Townsend et al. 2006). There may be an increased risk of interactions between sea turtles and fishing gear in areas where these oceanographic features occur simply because there are possibly more sea turtles and more fishing gear present, which increases the potential for interactions. However, at present we are unable to determine if any of these oceanographic features affect the likelihood of interactions between sea turtles and the fisheries in this Opinion. As discussed later on in this section, variables such as latitude, bottom depth, and sea surface temperature have been correlated with sea turtle interaction rates with gillnet and bottom trawl gear in the Mid-Atlantic (Murray 2018, 2020).

Given the seasonal distribution of sea turtles and the times and areas when the ten fisheries operate, green, Kemp’s ridley, loggerhead, and leatherback sea turtles are likely to overlap with operation of the fisheries primarily from May through November. Loggerhead interactions are possible year-round in the southern portion of the Mid-Atlantic (Murray and Orphanides 2013). Interactions with other sea turtle species outside these months and in other portions of the action area are certainly possible, albeit at lower frequencies.

7.3.1.2. Existing Information on Interactions with Sea Turtles

The discussion of sea turtle interactions that follows will focus on trawl, gillnet, trap/pot, and both commercial and recreational hook and line gear. Sea turtles incidentally captured or entangled in these types of fishing gear must be reported to NMFS on VTRs that are required for all federal fisheries except the American lobster and Jonah crab fisheries. At present, compliance with the requirement for federally-permitted fishermen to report sea turtle interactions on their VTRs is believed to be very low (as evidenced by the lack of reported interactions that have been documented on vessels with observers in recent years). Without reliable VTR reporting of sea turtle interactions, we are using information on bottom trawl, gillnet, and hook and line interactions collected through the observer programs managed by the NEFSC's Fishery Sampling Branch. These programs collect, process, and manage data and biological samples obtained by trained observers during commercial fishing trips throughout the New England and the Mid-Atlantic regions. For trap/pot gear interactions, we also reviewed sea turtle entanglement data that has been collected through and provided by GAR STDN because the observer programs have observed very few trap/pot trips until recently.

Past observed interactions of sea turtles in these four gear types were reviewed in the 2002 Atlantic deep sea red crab, 2013 batched fisheries, and the 2014 American lobster opinions. Updated information is provided herein. The number of reported interactions is a fraction of the total amount occurring, which is unknown for certain species and gears. However, there are model-based bycatch estimates available for sea turtles in both bottom trawl and gillnet fisheries in the Mid-Atlantic and Georges Bank, which provide an estimate of the total number of encounters based on an extrapolation of observed interactions (Murray 2018, 2020). For these gears, only interactions of green sea turtles in gillnet gear were observed too infrequently throughout Georges Bank and the Mid-Atlantic to support model-based bycatch estimates (Murray 2018, 2020). Although included in the study area for the Murray (2018, 2020) bycatch analyses, there are no estimates of sea turtle bycatch in the Gulf of Maine due to a lack of observed bycatch events. There are two records of loggerhead sea turtles observed captured in gillnet gear in the Gulf of Maine, one in 2010 and one it 2018.

The majority of interactions between sea turtles and fisheries considered in this Opinion have occurred south of the Gulf of Maine; this is likely because the distribution of sea turtles correlates with warmer water temperatures, resulting in greater densities of sea turtles south of Cape Cod. The spatial distribution of sea turtles off southern New England and in the Mid-Atlantic is coincident with several fisheries that may target or incidentally land fish species managed under the ten FMPs in this Opinion. As indicated above, the vast majority of sea turtle interactions with the trawl and gillnet components of these fisheries involve loggerheads (Murray 2018, 2020).

From 2014-2018 (the most recent 5-year period that has been statistically analyzed for trawls), NEFOP observers documented 50 loggerhead sea turtle interactions in U.S. Atlantic bottom trawl gear[36], 48 of which occurred in the Mid-Atlantic[37]. The ASM Program documented no sea turtles over this period. Observers also recorded five Kemp's ridleys, three leatherbacks, and two green sea turtles in bottom trawl gear from 2014-2018 in the Mid-Atlantic and on Georges Bank. The majority (83 percent) of the observed interactions occurred between July and October. Interaction rates were stratified by region, latitude zone, season, and depth. Within each stratum, observed interaction rates were multiplied by total days fished from VTR trips to calculate the estimated number of sea turtle interactions. For VTR trips with TEDs, estimated interactions of hard-shelled sea turtle species were proportioned into observable interactions (those that passed through the TED into the codend), and unobservable/quantifiable interactions (those that escaped out through the TED opening). On TED trips in each stratum, observable interactions were 3 percent of total estimated interactions, and unobservable/quantifiable interactions were 97 percent of total estimated interactions, based on a 97 percent experimental exclusion rate (Watson Jr. 1981 as cited in Murray 2020). Total observable mortalities were estimated by applying the mortality rate (50 percent) for turtles observed in trawl gear interactions from the most recent time series (2013-2017) available at the time (Murray 2020, Upite et al. 2018) to the total estimated observable interactions. The mortality rate for unobservable yet quantifiable interactions was assumed to be 0 percent (Murray 2015b, 2020). As described above, these are animals that escaped through the TED opening and are expected to survive.

The highest loggerhead interaction rate (0.43 turtles/day fished) was in waters south of 37º N (approximately Virginia Beach, Virginia) during November to June in waters greater than 164 ft (50 m) deep (Murray 2020). This is mostly south of the Greater Atlantic Region where the vast majority of the effort in the fisheries in the Opinion occurs. The greatest number of estimated interactions occurred in the Mid-Atlantic region north of 39º N (approximately Cape May, New Jersey), during July to October in waters less than 164 ft (50 m) deep, due to a greater amount of commercial effort in this stratum compared to those farther south. Within each stratum, interaction rates for non-loggerhead species were lower than rates for loggerheads (Murray 2020).

From 2014-2018, 571 loggerheads (CV=0.29, 95% CI=318-997) were estimated to have interacted with bottom trawl gear in the U.S. Mid-Atlantic while 12 loggerheads (CV=0.70, 95% CI=0-31) were estimated to have interacted with bottom trawls on Georges Bank (Murray 2020). The total number of loggerhead sea turtle interactions across both regions was equivalent to 182 adults[38]. Murray (2020) estimated mortality by applying a mortality rate of 50 percent (from the most recent time series (2013-2017) available at the time of the analysis) to observable interactions and a mortality rate of 0 percent to unobservable but quantifiable interactions. Using these rates, an estimated 272 loggerhead sea turtles (87 adult equivalents) have died from these

[36] Takes in the southern Mid-Atlantic shrimp twin trawl fishery were not included in the bycatch analysis or this Opinion because takes in this fishery are estimated by the NMFS Southeast Region and Northeast observers no longer observe this fishery. Additionally, authorization of the Mid-Atlantic shrimp twin trawl fishery is not part of the proposed action under consideration in this consultation.
[37] One of these included a sea turtle that could not be identified to species, but for this analysis it was presumed to be a loggerhead based on characteristics described by observers. The observer noted it was "dark brown, tannish with 5 vertebral scutes and an estimated length of [36 inches] 91cm".
[38] Explained in more detail on pg. 275 adult equivalence considers a turtle's reproductive value (RV), defined as the contribution of an individual in an age class to current and future reproduction.

interactions. In the Mid-Atlantic, 38 loggerheads were estimated to have been excluded by TEDs (i.e., unobservable but quantifiable interactions).

From 2014-2018, a total of 46 Kemp's ridley (CV=0.45, 95% CI=10-88) and 16 green (CV=0.73, 95% CI=0-44) sea turtles were estimated to have interacted with bottom trawl gear in the Mid-Atlantic, of which 23 and 8 resulted in mortality, respectively. There were no observed interactions between bottom trawl gear and Kemp's ridley sea turtles on Georges Bank. During this period, 20 (CV=0.72, 95% CI = 0-50) and 6 (CV=1.0, 95% CI=0-20) leatherback interactions were estimated to have occurred in the Mid-Atlantic and on Georges Bank, respectively, which resulted in 13 total mortalities. There were 0 Kemp's ridley, green, and leatherback sea turtles estimated to have been excluded by TEDs (i.e., unobservable but quantifiable interactions) (Murray 2020).

Bycatch estimates are also available for sink gillnet gear. From 2012-2016 (the most recent 5-year period that has been statistically analyzed for gillnets), fisheries observers reported a total of 27 loggerhead, 8 Kemp's ridley, 2 green, 2 leatherback, and 9 unidentified hard-shelled sea turtles incidentally caught in U.S. Mid-Atlantic and Georges Bank gillnet gear. Of these, 1 Kemp's ridley, 2 green, and 1 unidentified hard-shelled species were observed inside the sounds in North Carolina. These turtles were excluded from the bycatch rate calculations in Murray (2018) because they were outside the study region. Most (93 percent) of the loggerhead interactions occurred between 40º N and 41.5º N during June through September. In this same region, 5 Kemp's ridley interactions occurred during July through November. In addition, 3 Kemp's ridley interactions occurred around 35º N in April, June, and December. Both green sea turtle interactions occurred inside North Carolina sounds, one in March and the other in September. Both leatherbacks were observed around 40° N in November and December. Unidentified hard-shelled turtle interactions occurred between 35º N and 41.6º N from May to September (Murray 2018).

Murray (2018) estimated that from 2012-2016, sink gillnet fisheries in the mid-Atlantic and Georges Bank bycaught 705 loggerheads (CV=0.29, 95% CI over all years: 335-1116), 145 Kemp's ridleys (CV =0.43, 95% CI over all years: 44-292), 27 leatherbacks (CV =0.71, 95% CI over all years 0-68), and 112 unidentified hard-shelled turtles (CV=0.37, 95% CI over all years (64-321). Of these, mortalities were estimated at 557 loggerheads, 115 Kemp's ridley, 21 leatherbacks, and 88 unidentified hard-shelled sea turtles. Total estimated loggerhead bycatch was equivalent to 19 adults. The highest bycatch rate of loggerheads occurred in the southern Mid-Atlantic stratum in large mesh gear during November to June. Though only one sea turtle was observed in this stratum, observed effort was low, leading to a high bycatch rate. Bycatch rates of all other species were lower relative to loggerheads. Highest estimated loggerhead bycatch occurred in the northern Mid-Atlantic from July to October in large mesh gears due to the higher levels of commercial effort in the stratum. Mean loggerhead bycatch rates were ten times those of Kemp's ridley bycatch rates in large mesh gear in the northern Mid-Atlantic from July to October (Murray 2018).

Documented sea turtle interactions through 2019 with trawl and gillnet gear after the period analyzed in Murray (2018) and Murray (2020), respectively, are presented for additional reference (Table 66, Table 67); however, they are not yet included in any model-based estimates of sea turtle bycatch in the U.S. Mid-Atlantic or Georges Bank. With the exception of green sea turtles in gillnets, the model-based estimates of annual bycatch published in Murray (2018, 2020) represent the best available information for and analysis of bycatch in the bottom trawl and gillnet components of the ten fisheries assessed in this Opinion. For green sea turtle incidental bycatch in gillnet gear, raw fisheries observer data represents the best available information.

Table 66: Documented bycatch of sea turtles in trawl gear recorded by the NEFOP and ASM Program in 2019 (i.e., since the most recent bycatch estimate (Murray 2020)), along with the most landed commercial species (by hail weight) per trip[39] (NMFS NEFSC Observer Program, unpublished data).

	Horseshoe crab	**Scup**	**Sea scallop**	**Squid, Atlantic longfin**	**Summer flounder**
Loggerhead	1	1	1	2	2
Unidentified				1	

Table 67: Documented bycatch of sea turtles in gillnet gear recorded by the NEFOP and ASM Program from 2017-2019 (i.e., since the most recent bycatch estimate (Murray 2018)), along with the most landed commercial species (by hail weight) per trip[39]. Gillnet gear includes fixed or anchored sink and drift sink gear (NMFS NEFSC Observer Program, unpublished data).

	Atlantic menhaden	**Black drum**	**Bluefish**	**King mackerel**	**Monkfish**	**Skate, unknown**	**Smooth dogfish**	**Spiny dogfish**	**Spot**	**Striped bass**	**Winter skate**
Green		1								1	1
Kemp's ridley				1			3	1			
Loggerhead			1		5	1			1		
Leatherback	1										1
Unidentified					1						1

While it may be informative to assess the number of green sea turtles observed captured on gillnet trips when the majority of the landings were any of the species included in the ten FMPs, using this number as the estimated number of interactions would underestimate in two ways. First, green sea turtles could have been captured on trips where these species were part of the catch, but constituted less than the majority of the catch. Second, these captures are an underestimate given that they are only observed captures and we are not currently able to extrapolate this number to generate an estimate of total bycatch. In order to partially compensate for this underestimate and for the purposes of estimating interactions of green sea turtles with gillnet gear authorized under the ten FMPs in this Opinion, we use all observed interactions from 2010-2019, regardless of the most landed commercial species (Table 68). We are using the most

[39] For sea turtles brought on board dead, NMFS evaluates whether the mortality is attributed to the current interaction based on the postmortem condition and other evidence, such as entanglement in gear or concurrent abnormalities (NMFS 2017f). Only mortalities attributed to the current interaction are included here.

recent 10 years as we believe this is most reflective of the fisheries operations going forward. We then average the number of interactions per year as shown in Table 69.

Table 68: Documented bycatch of green and unknown sea turtles in gillnet gear recorded by the NEFOP and ASM Program from 2010-2019, along with the most landed commercial species (by hail weight) per trip. Gillnet gear includes fixed or anchored sink and drift sink gear (NMFS NEFSC Observer Program).

	Black drum	Hickory shad	Monkfish	Skate winter	Southern flounder	Striped bass	Summer flounder	Spanish mackerel
Green sea	1	1		1	1	1		
Unidentified			6	3			1	2

Table 69: Documented bycatch and annual average of green and unidentified sea turtles in gillnet gear recorded by the NEFOP from 2010-2019 (NMFS NEFSC Observer Program, unpublished data).

	Documented # of bycatch in gillnet gear	Documented # of bycatch/year in gillnet gear
Green sea turtle	5	0.5
Unidentified sea turtle	12	1.2

As described above, fisheries using trawl and gillnet gear incidentally capture sea turtles, and some of these interactions are lethal. Hard-shelled sea turtles foraging on the bottom or swimming through the water column in areas where the trawl fisheries operate are at risk. Leatherbacks may also be captured in trawl fisheries. Tagging studies have shown that leatherback sea turtles stay primarily within the water column rather than near the bottom (Dodge et al. 2014, James et al. 2005c). Given this and their largely pelagic life history (CETAP 1982, NMFS and USFWS 1992, Rebel 1974), interactions between leatherbacks and bottom trawl gear are expected to occur when the gear is traveling through the water column versus on the bottom.

Potential sea turtle interactions with sink gillnets are most likely to occur with hard-shelled sea turtles since these species are more likely to be found near the bottom where the netting of the gear is found. However, pelagic leatherbacks may also become entangled in the gear. Sea turtles are unlikely to be able to break free of entangling fishing gear and are thus vulnerable to drowning from forced submergence, although some have been recovered alive in sink gillnets.

The American lobster, red crab, Jonah crab, black sea bass, and scup fisheries use trap/pot gear. Sea turtles have been entangled in lobster, finfish, blue crab, and whelk/conch trap/pot gear (GAR STDN, unpublished data). Most of these fisheries use similar gear configurations and fishing methods. There is limited information on entanglements in these gears from the observer and VTR data. While the NEFSC Observer Program documented a leatherback entanglement in lobster gear in 2014, observer coverage in these fisheries has been low. From 2009-2020, one leatherback was also reported entangled via VTR in federal lobster gear and is included in the GAR STDN data below. VTR reported interactions do not accurately indicate the frequency of sea turtle interactions.

Most reports of entanglements in trap/pot gear are documented by the GAR STDN. The GAR STDN operates as an event response network, not as an active observer program. The reports are opportunistic (e.g., from private boaters, fishermen, USCG, state agencies (e.g., Maine Marine Patrol, Massachusetts Environmental Police) and local harbormasters). The level of reporting from the public depends on many factors, including the location and visibility of the turtle and the knowledge of the public regarding who to call when reporting an entanglement. Additionally, since the majority of entanglements are reported by recreational boaters, these data may be skewed to more coastal entanglements in waters that are easily accessible and highly utilized by boaters. Reports may also be skewed towards entanglements at the surface of buoy lines due to those entanglements being visible. Given the limitations on the GAR STDN dataset, it is difficult to correlate the number of entanglements reported to the GAR STDN to the actual number of entanglements that are occurring in coastal and offshore waters. The data presented below are a summary of the GAR STDN entanglement data. Since this dataset is the most complete and best available consolidation of sea turtle entanglement data in the action area, it will be used to estimate sea turtle interactions in the fisheries.

In terms of commercial hook and line gear, only the spiny dogfish and multispecies fisheries have a portion of landings attributed to hook and line gear (namely bottom longlines and handlines). Sea turtle bycatch has been documented in commercial hook and line fisheries, notably the pelagic longline fisheries (Swimmer et al. 2017). Loggerheads and Kemp's ridleys are known to investigate and bite baited hooks according to reports from commercial fishermen fishing for reef fish and sharks with both single rigs and bottom longlines (NMFS SEFSC 2001, TEWG 2000). However, no documented interactions of sea turtles have been recorded in the commercial Northeast bottom longline or handline fisheries from 2002-2019 (NEFSC observer program, unpublished data). Due to the lack of observed interactions in both the spiny dogfish and multispecies hook and line fisheries and because hook and line gear accounts for a small portion of the effort and landings for each fishery (less than 16 percent), we anticipate that interactions with sea turtles are extremely rare and unlikely.

Data on the capture of sea turtles in recreational hook and line fisheries is limited. Hooked sea turtles have been reported by the public fishing recreationally from boats, piers, beaches, banks, and jetties (TEWG 2000). Most sea turtle captures on rod and reel, as reported to the stranding network, have occurred during pier fishing. Fishing piers are suspected to attract sea turtles that learn to forage there for discarded bait and fish carcasses. The amount of persistent debris, including monofilament line, fishing tackle, and other manufactured items, is higher around piers, posing an additional threat to sea turtles in the area. These locations are outside the area where the federal fisheries operate.

Hard-shelled and leatherback sea turtles are known to ingest baited hooks or have their appendages snagged by hooks, both of which have been recorded in the STSSN database. Some will break free on their own and escape, possibly with embedded/ingested hooks and/or trailing line, which may cause post-release injury or death. Others may be cut free by fishermen and intentionally released. Though it is assumed that most sea turtles hooked by recreational fishermen are released alive, deceased sea turtles have been documented with hooks in their digestive tract.

In addition to interactions reported to the STSSN, two sea turtle interactions with handline gear were reported on VTRs by party/charter vessels. Of these, one interaction was reported in federal waters where the fisheries in this Opinion operate, and the species kept was weakfish. NMFS has also attempted to assess the extent of interactions between recreational anglers and sea turtles through a survey-based pilot study from 2012-2013 that included shore-based, private vessel, and charter/headboat fishing effort in waters off the southeast Atlantic. However, this study was limited to one state.

The recreational bluefish fishery accounts for approximately 70-80 percent of total bluefish landings. Rod and reel, handline, trap/pot, and spear gear are used in the recreational fishery, with rod and reel being the predominant gear type used. Since the recreational fishery receives 80 percent of the annual bluefish quota and charter/recreational boats are commonly found throughout the action area, a significant amount of hook and line fishing occurs for bluefish. However, data from the Marine Recreational Information Program (MRIP, see https://www.fisheries.noaa.gov/recreational-fishing-data/recreational-fishing-data-and-statistics-queries) from 2015-2019 indicate that only a small percentage (~2.4 percent of recreational catch and <1 percent of effort where bluefish is the primary or secondary target) of recreational fishing activity for bluefish occurs in federal waters where NMFS directly regulates the fishery. In state waters, the federal FMP sets the overall quota, but authorization and management of the recreational fishery is at the state level.

The bluefish fishery is the fishery with the largest recreational component. As described in section 4.1, bluefish are caught using a technique called jigging. Sea turtles are unlikely to be snagged by jigged gear as it is deployed near the surface and constantly reeled back to the boat. It is possible a sea turtle could become snagged if it comes into contact with the jigged hook, but the chances of that occurring are extremely low. Presently, there are no data sets available to provide estimates of incidental take from hook and line gear used in these federal fisheries when they are operating in federal waters.

The probability of hooking or entanglements in recreational hook and line gear in federal waters is difficult to ascertain and very little data are available for the U.S. Atlantic to analyze impacts from this type of interaction on individual animals. In addition, it may be difficult to tell if the entangling gear is recreational or commercial depending on the gear present. Based on the lack of documented takes by hook and line fisheries in this Opinion and the fishing techniques used by the recreational bluefish fishery, we anticipate that interactions with recreational fisheries in federal waters and in fisheries in this Opinion are extremely rare and unlikely.

7.3.1.3. Estimating Interactions with and Mortality of Sea Turtles

Estimating Interactions

As described earlier in this Opinion, Murray (2018, 2020) analyzes fisheries observer data and VTR data from fishermen to estimate the number of sea turtle interactions in bottom trawl and gillnet gear in U.S. Mid-Atlantic and Georges Bank waters that occurred over certain time periods (2014-2018 for trawls, 2012-2016 for gillnets). These reports on interactions represent the most accurate predictor of annual sea turtle interactions in U.S. bottom trawl and gillnet fisheries south of the Gulf of Maine to Cape Hatteras. For green sea turtles in gillnets, however, observer reports from the NEFSC observer/sea sampling database represent the best available information on annual bycatch in these fisheries. For trap/pot gear interactions with sea turtles, entanglement data from the GAR STDN represents the best available information on annual bycatch in the lobster, red crab, Jonah crab, black sea bass, and scup trap/pot fisheries.

Interactions with recreational and commercial hook and line gear in the fisheries in this Opinion are expected to be extremely rare and unlikely and are, thus, not addressed in this section.

The sea turtle bycatch estimate methods for trawls and gillnets (Murray 2018, 2020) estimated interaction rates for each sea turtle species with stratified ratio estimators. This method differs from previous approaches (Murray 2015b, Murray and Orphanides 2013, Warden 2011a), where rates were estimated using generalized additive models (GAMs). Ratio estimators are computationally simple with general application to many sampling designs (Cochran 1977, Murray 2020) and results may be similar to those using GAM or generalized linear models (GLM) if ratio estimators are stratified based on the same explanatory variables in a GAM or GLM model (Murray 2007, Murray and Orphanides 2013, Orphanides 2010).

Observer and commercial data were stratified by Ecological Production Unit (EPU), latitude zone, season, and depth, based on factors associated with loggerhead bycatch rates in previous trawl bycatch analyses (latitude, SST, depth) (Murray 2015b, Murray and Orphanides 2013, Warden 2011a). Within the Mid-Atlantic EPU, latitude zones included: Northern (>=37° N to the Mid-Atlantic boundary); Middle (<37º N and <39° N) and Southern (<=37° N). Season was used as a proxy for SST. Summer was defined as July – October and winter as November – June. Depth groups were defined as shallow (<= 164 ft (50 m)) or deep (> 164 ft (50 m)). Within the Georges Bank EPU, rates were stratified by season and depth groups. While only a few interactions occurred in the Georges Bank EPU, it was stratified as a separate region for a number of reasons. These include: (1) the EPUs are characterized by distinct patterns in oceanographic properties, fish distributions, and primary production (Ecosystem Assessment Program 2012); (2) previous analyses of turtle interactions delineated the "Mid-Atlantic" with the same boundaries, facilitating comparisons across time series; and (3) observer coverage is allocated separately across fleets operating in the Mid-Atlantic versus Northeast regions, of which Georges Bank is a part.

Previous NEFSC bycatch estimate reports assigned trips and associated bycatch to FMPs or individual species landed based on the distribution of landings for that trip. Trips in a certain time and area using trawls or gillnets were estimated to have a certain bycatch rate of loggerhead sea turtles (based on the observed interactions). In the estimate, the trip and its associated interactions (calculated using the bycatch rate) were assigned to multiple fisheries in a ratio that reflected the catch composition of that trip by weight (Murray 2013, 2015b, Warden 2011b). This method was meant to reflect how many of the fisheries that operate in the action area land several species on any given trip. As we have now batched all federal trawl and gillnet fisheries under GARFO jurisdiction into one Opinion, minus the scallop trawl fishery, we no longer need to partition bycatch by specific FMP except for scallops, which is a small percentage of the total trawl take estimate (Linden 2020).

Bottom Trawls

From 2014-2018, 571 loggerhead (CV=0.29, 95% CI=318-997), 46 Kemp's ridley (CV=0.45, 95% CI=10-88), 20 leatherback (CV=0.72, 95% CI = 0-50), and 16 green (CV=0.73, 95% CI=0-44) sea turtle interactions were estimated to have occurred in bottom trawl gear in the Mid-Atlantic region over the 5-year period. On Georges Bank, 12 loggerheads (CV=0.70, 95% CI=0-31) and 6 leatherback (CV=1.0, 95% CI=0-20) interactions were estimated to have occurred from 2014-2018. An estimated 272 loggerhead, 23 Kemp's ridley, 13 leatherback, and 8 green sea turtle interactions resulted in mortality over this period (Murray 2020). With the respective

95 percent CIs, it would be expected that anywhere from the low end to the high end of each sea turtle species could interact with trawl gear annually and that would be within the range of estimated interactions based on past records. For this Opinion, we are using the upper end of the 95 percent CI. In Murray (2020), this is an estimate of 1,028 loggerhead, 88 Kemp's ridley, 70 leatherback, and 44 green sea turtles over a 5-year period is the best available information on the anticipated number of interactions in the bottom trawl component of these fisheries in both state and federal waters.

To estimate the number of interactions in federal waters only, we used estimated interaction rates from Murray (2020) and VTR data from 2014-2018, the years of the Murray (2020) analysis. Using the number of trawl trips in federal waters (excluding Atlantic sea scallop trips), we estimate 954 loggerhead, 53 Kemp's ridley, 40 leatherback, and 32 green sea turtle interactions over a 5-year period is the best available information on bottom trawl interactions with sea turtles in the ten fisheries (Linden 2020). These estimates provide the best available information for determining the anticipated number of sea turtle interactions over future 5-year periods in the bottom trawl components of the ten fisheries. This represents the total number of interactions anticipated in the bottom trawl component of these fisheries and not just the number observed. We further believe that any interactions in bottom trawl gear that occur outside of the Mid-Atlantic and Georges Bank (i.e., in the Gulf of Maine) will be subsumed within this estimate as the upper ends of the 95 percent CIs (rather than the means) are being used. CIs for combined strata within the Mid-Atlantic and Georges Bank regions were obtained in through the summation of stratum-specific bycatch estimates (Murray 2020).

Gillnets
From 2012-2016, estimated bycatch of sea turtles in sink gillnet gear on Georges Bank and in the Mid-Atlantic was 705 loggerheads (of which 557 were mortalities) (CV = 0.29, 95% CI over all years: 335-1116), 145 Kemp's ridleys (115 mortalities) (CV=0.43, 95% CI over all years: 44-292), 27 leatherbacks (21 mortalities) (CV=0.71, 95% CI over all years: 0-68), and 112 unidentified hard-shelled sea turtles (88 mortalities) (CV=0.37, 95% CI over all years: 64-321) (Murray 2018). These estimates of sea turtle interactions with Mid-Atlantic and Georges Bank sink gillnet gear provide the best available information for determining the anticipated bycatch of sea turtles in that gear type in the action area. For this Opinion, we are using the upper ends of the 95 percent CIs and therefore estimate 1,116 loggerhead, 292 Kemp's ridley, 68 leatherback, and 321 unidentified hard-shelled sea turtle interactions over a 5-year period as the best available information on the anticipated number of interactions in the gillnet component of the ten fisheries. This represents the total number of interactions expected over future 5-year periods in the gillnet component of these fisheries and not just the number that may be observed. We further believe that any sea turtle interactions in gillnet gear that occur outside of the Mid-Atlantic and Georges Bank (i.e., in the Gulf of Maine) will be captured within these estimates as the upper ends of the 95 percent CIs (rather than the means) are being used. As with trawl gear, we used VTR effort in federal waters to estimate interactions with the fisheries in this Opinion.

Looking solely at fishing effort in federal waters, we estimate 853 loggerhead, 196 Kemp's ridley, 52 leatherback, and 226 unidentified hard-shelled sea turtle interactions over a 5-year period is the best available information on gillnet interactions with sea turtles in the ten fisheries (Linden 2020). Of the estimated hard-shelled sea turtle interactions identified to species, loggerheads represent 81 percent (853/1,049) of the total interactions and Kemp's ridleys represent 19 percent (196/1,049). Thus, we anticipate that of the 226 unidentified sea turtle interactions over a 5-year period, 183 will be loggerheads and 43 will be Kemp's ridleys. That brings our federal waters totals to 1,036 loggerhead, 239 Kemp's ridley, and 52 leatherback interactions over a 5-year period for gillnet gear used in the ten fisheries.

There are no total bycatch estimates for green sea turtles in U.S. Mid-Atlantic and Georges Bank gillnet gear. The very low number of observed green sea turtle interactions in gillnet gear suggests that interactions with this species within the action area in the fisheries considered in this Opinion are rare. However, given the fact that observer coverage in these fisheries is much less than 100 percent, it is likely that interactions with green sea turtles have occurred but were not observed or reported. Given effort in the gillnet fisheries as a whole and the seasonal overlap in distribution of this species with the operation of gillnet gear, green sea turtles are likely to occasionally interact with gillnet gear.

From 2010-2019, an annual average of 0.5 green sea turtles and 1.2 unidentified turtles were captured in gillnet gear (Table 69). Rounding up, we anticipate the take of 1 green sea turtle annually. While it is more likely that the unknown sea turtles are loggerheads or Kemp's ridleys, it is possible that, in any given year, a green sea turtle could be captured and not identified to species. Therefore, we are adding 1 unknown turtle to the anticipated capture number to account for this possibility. This gives a total of ten captures (average of two annually) in gillnet gear over a 5-year period.

Trap/Pot Gear

When calculating the sea turtle interaction rates for trap/pot gear used in the ten fisheries in this Opinion, we evaluated GAR STDN vertical line stranding and entanglement records documented during 2010-2019, the most recent 10-year period, in state and federal waters. We believe this approach is reasonable for a number of reasons. The species of sea turtles that occur in the action area are all highly migratory and found in both state and federal waters. Trap construction requirements are very similar in the state and federal fisheries, and effort throughout the seasons is similar. The vast majority of both state and federal trap/pot fishing effort occurs in the depth range where sea turtles are known to occur most frequently (0-120 ft (0-36 meters)); thus, none of the fisheries are known to have a disproportionate rate of sea turtle entanglement based on the distributions of sea turtles and fishing effort. Since the gear, timing, and distribution of effort with respect to sea turtle abundance are essentially the same in both state and federal waters, we believe the number of sea turtle entanglements reported to the GAR STDN is the best available data from which to estimate of sea turtle entanglements.

From 2013-2019, the NEFSC observer/sea sampling database included one reported interaction in pot gear in 2014 (a dead leatherback). Given the low level of observer coverage in these fisheries and the extremely small sample size of observer records for this gear type, these data will not be used to estimate sea turtle interactions with the fisheries.

An annual estimate of sea turtle interactions in fixed gear fisheries in federal waters was determined based on the number of confirmed and probable entanglement cases in the GAR STDN database from 2010-2019. Any of the estimates that produced fractional numbers were rounded up in the final estimates.

Between 2010 and 2019, 272 sea turtle entanglements in vertical line gear in state and federal waters of the Greater Atlantic Region (Maine through Virginia) were reported and classified with a probable or confirmed, high confidence rating. Although the action area of the fisheries considered in this Opinion extends to Key West, Florida, the trap/pot fisheries considered in this Opinion only operate in the Greater Atlantic Region. One case involved scallop aquaculture gear and another involved gear used by a vessel conducting research. These two cases will not be considered as part of the analysis. Of the 270 cases assessed, 255 involved leatherback sea turtles and 15 involved loggerhead sea turtles. For leatherbacks, there were 238 records in state waters, and 17 in federal waters. This includes four cases where the latitude and longitude were unknown, but the general location description or fishery indicated state waters. These four interactions were assigned to state fisheries based on the location and/or fishery description. For loggerheads, 14 records were in state waters and 1 in federal waters. There has only been one green and no Kemp's ridley sea turtles documented in the STDN database. The green sea turtle was entangled in conch gear in state water in 2007. Small turtles such as Kemp's ridley or smaller juvenile hard-shelled turtles likely do not have the strength to wrap fixed fishing gear lines around themselves (Sampson, pers comm January 16, 2020). Given the lack of documented interactions and the size of Kemp's ridley and green sea turtles in the areas where the trap/pot fisheries considered in this Opinion, interactions with these species in trap/pot gear are extremely rare and, therefore, not estimated below.

Leatherback Sea Turtles
Given differences in information available for the different trap/pot fisheries, we estimate leatherback sea turtle entanglements in federal lobster/Jonah crab traps/pot fisheries separately from fish trap/pot fisheries. Additionally, given the red crab fishery's offshore location and absence of GAR STDN data for sea turtle interactions with gear used in this fishery, we evaluate this fishery separately. We then combine the three estimates to produce an overall estimate of annual sea turtle entanglements, by species, expected in the fisheries in the Opinion.

Lobster and Jonah Crab*:* The Jonah crab FMP was implemented in federal waters in 2019 and only allows previously permitted lobster traps to target Jonah crab. Therefore, the entanglement estimate for the Jonah crab fishery is contained within the estimate for the lobster fishery.
The GAR STDN receives the majority of reports from private boaters and recreational fishermen who encounter entangled turtles in the water. These reports may come directly from the reporting individual or routed through the USCG, state agencies (e.g., Maine Marine Patrol, Massachusetts Environmental Police) or local harbormasters. It is likely that the opportunistic GAR STDN data is biased towards state waters given the high number of recreational vessels in state waters. For this analysis, entanglements are considered to occur at the same rate in the federal and state fisheries. Therefore, we will apply information on entanglement rates from observed state fishery entanglements to the federal portion of the fishery. While the interactions in federal waters are described above, they are not used in this analysis as they are contained within the overall estimate below.

Gear may be verified to fishery through the buoy/gear identification numbers, which can be traced in the various state agency and federal permit systems. The American lobster fishery occurs in state and federal waters by vessels with state and/or federal permits. Of the total effort in state and federal waters, approximately 20 percent of the overall lobster fishery operates their gear in federal waters (ASMFC 2009, NMFS 2014b) The GAR STDN entanglement data involved three different scenarios for gear involved in entanglements: 1) one gear set permitted to one fishery; 2) one gear set permitted in more than one fishery; and 3) two different gear sets permitted to two different fisheries. To ensure our estimate is conservative to the species, entanglements involving multiple permits or gear types were attributed to target species based on the following criteria:

1. If gear is permitted in one fishery, interactions are assigned to the species for which the permit is held.
2. If gear is permitted in more than one fishery:
    a. If one species is federally managed (e.g., lobster) and the other (e.g., conch) is not, interaction is assigned to the federally managed species.
    b. If both species are federally managed, 0.5 interactions are assigned to each species.
3. If two gears sets in two different fisheries are involved:
    a. If target species known, 0.5 interactions are assigned to each species.
    b. If one species known and the other is unknown, 0.5 interactions are assigned to the known species and 0.5 to unknown.

Using these criteria, 81 leatherback entanglements from 2010-2019 in state waters were confirmed to the lobster fishery. There were also 119.5 interactions in unknown gear. The 0.5 results from an interaction in which two gear sets were involved; one of these gear sets was unknown. The percentage (68.4 percent) of all cases with identified gear (118.5) in state waters that proved to be lobster trap/pot gear (81) was applied to the unknown gear entanglement total (119.5) to estimate the total unknown gear presumed to be lobster trap/pot gear. Therefore, 81.7 (68.4 percent of 119.5) of the interactions in unknown gear are presumed to be lobster gear. This results in 163 (=81+81.7) interactions with lobster gear over the 10 years, with an annual average of 16.3.

As described above, approximately 80 percent of the gear is in state waters. Therefore, the 163 interactions represent 80 percent of all interactions. Applying this, we anticipate the total take in state and federal waters of 203.75 (=163/0.8) leatherbacks over ten years. The annual average in state and federal waters is 20.38 takes. Given that 20 percent of the lobster fishing effort is in federal waters, we estimate that 4.08 (20 percent of 20.28) leatherback sea turtles will become entangled in federal lobster gear each year. Since part of a sea turtle cannot be taken, this estimate will be rounded up to five leatherback sea turtle entanglements in the federal lobster and Jonah crab fisheries annually.

Black Sea Bass and Scup: Using the criteria above, 18.5 leatherback entanglements in state waters from 2010-2019 (GAR STDN, unpublished data) were assigned to fish trap/pot gear. The percentage (15.6 percent) of all cases with identified gear (118.5) in state waters that proved to be fish trap/pot gear (18.5) was applied to the unknown gear entanglement total (119.5) to estimate the total unknown gear presumed to be fish trap/pot gear. Therefore, 18.6 (15 percent of 119.5) entanglement events with unknown gear in state waters are presumed to have involved fish trap/pot gear. Adding this (18.6) to the confirmed cases (18.5) results in 37.1 (annual average of 3.71) estimated leatherback entanglements in fish trap/pot gear.
Effort distribution data between state and federal waters in fish trap/pot gear used in the fisheries is not available. The fisheries and leatherbacks overlap in both state and federal waters, and we believe that interactions are equally likely in both areas. Additionally, the opportunistic GAR STDN data are considered biased towards state waters, and entanglements are considered to occur at the same rate in the federal and state fisheries. Therefore, we assume that the same number of interactions (3.71 annually) will occur in federal waters. The actual number of entangled leatherbacks per year may differ from this estimate; however, the actual number of

entanglements cannot be extrapolated from the existing data. Assuming effort in the federal fisheries is equal to effort in the state fisheries and entanglements in the federal fisheries occur at the same rate of the observed entanglements in the state fisheries, we anticipate four leatherback sea turtle interactions annually in the fish trap/pot component of the fisheries.

Red Crab: Leatherback sea turtles may be found in the area where red crab gear is set. Documentation of leatherback sea turtle interactions with red crab trap/pot gear has not occurred. Between 2016 and 2019, there were 263 red crab directed trips. Of those, 57 trips had observers onboard and did not observe any leatherback sea turtle interactions with red crab gear. Observer coverage ranged from 13 percent to 27 percent annually (NOAA Fisheries, unpublished data). Despite the absence of reported interactions of leatherbacks with red crab trap/pot gear, the possibility exists that interactions will occur. Red crab gear is configured and set in a manner comparable to lobster gear; a gear type known to be an entanglement risk to sea turtles. We realize that more leatherbacks might be entangled than are actually reported. However, there is not information available to estimate these; therefore, we anticipate one leatherback sea turtle interaction annually in trap/pot gear used in the red crab fishery.

Combined Estimate*:* Using the methods above, we estimate that there will be five leatherback interactions in federal lobster pot gear, four in fish pot gear, and one in red crab gear. Over a 5-year period, we anticipate 50 leatherback interactions in trap/pot gear used in the fisheries considered in this Opinion.

Loggerhead Sea Turtles

From 2010-2019, 15 loggerhead sea turtles were reported entangled in trap/pot gear. One of these interactions was confirmed to lobster gear in federal waters. The remainder were in state water fisheries. Six of these were in blue crab gear, 5 in conch gear, and 3 in unknown gear. The interactions with gears that may be used by fisheries in this Opinion occurred in lobster gear in 2017 (1) and in unknown gear in 2013 (1), 2014 (1), and 2019 (1).

To estimate the annual interactions, we used the highest number (1) of annual documented loggerhead entanglements per year from 2010-2019 that may have been trap/pot gear used in the fisheries (lobster and unknown gear). Although the actual number of loggerheads entangled in trap/pot gear per year may be larger, it cannot be extrapolated from the existing GAR STDN data. As a result, we have determined that the maximum number of annual interactions from 2010-2019 represents the best available scientific and commercial data on the number of loggerhead interactions anticipated in the trap/pot component of the federal fisheries annually. We believe that this approach is consistent with NMFS' directive to provide the "benefit of the doubt" to threatened and endangered species. Therefore, we anticipate one loggerhead sea turtle interactions annually in trap/pot gear used in the fisheries.

Estimating Mortalities

Sea turtle interactions with gillnet, bottom trawl, and trap/pot gear likely result in a higher level of sea turtle mortality than is evident due to mortality after a turtle is released alive (post-interaction mortalities). Injuries suffered by sea turtles interacting with these gear types fall into two main categories: (1) submergence injuries characterized by an absence or obvious reduction in breathing and consciousness with no other apparent injury, and (2) contact injuries characterized by entanglement of the head, flippers, and/or other body parts in the gear. The following information is provided as an assessment of the extent of these types of injuries likely to occur to sea turtles affected by the operation of these fisheries. It should be noted that the severity of sea turtle submergence injuries from trawl gear interactions will likely be less if the turtle is interacting with a trawl equipped with a TED rather than a trawl without one.

Sea turtles forcibly submerged in any type of restrictive gear eventually suffer fatal consequences from prolonged anoxia and/or seawater infiltration of the lung (Lutcavage and Lutz 1997). Studies examining the relationship between tow duration and sea turtle mortality in the shrimp trawl fishery show that mortality was strongly dependent on trawling duration (Epperly et al. 2002, Henwood and Stuntz 1987, NRC 1990, Sasso and Epperly 2006). The results of these studies were comparable. In general, tows of short duration have little effect on the likelihood of mortality for sea turtles caught in the trawl gear. Intermediate tow durations result in a rapid escalation to mortality, and eventually reach a plateau of high mortality, but will not equal 100 percent as a turtle caught within the last hour of a long tow will likely survive (Epperly et al. 2002, Henwood and Stuntz 1987, NRC 1990, Sasso and Epperly 2006). The stress of being captured in a trawl is greater in cold water than in warm water (Epperly et al. 2002, Sasso and Epperly 2006). Epperly et al. (2002) gave the example that a 40 minute tow in the summer time was predicted to have a 3 percent mortality rate whereas a 40 minute tow in the winter time was predicted to have a 5 percent mortality rate. To achieve a negligible mortality rate (defined by NRC as <1 percent), tow duration for both seasons would have to be less than 10 minutes (Epperly et al. 2002, Sasso and Epperly 2006). However, in both seasons, a rapid escalation in the mortality rate did not occur until after 50 minutes (Sasso and Epperly 2006) as had been found by Henwood and Stuntz (1987). Although the data used in the 2006 analysis were specific to bottom otter trawl gear in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries, the authors considered the findings to be applicable to the impacts of forced submergence in general (Sasso and Epperly 2006).

Metabolic changes that can impair a sea turtle’s ability to function can occur within minutes of a forced submergence. Most voluntary dives appear to be aerobic, showing little if any increases in blood lactate and only minor changes in acid-base status. The story is quite different, however, in forcibly submerged sea turtles, where oxygen stores are rapidly consumed, anaerobic glycolysis is activated, and acid-base balance is disturbed, sometimes to lethal levels (Lutcavage and Lutz 1997). Forced submergence of Kemp’s ridley sea turtles in shrimp trawls resulted in an acid-base imbalance after just a few minutes (times that were within the normal dive times for the species) (Stabenau et al. 1991). Conversely, recovery times for acid-base levels to return to normal may be prolonged. Henwood and Stuntz (1987) found that it took as long as 20 hours for the acid-base levels of loggerhead sea turtles to return to normal after capture in shrimp trawls for less than 30 minutes. This effect is expected to be worse for sea turtles that are recaptured before metabolic levels have returned to normal.

Tows by trawl vessels are usually around one to two hours in duration. However, Murray (2008) found that tow durations of bottom otter trawl gear that resulted in sea turtle bycatch ranged from 0.5 to over 5 hours. Shortened tow durations in some fisheries, which have been used to limit large amounts of non-target fish species bycatch, should help to reduce the risk of death from forced submergence for sea turtles caught in trawls, but they do not eliminate the risk. For trawl fisheries, assuming that the mortality rate for sea turtles from forced submergence is comparable to that measured for the shrimp fishery (Epperly et al. 2002, Sasso and Epperly 2006), sea turtles may die as a result of capture and forced submergence in trawl gear, especially if they are caught at the beginning of long tows.

There are far fewer studies on the effects of forced submergence in gillnets than there are for trawls. However, the risk of a sea turtle drowning as a result of entanglement in gillnet gear is assumed to be greater compared to trawl gear, as gillnets are often left to soak for extended periods of time (i.e., days rather than hours) and are usually anchored to the seafloor. If a sea turtle is caught in a gillnet soon after it is set and is unable to surface for air, the likelihood of mortality is high, as a fisherman may not be back to retrieve it for several days. Soak times for gillnets in which sea turtles were captured from 2012-2016 ranged between 0.2 and 264 hours (Murray 2018).

Mortality calculations
Prior to 2013, the best available information on sea turtle mortality was the number of dead sea turtles documented by the NEFOP and ASM programs and/or reported in the NEFSC bycatch estimates (Murray 2008, 2009a, Warden 2011a). Based on the descriptions provided by fisheries observers, it seemed probable that some injured sea turtles observed captured in commercial fishing gear and that were returned to the water alive would have subsequently died as a result of those injuries. We recognized the need to expand guidance originally developed for the scallop dredge fishery to attempt to encompass other Greater Atlantic Region gear types (e.g., gillnet, trawl) and a wide range of sea turtle injuries and to use a consistent approach for assessing post-release survival.

In November 2009, NMFS GARFO and the NEFSC hosted a workshop to discuss sea turtle injuries in regional fishing gear and associated post-interaction mortality. The workshop convened various experts in sea turtle veterinary medicine, health assessment, anatomy, and/or rehabilitation. The information gathered by individual participants at this workshop was then used by NMFS to develop technical guidelines for assessing sea turtle injuries in Northeast fishing gear (Upite 2011). In 2015, to promote national consistency when assessing post-interaction mortality in trawl, net, and trap/pot gear, NMFS convened an expert working group of veterinarians, sea turtle biologists, observer program experts, and resources managers to inform the development of national criteria (Stacy et al. 2015). Subsequent to the workshop, NMFS developed national guidance on assessing post-interaction mortality (NMFS 2017f). Each year, NMFS reviews records of incidental capture in trawl, net, and trap/pot to determine the post-interaction mortality in these gears (Upite et al. 2019). Based upon the best available scientific and commercial data, we believe that these guidelines are reasonable measures of what to expect for sea turtles captured by fishing gear and associated post-interaction mortality.

Workgroup members annually review each observer record and first determine if the injury was a result of the fishery interaction (haul/set/tow), interpreted as a “fresh” injury, using the guidance and expert opinion. If fresh, then the members use the national criteria to place the turtle into one of the three categories with the identified post-release mortality rates or provide justification for a 100 percent mortality determination. After the determinations are finalized, the records are separated by gear type. Based upon the percent probability of mortality and numbers of turtles in each category, turtle mortalities are calculated for each category by gear type. The number of dead turtles is then combined to obtain an overall mortality number by gear type, and the mortality percentage (number of dead turtles/number of total observations) is calculated (Upite et al. 2019). The associated mortality rates (10 percent or 20 percent (depending on depth of the fishing operation), 50 percent, 80 percent) for the three categories factor in any potential variations in species differences. As the criteria apply to all sea turtle species and life phases

(NMFS 2017f), all species are combined in the mortality rate estimates for each gear type Upite, 2019 #3097}. Therefore, the resulting mortality percentages apply to all sea turtle species.

Mortality percentages are calculated over a rolling 5-year period. The mortality percentages for the four most recent 5-year periods (Upite et al. 2019; Memorandums from Carrie Upite, Sea Turtle Recovery Coordinator, to Jennifer Anderson, ARA for Protected Resources, February 26, 2020 and April 26, 2021), which overlap with the bycatch estimates, are presented in Table 70. These post-interaction mortality rates are consistent with those calculated over the past several 5-year periods dating back to 2006-2010.

Table 70: Rolling 5-year mortality percentages by gear type

	Trawl	Gillnet	Vertical Line
2012-2016	47%	78%	53-59%
2013-2017	48%	73%	55-61%
2014-2018	46%	71%	60-64%
2015-2019	43%	64%	57-62%

The ten fisheries assessed in this Opinion primarily use sink gillnet, bottom otter trawl, and trap/pot gear. For this Opinion, we are using the highest percentage over these periods to account for variability in mortality rates and to be conservative to the species. Therefore, we assume that 78 percent of gillnet interactions and 64 percent of vertical line interactions will result in mortality. For trawl gear, Murray (2020) estimated 50 percent mortality based on the most recent time series (2013-2017) available at that time. While the methodology was the same (Upite et al. 2019), this rate is slightly higher than the mean mortality rate in trawl gear reported for 2013-2017 (Table 70) because takes in shrimp twin trawl gear were excluded from the data considered by Murray (2020). The fisheries considered in this Opinion do not use shrimp twin trawl gear; therefore, it is reasonable to use the mortality estimate from Murray (2020). In addition, using the highest percentage is the most conservative to the species. Murray (2020) also assumed that the mortality rate for unobservable yet quantifiable interactions (i.e., the turtle passed through a TED) was 0 percent. To be conservative, we are assuming that all sea turtle takes in trawls will be in non-TED equipped gear and that the mortality rate of 50 percent is the worst case scenario experienced by each species. This is a reasonable assumption given that only 5.7 percent of the estimated interactions in bottom trawls from 2014-2018 involved sea turtles escaping out of a TED opening, all of which were loggerheads (Murray 2020).

As described above, we are using the upper end of the confidence intervals in the bycatch estimates for trawl and gillnet gear to estimate interactions with sea turtles. We then apply the post-interaction mortality rates by gear type to estimate the number of interactions that are expected to result in mortalities (Table 71). Based on bottom trawl interaction rates from 2014-2018 (Murray 2020) and using the number of bottom trawl trips in federal waters (excluding Atlantic sea scallop trips) during that period, we estimate 954 loggerhead, 53 Kemp's ridley, 40 leatherback, and 32 green sea turtle interactions over a 5-year period (Linden 2020). Using a mortality rate of 50 percent and rounding up, we estimate 477 loggerhead, 27 Kemp's ridley, 20 leatherback, and 16 green sea turtle mortalities in trawl gear used in the fisheries considered in this Opinion.

Based on interaction rates from 2012-2016 (Murray 2018) and using the number of gillnet trips in federal waters during that period, we estimate 853 loggerhead, 196 Kemp's ridley, 52 leatherback, and 226 unidentified hard-shelled sea turtle interactions over a 5-year period (Linden 2020) interactions. As described above (see 7.3.1.3, gillnets), we anticipate that of the 226 unidentified sea turtle interactions, 183 will be loggerheads and 43 will be Kemp's ridleys. Therefore, 1,036 loggerhead, 239 Kemp's ridley, and 52 leatherback interactions over a 5-year period will occur in gillnet gear used in the ten fisheries. Using a mortality rate of 78 percent and rounding up, 808 loggerhead, 187 Kemp's ridley, and 41 leatherback interactions will result in mortality. For green sea turtles, we anticipate the take of two interactions with gillnet gear annually or 10 interactions over a 5-year period. Using the 78 percent mortality, we anticipate that 8 (rounded up from 7.8) interactions will be lethal over the five year period.

We anticipate 50 leatherback sea turtle interactions over a 5-year period in all trap/pot gear used in the ten fisheries. To estimate total mortality, the Northeast Sea Turtle Injury Workgroup reviewed sea turtle entanglement cases in the GAR STDN database that were identified as involving vertical fishing line (Memorandum from Carrie Upite, Sea Turtle Recovery Coordinator, to Jennifer Anderson, ARA for Protected Resources, February 26, 2020). While most vertical line entanglements involve trap/pot gear, this cannot always be conclusively determined, so mortality rates were presented for "vertical fishing line." To incorporate all of the entanglement-related mortality data, estimated mortality rates included cases that assumed the sea turtles were deceased. To be conservative towards the species, we used the highest mortality rate (64 percent) from the four most recent 5-year periods (Memorandums from Carrie Upite, Sea Turtle Recovery Coordinator, to Jennifer Anderson, ARA for Protected Resources, February 26, 2020 and April 26, 2021). Using a mortality percentage of 64 and rounding up, we anticipate that 32 leatherback interactions in trap/pot gear used in the 10 fisheries will result in mortality over a 5-year period. We also anticipate five loggerhead sea turtle interactions over a 5-year period in trap/pot gear used in the ten fisheries. As described above, the mortality percentages apply to all species; therefore, using the 64 percent mortality percentage and rounding up, we anticipate that 4 loggerhead interactions could be lethal.

Age Class of Sea Turtles Interacting with Fishing Gear

Leatherback sea turtles

The TEWG specifies that sub-adults range from 39.4-57.1 inches (100-145 cm) and adults are >57.1 inches (145 cm) CCL (TEWG 2007). Stranding, sighting, and tracking records suggest that both adult and immature leatherback sea turtles occur within the action area where the fisheries operate (James et al. 2005a, James et al. 2005c, NMFS and USFWS 1992, NMFS SEFSC 2001). Immature and sexually mature leatherback sea turtles are known to be captured in trap/pot gear. Using the formula in Avens et al. (2009) to convert SCL to CCL, leatherbacks entangled in trap/pot gear from 2010-2019 ranged from 48.6-68.7 inches (123.5-174.6 cm CCL) (GAR STDN, unpublished data). Although there were no measurements recorded for leatherbacks captured in Mid-Atlantic and Georges Bank gillnet gear from 2012-2016 (Murray 2018), there were two leatherbacks captured in trawl gear in the region from 2014-2018 that were 55.9 inches (142 cm) and 87.8 inches (223 cm) CCL, respectively (Murray 2020). Therefore, either immature or sexually mature leatherback sea turtles could interact in gillnet, trawl, or trap/pot gear since both age classes occur in areas where the ten fisheries operate.

Loggerhead sea turtles

The 2008 recovery plan identifies five life stages for loggerhead sea turtles: (1) hatchling: 4 centimeters CCL, 1-5 days; (2) post-hatchling: 1.6-2.4 inches (4-6 cm) CCL, <6 months; (3) oceanic juvenile: 3.3-25.2 inches (8.5-64 cm) CCL, 7-11.5 years; (4) neritic juvenile: 18.1-34.3 inches (46-87 cm) CCL, 13-20 years; and (5) adult male/female: 32.7 inches (>83 cm) CCL and >34.4 inches (>87 cm) CCL (respectively), >25 years for females (NMFS and USFWS 2008). From 1995-2016, loggerhead sea turtles captured in gillnets and measured (n=25) ranged from 20.5-39.8 inches (52-101 cm) CCL (Murray 2009a, 2013, 2018). From 2009-2018 in bottom trawl gears, loggerhead ranged from 19.3-46.8 inches (49-119 cm) CCL (n=126 turtles) (Murray 2015b, 2020). Measurements may not be collected when an animal was not brought on board for sampling (e.g., fell out of net), when an observer was off-watch, or when the interaction was observed by an at-sea monitor, who is not required to collect biological information from observed bycatch. Data is also available from the STDN on the size of loggerheads entangled in vertical lines. Loggerheads entangled in blue crab, conch, and unknown gear from 2010-2019 ranged from 26.7-48.0 inches (67.8 cm-121.9 cm) SCL. Converting SCL to CCL (Teas 1993), the CCLs ranged from 28.7-51.2 inches (73-130 cm). Measurements are not available for animal entangled in lobster trap/pot gear. Both neritic juveniles (sexually immature) and adults are captured in trawl, gillnet, and trap/pot gear.

Estimates of adult equivalents are also informative in assessing impacts to populations. Adult equivalence considers a turtle's reproductive value (RV), defined as the contribution of an individual in an age class to current and future reproduction. It translates the loss of individual turtles into the number of adults expected based on the likelihood the individual will survive to adulthood and reproduce. Compared to individual losses, monitoring adult-equivalent losses from fisheries interactions can be a more informative metric to assess population-level impacts (Haas 2010).

The most recent estimates of bycatch in gillnet (Murray 2018) and trawl (Murray 2020) includes turtle sizes and adult equivalents for loggerheads. From 2012-2016, observed loggerheads caught in Mid-Atlantic and Georges Bank gillnet gear, for which measurements could be taken, were neritic juveniles, and ranged between 21.3-27.2 inches (54.0-69.0 cm) CCL (n=11 turtles). Estimated interactions across Georges Bank and the Mid-Atlantic in gillnet gear were equivalent to 19 adults (Murray 2018). Due to the low sample of measured turtles in the analysis, limited information was available to calculate adult equivalent bycatch, and a larger sample of measured turtles including turtles from other strata would have helped provide a more accurate measure of adult equivalents (Murray 2018). Size classes of loggerheads observed captured in Mid-Atlantic and Georges Bank trawl gear from 2014-2018 spanned both juvenile and adult life stages, ranging from 20.1-46.8 inches (51.0-119.0 cm) CCL (n=38 turtles). From 2014-2018, the total number of loggerhead sea turtle interactions across Georges Bank and the Mid-Atlantic in bottom trawl gear was equivalent to 182 adults (Murray 2020). Based on these observer measurements and the known distribution of loggerhead sea turtles captured in other U.S. Atlantic coastal fisheries, we expect that both juvenile and adult loggerheads may be captured in gear used by these ten fisheries because both life stages are present within the action area.

Kemp's sea turtles

The post-hatchling stage for Kemp's ridley sea turtles was defined by the TEWG as Kemp's ridleys of 2-7.9 inches (5-20 cm) SCL, while turtles 7.9-23.6 inches (20-60 cm) SCL were considered to be benthic immature (TEWG 2000). Converting SCL to CCL (Teas 1993), post-hatchling Kemp's ridley sea turtles range from 2.1-8.3 inches (5.3-21.2 cm) and benthic immature turtles range from 8.3-25.0 inches (21.2-63.5 cm) CCL. Length at sexual maturity was more recently reported to vary considerably, ranging from 18.5-24.0 inches (47.0 to 61.0 cm) CCL (Bjorndal et al. 2014). Benthic immature turtles are those animals that have recruited to coastal benthic habitat. Mid-Atlantic and coastal New England waters are known to be developmental foraging habitat for immature Kemp's ridley sea turtles, while adults have been documented from waters and nesting beaches along the South Atlantic coast of the U.S. and in the Gulf of Mexico (Morreale and Standora 2005, Musick and Limpus 1997, TEWG 2000). Kemp's ridley turtles captured in Mid-Atlantic and Georges Bank gillnet gear from 2012-2016 ranged from 11.6-14.6 inches (29.5 to 37.0 cm) CCL (n=5 turtles), sizes considered to be juveniles (Bjorndal et al. 2014, TEWG 2000). Size classes of Kemp's ridleys observed captured in Mid-Atlantic and Georges Bank trawl gear from 2014-2018 ranged from 8.93-11.7 (22.7-29.7 cm) CCL (n=3 turtles), also in the size range of juvenile turtles. Given the life history of the species and the above bycatch records in recent years, we expect that only juvenile Kemp's ridley sea turtles are likely to interact with gear used in these fisheries.

Green sea turtles

Hirth (1997) defined a juvenile green sea turtle as a post-hatchling up to 15.7 inches (40 cm) SCL. A sub-adult was defined as green sea turtles from 16.1 inches (41 cm) through the onset of sexual maturity, and sexual maturity was defined as green sea turtles greater than 27.6-39.4 inches (70-100 cm) SCL (Hirth 1997). As they are for Kemp's ridleys, Mid-Atlantic waters are recognized as developmental habitat for juvenile green sea turtles after they enter the benthic environment (Morreale and Standora 2005, Musick and Limpus 1997). Green sea turtles observed captured in Mid-Atlantic and Georges Bank gillnet gear from 2012-2016 measured 10.2 and 11.8 inches (26.0 and 30.0 cm) CCL, which are considered juveniles. Two additional green sea turtles were captured in Mid-Atlantic and Georges Bank trawl gear from 2014-2018 and measured 10.1 and 12.2 inches (25.6 and 31.0 cm) CCL, respectively, also within the juvenile size range. However, nesting individuals are known to occur and feed in the Mid-Atlantic on occasion. A green sea turtle nest was documented in Delaware in 2011 and nests have also been recorded previously in North Carolina and Virginia (Hawkes et al. 2005), https://dwr.virginia.gov/blog/sea-turtles-in-virginia/, https://www.delawareonline.com/story/news/2018/10/26/rare-delaware-sea-turtle-nest-could-sign-climate-changed/1759869002/). Thus, we expect that both juvenile and adult green sea turtles are likely to interact with gear used in these fisheries.

7.3.2. *Vessel Strikes*

Vessels participating in the fisheries in the Opinion pose a potential threat to sea turtles when transiting to and from fishing areas and when moving during fishing activity. The degree of threat varies by vessel type (planing vs. displacement hull), vessel speed (Hazel et al. 2007, Work et al. 2010), sea turtle distribution and density in relation to vessel traffic (co-occurrence), sea turtle behavior, and environmental conditions (e.g., sea state, visibility). When sustaining injuries from vessels, sea turtles may be struck by the hull or by some portion of the steering or propulsion system. In fact, the most commonly recognized injuries are from propellers (Foley et al. 2019). Records from the Sea Turtle Stranding and Salvage Network (STSSN) show that both juvenile and adult sea turtles are subject to vessel strikes (NMFS STSSN database, unpublished data). Any of the sea turtle species can occur at or near the surface in open-ocean and coastal areas, whether resting, feeding or periodically surfacing to breathe. Therefore, green, Kemp's ridley, loggerhead, and leatherback sea turtles may all be struck by vessels operating in the fisheries in the Opinion, with any strike resulting in possible injury or mortality to the animal.

The proportion of vessel-struck sea turtles that survive is unknown. In some cases, it is not possible to determine whether documented injuries on stranded animals resulted in death or were post-mortem injuries. However, the available data indicate that post-mortem vessel strike injuries are uncommon in stranded sea turtles. Based on data from off the coast of Florida, there is good evidence that when vessel strike injuries are observed as the principle finding for a stranded turtle, the injuries were both ante-mortem and the cause of death (Foley et al. 2019). Foley et al. (2019) found that the cause of death was vessel strike or probable vessel strike in approximately 93 percent of stranded turtles with vessel strike injuries. Sea turtles found alive with concussive or propeller injuries are frequently brought to rehabilitation facilities; some are later released and others are deemed unfit to return to the wild and remain in captivity. Sea turtles in the wild have been documented with healed injuries so at least some sea turtles survive without human intervention.

To analyze the effects of vessels operating in the fisheries in this Opinion on sea turtles, we evaluated the best available scientific and commercial data on vessel traffic and sea turtle strandings. Here, we summarize the analysis we conducted (see Memorandum from Jennifer Anderson, ARA for Protected Resources to The File, December 23, 2020 for the detailed analysis). Vessel types that occur in the action area include fishing, recreational, and commercial (e.g., cargo, military, passenger, tankers, tug-tow) vessels. However, data is limited on the use of the area by commercial vessels other than fishing vessels and, therefore, this information was not used in the analysis. Review of the fishing footprints data[40], Northeast Ocean data portal[41], Mid-Atlantic Ocean data portal[42], and VTR data shows that the large majority of vessel traffic for the fisheries in this Opinion occurs from Virginia north. Therefore, our analysis focuses on the GAR. In addition, recreational boating surveys are available for 2012 and 2013 (Monmouth University 2016, Starbuck and Lipsky 2012). These and VTR data are the best available data on vessel use of the area. Based on the data provided in the recreational surveys (Monmouth University 2016, Starbuck and Lipsky 2012), we estimate that 13,082,108 recreational trips were taken from Maine to Virginia during May through October in each year from 2012 to 2013. To better understand the overall vessel traffic in the GAR in 2012 and 2013, VTR data for commercial fishing trips (for fisheries in the Opinion and outside it) were also queried over this time frame. This resulted in an average of 240,365 trips reported on VTRs from May through October in 2012 and 2013. Combining these estimates, a minimum of 13,322,473 trips were taken each year from 2012 to 2013. This provides us with an estimated minimum number of trips taken from May through October in these years. While turtles are generally present in the GAR from May through November, data on vessel trips in November was, with the exception of VTR data, not available.

Taking into consideration the information above, data provided by recreational boating surveys in 2012 and 2013, as well as VTR data, provide the best available scientific and commercial data to estimate the rate of sea turtles struck annually by vessels operating from Maine to Virginia.

[40] https://nefsc.noaa.gov/read/socialsci/fishing-footprints.php
[41] https://www.northeastoceandata.org/
[42] https://portal.midatlanticocean.org/

Therefore, we used STSSN stranding data from 2012 and 2013 to estimate the number of sea turtles struck annually. There were 173 stranded sea turtles with propeller marks or evidence of watercraft injury during this period.[43] This includes animals that are alive and dead. For dead turtles, injuries can occur ante- or post-mortem. As described above, Foley et al. (2019) found that in 93 percent of stranded turtles with evidence of vessel strike, the injury occurred ante-mortem and was the cause of death. Using this, we presume that 7 percent of the animals that stranded dead may have received the injuries post-mortem; therefore, the number of that stranded dead was adjusted down. This resulted in 162 strandings due to vessel strikes in 2012 and 2013. Not all sea turtles that are injured or die at sea will strand; studies estimate that up to 7-27 percent of at-sea mortalities will strand (Epperly et al. 1996, Murphy and Hopkins-Murphy 1989). To account for vessel-struck animals that do not strand, we corrected the number of reported strandings with the detection value of 17 percent (the mid-point between the estimates provided in Murphy and Hopkins-Murphy (1989) and Epperly et al. (1996)). This results in an estimate of 476 sea turtles stranding due to vessel strikes from Maine through Virginia from May through November each year (2012 and 2013).[44]

Using the minimum number of trips taken by recreational and commercial fishing vessels in 2012 and 2013 (i.e., 13,322,473 trips) and the estimated number of sea turtles stranding due to vessel strikes (i.e., 476 sea turtles), we estimate that one turtle is struck every 27,988 (= 13,322,473/476) trips[45]. Applying this rate to the trips taken by vessels in the fisheries in the Opinion, we estimated the number of sea turtles struck by these vessels. From 2015-2019, commercial fishing vessels operating in the fisheries considered in the Opinion reported trips on their VTRs. We believe that this most recent period provides the best estimate of trips that will be taken in future years over the course of the Opinion. In these years, an annual average of 65,330 trips were taken from May through November each year north of Virginia. While the rate calculated here (i.e., 1 turtle/27,988 trips) is based on vessel data from May through October, we are applying the interaction rate to fishing trip data from May through November to assess interactions during the time sea turtles are present in the GAR. This is appropriate given that we do not have information to suggest that the rates would be different in November, the data being used is intended to give a gross estimate of interactions, and the data from May through October represents the best available data since November data are not available. Applying the rate above, we estimate that 2.3 (=65,330/27,988) sea turtles of any species would be struck annually by vessels operating in the fisheries considered in the Opinion. Given that a partial turtle cannot be taken, we estimate that 3 interactions (lethal or non-lethal) may occur annually due to the operation of the fisheries or 15 interactions (lethal or non-lethal) in a 5-year period (Table 71). These vessel strikes could involve any of the four sea turtle species.

[43] Of the 173 stranded sea turtles with propeller marks, 22 were leatherbacks, 3 were green, 23 were Kemp's ridley, 124 loggerhead, and one was an unknown turtle species.

[44] The estimated number of turtles struck each year in 2012 and 2013 applies to all vessels (i.e., fishing, recreational, cargo, ferries, etc.) operating in the area.

[45] This is a conservative estimate of the number of turtles struck by fishing vessel in federal waters because (1) the total number of trips for all vessels used to estimate turtle struck per trip is an underestimate and (2) the number of trips by federally-permitted vessels includes trips those vessels took in state waters.

Table 71: Anticipated sea turtle interactions (mortalities) with gillnet, trawl, and trap/pot gear and vessels operating in the fisheries over a 5-year period

Species	Gillnet Interactions (Mortality)	Trawl Interactions (Mortality)	Pot/Trap Interactions (Mortality)	Vessel Interactions (Mortality)
Green	10 (8)	32 (16)		15 (15) any combination of species
Kemp's ridley	239 (187)	53 (27)		
Loggerhead	1,036 (808)	954 (477)	5 (4)	
Leatherback	52 (41)	40 (20)	50 (32)	

7.3.3. ***Prey***

Sea turtle prey items such as horseshoe crabs, other crabs, whelks, and fish are removed from the marine environment as fisheries bycatch in one or more of the ten fisheries under consultation. None of these are typical prey species of leatherback sea turtles or of neritic juvenile or adult green sea turtles (the age classes anticipated to occur in continental shelf waters where the fisheries operate) (Bjorndal 1985, 1997, Mortimer 1982, NMFS and USFWS 1992, Rebel 1974). Therefore, the ten fisheries will not affect the availability of prey for leatherback and green sea turtles in the action area.

Neritic juveniles and adults of both loggerhead and Kemp's ridley sea turtles are known to feed on species (Burke et al. 1994, Burke et al. 1993, Dodd 1988, Keinath et al. 1987, Lutcavage and Musick 1985, Morreale and Standora 2005, Seney and Musick 2005, Seney and Musick 2007) that are caught as bycatch in numerous fisheries. In a study of the diet of loggerhead sea turtles in Virginia waters from 1983-2002, Seney and Musick (2007) found a shift in the diet of loggerheads in the area from horseshoe and blue crabs to fish, particularly menhaden and Atlantic croaker. The authors suggested that a decline in the crab species have resulted in the shift and loggerheads are likely foraging on fish captured in fishing nets or on discarded fishery bycatch (Seney and Musick 2007). The physiological impacts of this shift are uncertain; although, it was suggested as a possible explanation for the declines in loggerhead abundance noted by Mansfield (2006). Preliminary data from stranded loggerheads in Virginia from 2008-2012 suggests a return to a more traditional diet, with large whelks, decapod crustaceans, and horseshoe crabs constituting approximately 80 percent of the prey items. While differences in turtle size and geographic distribution compared to the earlier studies appear to be a factor, the authors suggest that reductions in blue and horseshoe crab harvest limits since the early 2000s may have increased the availability of these prey species to loggerhead sea turtles (Barco et al. 2015). A preliminary analysis of the GI contents of stranded Kemp's ridley sea turtles from 2010-2013 showed their diet was similar to the 1983-2002 diet. However, insects were recorded for the first time and horseshoe crabs and mud snails were consumed more frequently compared to the earlier years. Fish, first recorded in the Kemp's ridley diet in 2000, remained an important component (Barco et al. 2015, Seney et al. 2015). In addition, while the fisheries that target crab species may be impacting loggerheads and Kemp's ridleys by reducing available prey, the crabs caught as bycatch are expected to be returned to the water alive, dead, or injured to the extent that the organisms will shortly die. Injured or deceased bycatch would still be available as prey for sea turtles, particularly loggerheads, which are known to eat a variety of live prey as well as scavenge dead organisms. Given this information, it is extremely unlikely the fisheries will have an effect on sea turtle prey and, therefore, effects are discountable.

7.3.4. ***Habitat***

As described above, geologic structures generally recover more quickly from bottom trawling on mud and sand substrates than on cobble and boulder substrates; while biological structures recovered at similar rates across substrates (see Appendix D in NEFMC 2016b, 2020b). The foraging distribution of Kemp's ridley, loggerhead, and green sea turtles in Mid-Atlantic and New England waters as far north as approximately Cape Cod and Georges Bank, do not typically occur in gravel habitats. In addition, green sea turtles forage in seagrass where the fisheries in this Opinion do not operate. Leatherback sea turtles have a broader distribution in New England waters, which may include clay outcroppings, but are pelagic feeders, and would be less impacted by alterations to benthic habitat. For these reasons, and the lack of any evidence that fishing practices affect habitats in degrees that harm or harass ESA-listed species, we find that while continued fishing efforts by the fisheries may potentially alter benthic habitats, these habitat alterations will be too small to be meaningfully measured or detected and will, therefore, have an insignificant effect on ESA-listed sea turtles.

7.4. Effects to Atlantic Sturgeon

7.4.1. ***Gear Interactions***

Gear types used in the fisheries considered in this Opinion known to interact with Atlantic sturgeon include trawl and gillnets. It is also possible that bottom longline gear could hook Atlantic sturgeon while foraging, but there have been no reported interactions. Entanglement or capture of Atlantic sturgeon in trap/pot gear is extremely unlikely. A review of all available information resulted in several reported captures of Atlantic sturgeon in trap/pot gear in Chesapeake Bay as part of a reward program for Maryland, yet all appeared to be juveniles no greater than two feet in length. In addition, there has been one observed interaction, in 2006, on a trip where the top landed species was blue crab (NEFSC observer/sea sampling database, unpublished data). No incidents of trap/pot gear captures or entanglements have been reported in any of the ten federal fisheries under consultation.

7.4.1.1. Factors Affecting Atlantic Sturgeon Interactions

Diets of sub-adult and adult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (ASSRT 2007, Bigelow and Schroeder 1953a, Collins et al. 2008, Guilbard et al. 2007, Haley 1998, Hatin et al. 2002, Johnson et al. 1997, Novak et al. 2017, Savoy 2007). Because of their size, body design, and the benthic nature of their invertebrate prey, it is likely that feeding sub-adult and adult Atlantic sturgeon could swim into or become entangled in the mesh of sink gillnet gear or be captured by bottom otter trawl gear operating in the action area.

While migrating, Atlantic sturgeon may be present throughout the water column and could interact with trawl gear while it is moving through the water column. Atlantic sturgeon interactions with gillnet and bottom trawl gear are likely at times when and in areas where their distribution overlaps with the operation of the fisheries. Atlantic sturgeon also may encounter hooks from both hook-and-line gear and longline gear while traveling through the water column.

Oceanic habitat use of sub-adult Atlantic sturgeon was examined by Dunton et al. (2010) by identifying their spatial distribution using five fishery-independent surveys. They found areas near the mouths of large bays (Chesapeake and Delaware) and estuaries (Hudson and Kennebec rivers) had higher concentrations of individuals during the spring and fall (Dunton et al. 2010).

Similarly, Breece et al. (2018) found Atlantic sturgeon occur at higher concentrations at the Delaware mouth from late spring through fall. This work also suggested that shallower waters, warmer bottom temperatures, and areas to the eastern portion of Delaware Bay were predictive of residency, while movement was predicted by increased depth, cooler bottom temperatures, and areas toward the western part of the Bay (Breece et al. 2018). In a study, matching fisheries independent biotelemetry observations of Atlantic sturgeon with daily satellite observations found that depth, day-of-year, sea surface temperature, and light absorption by seawater were the most important predictors of Atlantic sturgeon occurrence (Breece et al. 2017). A recent analysis suggests that Atlantic sturgeon may select for co-varying environmental properties (i.e., ocean color and seas surface temperature) than geographical location (Breece et al. 2016). Atlantic sturgeon may experience higher levels of harm from bycatch during seasonal aggregations (Dunton et al. 2015) or migration (Breece et al. 2017).

Factors currently thought to affect Atlantic sturgeon interactions with fishing gear and mortality due to fishing gear include: (1) gear type; (2) location and depth of gear, (3) water temperature, (4) gear characteristics (i.e., mesh size, use of tie-downs on gillnets), (5) soak/tow duration, and (6) geographic formations and environmental factors that influence placement of fishing gear and sturgeon movements. Atlantic sturgeon bycatch in ocean fisheries has been documented in all four seasons with higher numbers of interactions in November and December in addition to April and May (Miller and Shepard 2011). Mortality is also correlated to higher water temperatures, the use of tie-downs, increased soak times (>24 hours), and areas of concentrated occurrence such as overwintering areas (ASMFC 2007). Most observed sturgeon deaths occur in sink gillnet fisheries. For otter trawl fisheries, Atlantic sturgeon bycatch incidence is highest in shallower depths in June (ASMFC 2007).

Recently, a number of designated wind energy areas off the U.S. Atlantic coast have been surveyed for Atlantic sturgeon occurrence and their findings generally support the above information on sturgeon movements and seasonality. In the New York Wind Energy area, a recent study showed that acoustic detections of Atlantic sturgeon were highly seasonal and peaked from November through January. Conversely, fish were relatively uncommon or entirely absent during the summer months (July-September) (Ingram et al. 2019). In the Delaware Wind Energy Area, Atlantic sturgeon were detected during all months of the year; however, their occurrence was lowest in August, and highest in November and December (Haulsee et al. 2020). In the Maryland Wind Energy Area, Atlantic sturgeon incidence was highest in the spring and fall and tended to be biased toward shallow regions and warmer waters (Rothermel et al. 2020).

7.4.1.2. Existing Information on Interactions with Atlantic Sturgeon

Sub-adult and adult Atlantic sturgeon may be present in the action area year-round. For sink gillnets, higher levels of Atlantic sturgeon bycatch have been associated with depths of less than 131 ft (40 m), mesh sizes of greater than 10 inches (25 cm), and the months of April and May (ASMFC 2007). For otter trawl fisheries, the highest incidence of Atlantic sturgeon bycatch have been associated with depths less than 98 ft (30 m) (ASMFC 2007). More recently, over all gears and observer programs that have encountered Atlantic sturgeon, the distribution of haul depths on observed hauls that caught Atlantic sturgeon was significantly different from those that did not encounter Atlantic surgeon (KS test: $D = 0.60$, $p < 0.001$) with Atlantic sturgeon encountered primarily at depths less than 66 ft (20 m) (ASMFC 2017). Atlantic sturgeon captures in both state and federal waters are reported by observers and have been included in the NEFSC observer/sea sampling database since 1989, even though they were not listed under the ESA until 2012.

We have reviewed available bycatch information and have found that Atlantic sturgeon are frequently reported to interact with both gillnet and trawl gear throughout the action area (ASMFC 2007, 2017, Miller and Shepard 2011, Stein et al. 2004a). The above-mentioned studies have examined Atlantic sturgeon bycatch as well as mortality in commercial gillnet and trawl gear along the U.S. Atlantic coast.

Fishing records collected by onboard observers for 1989-2000 showed that, at that time, the highest levels of bycatch occurred in fisheries using sink gillnets (targeting spiny dogfish, monkfish, and Atlantic cod) and that bycatch was higher in the southern parts of the fisheries (Stein et al. 2004a). The mortality rate for Atlantic sturgeon captured in sink gillnets was 22 percent, and the peak occurred in winter and spring (Stein et al. 2004a). Inshore drift gillnets also showed high capture rates for Atlantic sturgeon, peaking in April, and mortality was calculated to be 10 percent. Otter trawls also accounted for high levels of bycatch, with bycatch peaking in winter and late spring, but there were no observed mortalities. However, the effect of fishing gear may last beyond contact and release (Boreman 1997, Clark and Hare 1998, Kynard 1997, Stein et al. 2004a). The review of bycatch data (Stein et al. 2004b) suggested that the following factors may affect bycatch rates:

1. Differences in regional temperatures that affect movements and migration patterns, thus, affecting the amount of time sturgeon spend in the marine environment where fishing is occurring, particularly for the sub-adult and non-spawning adults.
2. Geographic formations, such as the narrow continental shelf at the Mid-Atlantic Bight, that affect foraging sturgeon and fishing gear use, bringing them into closer contact.

The analysis also found that 85 percent of all recorded sturgeon bycatch involved the following targeted species: monkfish, spiny dogfish, Atlantic cod, summer flounder, American shad and scup (Stein et al. 2004b). It should be noted that gear may also have been a factor as some gear used to target other species may be easier for sturgeon to break through, resulting in encounters with the gear that the animal is able to break out of. Bycatch was at its lowest in the summer months.

The ASMFC's Technical Committee issued a 2007 report on the estimated bycatch of Atlantic sturgeon in coastal Atlantic commercial fisheries of New England and the Mid-Atlantic using different methodology and a different time frame (2001-2006) than Stein et al. (2004b) used. While not directly comparable, both studies found that deaths were infrequent in the otter trawl observer dataset. The ASMFC report found substantially lower bycatch in both gillnet and otter trawl datasets, and substantially lower mortality in sink gillnets (13.8 percent as compared to 22 percent reported for the earlier period) (ASMFC 2007).

It is important to note that observer coverage, on which this data is based, varies across fisheries. However, some patterns did emerge among the factors associated with mortality in sink gillnets: tie-downs, mesh sizes, water temperature, and soak times (ASMFC 2007). Tie-downs increase the mesh to area ratio within a given space by reducing the vertical profile of the net and create "bags" in the gear between each vertical line (ASMFC 2007).

- Larger mesh sizes, particularly the 12-inch (30.5 cm) mesh, showed high mortality rates
- Longer soak times increased bycatch and mortality

- Warmer water temperatures resulted in higher mortalities
    - In warmer waters, soak times of >24 hours resulted in 40 percent mortality and soak times of <24 hours resulted in 14 percent mortality
- Significant positive associations with higher mortalities and warmer water combined with tie-downs, as well as longer soak times combined with tie-downs.

The third study examined otter trawl and sink gillnet data from the NEFOP and ASM programs that was collected from 2006 to 2010. This study expanded the frequency of encounters by using total landings recorded on VTRs (Miller and Shepard 2011).

Miller and Shepard (2011) also characterized observed and estimated sturgeon takes by division and quarter, as well as provided annual and total predicted takes and relative influence of FMP species groups to annual take estimates. The fisheries using sink gillnet gear with the highest predicted take rates were monkfish, skate and flounder/scup/black sea bass. The fisheries with the highest predicted take rates using otter trawls were flounder/scup/black sea bass, skate, and squid/mackerel/butterfish. The NEFSC study reported a higher rate of Atlantic sturgeon mortality in otter trawls than the previous two studies.

For Atlantic sturgeon, the model-based estimates of annual bycatch in gillnet and bottom trawl gear published in ASMFC (2017) represent the best available information for and analysis of bycatch in the ten fisheries assessed in this Opinion, yet unlike the estimates for sea turtles, they cannot be apportioned into state and federal waters estimates at this time. From 2011-2015, the average annual bycatch of Atlantic sturgeon in bottom otter trawl gear was 777.4 sturgeon under the best fit model. From 2011-2015, the average annual bycatch of Atlantic sturgeon in gillnet gear was 627.6 sturgeon under best fit model (ASMFC 2017).

The best performing model for each gear type was applied to VTRs to predict Atlantic sturgeon bycatch across all trips. The total bycatch of Atlantic sturgeon from bottom otter trawls ranged between 624-1,518 fish over the 2000-2015 time series. The proportion of the encountered Atlantic sturgeon recorded as dead ranged from 0-18 percent (average 4 percent) This resulted in annual dead discards ranging from 0-209 fish. The total bycatch of Atlantic sturgeon from gillnets ranged from 253-2,715 fish. The proportion of Atlantic sturgeon recorded as dead ranged from 12-51 percent (average 30 percent), resulting in annual dead discards ranging from 110-690 fish. Otter trawls and gillnets caught similar sizes of Atlantic sturgeon, with most fish in the 3.3-6.6 ft (100-200 cm) total length range, although both larger and smaller individuals were captured.

The distribution of haul depths on observed hauls (all gears) that caught Atlantic sturgeon was significantly different from those that did not encounter Atlantic sturgeon (KS test: $D = 0.60$, $p < 0.001$). Atlantic sturgeon were encountered primarily at depths less than 66 ft (20 m). The distribution of SST on observed hauls encountering Atlantic sturgeon was also significantly different from those not encountering Atlantic sturgeon (KS test: $D = 0.14$, $p < 0.001$), with Atlantic sturgeon primarily encountered at water temperatures of approximately 7.2–15.6 ºC (ASMFC 2017).

Although they are not yet included in any model-based estimates of Atlantic sturgeon bycatch, documented Atlantic sturgeon interactions with gillnet and bottom trawl gear from 2016-2019 are included below for additional reference (Table 72). The number of observed takes is affected by the level and spatial extent of observer coverage. A review of observer coverage by SBRM year and SBRM fleet indicates that there was variation between the fleets. When averaging the coverage achieved across the gear type (excluding fleets with pilot coverage), observed sea days averaged between 10.2 percent and 13.2 percent for trawl fleets and between 9.8 percent to 12.7 percent for gillnet fleets for SBRM 2017 (July 2015 through June 2016) through SBRM 2020 (July 2018 through June 2019). Observed trips for these fleets averaged between 8.8 percent and 12.4 percent for the trawl fleets and 9.6 percent to 12.7 percent for the gillnet fleets during this period (Hogan et al. 2019, Wigley et al. 2021) The observed numbers are generally in line with the documented bycatch levels from the previous 5-year period of 2011-2015 for which bycatch estimates for gillnet and trawl gear have been calculated.

Table 72: Documented bycatch of Atlantic sturgeon in bottom otter trawl (fish) and gillnet gear recorded during the NEFOP and ASM programs from 2016 through 2019. Gillnet gear includes fixed or anchored sink, drift sink, anchored floating, and drift floating gillnets.

Year	Documented # of bycatch in bottom otter trawl gear	Documented # of bycatch in gillnet gear
2016	70	190
2017	82	122
2018	188	121
2019	58	154

7.4.1.3. Estimating Interactions with and Mortalities of Atlantic Sturgeon

Interactions are typically described in terms of impacts (e.g., exposure to increased water temperature resulting in injury, loss of access to spawning grounds, or capture) to individual fish, and the life stage is identified, when possible. Each of the five DPSs of Atlantic sturgeon are considered a separate species under the ESA; therefore, we must attribute the fish taken to the appropriate DPS. Several papers have been prepared since the ESA listing which provides the methodology and data that is used to make these assignments (Damon-Randall et al. 2013, Kazyak et al. 2020, Wirgin et al. 2018).

The primary cause of Atlantic sturgeon interactions with the fisheries in this Opinion is the deployment of particular gears in specific areas and times. While attempts to quantify interactions by FMP may, at times, be necessary for regulatory purposes, quantifying this linkage between individual FMPs and sturgeon interactions is difficult because of the nature of fishing in the Greater Atlantic Region that results in a trip landing species across multiple FMPs. The NEFSC conducted several analyses of sturgeon bycatch data in an attempt to categorize interaction rates by commercially sought species groups (i.e., FMP species groups or proxies to FMP species groups). At the conclusion of their efforts, the NEFSC found that partitioning discard encounters to FMPs is not particularly informative due to the high likelihood of inappropriately attributing interactions. Batching the ten FMPs into this single consultation allows us to identify, analyze, and address interactions of Atlantic sturgeon more holistically by gear type, area, and time.

The ASMFC (2017) Atlantic sturgeon stock assessment analyzed fishery observer and VTR data to estimate Atlantic sturgeon interactions in gillnet and otter trawl gear in the Mid-Atlantic and New England regions from 2000-2015, the timeframe which included the most recent, complete data at the time of the report. This report represents the most accurate predictor of annual Atlantic sturgeon interactions in the fisheries. We chose to use the most recent 5-year period of

2011-2015 as the best available information on Atlantic sturgeon bycatch in trawl and gillnet gear used in the fisheries, as it is the period most accurately resembles the current fisheries.

While previous estimates of sturgeon discards (ASMFC 2007, Stein et al. 2004a) used ratio estimators, later analysis indicated that these ratio estimators may not be sufficient because sturgeon encounters within defined spatial and temporal strata were more heterogeneous than desirable for a ratio estimator. In addition, an examination of observer data indicated that the species mix within a trip may be a better predictor of Atlantic sturgeon encounter rates than the traditional variables (e.g., mesh and gear) used to describe a stratum and that a model-based approach may help resolve some of the heterogeneity within a stratum. Accordingly, a GLM framework was developed to estimate Atlantic sturgeon discards in federal waters (Miller and Shepard 2011).

This GLM framework was used to estimate Atlantic sturgeon discards for the ASMFC (2017) benchmark assessment. The model estimated Atlantic sturgeon takes on each trip as a function of the trip-specific species mix, year, and quarter. In Miller and Shepard (2011), the species mix considered was comprised of those species currently managed with federal FMPs. However, in the ASMFC (2017) assessment, the species considered as covariates were those species with the highest catch on observed hauls encountering Atlantic sturgeon. More specifically, the total hail weights were estimated for all individual species on hauls that encountered Atlantic sturgeon and the species included as covariates were those whose cumulative sums represented 95 percent of the total hail weights on these hauls. Depth and mesh were examined as potential covariates; however, these variables were not included because they were often missing and can change substantially over the course of a trip. The composition of species landed on a trip was thought to be a proxy for differences in mesh size and depth (ASMFC 2017).

To predict Atlantic sturgeon take for all commercial landings, landings from each trip between 2000 and 2015 in the GARFO VTR database were determined for each species covariate. Using the estimated coefficients from the best performing model for each gear, the expected Atlantic sturgeon take was predicted for each VTR trip where information was available on whether the species was landed, and, if necessary, year and quarter. Total annual discard estimates were the sums of all predictions from the best-performing model for trips made in the relevant year (ASMFC 2017).

To estimate dead bycatch, GLMs were fit to data based only on those Atlantic sturgeon encounters where individuals were recorded as dead. These models, however, resulted in nonsensical estimates for the total expected Atlantic sturgeon take when expanded to the VTR trips, presumably due to low sample sizes (ASMFC 2017). As a result, dead discards were estimated by calculating the proportion of observed Atlantic sturgeon recorded as dead and applying this proportion to the total take estimate (ASMFC 2017).

The best performing model for each gear type was applied to VTRs to predict Atlantic sturgeon take for all trips. The total bycatch of Atlantic sturgeon from bottom otter trawls ranged from 624-1,518 fish over the time series. The proportion of the encountered Atlantic sturgeon recorded as dead ranged between 0-18 percent and averaged 4 percent. This resulted in annual dead discards ranging from 0-209 fish. The total bycatch of Atlantic sturgeon from sink and drift gillnets ranged from 253-2,715 fish. The proportion of Atlantic sturgeon recorded as dead ranged between 12-51 percent and averaged 30 percent, resulting in annual dead discards ranging from 110-690.

Table 73: Estimated interactions of Atlantic sturgeon by gear type

	Total Interactions	Otter Trawl		Gillnet	
		#	%	#	%
2011	1,306	892	68.3	414	31.7
2012	1,013	760	75.0	253	25.0
2013	2,640	894	33.9	1,746	66.1
2014	1,424	717	50.4	707	49.6
2015	1,309	624	47.7	685	52.3
Average	1,405	777.4	55.3	627.6	44.7

Otter Trawls

Based on data collected by observers for reported Atlantic sturgeon captures in bottom otter trawl gear, the ASMFC (2017) report estimated the average annual bycatch of Atlantic sturgeon in bottom otter trawl gear during 2011-2015 to be 777.4 individuals (Table 73). For the purposes of this Opinion, we are rounding the annual average of 777.4 to 778 since a partial sturgeon take is not possible. This estimate of Atlantic sturgeon bycatch in bottom otter trawl gear provides the best available information for determining the anticipated number of Atlantic sturgeon interactions per year in the bottom trawl components of the ten fisheries. Therefore, we expect that, on average over a 5-year period, 778 Atlantic sturgeon will be taken in bottom trawls each year. This represents the total number of interactions we are expecting on average annually in the bottom trawl component of these fisheries and not just the number observed. This is likely to be an overestimate for bycatch in federal waters as we cannot partition bycatch between state and federal waters at this time.

Gillnets

From 2011 to 2015, the average annual bycatch estimate of Atlantic sturgeon in Northeast and Mid-Atlantic gillnet gear was 627.6 individuals (Table 73) (ASMFC 2017). For the purposes of this Opinion, we are rounding the annual average of 627.6 to 628 since a partial sturgeon take is not possible. These estimates of Atlantic sturgeon interactions with Northeast and Mid-Atlantic gillnet gear provide the best available scientific and commercial data for determining the anticipated bycatch of Atlantic sturgeon in that gear type in the action area. Thus, an annual average of 628 Atlantic sturgeon per year is the best available information on the anticipated number of interactions in the gillnet component of these fisheries. In other words, over the course of the time period of the proposed action, we expect that, on average over a 5-year period, 628 Atlantic sturgeon will be taken in gillnets each year, but that we expect more or less in each individual year. This represents the total number of interactions we are expecting annually in the gillnet component of these fisheries and not just the number observed. This may be an overestimate of bycatch in federal waters as we cannot partition bycatch between state and federal waters at this time. One thing to note from the above table is that there was a large range in gillnet takes over the 2011-2015 time period, with a large number of interactions in 2013 (1,746) and a small number in 2012 (253). Although the observed bycatch was very similar for some years, the modeled results for the estimated take appear quite dissimilar, perhaps because of differences in the amount of unobserved fishing effort that fit the model parameters. The bycatch estimate for gillnets could also be an underestimate if any future years mimic 2013 where the distribution of fishing effort and sturgeon likely overlapped to a large degree.

Mortalities and Age Classes of Atlantic Sturgeon

NEFOP data from Miller and Shepherd (2011) indicates that mortality rates of Atlantic sturgeon caught in otter trawl gear and gillnet gear is approximately 5 percent and 20 percent, respectively. The ASMFC (2017) report did not provide updated mortality rates for trawls and gillnets, so we are using the values from Miller and Shepard (2011) as the best available scientific information. As explained in the Status of Species section, the range of all five DPSs overlaps and extends from Canada through Cape Canaveral, Florida. Atlantic sturgeon originating from all five DPSs use the action area. We have considered the best available information from a recent mixed stock analysis done by Kazyak et al. (2020) to determine from which DPSs individuals in the action area are likely to have originated. The authors used 12 microsatellite markers to characterize the stock composition of 1,704 Atlantic sturgeon encountered across the U.S. Atlantic Coast, to provide an enhanced understanding of life history for this species, its exposure to anthropogenic threats, and to support efforts to understand the relative abundance of specific stocks. We have determined that when evaluating the entire action area, which most closely aligns with their GARFO study region, Atlantic sturgeon throughout likely originate from the five DPSs at the following frequencies: New York Bight 71.4 percent; Chesapeake Bay 10.7 percent, Gulf of Maine 8.7 percent, South Atlantic 5.6 percent, and Carolina 2.6 percent, and Canada 1.0 percent (Table 74). Therefore, this represents the best available information on the likely genetic makeup of individuals occurring throughout the action area. The genetic assignments have corresponding 95 percent confidence intervals. However, for purposes of section 7 consultation, we have selected the reported values without their associated confidence intervals. The reported values, which approximate the mid-point of the range, are a reasonable indication of the likely genetic makeup of Atlantic sturgeon in the action area. These assignments and the data from which they are derived are described in detail in Kazyak et al. (2020).

Table 74: Estimated mortalities by DPS for the batched FMPs based on NEFOP data 2011-2015. DPS percentages listed are the point values representing the genetics mixed stock analysis results.

Sink Gillnets

	% Mortality	Estimated Mortalities
Annual average (628)	0.20	125.6
GOM (8.7%)		10.9
NYB (71.4%)		89.7
CB (10.7%)		13.4
Carolina (2.6%)		3.3
SA (5.6%)		7.0
Canada (1.0%)		1.3

Otter Trawls

	% Mortality	Estimated Mortalities
Annual average (778)	0.05	38.9
GOM (8.7%)		3.4
NYB (71.4%)		27.8
CB (10.7%)		4.2
Carolina (2.6%)		1.0
SA (5.6%)		2.2
Canada (1.0%)		0.4

Total

	Estimated Mortalities
Annual average (1,406)	164.5
GOM (8.7%)	14.3
NYB (71.4%)	117.5
CB (10.7%)	17.6
Carolina (2.6%)	4.3
SA (5.6%)	9.2
Canada (1.0%)	1.7

7.4.2. *Vessel Strikes*

Based on the best available information, vessel strikes are a significant threat to Atlantic sturgeon (77 FR 5880 and 77 FR 5914; February 6, 2012). Given that Atlantic sturgeon sub-adults and adults from all DPSs use ocean waters from Labrador, Canada to Cape Canaveral, Florida, as well as estuaries of large rivers along the U.S. East Coast, activities affecting these water bodies are likely to impact more than one Atlantic sturgeon DPS.

The exact number of Atlantic sturgeon that die due to vessel strikes is unknown. The factors relevant to determining the risk to Atlantic sturgeon from vessel strikes are currently unknown, but may be related to size and speed of the vessels, navigational clearance (i.e., depth of water and draft of the vessel) in the area where the vessel is operating, and the behavior of Atlantic sturgeon in the area (e.g., foraging, migrating, etc.). While we have some information on the number of mortalities in the Delaware and James rivers that are thought to be due to vessel strikes (Balazik et al. 2012c, Brown and Murphy 2010) we are not able to use those numbers to extrapolate effects throughout one or more DPS. This is because of (1) the small number of data points and (2) lack of information on the percent of incidences that the observed mortalities represent. While vessel strikes are believed to be a threat in several rivers as noted in the *Status of the Species* and *Environmental Baseline* sections above, we do not have information that suggests that Atlantic sturgeon are struck by vessels in the open marine environment of the action area where the vessels participating in these fisheries are operating. The risk of strike is expected to be considerably less in the Atlantic Ocean than in rivers. This is because of: (1) the greater water depths in ocean areas, which increases the space between bottom oriented sturgeon and vessel propellers and hulls, (2) a lack of obstructions or constrictions that would otherwise restrict the movement of sturgeon, and (3) the more dispersed nature of vessel traffic and more dispersed distribution of individual sturgeon which reduces the potential for co-occurrence of individual sturgeon with individual vessels. Given the greater depths in the vast majority of the action area (with the exception of nearshore areas where vessels will dock) and that sturgeon most often occur at or near the bottom while in the action area, the potential for co-occurrence of a vessel and a sturgeon in the water column is extremely low even if a sturgeon and vessel co-occurred generally. All of these factors are expected to decrease the likelihood of an encounter between an individual sturgeon and a vessel and also increase the likelihood that a sturgeon would be able to avoid any vessel. Based on these factors and the lack of any information to suggest that Atlantic sturgeon are struck and killed by vessels in the marine environment, vessel strikes in the action area are extremely unlikely to occur during vessel transits or fishing operations and, therefore, the effects are discountable.

7.4.3. ***Prey***

Diets of adult and migrant sub-adult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (ASSRT 2007, Bigelow and Schroeder 1953a, Guilbard et al. 2007, Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (ASSRT 2007, Bigelow and Schroeder 1953a, Guilbard et al. 2007).

Sink gillnets are anchored to the bottom and fish in the lower one-third of the water column. Although sink gillnets are anchored to the seafloor, several studies have found that gillnet gear has little or low impact on bottom habitat (GBCHS 2008, Morgan and Chuenpagdee 2003, Northeast Region Essential Fish Habitat Steering Committee 2002). In an analysis of effects to habitat from fishing gears, mud and sand habitats were found to recover more quickly than courser substrates (see Appendix D in NEFMC 2016b, NEFMC 2020b). Any negative effect from gillnets would vary between fishing habitats, with very low levels of damage on sand, some damage lasting a few days on mud, and more lasting damage on hard bottom clay habitats (Northeast Region Essential Fish Habitat Steering Committee 2002). Sink gillnets are therefore expected to have discountable effects on Atlantic sturgeon prey.

The effects of bottom trawls on benthic community structure have been the subject of a number of studies. In general, the severity of the impacts to bottom communities is a function of three variables: (1) energy of the environment, (2) type of gear used, and (3) intensity of trawling. High-energy and frequently disturbed environments are inhabited by organisms that are adapted to this stress and/or are short-lived and are unlikely to be severely affected, while stable environments with long-lived species are more likely to experience long-term and significant changes to the benthic community (Johnson 2002, Kathleen A. Mirarchi Inc. and CR Environmental Inc. 2005, Stevenson et al. 2004). Modern otter trawls are lighter than older trawls and scallop dredges, and cause less disturbance to benthic communities, but many older-style beam trawls are still in use (Kathleen A. Mirarchi Inc. and CR Environmental Inc. 2005).The intensity of trawling also affects benthic communities, and significant loss of large sessile epifauna from hard substrates has been demonstrated (Kathleen A. Mirarchi Inc. and CR Environmental Inc. 2005, Stevenson et al. 2004). A majority of studies has found that trawling on mud bottoms decreases the species richness, diversity, abundance, and biomass (Johnson 2002, Stevenson et al. 2004). However, a Massachusetts Bay trawling study found no difference between the species composition in trawled and control lanes, but found that faunal density was slightly higher in the trawled lanes (Kathleen A. Mirarchi Inc. and CR Environmental Inc. 2005). While there may be some changes to the benthic communities on which Atlantic sturgeon feed as a result of bottom trawling, there is no evidence the bottom trawl activities of the ten fisheries have a negative impact on availability of Atlantic sturgeon prey.

The trap/pot gear used in the lobster, red crab, Jonah crab, black sea bass, and scup fisheries is considered to have low impact to bottom habitat, and is unlikely to incidentally capture Atlantic sturgeon prey. Hook-and-line gear is also unlikely to affect prey, as it has little effect on bottom habitat and is unlikely to incidentally capture Atlantic sturgeon prey. Currently, there is no indication that Atlantic sturgeon are food-limited or that commercial fisheries might negatively impact their food availability, given the diversity of their diets. Given this information, it is extremely unlikely the fisheries will have an effect on Atlantic sturgeon prey and, therefore, effects are discountable.

7.4.4. *Habitat*

Atlantic sturgeon use the action area as a migratory route and for overwintering and likely foraging. Within the marine range of Atlantic sturgeon, several marine aggregation areas (see papers listed below for definitions of aggregation areas) have been identified adjacent to estuaries and/or coastal features formed by bay mouths and inlets along the U.S. eastern seaboard. Depths in these areas are generally no greater than 82 ft (25 m) (Dunton et al. 2010, Erickson et al. 2011, Laney et al. 2007, Stein et al. 2004b). Additional studies are still needed to understand why aggregations are found at these particular sites. The sites likely serve different purposes; there is some indication that they may serve as thermal refuges, wintering sites, or marine foraging areas (Dunton et al. 2010, Erickson et al. 2011, Stein et al. 2004b). The following known marine aggregation sites are generally inshore of federal waters where the fisheries operate:

- Waters off North Carolina, including Virginia/North Carolina border (Laney et al. 2007);
- Waters off the Chesapeake and Delaware Bays (Dunton et al. 2010, Erickson et al. 2011, Oliver et al. 2013, Stein et al. 2004b)
- New York Bight (e.g., waters off Sandy Hook, New Jersey, and Rockaway Peninsula, New York) (Dunton et al. 2015, Erickson et al. 2011, O'Leary et al. 2014, Stein et al. 2004b)
- Massachusetts Bay (Stein et al. 2004b)
- Long Island Sound (Bain et al. 2000, Savoy and Pacileo 2003, Waldman et al. 2013);
- Connecticut River Estuary (Waldman et al. 2013);
- Kennebec River Estuary (Wippelhauser 2012, Wippelhauser and Squiers 2015).
- Mouth of the Saco River (Novak et al. 2017)

While there may be some overlap in aggregations and fishing effort, we have no information that indicates negative effects on Atlantic sturgeon prey items, although foraging, overwintering, and migrations may be temporarily disturbed by the use of bottom fishing gear. However, any disturbance will not rise to the level of harm or harassment as there is plenty of room in the open ocean environment for Atlantic sturgeon to move around obstacles such as nets with little expenditure of energy. Gillnet gear may also impede Atlantic sturgeon migrations, but the effects are also expected to be insignificant for the same reasons above. Compared to the overall size of the action area, the amount of area occupied by commercial and recreational fishing gear at any point in time is extremely small. Given this information, any habitat alterations will be too small to be meaningfully measured or detected and will, therefore, have insignificant effects on Atlantic sturgeon.

7.5. Effects to Atlantic Salmon

7.5.1. *Gear Interactions*

7.5.1.1. Factors Affecting Atlantic Salmon Interactions

Atlantic salmon in the ocean are pelagic and highly surface oriented (Kocik and Sheehan 2006, Renkawitz and Sheehan 2012). The preferred habitat of post-smolt salmon in the open ocean is principally the upper ten meters of the water column (Baum 1997, ICES 2005), although there is evidence of forays into deeper water for shorter periods. Adult Atlantic salmon demonstrate a wider depth profile (ICES 2005), but overall salmon tend to be distributed in the surface layer, and all fisheries covering this part of the water column are considered to have a potential to intercept salmon. Due to these factors and the limited abundance of Atlantic salmon in the action area, they are not typically caught in the ten fisheries under discussion as these fisheries primarily focus on bottom-dwelling species in areas such as the Gulf of Maine where Atlantic salmon are most prevalent.

While migrating, Atlantic salmon may be present throughout the water column and could interact with bottom trawl and gillnet gear. All observed takes of Atlantic salmon in the ten fisheries that have been recorded by the NEFOP and ASM programs since 1989 have occurred in bottom trawls or gillnets. Atlantic salmon interactions with bottom trawl and gillnet gear are likely to occur at times when and in areas where their distribution overlaps with the operation of the fisheries. Atlantic salmon also may encounter hooks from both hook-and-line and longline gear while traveling through the water column, although commercial interactions in federal waters have not been documented and, hence, are extremely unlikely to occur and therefore discountable.

7.5.1.2. Description of Existing Information on Interactions with Atlantic Salmon

Adult Atlantic salmon may be present in the action area year round. However, fishermen permitted under the ten fisheries in this Opinion rarely capture them in the marine environment. NEFOP data from 1989-2019 show records of incidental bycatch of Atlantic salmon in seven of the 31 years, with a total of 15 individuals caught, nearly half of which (seven) occurred in 1992. There is no information available on the genetics of these bycaught Atlantic salmon, so we do not know how many of them were part of the GOM DPS. It is likely that some of these salmon, particularly those caught south of Cape Cod, may have originated from the stocking program in the Connecticut River. Those Atlantic salmon caught north of Cape Cod and/or in the Gulf of Maine are more likely to be from the GOM DPS.

Of the observed incidentally caught Atlantic salmon, ten were listed as “discarded,” which is assumed to be a live discard (Kocik, pers comm.; February 11, 2013). Five of the 15 were documented as lethal interactions. The incidental takes of Atlantic salmon occurred in bottom otter trawls (4) and gillnets (11). Observed captures occurred in March (2), April (2), May (1), June (3), August (1), and November (6).

7.5.1.3. Estimating Interactions with and Mortality of Atlantic Salmon

Due to the low number of observed interactions and the low number of Atlantic salmon in the action area, it is expected that interactions between the ten fisheries and Atlantic salmon will be low in any given year.

There are no model-based bycatch estimates for Atlantic salmon in bottom trawl or gillnet gear. The very low number of observed Atlantic salmon interactions in bottom trawl and gillnet gear as reported in the NEFSC observer/sea sampling database suggests that interactions within the action area are rare events. However, given the fact that observer coverage in these fisheries is much less than 100 percent, additional interactions with Atlantic salmon may have occurred, but were not observed or reported. In the most recent 3-year report on the Standardized Bycatch Reporting Methodology (SBRM years 2018-2020), the percentages of observed trips, in terms of number of trips, varied by fleet and ranged from approximately 1.5 percent to 40 percent for the different trawl and gillnet fleets in New England and mid-Atlantic that use gears considered in this Opinion (Wigley et al. 2021). In the prior report (SBRM years 2015-2017), the percentages of observed trips also varied by fleet and ranged from approximately 2 percent to 23 percent for the different trawl and gillnet fleets, excluding 1 fleets for which 100 percent observer coverage was a requirement (New England small mesh haddock separate trawl) (Hogan et al. 2019). It should be noted that fleets are defined as a region, gear type, mesh group, access area, and trips category combination and not by FMP. Due to the effort in the fisheries as a whole, and the seasonal overlap in distribution of these species with operation of bottom trawl and gillnet gear, a small number of Atlantic salmon may interact with both gear types.

In coming up with an estimate for Atlantic salmon interactions in the ten fisheries, we have chosen to look primarily at incidental takes recorded over the most recent ten year timeframe of 2010-2019, as those years most accurately reflect current effort trends and gear use in the fisheries, the current biological environment in the action area, and encompass years following the "regime shift" of low marine survival for Atlantic salmon that began in the early 1990s and has persisted to date.

A review of the NEFOP and ASM observer records from 2010-2019 reveals that there were no reported takes in bottom trawl gear. However, prior to 2010 there were four incidental takes that occurred in bottom trawl gear (one in 1992, one in 2004, and two in 2005). Thus, we anticipate that incidental takes of Atlantic salmon in bottom trawl gear could occur in future years.

A review of the NEFOP and ASM observer records from 2010-2019 reveals that there were three reported takes Table 75 in gillnet gear (one in 2011 and two in 2013). The average annual number of Atlantic salmon captures in gillnet gear in the action area documented through the NEFOP and ASM data is 0.30. The three documented incidental captures of Atlantic salmon from 2010-2019 occurred in the multispecies and spiny dogfish gillnet fisheries during the spring and summer months (April, June, and August) in the Gulf of Maine (NMFS statistical areas 513 and 515). These interactions occurred in federal waters.

Table 75: Observed salmon takes in gillnet gear from 2010-2019

Year	Month	Fishery	Gear	Stat Area
2011	June	Spiny Dogfish	Sink Gillnet	513
2013	April	Multispecies	Sink Gillnet	515
2013	August	Multispecies	Drift Gillnet	513

Considering the most recent ten years of data on Atlantic salmon interactions in the ten fisheries, in addition to the historic records dating back to 1989, we believe that there will be two Atlantic salmon interactions every five years in either trawl or gillnet gear. This is because there were three reported interactions in these gear types combined from 2010-2019, which averages to 1.5 every five years. Since the capture of a partial Atlantic salmon is not possible, since trawl interactions have been known to occur in previous years, and to give the benefit of the doubt to the species, we are rounding 1.5 up to 2. To give further benefit of the doubt to the species, we expect that all captures of Atlantic salmon in the ten fisheries will be GOM DPS fish, as all recent interactions in the past ten years have occurred in the Gulf of Maine. Although it is possible for some fish to be from non-listed Canadian or Saco, Merrimack, or Connecticut River stocks, we do not have any genetic information at present to determine the likely percentage breakdown of takes amongst those populations. Therefore, we are assuming that any future captures will be GOM DPS fish unless otherwise identified by genetic sampling.

In regards to the life stages of Atlantic salmon that may be captured, we anticipate that all captured salmon will either be post-smolts or adults. This is based on the size ranges (length and weight) of the individuals that have been captured over the past ten years (31-34 inches (79-87 cm), 7-11 lbs (3-4 kg). Smaller individuals (2-4 lbs (0.9-1.8 kg) in weight) have been captured in prior years dating back to 1989, but none in that size range since 2005.

Estimated Mortality

Of the three reported interactions in the fisheries from 2010-2019, two were recorded as dead and one was released alive. For the fish that was released alive, its post-capture and release condition is not known, and there is the potential that that fish could have been injured or ultimately died as a result of the capture, handling, and release. The primary contributing factors to stress and death from handling are differences in water temperatures (between the ocean and wherever the fish are held), dissolved oxygen conditions, the amount of time that fish are held out of the water, and physical trauma. Stress on Atlantic salmon increases rapidly from handling if the water temperature is too warm or dissolved oxygen is below saturation.

Of the 15 total reported interactions with Atlantic salmon in the ten fisheries dating back to 1989, at least five, and possibly more, resulted in mortalities. Ten are listed as "discarded" in the NEFSC observer/sea sampling database and are assumed to have been discarded alive, however, the ultimate condition of the fish is not known. Of the 11 documented takes in gillnet gear, three were dead (27 percent), while eight were discarded presumed alive (73 percent). Of the four documented takes in bottom otter trawl gear, two were dead (50 percent) and two were discarded presumed alive (50 percent).

For the purposes of this Opinion, to give benefit of the doubt the species, and due to the potential for stress, injury and death post-capture, we will assume that both of the interactions in bottom trawl or gillnet gear in the ten fisheries over a 5-year period will result in mortality.

7.5.2. *Vessel Strikes*

Vessel strikes are not known to be a major threat to Atlantic salmon in the Gulf of Maine. The threats assessment done for Atlantic salmon as part of the 2009 endangered listing of the expanded GOM DPS did not list vessel strikes as a high priority threat (74 FR 29344; June 19, 2009). There are no known reports of vessel strike injuries. In addition, the co-occurrence of Atlantic salmon and vessels in the action area is very low. As described in the Status of the Species (section 4), Atlantic salmon occur in a portion of the action area, further limiting the co-occurrence. In a review of the literature on avoidance of vessels by fish, De Robertis and Handegard (2013) summarized how fish react to approaching vessels and considered the mechanisms that might influence whether fish that have detected the presence of a vessel will react. They found that when fish are observed to react to moving research vessels, the reaction is generally consistent with an avoidance response. Potential stimuli to which the fish react to include low frequency sound, sound pressure levels, visual cues, ship bow wave, particle motion or acceleration, and stimulated bioluminescence. While none of the studies reviewed by De Robertis and Handegard (2013) were specific to salmon, a study completed by Knudsen et al. (1992), which assessed juvenile Atlantic salmon awareness and avoidance responses to sound, did support many of findings in De Robertis and Handegard (2013). The auditory system of fish are sensitive to particle motion and low frequency sounds (De Robertis and Handegard 2013) and in the study completed by Knudsen et al. (1992), salmonids showed a strong awareness/avoidance response to 5-10 Hz sounds, and particle velocities of 10^{-2} m s^{-2}. Low frequency sounds are introduced into the marine environment by vessels transiting to and from shore. Specifically, low frequency particle motion is produced by the displacement of water as vessels move through the water column, combined with the low frequency excitation of the hull caused by the vessel machinery (De Robertis and Handegard 2013). Given the infrequent presence of Atlantic salmon in the action area, their low density when they do occur, the relatively small numbers of fishing vessels, and the likelihood of an avoidance response if encountered/approached by a vessel, the risk of a vessel interaction with a salmon is extremely unlikely, and therefore, discountable.

7.5.3. ***Prey***

Upon completion of the physiological transition to salt water, post-smolt Atlantic salmon grow rapidly and have been documented to move in small schools loosely aggregated close to the surface (Dutil and Coutui 1988). After entering into the nearshore waters of Canada and the U.S., post-smolts become part of a mixture of stocks of Atlantic salmon from various North American streams. Their diet includes invertebrates, amphipods, euphausiids, and fish (Fraser 1987, Hislop and Shelton 1993, Hislop and Youngson 1984, Jutila and Toivonen 1985). Results from a 2001-2005 post-smolt trawl survey in Penobscot Bay and the nearshore waters of the Gulf of Maine indicate that Atlantic salmon post-smolts are prevalent in the upper water column (Sheehan et al. 2005).

Most of the GOM DPS-origin salmon spend two winters in the ocean before returning to streams to spawn. Aggregations of Atlantic salmon may still occur after the first winter at sea, but most evidence indicates that they travel individually (Reddin 1985). At this stage, Atlantic salmon primarily eat fish, feeding upon capelin, herring, and sand lance (Hansen and Pethon 1985, Hislop and Shelton 1993, Reddin 1985).

The majority of the fishing gears utilized by the ten fisheries operate on or very near the bottom. Fish species caught in these gears are species that live in benthic habitat (on or very near the bottom) such as flounders. Schooling fish, such as herring, capelin, and sand lance, occur within the water column, and therefore, with the exception of the mackerel/squid/butterfish fishery, the operation of the fisheries will not affect the availability of prey for foraging post-smolt and adult Atlantic salmon. Although small schooling fish species (including mackerel) may be caught in net gear targeting mackerel/squid/ butterfish, we have found no information that indicates this causes significant impacts to the GOM DPS of Atlantic salmon. Given this information, it is extremely unlikely the fisheries will have an effect on Atlantic salmon prey and, therefore, effects are discountable.

7.5.4. ***Habitat***

Atlantic salmon also use the action area as a migratory route and for foraging. Aggregations of Atlantic salmon may occur both at the post-smolt stage and after their first winter at sea, but most evidence indicates that they travel individually as adults (Reddin 1985). Foraging and travel activity may be temporarily disturbed by the use of bottom fishing gear, but the effects are expected to be extremely unlikely, and therefore discountable, as Atlantic salmon that occur in ocean waters are primarily pelagic rather than benthic. As a result, any potential overlap between salmon and bottom fishing gear will be minimal and occur on only brief occasions when the gear is being lowered to the ocean floor or pulled back up to the surface. Furthermore, in the open ocean there is ample room for highly mobile salmon to move around slow moving obstacles such as bottom trawling gear and, thus, any effects are likely to be too small to be meaningfully measured or detected (i.e., are insignificant). Immobile gillnet and trap/pot gear may also impede Atlantic salmon travel, but the effects are also expected to be insignificant and discountable for similar reasons related to salmon mobility and ample room in the open ocean to maneuver. There is designated critical habitat for Atlantic salmon in proximity to the action area and physical and biological features have been identified in the freshwater and estuarine environment (74 FR 29300; June 19, 2009). However, although successful marine migration is essential to the survival and recovery of GOM DPS Atlantic salmon, we were not able to identify the essential features of marine migration and feeding habitat or their specific locations at the time critical habitat was designated.

7.6. Effects to Giant Manta Rays

7.6.1. *Gear Interactions*

As described in section 4.2.3.3, giant manta rays occur in coastal, nearshore, and pelagic waters off the U.S. east coast. Although sightings north of Cape Hatteras are rare, giant manta rays have been observed as far north as New Jersey, usually found in water temperatures between 19 and 22 °C (Miller and Klimovich 2017). Giant manta rays are potentially susceptible to capture by trawl, longline, vertical longline, and gillnets based on records of their capture in fisheries using these gear types. Given the occasional occurrence and water temperature preferences of giant manta rays off the U.S. Atlantic coast from Florida to New Jersey, the distribution of giant manta rays is likely to overlap with most of the fisheries under the proposed actions. This is confirmed by the past captures of manta rays in commercial fisheries using similar gear types as evidenced by NEFOP incidental take data.

We have reviewed collections from the different gear types and noted giant manta rays have been collected during the fisheries activities in the past 18 years by two gear types – trawls and gillnet. Based on this information, we believe that trawl and gillnet are the only gears used in the fisheries that may adversely affect giant manta rays. We believe the potential risk from the other gear types is discountable.

7.6.1.1. Factors Affecting Giant Manta Ray Interactions by Gear Type

Spatial Overlap of Fishing Effort and Giant Manta Ray Abundance

The spatial and temporal overlap of giant manta rays with fishing effort is a factor that affects the likelihood of these species becoming entangled in gillnet gear or captured in trawl gear. The more abundant the animals are in a given area where fishing occurs, the greater the probability that one of them will interact with gear. The temporal distribution of fishing effort and giant manta ray abundance may also be a factor.

Species Morphology

The conditions faced by manta rays during the different phases of capture in fishing operations include traumatic handling practices (lifting up by the gills or dragging on the deck and/or towing). Giant manta rays may also be exposed to physical contact with hard objects, the harsh harvesting process of removing it from the fishing gear and removal from the water (lack of oxygen, exposure to the sun and organs crushed because of the weight of gravity). Manta rays are large; thus, it can be extremely difficult to lift them back into the water.

Environmental Conditions
Water temperature may play a role in the timing of giant manta ray migrations and presence at aggregation sites. More research is needed to understand the movements of giant manta rays and potential interactions with gillnets during various times of the year. Initial studies seem to show a seasonal component to their movements.

7.6.1.2. Description of Existing Information on Interactions with Giant Manta Rays

NMFS' observers document each interaction with a Mobulid ray by species when possible. Observations historically included giant manta ray, *Mobula birostris*; Atlantic devil ray, *Mobula hypostoma*; and manta, unknown Mobulidae (any manta and devil ray species that could not be confirmed to species). Because of the unique form and cephalic lobes adjacent to the mouth of manta and devil rays, it is unlikely but possible that these records would have been listed more generally as a stingray, unknown; or a ray, unknown. Historically, many Mobulidae species may have been identified as giant manta rays because observers were provided with the Peterson Field Guide of Atlantic Coast Fishes (1986) as a primary resource for species identification, and the giant manta ray was the only large Mobulidae species shown (L. Kellogg, pers. comm. March 25, 2019). In 2015, NMFS NEFSC re-evaluated photo records of Mobulidae species and found that numerous historic records that were originally identified as giant manta rays were actually other *Mobula* species. Thus, historic records that did not include photos, or where photos were not detailed enough to determine a species, were then classified as ray, manta, unknown (Mobulidae), an unresolved Mobulidae species.

An annual estimate of giant manta ray interactions was determined based on the number of observed cases in the NEFSC observer/sea sampling database from 2010-2019. Observed interactions from 2010-2019 between giant manta rays and unknown ray species in gear types used in the fisheries are listed in Table 76 (NEFSC observer/sea sampling database, unpublished data). *Mobula* records confirmed as species other than giant manta ray species are not considered in this analysis. From 2010–2019, two records in federal waters were confirmed by photo to be giant manta rays. Two Mobulidae records in state waters were not able to be identified to species. Given that these unknown ray species may have been giant manta rays, for the purpose of this analysis, we assume that both unidentified rays were giant manta rays. All four interactions occurred off North Carolina. These captures are an underestimate given that they are only observed captures, and we are not currently able to extrapolate this number to generate an estimate of total bycatch. In order to partially compensate for this underestimate and for the purposes of estimating interactions under the ten FMPs in this Opinion, we used all observed interactions from 2010-2019.

Table 76: Observed interactions with *Mobula birostris* and unknown Mobulida

Year	Species	Gear
2014	Giant Manta Ray	Bottom Otter Trawl
2014	Giant Manta Ray	Bottom Otter Trawl
2015	Unknown Mobulida	Gillnet
2015	Unknown Mobulida	Gillnet

7.6.1.3. Estimating Interactions with and Mortality of Giant Manta Rays

From 2010 through 2019, there were two (unidentified) giant manta rays observed in gillnet gear and two giant manta rays observed in trawl gear (Table 76). The number of interactions occurring annually is variable and influenced by sea temperatures, species abundances, fishing effort, and other factors that are difficult to predict. Because of this variability, it is unlikely that giant manta rays will be consistently impacted year after year. For example, there were two observed giant manta ray captures in gillnet gear in one year and other years with zero. As a result, estimating interactions using 1-year estimates is largely impractical. For these reasons, we believe that four interactions may occur over a 5-year time period. Therefore, we have determined that an annual average of 0.8 giant manta rays may be captured in gillnet gear or trawl gear used in the fisheries.

In the four cases observed between 2010-2019, the records indicate all animals were encountered alive and released alive. Additionally, all of the giant manta ray interactions in gillnet or trawl gear recorded in the NEFSC observer/sea sampling database (13 between 2001 and 2019) indicate the animals were encountered alive and released alive. Furthermore, during 2005-2012, ten giant manta rays caught in gillnet gear used in the Gulf of Mexico and South Atlantic Coastal Migratory Pelagic Fishery were observed to be released alive. Therefore, we expect that nearly all giant manta ray interactions with gear used in the fisheries will be released alive within the same general area and survive the interaction. However, details about specific conditions such as injuries, damage, time out of water, how the animal was moved or released, or behavior on release is not always recorded. While there is currently no information on post-release survival, NMFS Southeast Gillnet Observer Program observed a range of 0 to 16 giant manta rays captured per year between 1998 and 2015 and estimated that approximately 89 percent survived the interaction and release (see NMFS reports available at http://www.sefsc.noaa.gov/labs/panama/ob/gillnet.htm) . If 89 percent of the giant manta rays anticipated to interact with gears used in the fisheries survive, it is reasonable to expect one giant manta ray mortality in gillnet or trawl gear over a 10-year period. Therefore, we expect no more than one mortality as a result of interactions with gear used in the fisheries every 10 years.

7.6.2. *Vessel Strikes*

As giant manta ray aggregation sites are sometimes in areas of high maritime traffic, giant manta rays are potentially at risk of being struck by vessels (Miller and Klimovich 2017). Along Florida's southeast coast, five giant manta rays have been struck by vessels from 2016-2019; these individuals had injuries (i.e., fresh or healed dorsal surface propeller scars) consistent with a vessel strike. Available information indicates the threat of vessel strike on giant manta ray is predominantly an issue in shallow, coastal waters and in proximity to inlets where giant manta ray frequent, likely to facilitate feeding (NMFS 2020d). Yet, few instances of confirmed or suspected mortalities of giant manta ray attributed to vessel strikes (e.g., via strandings) have been documented. This lack of documented mortalities could also be the result of other factors that influence carcass detection (i.e., wind, currents, scavenging, decomposition etc.) (NMFS 2020d). In addition, manta rays appear to be able to heal from wounds very quickly, while high wound healing capacity is likely to be beneficial for their long-term survival, the fitness cost of injuries and number of vessel strikes occurring may be underestimated (McGregor et al. 2019).

While there is evidence of vessel interactions in nearshore aggregation areas where giant manta rays and vessels may be concentrated, we believe vessels used in the proposed action are extremely unlikely to strike a giant manta ray. While giant manta rays can be frequently observed traveling just below the surface and will often approach or show little fear toward vessels, few instances of strandings of giant manta ray are attributed to vessel strike injury. They also appear able to move fast enough to avoid most moving vessels, as anecdotally video evidence shows high speed vessels passing over giant manta rays and the ray being able to maneuver away in time to avoid the interaction (NMFS 2020d). In addition, vessels associated with the proposed action do not transit through giant manta ray aggregation areas. As a species more prevalent in the southern waters of the action area, the overlap of giant manta rays and vessels used by the fisheries considered in this Opinion is limited, and there is a low density of giant manta rays in the area these vessels primarily operate. Given that (1) giant manta rays have limited distribution in the vast majority of the areas the vessels in this Opinion operate, (2) giant manta rays are highly mobile, and (3) there are very limited reports of vessel interactions with giant manta rays, we have determined that interactions between vessels and giant manta rays are extremely unlikely to occur. Thus, the effects to giant manta rays from fishing vessels used in the federal fisheries are discountable.

7.6.3. ***Prey***

Giant manta rays are filter-feeders and generalist carnivores that feed on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderate sized fishes (Bigelow and Schroeder 1953b, Burgess et al. 2016, Miller and Klimovich 2017, Stewart et al. 2017). However, planktonic organisms appear to comprise the majority of the diet for giant manta rays. Foraging is rarely observed in U.S. waters, and the available data do not indicate any specific areas that appear to be used for foraging purposes within waters under U.S. jurisdiction (84 FR 66652, December 5, 2019). Overall, the best available information indicates that giant manta rays will feed on a variety of planktonic organisms and are not limited by the required presence of a specific prey species for successful foraging to occur (84 FR 66652, December 5, 2019). Additionally, planktonic organisms are extremely small and will pass through or around the fishing gears rather than be captured on or in them. Based on this information, we have determined that the effects of the fisheries on the availability of planktonic organisms for foraging giant manta rays are likely so small that they cannot be meaningfully measured, detected, or evaluated, and, therefore, insignificant.

7.6.4. ***Habitat***

Recent manta ray research in U.S. waters has documented the presence of juvenile giant manta rays off the east coast of Florida, suggesting the existence of juvenile and potential manta ray nursery habitat (84 FR 66652, December 5, 2019). While we have evidence of the presence and use of specific areas by juvenile giant manta rays, the available information does not allow us to identify any physical or biological features within these areas that are essential to support a manta ray nursery habitat. As described above, bottom trawl is the only gear type that has the potential to adversely affect bottom habitat in the action area. Given that the bluefish fishery is the only fishery to overlap with the habitat off Florida, that this overlap is limited by the low effort in this area, and the fishery off Florida is primarily gillnet and hook-and-line (gear types we don’t expect to adversely affect bottom habitat), we have determined that the effects of the fisheries on the essential habitat for giant manta rays are extremely unlikely and therefore, discountable.

7.7. Summary of Anticipated Interactions with ESA-listed Species

Large Whales

Based on the analysis above, we anticipate that 9.14 percent of the right whale population will become entangled each year. We estimated that between 2010 and 2018, entanglements resulted in an annual average of 4.7 M/SIs to right whales entangled in U.S. federal fishing gear. These interactions will be reduced with the implementation of Phase 1 of the Framework (i.e., the ALWTRP rule), which will reduce risk to right whales in American lobster and Jonah crab trap/pot fisheries by 60 percent. Based on the distribution of risk in state and federal waters, this will reduce M/SI from entanglements in gear used in the U.S. federal fisheries to annual average of 2.69. The Framework will further reduce the annual average of M/SI to 2.61 in 2023, 1.04 in 2025, and 0.136 in 2030. We also estimate that the operation of the U.S. federal fisheries with the ALWTRP measures implemented will entangle an annual average of 1.89 fin whales. resulting in an annual average of 1.08 M/SI; 1 sei whale, resulting in 1 M/SI; and 1 sperm whale, resulting in 1 M/SI.

Sea Turtles

Based on the information above, we anticipate 1,995 loggerhead sea turtles from the Northwest Atlantic DPS (1,036 in gillnet, 954 in trawls, and 5 in trap/pot) will interact gear utilized in the ten fisheries assessed in this Opinion every five years. Of these, 78 percent (808) of the interactions in gillnet gear 50 percent (477) of the interactions in bottom trawl gear, and 64 percent (4) of the interactions in trap/pot over a 5-year period are expected to lead to mortality. Therefore, 1,289 of the 1,995 loggerhead sea turtles that interact with these fisheries every five years are expected to die or sustain serious injuries leading to death or failure to reproduce.

We anticipate 142 leatherbacks (52 in gillnets, 40 in trawls, and 50 in trap/pots) will interact with gear utilized in the ten fisheries assessed in this Opinion every five years. Of these, 78 percent (41) of the interactions in gillnet gear, 50 percent (20) of the interactions in bottom trawl gear, and 64 percent (32) of the interactions in trap/pot gear over a 5-year period are expected to lead to mortality. Therefore, 93 of the 142 leatherback sea turtles that interact with these fisheries every five years are expected to die or sustain serious injuries leading to death or failure to reproduce.

We anticipate 292 Kemp's ridleys (239 in gillnets and 53 in trawls) will interact with gear utilized in the ten fisheries assessed in this Opinion every five years. Of these, 78 percent (187) of the interactions in gillnet gear and 50 percent (27) of the interactions in bottom trawl gear over a 5-year period are expected to lead to mortality. Therefore, 214 of the 292 Kemp's ridley sea turtles that interact with these fisheries every five years are expected to die or sustain serious injuries leading to death or failure to reproduce.

We anticipate 42 green sea turtles (10 in gillnet and 32 in trawls) will interact with gear in the ten fisheries assessed in this Opinion every five years. Of these, 78 percent, (8) of the interactions in gillnet gear and 50 percent (16) of the interactions in bottom trawl gear over a 5-year period are expected to lead to mortality. Therefore, 24 of the 42 green sea turtles that interact with these fisheries every five years are expected to die or sustain serious injuries leading to death or failure to reproduce.

In addition, we anticipate that three sea turtles of any species may be struck by vessels operating in these fisheries each year and that these interactions may result in mortality. This results in 15 mortalities over a 5-year period. The total number of anticipated sea turtle interactions with fishing gear and vessels in the ten fisheries addressed in this Opinion over a 5-year period is summarized in Table 71.

Atlantic Sturgeon
Based on the history of documented interactions with commercial fishing gear and largely on the results of the ASMFC (2017) Atlantic sturgeon stock assessment, we anticipate 1,406 interactions annually between Atlantic sturgeon and otter trawls and gillnets used in the ten fisheries. Of those interactions, 628 are expected from gillnet gear and 778 are expected to be from otter trawls.

NEFOP data indicates that average mortality rates of Atlantic sturgeon caught in otter trawl gear and gillnet gear across the federal fisheries is approximately 5 percent and 20 percent, respectively. Using those percentages and results from the genetics mixed stock analysis, we have been able to estimate the number of sub-adult and adult interactions and mortalities with each gear type per DPS. A summary of the annual anticipated Atlantic sturgeon interactions in the ten fisheries addressed in this Opinion is summarized by gear type Table 77.

Table 77: Anticipated Atlantic sturgeon mortalities by gear type and DPS in the ten fisheries each year.

	Gillnet Interactions	Gillnet Mortalities	Trawls Interactions	Trawl Mortalities
GOM DPS	54.6	10.9	67.7	3.4
NYB DPS	448.4	89.7	555.5	27.8
CB DPS	67.2	13.4	83.2	4.2
Carolina DPS	16.3	3.3	20.3	1.0
SA DPS	35.2	7.0	43.6	2.2
Canada	6.3	1.3	7.8	0.4
USA DPS Sum	621.7	124.3	770.3	38.6

Atlantic Salmon
Based on the known distribution of GOM DPS Atlantic salmon yet the lack of genetic information on the fish involved in interactions from 2010-2019, we are taking a precautionary approach by assuming that all the interactions were with GOM DPS Atlantic salmon. Based on past data, we anticipate two GOM DPS Atlantic salmon interactions every five years in either trawl or gillnet gear. Both of these interactions may result in mortality.

Giant Manta Rays
Based on observer data from 2001 through 2019, we estimate that four giant manta rays will interact with the trawl or gillnet gear over a 5-year period. One of these interactions (either in trawl or gillnet) may result in mortality in a 10-year period.

7.8. NEFMC's Omnibus Essential Fish Habitat (EFH) Amendment 2

As described in the proposed action, NMFS implemented approved regulations for the New England Fishery Management Council's Omnibus Essential Fish Habitat (EFH) Amendment 2 (Habitat Amendment or Amendment) on April 9, 2018 (68 FR 15240). In this section, we evaluate how the implementation of the Habitat Amendment affected the operation of the fisheries in this Opinion, as well as fisheries outside the Opinion, and whether any changes in the operation of the fisheries changed the anticipated effects on ESA-listed species considered above or in previous consultations. For fisheries outside of this Opinion (i.e., Atlantic sea scallop, Atlantic herring, surfclam/ocean quahog, and tilefish), only those that are likely to adversely affect listed species (e.g., interactions between fishing gear and listed species have been documented) are considered in this evaluation. The Atlantic sea scallop fishery operates in the GAR. Interactions between scallop fishing gear (scallop dredge or trawl) and listed species have been observed, with interactions often resulting in the injury or mortality to the animal. Given this, the Atlantic sea scallop fishery, along with the other fisheries in this Opinion, will be considered in the following assessment. As provided in section 2, the Atlantic herring (primarily purse seine and mid-water trawl gear), surfclam/ocean quahog (primarily hydraulic clam dredge gear), and tilefish (primarily bottom longline and rod/reel gear) FMPs operate in the GAR; however, interactions with ESA-listed species in these fisheries have not been documented, are extremely unlikely, or the gear is not known to interact with listed species or critical habitat.[46] As the Amendment did not implement measures that changed the overall nature and operation of these fisheries (i.e., gear and area fished; see Appendix 2), the Amendment did not introduce effects to listed species that have not been previously considered in prior consultations.[47] Given this, the underlying consultations and determination of effects for these fisheries remain valid. These fisheries and their associated gear types (i.e., purse seine, mid-water trawl, hydraulic clam dredge, bottom longline, and /or rod and reel), therefore, will not be considered further in this evaluation. the animal. Given this, the Atlantic sea scallop fishery, along with the other fisheries in this Opinion, will be considered in the following assessment.

The Habitat Amendment revised the EFH and habitat areas of particular concern (HAPC) designations; revised, removed, or created habitat management areas (including gear restrictions); established dedicated habitat research areas (DHRA); established, maintained, or removed spawning protection areas; and, implemented administrative measures (i.e., framework adjustment and monitoring measures). However, as provided in section 3.3, on October 28, 2019, the court enjoined NMFS from allowing gillnet fishing within the Nantucket Lightship (NLS) and Closed Area 1 (CA I) Groundfish Closure Areas (CLF v. Ross, Civil Action No. 18-1087 (JEB)). Per this Order, on December 17, 2019, NMFS issued a rule suspending the Amendment's opening of the NLS and CA I Groundfish Closure Areas to gillnet fishing. All other measures implemented by the Amendment remain in place.

As habitat management measures, pre-or post- Habitat Amendment, have been implemented in various portions of the Gulf of Maine (GOM), Georges Bank (GB), and Southern New England (SNE), it is important to define these sub-regions. For the purposes of this assessment, the sub-regions are defined by ecological sub-regions or (EPU) (Ecosystem Assessment Program 2012). Specifically, four primary EPUs have been identified in the Northeast U.S. Continental Shelf

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[46] ESA section 7 consultation has been completed on the Atlantic herring (February 9, 2010), Surfclam/ocean quahog (January 2, 2020), and Tilefish FMPs (October 27, 2017). NMFS determined that these fisheries may affect, but are not likely to adversely affect ESA-listed species or designated critical habitat. As the Amendment did not implement measures that changed the nature and operation of these fisheries, the Amendment did not introduce effects to listed species that have not been previously considered in prior consultations. Given this, the underlying consultations and determination of effects for these fisheries remain valid. Therefore, these fisheries remained covered by the consultations issued on February 9, 2010 (Atlantic herring FMP); January 2, 2020 (Surfclam/ocean quahog FMP); and, October 27, 2017 (Tilefish FMP).

[47] See footnote 1.

Large Marine Ecosystem: GOM, Scotian Shelf, GB, and Mid-Atlantic Bight (MAB) (Figure 54). The boundaries of the EPUs are based upon regions of the Northeast U.S. Continental Shelf Large Marine Ecosystem that have distinct patterns in oceanographic properties, primary production, and fish distribution (Ecosystem Assessment Program 2012). Although the SNE sub-region is not defined as an EPU, it is included in our analysis below. The approximate location of the SNE sub-region overlaps the Mid-Atlantic and GB EPUs (i.e., see overlap in statistical areas 538, 537, 526). Taking this into consideration, and based on input from SFD and the Councils, this assessment will consider statistical area 539 and portions of 538, 537, and 526 as the SNE sub-region.

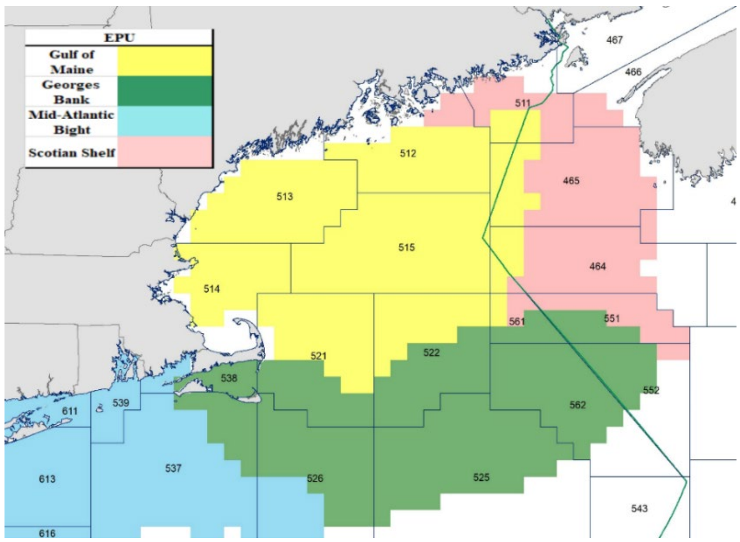

Figure 54: Ecological Production Units of the Northeast U.S. Continental Shelf Large Marine Ecosystem

Given the size and complexity of the Habitat Amendment, the analysis is broken down into the following categories of habitat management: (1) habitat management measures; (2) EFH and HAPC; (3) spawning protection measures; and, (4) framework adjustments and monitoring.

In order to comprehensively evaluate the measures implemented by the Habitat Amendment, including those NMFS suspended (see section 3.3), the following analysis will compare the impacts resulting from the implementation of all approved measures to impacts occurring before the measures were implemented. To do this, we first assess what changes occurred by region and what affect this would have on fishing effort. We then assessed the effects to protected species that may result from any changes in the fishery as a result of the Habitat Amendment.

More specifically, we used the following methods for each region. As actions authorized in the Habitat Amendment have the potential to change fishing behavior (e.g., shifts in effort), the following analysis will describe how effort is expected to change in response to the measures authorized in the Amendment and whether these changes, if they occur, equate to new or elevated risks to listed species. To inform this analysis, we consider:

1. fishing effort and behavior (e.g, area fished) in the GOM, on GB, and in SNE pre-and post- implementation of the Habitat Amendment. This includes assessing any shifts in effort, with the term “localized” referring to shifts within the same general area within the EPU or a habitat or groundfish management area.

2. changes to, or removal of, management area boundaries; fishing gear restrictions within management areas; and fishing gear types predominantly used in and around the designated habitat management areas pre-and post-Amendment (NEFMC 2016b);
3. listed species distribution in the GOM, on GB, and in SNE. Specific emphasis is placed on the overlap, in time and space, with the designated habitat management areas pre- and post-implementation of the Habitat Amendment); and,
4. documented records of listed species gear interactions.

Fishing effort can be defined in a number of ways (e.g., number of permits, trips, days fished, amount of landings). Given this, careful consideration is made when using the term "effort" and in inferring any changes to fishing effort post-Amendment. When assessing risks to protected species, the quantity of gear in the water (e.g., number of vertical lines, gillnets, bottom trawls), gear soak/tow duration, and the temporal and spatial overlap of the gear and protected species is considered. This analysis evaluates the distribution and quantity of gear in the water, as well as the number of trips as a proxy for fishing effort. In terms of evaluating the distribution and quantity of gear in the water, only those gear types identified in this section and in section 7 as likely to adversely affect listed species are considered in the assessment (i.e., trap/pot, sink gillnet, bottom trawl, and scallop dredge).

To evaluate fishing effort for all gear types except trap/pot, we reviewed data provided on the Northeast Ocean Data Portal[48] (1996 through 2015), the NEFSC's Fishing Footprints[49] (1996 through 2015), and more recent VTR data (2016-2019) (NMFS unpublished data, Appendix 2). When applicable, NMFS also used information from NMFS' Marine Mammal Stock Assessment Reports (SARS).[50] The SARS provide fishery information for those fisheries that interact with marine mammals. In general, these data sources showed similar patterns of fishing effort. When differences were apparent, we have described the differences below. For recent VTR data, both pre-Amendment (September 1, 2016, through March 31, 2018) and post-Amendment (April 1, 2018, through October 31, 2019) data was evaluated. The number of trips reported on VTRs in specified regions and by gear type was used to assess whether there were shifts in effort after implementation of the Amendment. While VTR data prior to 2016 is available, these specific timeframes were chosen to allow us to compare similar periods (approximately 572 days). Each of these time periods include one complete fishing year and part of another. As fishing effort can vary seasonally, sampling data in the midst of a fishing year can result in an incomplete picture of overall effort for that fishing year. Depending on the fishery and the targeted stock, there may be months when there is more directed effort than others. Given the variable timeframes considered as pre- or post- Amendment, it is possible that the VTR data sets considered to be pre- or post- Amendment, do not equally capture overall effort a fishery over the designated timeframe. As a result, apparent changes in effort may be due to sampling, and not necessarily a result of the measures implemented under the Amendment. In these cases, we also describe the fishery and potential shifts qualitatively. Figures provided in Appendix 2 depict the distribution of trips in the GOM, GB, and SNE pre- and post- Amendment.

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[48] https://www.northeastoceandata.org/
[49] https://fish.nefsc.noaa.gov/read/socialsci/fishing-footprints.php
[50] See Appendix III (Fishery Descriptions) in the marine mammal SARS: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessments

Quantifying federal trap/pot gear effort from these data sources at fine spatial scales is difficult given data necessary to quantify this effort (e.g., VMS or VTR) is often lacking. Recent tools developed by the NEFSC, in support of the ALWTRT, have helped inform the spatial resolution of the lobster and Jonah crab trap/pot fisheries; American lobster is the predominant trap/pot fisheries in the Northeast Region (i.e., GOM, GB, and SNE). The NEFSC's DST assists managers, decision makers, and stakeholders visualize and understand the spatiotemporal distribution of trap/pot gear and the spatiotemporal overlap of this gear with large whales. This assessment uses the information and results from the NEFSC's Decision Support Tool as the best available information on the distribution of trap/pot effort in the GOM, GB, and SNE. The Decision Support Tool estimates the number of traps per square mile. This is the best available estimate of trap/pot effort in waters off the northeast United States. This output is found in the draft Environmental Impact Statement on the proposed ALWTRT measures (NMFS 2020b)[51].

After evaluating the fishing effort (magnitude and distribution) in each region and how it may have changed with the implementation of the Habitat Amendment, we assess the risks to ESA-listed species. To assess risk, we take into consideration our analysis of effort pre- and post-Amendment, ESA-listed species distribution (see *Status of the Species*), and documented interactions. This informs the degree of overlap between listed species and fisheries in each region. We then identify which gears pose a risk to listed species. This information is used to inform how risks to ESA-listed species may have changed with the implementation of the Habitat Amendment. In assessing risk, we consider how the level of risk affected by changes in the:

1. temporal and spatial overlap of the gear and a protected species.
2. quantity of gear in the water (e.g., number of vertical lines, gillnets, bottom trawls).
3. gear soak/tow duration.

Finally, we consider the impacts across all regions and assess whether additional (or less) risk is anticipated from the implementation of the Habitat Omnibus Amendment.

7.8.1. *Habitat Management Areas (Habitat and Groundfish Closed Areas)*

Habitat management areas include Habitat and Groundfish Closed Areas, which restrict the use of certain gears to minimize, to the extent practicable, adverse[52] effects of fishing on EFH. Gear used in the Northwest Atlantic fisheries fall in two major categories: mobile gear (e.g., bottom trawls, scallop dredge) and fixed gear (e.g., trap/pot, sink gillnet). Changes to restrictions of gear fished in designated habitat management areas has the potential to directly or indirectly affect mobile and/or fixed gear fishing effort. Therefore, this assessment first evaluates how gear restrictions in these areas, pre- and post- Amendment, affect effort in the region. This information will then be used to assess interactions risks to listed species pre- and post-Amendment.

Between 1994 and 2004, habitat and groundfish closure areas were established in the GOM, GB, and SNE (Figure 55). These closure areas were designated to protect EFH and/or groundfish

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[51] In the DEIS for the proposed ALWTRP rule (NMFS 2020b), see appendix 3.1 for information and results of the DST.

[52] Specific to EFH, the Secretarial EFH guidelines (67 FR 2343, January 17, 2002) define an 'adverse effect' as any impact that reduces the quality and/or quantity of EFH, but only requires that actions be taken to prevent, mitigate, or minimize adverse effects from fishing, if they are both 'more than minimal' and 'not temporary'.

stocks by restricting particular fishing gears (e.g., bottom trawls, sink gillnets) (NEFMC 2016b); they were not designed to reduce interactions with listed species. Given that the closures have been in place for 16-26 years, patterns of fishing behavior and effort in the GOM, on GB, and in SNE are well-established, with effort often concentrating at the closure boundaries. This behavior, in turn, results in some gear types being more prevalent in certain areas of the GOM, on GB, or in SNE. Here, we describe the long-established behaviors that were present prior to the implementation of the Habitat Amendment and evaluate whether changes are expected due to the Amendment's implementation of measures that removed, modified, or maintained long-term closures in the three sub-regions (Figure 55).

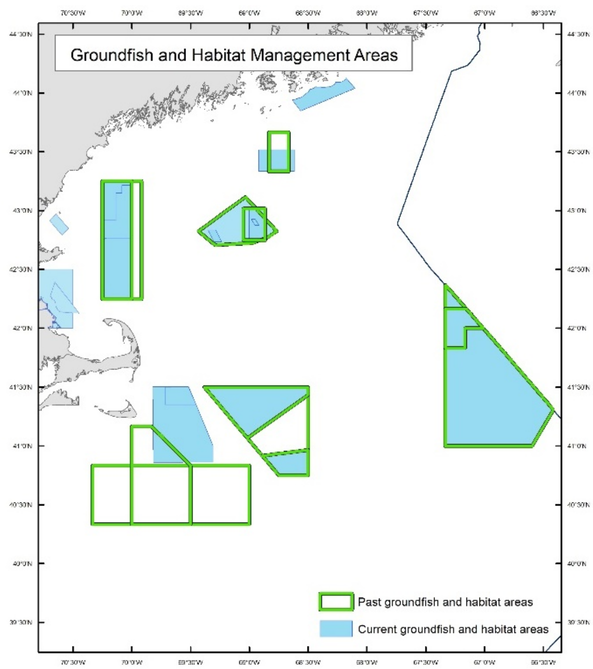

Figure 55: Habitat and groundfish closure areas pre- and post- Habitat Amendment

7.8.1.1. Measures in the Gulf of Maine (GOM)

Prior to the Habitat Amendment, the following Habitat and/or Groundfish Closure Areas existed in the GOM and provide an understanding of pre-existing fishing behavior and effort:

Western Gulf of Maine (WGOM) Habitat and Groundfish Closure Areas and Inshore Roller Gear Restricted Area

In the western portion of the GOM, an Inshore Roller Gear Restricted Area and the WGOM Habitat and Groundfish Closure Areas existed prior to the Amendment. The boundaries of the WGOM Habitat and Groundfish Closure Areas overlapped with the exception of the eastern boundary of the Groundfish Closure area which extended farther into the GOM (Figure 56). Prior to the Amendment, the following management measures applied:

- Inshore Restricted Roller Gear Area: maximum diameter of any part of the trawl footrope of 12 inches; fishing with all other gear types permitted in this area.
- WGOM Habitat Closure: Mobile bottom-tending gears (e.g., bottom trawl, scallop dredge) prohibited. Fishing with all other gear types (e.g., ,sink gillnet, trap/pot) permitted.

- WGOM Groundfish Closure: sink gillnet, bottom trawl, scallop dredge, and bottom longline gears prohibited. Pot/trap, mid-water trawl, purse seine, and clam dredge gears permitted.

In addition to the long-established habitat and groundfish closures, sequential (later termed rolling) closures to protect GOM cod have been implemented in the GOM since 1998, with the most recent adjustment in 2015 (63 FR 15326, March 31, 1998; 80 FR 25110, May 1, 2015). Based on our review of the available data, scallop dredge, sink gillnet, bottom trawls, and trap/pot gear are commonly used in this area of the GOM. Prior to the Amendment: (1) there was a predominance of scallop dredge effort on Stellwagen Bank near the southern boundary of the WGOM Habitat and Groundfish Closure Areas, as well as within the nearshore waters of Ipswich Bay; (2) sink gillnet effort was prevalent along the western and northeastern boundary of both the WGOM Habitat and Groundfish Closure Areas; and (3) bottom trawl operations were prevalent around all the boundaries of both areas. Pot/trap effort was prevalent throughout the WGOM (both within and outside of the Closure Areas). Depending on distance from shore, the number of traps per square mile varies, with inshore waters of the WGOM (within 3 miles from shore) containing approximately 10 to 1,000 traps per square mile, and waters in the WGOM greater than 3 miles from shore containing approximately 1 to 100 traps per square mile (NMFS 2020b). To summarize, prior to the Amendment fishing in and around the Closed Areas is dominated by bottom trawl, sink gillnet, scallop dredge, and trap/pot gear.

In the Western GOM, the Habitat Amendment implemented the following management measures (Figure 56):

- Maintained the Inshore Roller Gear Restriction. Gear restrictions and exemptions remain unchanged.
- Maintained the WGOM Habitat Closure (now termed Habitat Management Area (HMA)). Gear restrictions and exemptions remain unchanged.
- Aligned the eastern boundary of WGOM Groundfish Closure with the WGOM HMA. Gear restrictions and exemptions remain unchanged.
- Exempted shrimp trawling from the designated portion of the northwest corner of the WGOM Closure Areas (i.e., WGOM Habitat HMA and Groundfish Closure Area).
- Established the Stellwagen DHRA. The footprint of the DHRA is located within the WGOM Closure Areas and is closed to all commercial mobile bottom-tending, sink gillnet, and demersal longline gears. All of these are also prohibited by the WGOM Groundfish Closure restrictions.

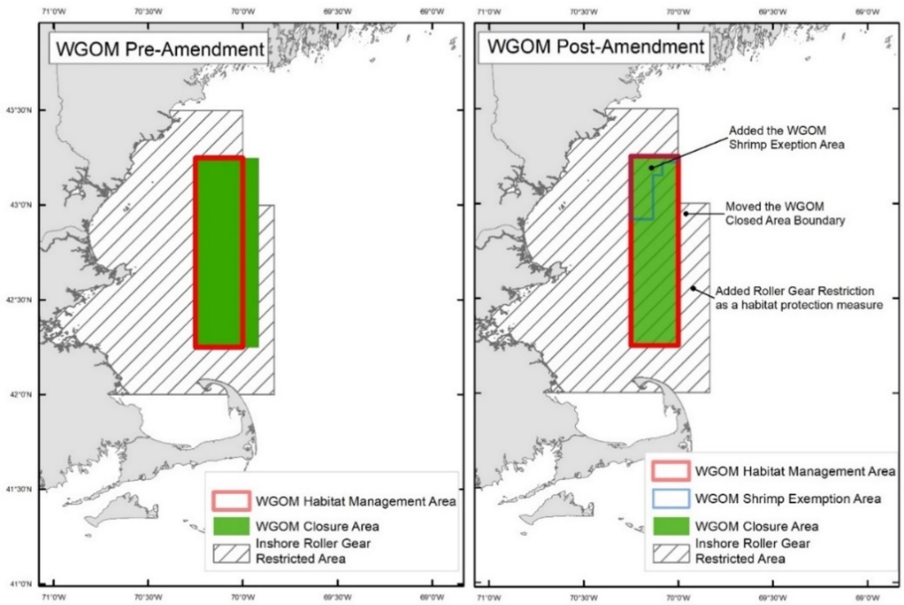

Figure 56: Pre- and post- habitat and groundfish management areas in the WGOM

Cashes Ledge Habitat and Groundfish Closure Areas

Prior to the Amendment, in the central GOM (CGOM), the boundaries of the Cashes Ledge Habitat Closure Area fell within the boundaries of the Cashes Ledge Groundfish Closure Area (Figure 57); during this time, the following management measures applied:

- Cashes Ledge Habitat Closure Area: Bottom trawl, scallop dredge, and clam dredge gears prohibited. Pot/trap, sink gillnet, mid-water trawl, purse seine, and bottom long-line gears permitted.
- Cashes Ledge Groundfish Closure Area: Sink gillnet, bottom trawl, scallop dredge, and bottom longline gears prohibited. Pot/trap, clam dredge, mid-water, purse seine gears permitted.

Prior to the Amendment, bottom trawl gear, followed by gillnet gear, is the primary gear type fished around the Closure Areas in the CGOM. Pot/trap effort was also common throughout the CGOM (both within and outside of the Closure Areas), with the number of traps per square mile estimated to be approximately ≤10 traps per square mile (Fig. 4.1.3.a in NMFS 2020b).

The Amendment implemented the following management measures in the CGOM (Figure 57):

- Maintained Cashes Ledge Groundfish Closure Area. Gear restrictions and exemptions remain unchanged.
- Shifted the western boundary of the Cashes Ledge HMA slightly to the east. The Cashes Ledge Habitat Closure Area (now the HMA) still resides in the Cashes Ledge Closure Area. Gear restrictions and exemptions in the Cashes Ledge HMA remain unchanged.
- Established the Ammen Rock HMA. This HMA is located inside the Cashes Ledge HMA. With the exception of lobster pot/tap gear, fishing with all other gear types is prohibited.
- Established the Fippennies Ledge HMA. This HMA is located inside the Cashes Ledge Groundfish Closure Area. All mobile bottom tending gear is prohibited.

Jeffreys Ledge Habitat Closure Area
Prior to the Amendment, in the CGOM, Jeffreys Ledge Habitat Closure Area (Figure 57) existed north of the Cashes Ledge Habitat and Groundfish Closure Areas; restrictions to mobile bottom tending gear were the same as those provided in the Cashes Ledge Habitat Closure Area. The review of fishing data consistently showed bottom trawl gear and trap/pot gear (both within and outside of the Closure Areas) being the predominant gear type fished around the Closure Area prior to the Amendment. In this region of the GOM, the number of traps per square mile varies by distance from shore. In general, the number of traps per square mile decreases with increased distance from shore. Coastal or nearshore (i.e., < 3 miles from shore) waters contain approximately 100 to 1,000+ traps per square mile in this area of the GOM, while more offshore waters (i.e., 3-12 miles from shore) contain approximately 10 to 100 traps per square mile (NMFS 2020b). There was little to no effort with other gear types (i.e., sink gillnet, dredge) in this region of the GOM prior to the Amendment.

The Habitat Amendment implemented the following management measures in this area of CGOM (Figure 57):

- Modified the Jeffreys Bank Habitat Closure Area (now termed HMA) boundaries. Gear restrictions and exemptions remain unchanged; and,
- Established the (small) Eastern Maine HMA; All mobile bottom tending gear is prohibited.

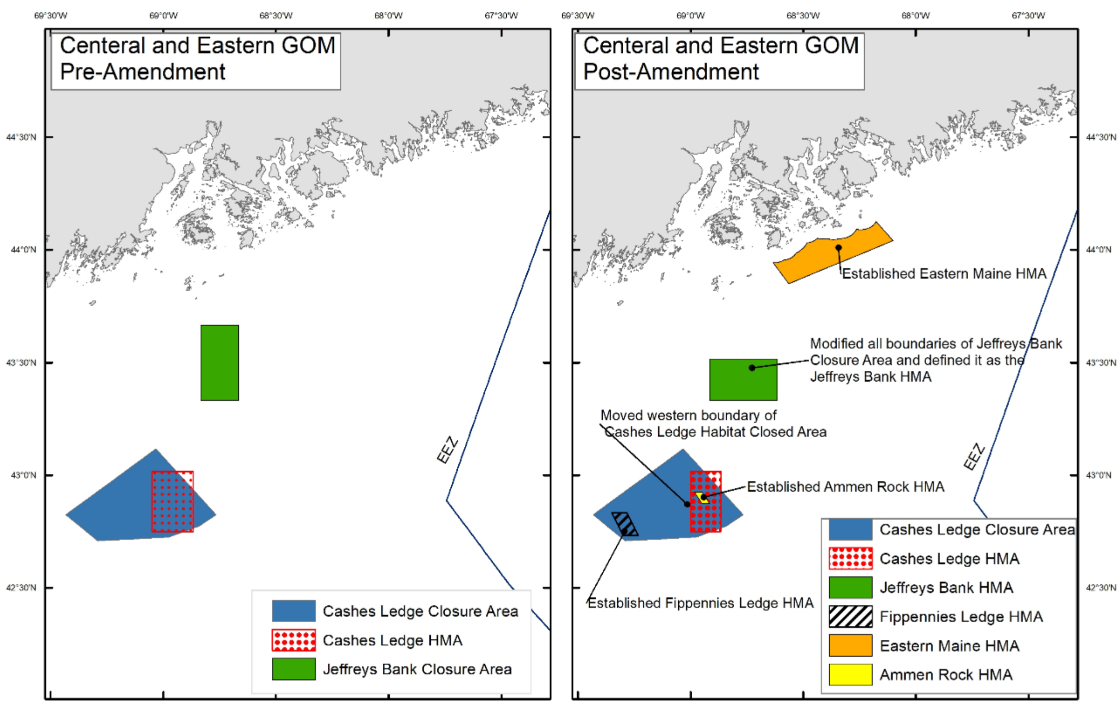

Figure 57: Pre- and post- habitat and groundfish management areas in the CGOM

7.8.1.2. *Effort in the Gulf of Maine Pre- and Post- Amendment*

There is the potential that the management measures implemented through the Amendment may change fishing effort in the GOM, relative to pre-existing conditions prior to the Amendment.

However, as described below and as provided in Appendix 2, little to no shifts and effort are expected.

The Amendment established the Stellwagen Bank DHRA, Ammen Rock HMA, Fippennies Ledge HMA, and EGOM HMA. The Amendment also modified the boundaries of the WGOM Groundfish Closure Area, Jeffreys Bank Closure Area (now termed HMA), and the Cashes Ledge HMA. The modification to these boundaries did not greatly change the overall footprint of these areas (Figure 56 and Figure 57); that is, they continue to fall within existing GOM closure areas or in portions of the GOM where the requirements implemented through the Amendment will have little to no effect to fisheries in this area. Given this, the management measures implemented by the Habitat Amendment provided little incentive for vessels to greatly change fishing behavior (e.g., area fished) or effort post-Amendment. For instance, VTR data showed that post-Amendment modification to the boundaries of the WGOM Groundfish Closure resulted in little to no change in distribution or level of gillnet or bottom trawl effort (see Appendix 2). Specifically, relative to pre-Amendment conditions, sink gillnet effort remained prevalent along the western and northeastern boundary of both the WGOM Habitat and Groundfish Closure Areas; bottom trawl effort also remained prevalent around all the boundaries of both areas. Overall, relative to pre-Amendment conditions in the WGOM, significant changes in gillnet or bottom trawl effort, post-Amendment, were not apparent (see Appendix 2). In addition, review of the VTR data showed that the designation of the Jeffreys Bank HMA resulted in a small shift in bottom trawl effort just to the west of the HMA (i.e., in the area once delineated as the Jeffrey's Bank Closure Area); however, even with this small shift, the overall level and distribution of bottom trawl effort remained relatively consistent with pre-Amendment conditions in the GOM, with little to no displacement of effort (see Appendix 2). To provide another example, in the area where the EGOM HMA is designated, fishing is primarily prosecuted with trap/pot and purse seine gear. Mobile bottom tending gear (e.g., shrimp/bottom trawl, clam dredge) is used minimally. As the EGOM HMA only restricts the use of mobile bottom tending gear, post-Amendment changes in fishing behavior or shifts in effort are not expected to occur in this area of the GOM.

Pot/trap operations in the GOM are also likely to remain similar to pre-Amendment conditions as the Amendment did not appear to create any incentive for effort to change. Pot/trap gear has never been restricted in the Habitat or Groundfish Closure Areas. Therefore, effort in these areas is expected to continue at levels similar to the effort pre-Amendment. In addition, as the fishing grounds for trap/pot fisheries are well established throughout the GOM and introduction of new effort would result in gear conflicts, it is not expected that post-Amendment, trap/pot fisheries will be forced to relocate from pre-existing fishing grounds due to other vessels (e.g., mobile bottom tending, sink gillnet) responding to the measures implemented by the Amendment. Similarly, given the established fishing grounds of trap/pot fisheries in the GOM, combined with the desire to avoid gear conflicts both within and outside of the fishery, any post-Amendment changes to habitat or groundfish management areas (i.e., opening or closing an area to bottom tending gear) are not expected to create incentive for large shifts in trap/pot effort within or around these affected areas or in the GOM overall. Should any shifts in pot trap effort occur post-Amendment, it is likely to be seen by those trap/pot vessels whose fishing grounds overlap with or are in close proximity to the management areas delineated or modified by the Amendment. Given this, any shifts in trap/pot effort are expected to be localized and not a result of a vessel attempting to gain access to an area that it previously had not fished.

Based on this analysis, there is no evidence that the measures implemented through the Habitat Amendment created incentive for vessels to change their operations. This is supported by: (1) the management measures implemented in the GOM through the Habitat Amendment resulted in little to no change in the overall footprint of pre-existing Groundfish or Habitat Closure Areas (Appendix 2) and (2) gear restrictions in the designated closure areas also remained relatively consistent. Relative to operating conditions prior to the Amendment, there was little to no change in area fished, quantity of gear set/towed, and/or placement of fishing gear (i.e., gillnet, trap/pot, bottom trawl) in the GOM (see Appendix 2).

7.8.1.3. Assessment of Risks to Protected Species

In the *Status of the Species* (see section 4.0), Table 42 identifies the ESA-listed species and critical habitat that occur in the action area and that may be adversely affected (e.g., there have been observed and documented interactions in the fisheries or with gear type(s) similar to those used in the fisheries) by the proposed action. Of the species identified in Table 42, only ESA listed species of large whales, sea turtles, Atlantic sturgeon, and Atlantic salmon are likely to occur in the GOM and overlap with the fisheries operating in this region. The following evaluation will considered gear interaction risks to these species. Based on this information, and our evaluation of fishing effort pre-and post-Amendment (see above), we then evaluate how these risks may have changed with the implementation of the Amendment.

Large Whales in the GOM: Occurrence and Gear Interaction Risks
As provided in the *Status of the Species*, North Atlantic right, fin, sei, and sperm whales are likely to occur in the GOM. North Atlantic right, fin, sei and sperm whales are likely to overlap with fisheries operating in the GOM. However, not all fisheries operating in this region pose an entanglement risk to these species. Based on more than 30 years of observed or documented interactions (see Marine Mammal SARs) between large whales and fishing gear in the Northwest Atlantic, the greatest entanglement risk to large whales is posed by fixed gear used in trap/pot or sink gillnet fisheries (Angliss and DeMaster 1998, Cassoff et al. 2011, Henry et al. 2017, Henry et al. 2015, 2016, Henry et al. 2020, Henry et al. 2019, Johnson et al. 2005, Knowlton and Kraus 2001, Sharp et al. 2019). Fin, sei, and North Atlantic right whales have died or been seriously injured from entanglement in fishing gear along the Gulf of Mexico Coast, U.S. East Coast, and Atlantic Canadian Provinces from 2012 to 2018; however, there have been no confirmed M/SI from entanglement in fishing gear for sperm whales since 2011[53] (Hayes 2019, Hayes et al. 2020, Henry et al. 2020, Henry et al. 2019, Waring et al. 2015, Waring et al. 2014).

In regards to other gear types fished in the GOM, such as bottom trawl and scallop dredge there have been no observed or documented interactions between North Atlantic right, fin, sei, and sperm whales and these gear types (Henry et al. 2017, Henry et al. 2015, 2016, Henry et al. 2020, Henry et al. 2019) (see also NMFS' Marine Mammal Stock Assessment Reports: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-

[53] A sperm whale was reported entangled in monkfish net on the Canadian Grand Banks in 2011, but was released alive and gear free (Ledwell 2012 as cited in Waring et al. 2015).

assessment-reports-region). Based on this, interactions between these gear types and listed species of whales are extremely unlikely.

Sea turtles in the GOM: Occurrence and Gear Interaction Risks
As described in the *Status of the Species*, hard-shelled and leatherback sea turtles arrive in the Gulf Maine in June and leave the area by the end of November. Fisheries operating in the GOM use gear types known to pose an interaction risk to sea turtles, including gillnet, trap/pot, scallop dredge, and bottom trawl gear (Murray 2015a, 2018, 2020) NEFSC observer/sea sampling database, unpublished data; GAR STDN, unpublished data). Although sea turtles are at risk of interacting with these gears, observed fishery interactions in this sub-region are rare. Review of observer records over the last 30 years show only two observed gear interactions with a sea turtle in the GOM (i.e., both loggerhead sea turtles and gillnet gear); near WGOM Closure areas (NEFSC observer/sea sampling database, unpublished data). There is also data available through the GAR STDN. Leatherback sea turtles are at risk of interacting with trap/pot gear, specifically the vertical lines associated with this gear type. Based on information provided by the GAR STDN (unpublished data), from 2009 through 2018, leatherback sea turtles have been reported entangled in vertical line gear, with multiple cases reported in the GOM and in Cape Cod Bay (GAR STDN, unpublished data). In addition, hard-shelled sea turtles can become entangled in the vertical lines associated with trap/pot gear; however, none have been reported in the GOM from 2009-2018 (GAR STDN, unpublished data).

Atlantic sturgeon in the GOM: Occurrence and Gear Interaction Risks
As described in the *Status of the Species*, Atlantic sturgeon (adult and sub-adult) from all DPSs, occur in the GOM, primarily inshore of the 164-ft (50-m) depth contour (Dunton et al. 2010, Laney et al. 2007, Stein et al. 2004a, b, Waldman et al. 2013, Wirgin et al. 2015b). However, they are not restricted to these depths and excursions into deeper (e.g., 250 ft (75 m) continental shelf waters have been documented (Collins and Smith 1997, Dunton et al. 2010, Erickson et al. 2011, Stein et al. 2004a, b, Timoshkin 1968).

Of the gear types used in the GOM, gillnet and bottom trawl gear pose the greatest interaction risk to this species. Numerous interactions between Atlantic sturgeon and gillnet or bottom trawl gear have been observed in the Northwest Atlantic, with many of these interactions observed in the GOM (NEFSC observer/sea sampling database, unpublished data). Observed or reported interactions between Atlantic sturgeon and other gear types fished in the GOM (i.e., trap/pot, scallop dredge) are rare to non-existent (NEFSC observer/sea sampling database, unpublished data); based on this, interactions between Atlantic sturgeon and these gear types are extremely unlikely.

Atlantic salmon in the GOM: Occurrence and Gear Interaction Risks
In general, smolts, post-smolts, and adult Atlantic salmon may be present in the GOM and coastal waters of Maine in the spring (beginning in April), and adults may be present throughout the summer and fall months. Therefore, Atlantic salmon are likely to overlap with fisheries operating in the GOM. Of the gear types used in the GOM, gillnet and bottom trawl gear pose the greatest interaction risk to this listed species. However, although interactions are possible with these gear types, observed interactions are rare (Kocik et al. 2014) (NEFSC observer/sea sampling database, unpublished data). Observed interactions between Atlantic salmon and other gear types fished in the GOM (i.e., trap/pot, scallop dredge) are non-existent (NEFSC

observer/sea sampling database, unpublished data); based on this, interactions between these gear types and Atlantic salmon are extremely unlikely.

Summary of Co-occurrence and Gears that Pose a Risk to Listed Species
Given the information above, ESA-listed species of large whales, sea turtles, and fish are at risk of interacting with several gear types fished in the GOM, with some gear types posing more or less of an interaction risk than others. Table 78 provides a summary of those gear types that pose the greatest entanglement risk to listed species of large whales, sea turtles, and fish.

Table 78: Fishing gear types that pose the greatest entanglement risk to ESA-listed species of large whales, sea turtles, and fish.

Listed Species Group	Gear Types
Large whales	Sink Gillnet; Pot/Trap
Sea turtles	Sink Gillnet; Bottom Trawl; Scallop Dredge; Pot/Trap
Atlantic sturgeon and Atlantic salmon	Sink Gillnet; Bottom Trawl

7.8.1.4. Assessment of Risk with the Implementation of the Habitat Amendment

The species considered in this Opinion overlapped with these gear types prior to the Amendment; therefore, pre-Amendment fishing conditions in the GOM (i.e., pre-Amendment) posed some level of gear interaction risks to listed species occurring in the GOM. Here, we evaluate whether these risks changed when the Amendment was implemented. Although the measures implemented resulted in changes to management areas in the GOM, there is no indication that these changes created incentive for fishing behavior or effort to change from pre-existing operating conditions. Relative to conditions prior to the Amendment, the management measures implemented in the GOM through the Habitat Amendment resulted in very little change to the overall footprint of pre-existing Groundfish and Habitat Closure Areas. At most, small localized shifts in effort were seen post-Amendment; however, overall there was little to no difference in the distribution (locally or regionally) and magnitude (e.g., quantity of gear fished and gear soak/tow duration) of effort in the GOM after implementation of the Habitat Amendment. There is also no evidence to suggest that the measures in the Amendment created an incentive for changes in effort to occur in the future.

Species occurrence and distribution may respond and adapt to changes in the marine environment (e.g., changes to prey availability, water temperature changes); however, the overall occurrence and broad scale distribution of these species in the GOM has remained relatively consistent from pre-Amendment to post-Amendment. Given the above, the degree of overlap between listed species and gear in future fishing years is expected to remain similar to the overlap prior to the Amendment. Although the measures implemented through the Amendment are not expected to remove or reduce interaction risks to listed species, they are also not expected to result in new or increased interaction risks to these species relative to conditions in the GOM prior to the Amendment.

7.8.1.5. Measures on Georges Bank (GB)

Prior to the Habitat Amendment, the following Habitat and/or Groundfish Closure Areas existed on GB; this information will serve to establish pre-existing fishing effort on GB, particularly in and around these Closure Areas:

Closed Area I (CA I) Habitat and Groundfish Closure Areas
Prior to the Amendment, the CA I Habitat and Groundfish Closure Areas were located at the western end of GB. At this time, the boundaries of the CA I Habitat Closure Area fell within the boundaries of the CA I Groundfish Closure Area (Figure 58). Prior to the Amendment, the following management measures applied:

- CA I (North and South) Habitat Closure Area: Mobile bottom-tending gear (e.g., bottom trawl, scallop dredge) prohibited. Pot/trap, sink gillnet, mid-water trawl, purse seine, and bottom longline gear permitted.
- CA I Groundfish Closure Area: Pot/trap, purse seine, and mid-water trawl gears permitted. With the exception of NE Multispecies vessels participating in special access programs (SAPs), all other vessels fishing with gear types capable of catching groundfish (e.g., sink gillnet, bottom trawl, bottom longline) or with clam dredge gear prohibited. Through SAPs, bottom longline gear was allowed to be set in portions of the Groundfish Closed Area. Scallop dredge gear was also allowed in specified portions of the Closed Area; specifically the area designated by the Scallop FMP's scallop rotational management program as the CA I Access Area.

Scallop dredge, gillnet, bottom trawl, and trap/pot gear were commonly used in this area of GB prior to the Amendment. The review of available fishing data consistently showed a predominance of scallop dredge gear in the waters surrounding the western boundary of the Closure Areas (e.g., Great South Channel). As noted above, scallop dredge gear can operate in the area designated by the Scallop FMP as the CA I Access Area (located within the central portion of the CA I Groundfish Closure Area) at certain times. Prior to the Amendment: (1) sink gillnet gear was predominant at the tip of the Northwestern corner of the CA I North Habitat Closure Area and extended northwest to the eastern shore of Cape Cod's National Shoreline (e.g., waters off of Chatham, Massachusetts); (2) bottom trawl gear was predominant on all of the CA I boundaries and was common in the waters between the CA I and CA II Closure Area; and (3) trap/pot effort was fished both within and outside of the Closure Areas, with the number of traps per square mile in this region of GB estimated to be approximately ≤10 traps per square mile (Figure 4.1.3.a in NMFS 2020b).

CA II Habitat and Groundfish Closure Areas
Prior to the Amendment, the CA II Habitat and Groundfish Closure Areas were located at the eastern end of GB. The boundaries of the CA II Habitat Closure Area fell within the boundaries of the CA II Groundfish Closure Area (Figure 58). Prior to the Amendment, the following management measures applied:

- CA II Habitat Closure Area: bottom trawl, scallop dredge, and clam dredge gears prohibited. Pot/trap, mid-water trawl, and purse seine permitted.
- CA II Groundfish Closure Area: Pot/trap, purse seine, and mid-water trawl gears permitted. With the exception of NE Multispecies vessels participating in a SAP, all other vessels fishing with gear types capable of catching groundfish (e.g., sink gillnet, bottom trawl, bottom longline) or clam dredge were prohibited. Through SAPs, bottom trawl and bottom longline gears could also be used in portions of the Closed Area. Scallop dredge gear was also allowed in specified portions of the Closed Area; specifically the area designated by the scallop FMPs scallop rotational management program as the CA II Access Area.

Prior to the Amendment, scallop dredge, bottom trawl, and trap/pot gear were commonly fished on GB; gillnet gear was used to a lesser extent. Review of available fishing data showed that prior to the Amendment, there was a predominance of scallop dredge gear in the waters surrounding the southern boundary of the Closure Areas, as well as within the southeastern corner of the CA II Groundfish Closure Area; this latter area is designated by the Scallop FMP as the CA II Access Area.[54] Prior to the Amendment, bottom trawl effort was common in the waters off the western and southern boundaries of the CA II Closure Areas and in the waters between the CA I and CA II Closure Areas. Pot/trap effort occurred both within and outside of the Closure Area (Figure 4.1.3 in NMFS 2020b), with the number of traps per square mile estimated to be approximately ≤10 traps per square mile. There was limited gillnet effort off the western boundary of the Closure Area prior to the Amendment.

Relative to pre-Amendment conditions in this area of GB, the Amendment implemented the following management measures:

- Removed CA I Habitat and Groundfish Closure Areas.[55]
- Established CA I North Seasonal Spawning Closure (formerly CA I North Habitat Closure). Between February 1 to April 15, the area is closed to commercial and recreational gears capable of catching groundfish (e.g., bottom trawl, sink gillnet). Fishing with scallop dredge gear is permitted year round;
- Established the GB DHRA (formerly CAI South Habitat Closure). This area is closed to mobile bottom-tending gears for at least three years and could be opened after a review of the research activities in the area;
- Established the Great South Channel HMA. With the exception of clam dredge gear, mobile bottom-tending gear is restricted throughout the HMA. Pursuant to 85 FR 29870 (May 19, 2020), three dredge exemption areas were established within the Great South Channel HMA: McBlair, Old South, and Fishing Rip. McBlair and Fishing Rip dredge exemption areas will be open to fishing for surfclams year round. Old South Dredge Exemption Area will be open for surfclam fishing from May 1 through October 31.
- Maintained the CA II Habitat and Groundfish Closure closures, with the same gear restrictions (Figure 58). The Council recommended modifications to both areas, but NMFS disapproved those changes (83 FR 15240, April 9, 2018).

[54] Under Scallop FMP's rotational management, Access Areas are either opened or closed annually to fishing based on scallop health, size, and biomass in specific Access Areas. Consideration of these factors are necessary to assess the potential distribution and level of scallop fishing effort on a year to year basis.

[55] Pursuant to a ___________, NMFS issued a rule (84 FR 68798, December 17, 2019) suspending the Amendment's opening of the Closed Area I Groundfish Closure Area to gillnet fishing and restoring prior regulations prohibiting gillnet gear from fishing in this area until further notice.

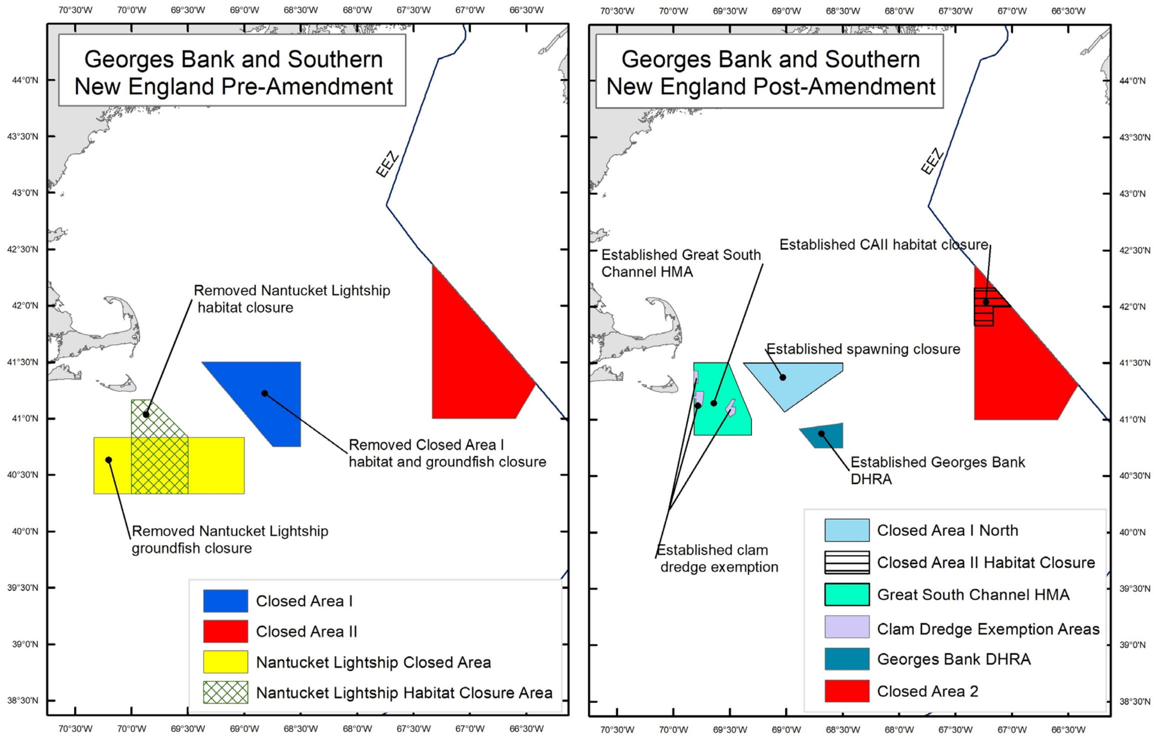

Figure 58: Pre- and post- habitat management and groundfish areas on GB (and SNE)

7.8.1.6. *Effort on Georges Bank Pre- and Post- Amendment*

The Habitat Amendment removed the CA I Habitat and Groundfish Closure Areas. The Amendment implemented the CA I North Seasonal Spawning Closure Area in the area formerly designated as the CA I North Habitat Closure and the GB DHRA in the area formerly designated as the CA I South Habitat Closure. There is the potential that the management measures implemented through the Amendment may change fishing effort in this area of GB. Figures provided in Appendix 2 depict the distribution of trips using sink gillnet, bottom trawl, or scallop dredge on GB pre- and post- Amendment.[56]

The removal of the CA I Habitat and Groundfish Closure Areas resulted in a shift of scallop dredge effort into the newly opened area post-Amendment. With this shift, the VTR data showed an increase in effort, particularly in the former CA I North Habitat Closure Area (Figure 59). As scallop dredge vessels were previously restricted from accessing the CA I Habitat Closure Areas, the opening of these areas post-Amendment created incentive for scallop dredge vessels to redirect effort into formerly closed areas. This opening; however, is not unrestricted. The Scallop FMP manages the scallop fishery through an Access Area (i.e., NLS, Mid-Atlantic, CA I, and

[56] On November 1, 2019, in preparation for the rule to re-close the NLS Groundfish Closure Area to gillnet fishing, NMFS notified gillnetters operating in the (former) NLS Groundfish Closure Area that all gillnet gear needed to be removed from the Closure Area. As a result, VTR data on gillnet vessels operating in this area between November 1, 2010 and December 16, 2019, was not considered as there was no active gillnet fishing in this area. The April 1, 2018, through October 31, 2019 time frame was considered in this analysis.

CAII) rotational management program to maximize scallop yield. This program restricts the level of effort in Access Areas on a year-to-year basis. Subsequent to the Habitat Amendment, the Scallop FMP expanded the CA I Access Area to encompass the western part of the CA I Closure that was removed by the Habitat Amendment.

When comparing the pre- to the post-Amendment maps provided in Figure 59, there is a shift in scallop dredge effort into the area formerly designated as the CA I North Habitat Closure.[57] Pre-Amendment (i.e., September 1, 2016, through March 31, 2018), the CA I Access Area was closed to scallop fishing (i.e., no allocated trips to the Access Area; 81 FR 26727 (May 4, 2016) and 82 FR 15144 (March 27, 2017)). This, combined with the prohibition of scallop dredge gear in the CA I North and South Habitat Closures resulted in scallop dredge effort outside of the CA I Closure and Access Area boundaries and concentrating primarily along the western boundaries of these areas. Post-Amendment, the removal of the CA I Closure Areas, combined with the (expanded) CA I Access Area being opened to scallop fishing, an increase in effort was seen in the area formerly designated as the CA I North Habitat Closure. While a localized increase in effort was seen in the area formerly designated as the CA I North Habitat Closure, there was no apparent increase in overall scallop dredge effort on GB in response to measures implemented post-Amendment.

In addition to the removal of the CAI Habitat and Groundfish Closure Areas, the Amendment also implemented the Great South Channel (GSC) HMA in a portion of GB that was accessible to mobile bottom tending gear, such as scallop dredges, pre-Amendment. As the HMA restricts the use of scallop dredge gear, existing scallop dredge vessels are likely to shift effort to: (1) just outside the boundaries of the HMA (accessing the available resources around the HMA's boundaries); and/or, (2) other pre-existing fishing grounds in portions of SNE (e.g., waters around or in NLS-Access Area (if opened) or CA I-Access Area (if opened; see above); this is supported by maps provided in Figure 59. However, as provided above, scallop fishing effort is overwhelmingly dictated by the Scallop FMPs rotational management program, which is based on annual scallop abundance estimates. Given this, where scallop vessels may shift to, as a result of the implementation of the GSC HMA, will be highly influenced by scallop abundance and resultant management measures implemented by the Scallop FMP. Below, the analysis assesses whether these changes in effort subsequent to the Habitat Amendment resulted in increased risks to protected species.

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[57] Between 2016 and 2019, the Scallop FMP's CA 1 Access Area was closed to scallop fishing in 2016 (80 FR 22119, April 21, 2015) and 2017 (82 FR 15155, March 27, 2017), and open to scallop fishing in 2018 (83 FR 17300, April 19, 2018) and 2019 (84 FR 11436, March 27, 2019).

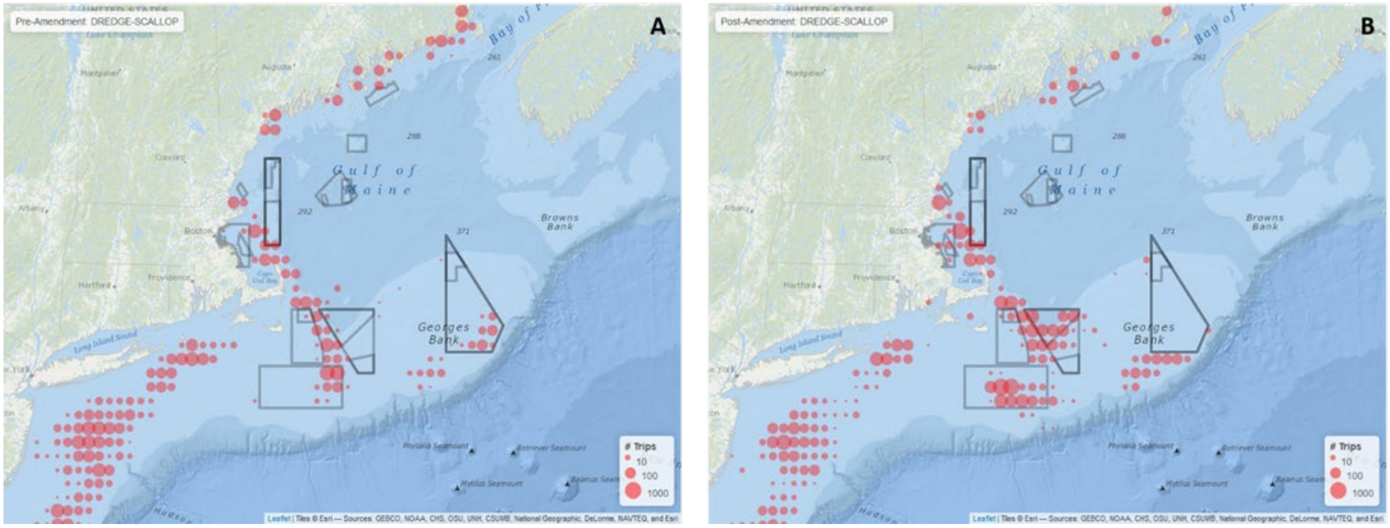

Figure 59: Scallop dredge VTRs pre-and post- Habitat Amendment. Figure A=Pre-Amendment (September 1, 2016-March 31, 2018) and Figure B=Post Amendment (April 1, 2018-October 31, 2019) (see Appendix 2)[58]

Sink gillnet gear is also fished to some extent on western GB. Pre-Amendment, gillnet gear was predominantly set near the northwestern corner of the CA I North Habitat Closure Area and extended west to the eastern shore of Cape Cod's National shoreline (e.g., waters off of Chatham, MA). Implementation of the GSC HMA did not restrict use of gillnet gear in the HMA. Therefore, this measure did not affect the distribution or level of effort with this gear type in this portion of GB. As result, the remainder of this assessment will focus on the effects of the Amendment's removal of the CA I Closure Areas on gillnet effort.

While the Habitat Amendment's removal of the CA I Closure Area has the potential to result in a shift in sink gillnet effort relative to conditions prior to the Amendment, large shifts are not expected. Depending on the health of target stocks in the area, there may be little incentive to shift into the newly opened areas. In general, vessels often concentrate gear along the boundaries of a restricted/closed area to take advantage of any spillover of a healthy target stock. The limited boundary effects seen by sink gillnet vessels in the region prior to the Amendment suggests that targeted stocks may be limited within the CA I Closure Areas, thereby discouraging gillnet effort around the Closed Area boundaries. Given that there may be limited availability of the target stock, the opening the CA I Closure Areas may provide little incentive for vessels to shift into this area of GB. In addition, with the expansion of the scallop CA I Access Area and the designation of the CA I North Seasonal Spawning Closure (formerly CA I North Habitat Closure), there may be little incentive to shift or sink gillnet effort into the now "opened" areas given the seasonal gear restrictions in the CA I North Spawning Closure and the potential for

[58] Figure 59 does not depict the boundaries of the Scallop FMP's Rotational Management Areas. The boundaries delineating the (prior) CA I Habitat and Groundfish Management Area are, in general, reflective of the area encompassed by the CA I Scallop Rotational Area. For details on the Scallop FMP's Rotational Management Area, see https://www.fisheries.noaa.gov/resource/map/atlantic-sea-scallop-managed-waters-fishing-year-2020.

gear conflicts associated with scallop dredge effort in the scallop CA I Access Area. This is supported by the VTR data (Figure 60).

Review of VTR data showed no apparent change in the distribution of sink gillnet effort in the GB region post-Amendment. Specifically, there was no large shift of gillnet effort locally or regionally (i.e., from other sub-regions) into the area formerly designated as CA I Habitat and Groundfish Closure Areas. If a large shift in effort had occurred post-Amendment, it is likely that this gear type would be more concentrated on GB, particularly in and around the former CA I Closure Areas. Review of the VTR data also showed no apparent increase in overall gillnet effort on GB post-Amendment. However, the VTR data did show a localized increase in gillnet effort in the waters just off Chatham, Massachusetts. As provided above, pre- or post-Amendment, this area has consistently been a site on GB where gillnet effort is predominant. During the time effort increased, there were several fishery management measures (described below) independent of this action that applied to gillnets in this area. These fishery management changes would have occurred whether or not the Habitat Amendment was implemented. Although the VTR data (Figure 60) capture a limited period of time post-Amendment, it is likely the pattern of fishing effort shown in the maps are reflective of future effort in the area.

Taking into consideration the above information, implementation of the Amendment resulted in small, localized shifts in effort by gillnet gear on GB; however, large shifts in effort from other regions outside of GB were not apparent . While a localized increase in gillnet effort was apparent on GB post-Amendment, review of the VTR data showed no evidence that overall gillnet effort increased as a result of the management measures implemented through the Habitat Amendment. Below, the analysis assesses whether the implementation of the Amendment resulted in increased risks to protected species.

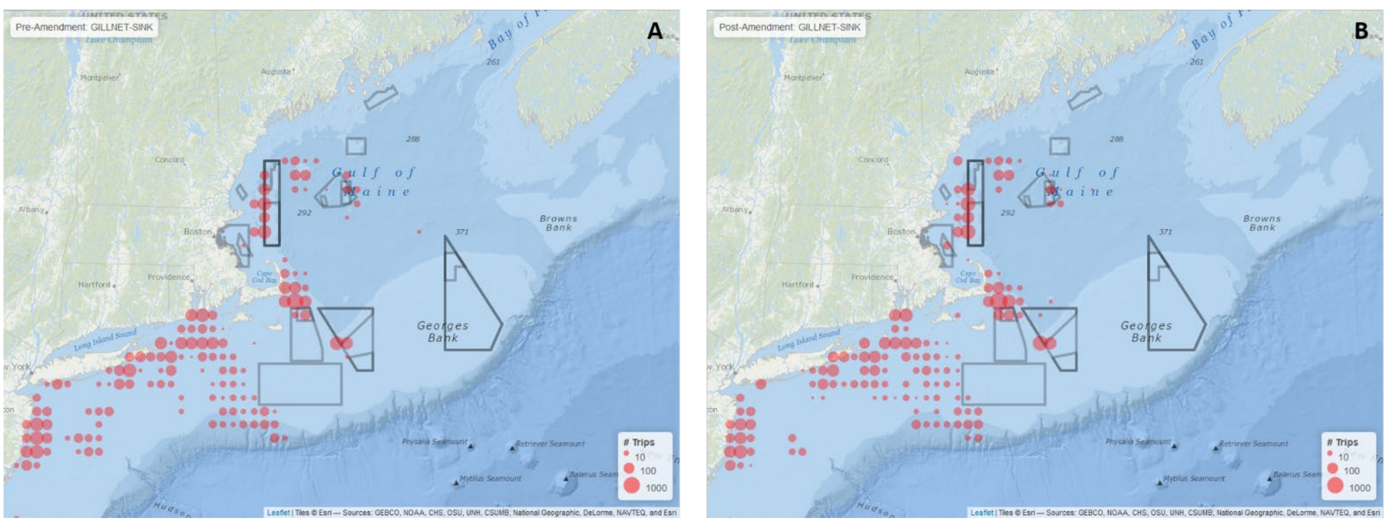

Figure 60: Sink gillnet VTRs pre-and post- Habitat Amendment. Figure A=Pre-Amendment (September 1, 2016-March 31, 2018) and Figure B=Post Amendment (April 1, 2018-October 31, 2019) (see Appendix 2)

Bottom trawl gear is also commonly fished on GB. Implementation of the GSC HMA through the Amendment restricted mobile bottom tending gear, such as bottom trawls, from areas on GB they could access pre-Amendment. Therefore, the Amendment's implementation of the GSC HMA and removal of the CA I Closure Areas has the potential to result in a shift in bottom trawl effort, however, relative to overall bottom trawl operating conditions prior to the Amendment, large shifts in effort are not expected. Specifically, any changes in bottom trawl fishing effort are likely to be localized, with existing bottom trawl effort redistributing into the now "opened" area, resulting in a change in the distribution of bottom trawl gear. This is supported by the VTR data (Figure 61). Prior to the Amendment, bottom trawl effort was concentrated along the northern boundaries of the CA I Closure Areas (Figure 61A), extending into the Great South Channel waters to the west of the Closure Area (i.e., area currently designated as the GSC HMA). Post-Amendment (Figure 61B), this effort shifted into the "opened" area, resulting in bottom trawl effort becoming more dispersed instead of concentrated in any one area of GB. Aside from this small, localized shift in effort, there is also no detectable shift in bottom trawl effort from other sub-regions (e.g., GOM, mid-Atlantic) post-Amendment as the VTR data (Figure 61) do not show an increased concentration of effort on GB. There is no evidence that the management measures implemented through the Habitat Amendment created an incentive to increase effort in this or other areas of GB.

Although VTR data post-Amendment (Figure 61B) captures a limited period of time, it is likely this pattern of bottom trawl fishing behavior and effort will remain similar to this in the future. However, as with other gears, the incentive to shift will be highly dependent on: (1) the status of the target stock(s) in the area; (2) competition for access to fishing grounds with vessels (e.g., scallop) that had pre-existing access to the area; and, (3) seasonal (i.e., CA I North Seasonal Spawning Closure) or year-long (i.e., GB DHRA) restrictions of when bottom trawl fishing can occur. Taking into consideration this information and the information provided in Figure 61, implementation of the Habitat Amendment resulted in small, localized shifts in bottom trawl effort on GB; however, large shifts in effort from other regions outside of GB were not apparent. In addition, any shifts in bottom trawl effort are not expected to equate to more bottom trawl gear and/or longer duration trawl tows than previously experienced in and around this area of GB.

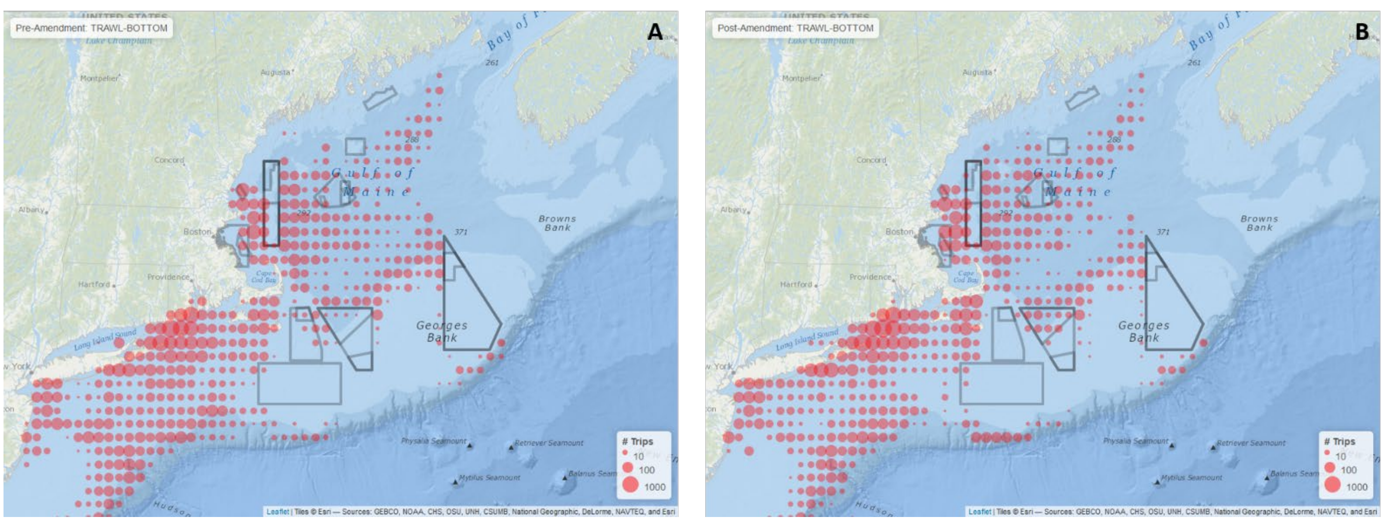

Figure 61: Bottom trawl VTR pre-and post- Habitat Amendment. Figure A=Pre-Amendment (September 1, 2016-March 31, 2018) and Figure B=Post Amendment (April 1, 2018-October 31, 2019) (see Appendix 2).

Pot/trap gear, the other gear type commonly found on GB, has never been restricted in Closed Areas designated on GB. As the GSC HMA did not restrict the use of trap/pot gear in the HMA and fishing with trap/pot gear was never restricted in the former CA I Closure Areas, it is not expected that the management measures implemented through the Amendment will result in any changes in the distribution or level (i.e., number of traps/trawls set) of trap/pot effort in this area of GB relative to pre-Amendment conditions. In addition, as the fishing grounds for trap/pot fisheries are well established on GB, to avoid introducing new gear conflicts, it is not expected that post-Amendment, trap/pot fisheries will be forced to relocate from pre-existing fishing grounds as a result of other vessels (using other gear types) responding to the management measures implemented by the Amendment. Similarly, given the established fishing grounds of trap/pot fisheries on GB, combined with the desire to avoid gear conflicts both within and outside of the fishery, any post-Amendment changes to habitat or groundfish management areas (i.e., opening or closing an area to bottom tending gear) are not expected to create incentive for large shifts in trap/pot effort within or around these affected areas, or on GB overall. Should any shifts in pot trap effort occur post-Amendment, it is likely to be seen by those trap/pot vessels whose fishing grounds overlap with or are in close proximity to the management areas delineated or modified by the Amendment. Given this, any shifts in trap/pot effort are expected to be localized and not a result of a vessel attempting to gain access to an area that it previously had not fish.

To summarize, given the relatively consistent distribution and level of effort seen on GB pre-and post-Amendment, there was no detectable shift in effort onto GB from other regions (e.g., GOM, Mid-Atlantic). Based on this and the information provided above, post-Amendment fishing operations on GB for most gears are likely to remain similar to pre-Amendment conditions, and therefore, the management measures implemented under the Habitat Amendment did not appear to create any incentive for effort with most gear types to increase on GB. There does appears to be a localized shift in gillnet, bottom trawl, and scallop dredge effort following the implementation of the Amendment. As described in more detail below, the increase in gillnet effort resulted from management measures other than the implementation of the Amendment, and trawl effort remained the same overall but dispersed locally. Therefore, effort across the region only appeared to increase for scallop dredge. The impacts of these changes on protected species will be considered below.

7.8.1.7. Assessment of Risks to Protected Species

In the *Status of the Species* (section 4.0), Table 42 identifies the ESA-listed species and critical habitat that occur in the action area and that may be adversely affected (e.g., there have been observed and documented interactions in the fisheries or with gear type(s) similar to those used in the fisheries) by the proposed action. Of the species identified in Table 42, ESA listed species of large whales, sea turtles, and Atlantic sturgeon are likely to occur on GB and overlap with the fisheries operating in this region. Given the distribution and migration patterns of Atlantic salmon (see *Status of the Species*), it is extremely unlikely this species will occur and overlap with fisheries operating on GB. In addition, as manta rays have only been observed as far north as New Jersey, this species is not expected to occur on GB. Based on this information and the information provided in the *Status of the Species*, the following evaluation will only considered gear interaction risks to listed species of large whales, sea turtles, and Atlantic sturgeon. Taking into consideration this information, and our evaluation of fishing effort pre-and post-Amendment (see above), we evaluate how these risks may have changed with the implementation of the Amendment.

Large Whales on GB: Occurrence and Gear Interaction Risks
As provided in *Status of the Species*, North Atlantic right, fin, sei, and sperm whales are likely to occur on and around GB. North Atlantic right, fin, sei and sperm whales are likely to overlap

with fisheries operating on GB; however, not all fisheries operating in this region pose an entanglement risk to these species. As described above, the greatest entanglement risk to large whales is posed by fixed gear used in trap/pot or sink gillnet fisheries (see section 7).

Observed or documented interactions between ESA-listed large whales and other gear types fished on GB, such as bottom trawl and scallop dredge are non-existent (Henry et al. 2017, Henry et al. 2015, 2016, Henry et al. 2020, Henry et al. 2019). Based on this, interactions between these gear types and North Atlantic right, fin, sei, and sperm whales are extremely unlikely to occur.

Sea turtles on GB: Occurrence and Gear Interaction Risks
As described in the *Status of the Species*, sea turtles arrive in this region in the summer (around June) and leave the area in the fall (generally by the end of November). Of the gear types fished on GB, gillnet, trap/pot, scallop dredge, and/or bottom trawl gear pose the greatest interaction risk to sea turtles (Murray 2011, 2015a, b, 2018, 2020, Murray and Orphanides 2013, Warden 2011a, b), NEFSC observer/sea sampling database, unpublished data; STDN, unpublished data). Although sea turtles are at risk of interacting with these gears, relative to the Mid-Atlantic, observed or reported fishery interactions in this region are infrequent (Murray 2011, 2015a, b, 2018, 2020, Murray and Orphanides 2013, Warden 2011a, b), NEFSC observer/sea sampling database, unpublished data; STDN, unpublished data). The infrequency of observed or reported sea turtle interactions on GB may, in part, be due to the degree of overlap between sea turtles and fisheries operating in this sub-region. Encounter rates of hard-shelled species of sea turtles are higher in the Mid-Atlantic relative to the GOM and GB (Murray 2018, 2020).

Atlantic sturgeon on GB: Occurrence and Gear Interaction Risks
As provided in the *Status of the Species*, Atlantic sturgeon (adult and sub-adult) from any DPS have the potential to occur on GB. However, the level of overlap between Atlantic sturgeon and fisheries operating in this region is likely low given that the species is more likely to occur inshore of the 164 ft (50 m) depth contour. Of the gear types used on GB, gillnet and bottom trawl gear pose the greatest interaction risk to this species. Numerous interactions between Atlantic sturgeon and gillnet or bottom trawl gear have been observed in the Northwest Atlantic (NEFSC observer/sea sampling database, unpublished data). However, relative to sub-regions such as the GOM or SNE, observed interactions between Atlantic sturgeon and gillnet or bottom trawl gear on GB are infrequent (NEFSC observer/sea sampling database, unpublished data); this is likely due to Atlantic sturgeon's preference for waters inshore of the 164 ft (50 m) depth contour. Observed or reported interactions between Atlantic sturgeon and other gear types fished on GB (i.e., trap/pot, scallop dredge) are rare to non-existent (NEFSC observer/sea sampling database, unpublished data); based on this, interactions between these gear types and Atlantic sturgeon are extremely unlikely.

Summary of Co-occurrence and Gears that Pose a Risk to Listed Species
Given the information above, ESA-listed species of large whales, sea turtles, and Atlantic sturgeon are at risk of interacting with several gear types fished on GB, with some gear types posing more or less of an interaction risk than others. Table 79 provides a summary of those gear types that pose the greatest entanglement risk to listed species of large whales, sea turtles, and Atlantic sturgeon.

Table 79: Fishing gear types that pose the greatest entanglement risk to ESA-listed species of large whales, sea turtles, and Atlantic sturgeon

Listed Species Group	Gear Types
Large whales	Sink Gillnet; Pot/Trap
Sea turtles	Sink Gillnet; Bottom Trawl; Scallop Dredge; Pot/Trap
Atlantic sturgeon	Sink Gillnet; Bottom Trawl

7.8.1.8. Assessment of Risk with the Implementation of the Habitat Amendment

As described above, there is no evidence that the Habitat Amendment changed fishing behavior or effort for trap/pot gear fished on GB. Relative to operating conditions on GB prior to the Amendment, there is little to no change in the area fished or the quantity of fishing gear set on GB post-Amendment. Species occurrence and distribution may respond and adapt to changes in the marine environment (e.g., changes to prey availability, water temperature changes); however, the overall occurrence and broad scale distribution of these species on GB has remained relatively consistent from pre-Amendment to post-Amendment. As the degree of overlap between listed species and these gear types on GB in future fishing years is expected to remain similar to the overlap prior to the Amendment, pre- or- post Amendment, the likelihood of an interaction occurring in any area of GB is expected to be the same.

Post-Amendment, fishing data did show a change in bottom trawl fishing effort on GB. Bottom trawl fisheries on GB overlapped with ESA-listed species prior to the Amendment, posing a level of interaction risk to these species. The changes in effort observed post-Amendment, therefore, have the potential to modify the preexisting level of risk. Specifically, listed species potentially affected by any post-Amendment changes in bottom trawl effort on GB are sea turtles and Atlantic sturgeon (Table 79). As described above, bottom trawl gear became more dispersed post-Amendment. While there were localized shifts in bottom trawl effort, overall bottom trawl effort on GB did not increase. Given that bottom trawl vessels are fishing the same general area as they had pre-Amendment, changes in bottom trawl operations (i.e., increase the amount of bottom trawl gear fished or increase the gear tow duration), due to implementation of the Amendment, are not expected. As described above, the overall occurrence and broad scale distribution of listed species of sea turtles and Atlantic sturgeon on GB has remained relatively consistent from pre-Amendment to post-Amendment. Therefore, the degree of overlap between listed species and bottom trawl gear in future fishing years is expected to remain similar to the overlap prior to the Amendment. While these species may move between areas of Georges Bank, the interaction risk is relatively constant across the area. Therefore, the local movements do not change the likelihood of an interaction.

Post-Amendment, fishing data showed that, although the overall magnitude of gillnet effort did not change on GB, there was a localized increase in gillnet effort in the waters off Chatham, Massachusetts. This area off Chatham is an established gillnet fishing ground for numerous fisheries targeting stocks such as monkfish, skate, and spiny dogfish. Vessels fishing with gillnet gear on GB overlapped with ESA-listed species prior to the Amendment, posing a level of interaction risk to these species. The changes in effort observed post-Amendment, therefore, have the potential to modify this pre-existing level of risk. Specifically, listed species potentially affected by any post-Amendment changes in gillnet effort on GB are large whales, sea turtles, and Atlantic sturgeon (Table 79). With respect to the removal of the CA I Closure Area and the implementation of the GSC HMA, the review of fishing data showed no indication that these changes caused a shift in gillnet effort, locally or regionally, and the relative distribution and overall magnitude of gillnet effort remained similar to pre-Amendment conditions. The localized increase in gillnet effort detected in the VTR data post-Amendment occurred in an area of GB that had never been closed to fishing, and in an area that has been, and continues to be, predominantly fished with gillnet gear. Given these factors, the cause for the localized increase in effort is not expected to be due to the measures implemented under the Amendment; this determination was further confirmed by policy analysts with NMFS Sustainable Fisheries Division. Between 2017 and 2018, numerous fishery management actions were authorized and implemented. Many of these actions (e.g., extra-large mesh gillnet sector exemptions from the Multispecies FMP, modifying the management uncertainty buffer in the Monkfish FMP, inclusion of a new stock in the Skate FMP) provided gillnet fishermen more opportunity to target and land managed fish stocks, specifically monkfish, skate, and dogfish, whose total stock biomass was high. As monkfish, skate, and spiny dogfish commonly occur in the waters off Chatham, Massachusetts, these waters are important fishing grounds for vessels operating out of nearby ports. Given this, it is likely that the fishery management regulations implemented between 2017 and 2018 provided incentive for existing vessels to take additional gillnet trips in this area of GB given the increased opportunity to land, rather than discard, these targeted species. Although there was an apparent increase in effort in this area of GB, given the increased biomass of these managed fish stocks and increased opportunity to land these species, it is likely that fishing efficiency increased in these waters. Rather than reflecting an increase in gear in the water, an increase in fishing efficiency can indicate a decrease in gillnet soak time as catch per unit effort (CPUE) increases with the increase in stock biomass. This, in turn, can result in an increase in trips taken to the fishing grounds to tend (haul back) gear and land fish; this is supported by the VTR data reviewed over this timeframe. An increase in CPUE and a decrease in gillnet soak duration reduces the potential for sea turtles, large whales, or Atlantic sturgeon to encounter and interact with this gear type. Given the fishery management measures taken in 2017 and 2018, the localized increase in gillnet effort on GB would have occurred regardless of the measures implemented under the Amendment.

While there were localized shifts gillnet effort post-Amendment, overall gillnet effort on GB did not increase as a result of the management measures implemented under the Habitat Amendment. The overall distribution of gillnet gear remained relatively the same, with small (insignificant) shifts in or around pre-Amendment fishing grounds on GB. Given that gillnet vessels are fishing the same general area as they had pre-Amendment, changes in gillnet operations (i.e., increase the amount of gillnet gear fished) or increases in the gear soak duration due to implementation of the Amendment are not expected. As described above, the overall occurrence and broad scale distribution of listed species of large whales, sea turtles, and Atlantic salmon on GB has remained relatively consistent from pre-Amendment to post-Amendment. Given the above, the degree of overlap between listed species and gillnet gear in future fishing years is expected to remain similar to the overlap prior to the Amendment. While these species may move between areas of Georges Bank, the interaction risk is likely relatively constant across the area. Therefore, the local movements do not change the likelihood of interactions.

Post-Amendment, fishing data also showed a localized change in scallop dredge effort on GB. Scallop vessels fishing with dredge gear overlapped with ESA-listed species prior to the

Amendment, posing a level of interaction risk to these species. The changes in effort observed post-Amendment, therefore, have the potential to modify this preexisting level of risk. Specifically, listed species potentially affected by any post-Amendment changes in scallop dredge effort on GB are sea turtles and Atlantic sturgeon, the only listed species known to interact with this gear type. In addition, there appeared to be an increase in the number of trips in the area formerly designated as the CA I North Habitat Closure (currently designated as the CA I Access Area under the Scallop FMP). Given the duration of time this area was closed to the scallop fishery pre-Amendment, a healthy biomass of scallops became established in the former CA I North Habitat Closed Area. Once opened via the scallop rotational area management program, vessels redirected effort into the area to harvest these scallops. With a high biomass of harvestable scallops, there is likely to be a decrease in a scallop dredge vessel's area swept as the catch per unit effort is higher. A decrease in area swept equates to scallop dredge gear being present for less time in the water/on the bottom, which in turn, reduces the potential for sea turtles or Atlantic sturgeon to encounter and interact with this gear type. As described above, the shift in effort post- Amendment was local with pre-existing effort located outside the boundaries of the former CA I Closure Area shifting inside the former Closure Areas. These are areas with similar likelihood of interaction. As a result, the observed shift is not expected to result in changes in encounter rates between listed species and gear, and therefore, sea turtle or Atlantic sturgeon bycatch on GB are likely to remain similar pre- and post-Amendment. In addition, species occurrence and distribution may respond and adapt to changes in the marine environment (e.g., changes to prey availability, water temperature changes); however, the overall occurrence and broad scale distribution of these species on GB has remained relatively consistent from pre-Amendment to post-Amendment. Given the above, the degree of overlap between listed species and gear in future fishing years is expected to remain similar to the overlap prior to the Amendment.

7.8.1.9. Measures in Southern New England (SNE)

Prior to the Habitat Amendment, the following Habitat and/or Groundfish Closure Areas existed in SNE; this information will serve to establish pre-existing fishing behavior and effort in SNE, particularly in and around these Closure Areas.

Nantucket Lightship (NLS) Habitat and Groundfish Closure Areas
Prior to the Amendment, the NLS Habitat and Groundfish Closure Areas were located south of Nantucket, MA, with a portion of the NLS Habitat Closure Area occurring within the boundaries of the NLS Groundfish Closure Area (Figure 62). Prior to the Amendment, the following management measures applied:

- NLS Habitat Closure Area: Bottom trawl, scallop dredge, and clam dredge gears prohibited. Pot/trap, sink gillnet, mid-water trawl, purse seine, and bottom longline gear permitted.
- NLS Groundfish Closure Area: Pot/trap, purse seine, clam dredge, and mid-water trawl gears permitted. With the exception of vessels belonging to the NE Multispecies sector program, all other vessels fishing with gear types capable of catching groundfish prohibited (e.g., sink gillnet, bottom trawl, bottom longline). Through sector exemptions

authorized by NMFS since 2013[59], sink gillnet, bottom trawl, and bottom longline gears could be set in portions of the Groundfish Closed Area (outside of the area overlapping the Habitat Closure). Clam and scallop dredge gear could also operate in specified portions of the Closed Areas (i.e., outside of the area overlapping the Habitat Closure).

Prior to the Amendment, scallop dredge, bottom trawl, sink gillnet, and trap/pot gears were commonly used in areas of SNE. Scallop dredge effort was concentrated within the Northeast corner of the NLS Groundfish Closure Area (which is a scallop Access Area) and in waters north of the NLS Closure Areas (i.e., Great South Channel, waters off the western boundary of CA I Closure Areas). Bottom trawl effort was common in waters off Rhode Island, southern Massachusetts (e.g., Nantucket Sound, waters south of Martha's Vineyard and Nantucket), and in waters to the west and south (i.e., along the continental shelf edge) of the Closure Areas; some effort was located along the Northwest corner of the Closure Areas. Sink gillnet effort was common along the western and south-western boundary of the NLS Groundfish Closed Area. In addition, through sector exemptions (see above), limited sink gillnet effort occurred within the boundaries of the NLS Groundfish Closed Area prior to the Amendment. Pot/trap effort was common in this area of SNE (both within and outside of the Closure Areas), with the number of traps per square mile in this area of SNE estimated to be approximately ≤10 traps per square mile (Fig. 4.1.3.a in NMFS 2020b).

Relative to pre-Amendment conditions in this area of SNE, the Amendment implemented the following management measures (Figure 62).

- Removed the NLS Habitat and Groundfish Closure Areas.[60]

[59] For sector exemptions authorized since 2013, see: 78 FR 25591, May 2, 2013; 78 FR 53363, August 29, 2013; 78 FR 76077, December 16, 2013; 79 FR 22043, April 21, 2014; 79 FR 23278, April 28, 2014; 80 FR 25143, May 1, 2015; 82 FR 39363, August 18, 2017 ; 82 FR 19618, April 28, 2017; 83 FR 18965, May 1, 2018; 83 FR 34492, July 20, 2018; 84 FR 17916, April 26, 2019; 85 FR 23229, April 27, 2020.

[60] Pursuant to a __________, NMFS issued a rule (84 FR 68798, December 17, 2019) suspending the Amendment's opening of the NLS Groundfish Closure Area to gillnet fishing and restoring prior regulations prohibiting gillnet gear from fishing in this area until further notice

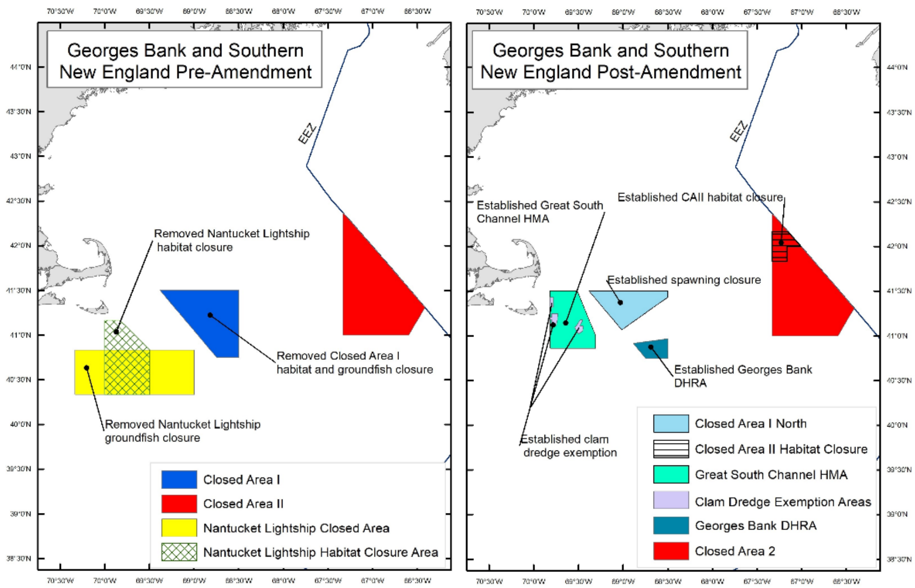

Figure 62: Habitat and groundfish management Areas in SNE (and on GB) pre- and post-Habitat Amendment

In addition, subsequent to changes in SNE from the Habitat Amendment, the Scallop FMP, adjusted and/or expanded the boundaries of the NLS Access Area post-Amendment (83 FR 17300, April 19, 2018; 84 FR 11436, March 27, 2019; 85 FR 17754, March 31, 2020). However, to date, the overall footprint of the Access Area has remained relatively the same. Relative to pre-existing operating conditions, there is the potential that the management measures implemented through the Amendment (and further changed by the Scallop FMP) may change fishing effort/behavior in this area of SNE. Figures provided in Appendix 2 depict the distribution of trips using sink gillnet, bottom trawl, and scallop dredge gear in SNE pre- and post- Amendment.

The removal of the NLS Habitat and Groundfish Closure Areas post-Amendment resulted in a shift of scallop dredge effort into the newly opened areas. VTR data (Figure 63) suggests that, with this shift, there was also an increase in effort, particularly in the former NLS Habitat Closure Area. As scallop dredge vessels were previously restricted from accessing the NLS Habitat Closure Area, the opening post-Amendment created an incentive for scallop dredge vessels to redirect effort into formerly closed areas. This opening; however, is not unrestricted as the level of scallop dredge effort is constrained by the Scallop FMP's rotational management program (e.g., expanding, and opening or closing an Access Area)[61].

In the pre-Amendment maps, the NLS Access Area was opened to scallop fishing. This, combined with the gear prohibitions in the NLS Habitat and Groundfish Closure Areas, explains the distribution of scallop dredge effort in and around the boundaries of these management areas

[61] Between 2016 and 2019, the Scallop FMP's NLS Access Area was closed to scallop fishing in 2016 (80 FR 22119, April 21, 2015) and opened to scallop fishing in 2017 (82 FR 15155, March 27, 2017), 2018 (83 FR 17300, April 19, 2018), and 2019 (84 FR 11436, March 27, 2019).

over this timeframe. The combination of opening the areas and the subsequent scallop actions likely resulted in the shift in scallop dredge effort to this area of SNE post-Amendment (Figure 63B). Given the proximity of the newly established GSC HMA to SNE, there may also be some localized shifts in effort from the GSC HMA to areas south of this HMA where there are pre-existing fishing grounds located around and, when open under the scallop rotational management program, in the NLS-Access Area; this is supported by maps provided in Figure 63. While there was an increase in effort in the area formerly designated as the NLS Groundfish Closure Area post-Amendment, there was no apparent increase in overall scallop dredge effort in SNE. Below, the analysis assesses whether this change in effort subsequent to the Habitat Amendment resulted in increased risks to protected species.

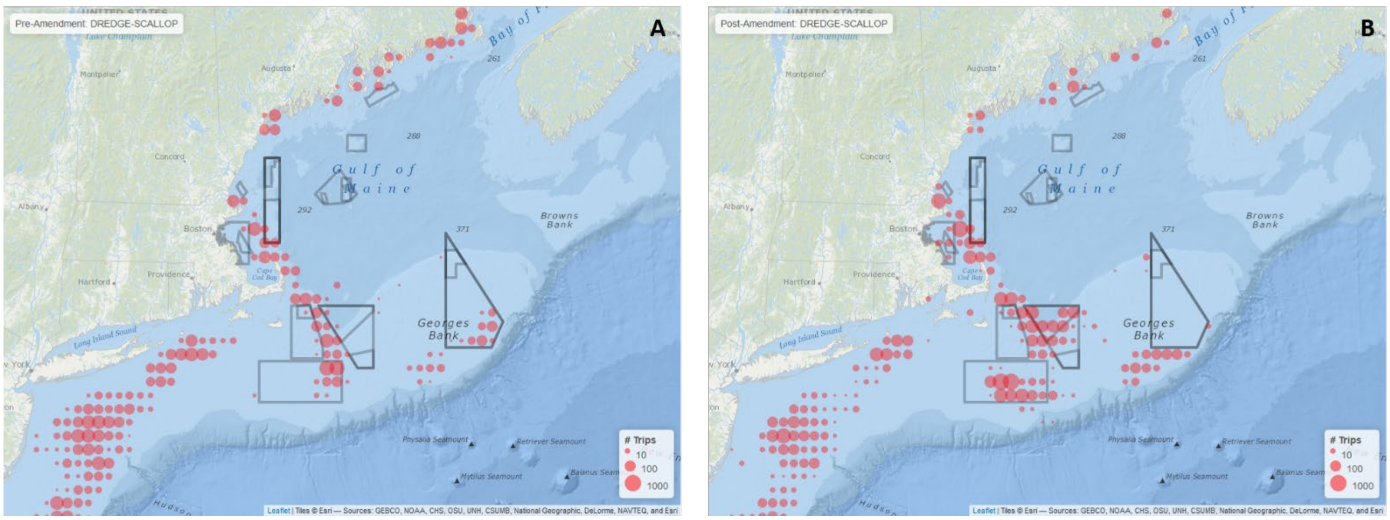

Figure 63: Scallop dredge VTRs pre-and post- Habitat Amendment. Figure A=Pre-Amendment (September 1, 2016-March 31, 2018) and Figure B=Post Amendment (April 1, 2018-October 31, 2019) (see Appendix 2)[62]

Prior to the Amendment, gillnet gear was commonly set along the western and southwestern boundary of the NLS Groundfish Closure Area. There were also exemptions to the NLS Groundfish Closure Area that allowed gillnet vessels, through sector exemptions, to fish in these areas prior to the Amendment. While the Amendment's removal of the NLS Closure Areas has the potential to result in a shift in gillnet effort, any shift is likely to be minor relative to overall gillnet operating conditions prior to the Amendment. Specifically, any changes in gillnet fishing effort are likely to be localized, with existing gillnet effort redistributing into the now "opened" area, resulting in a change in the distribution of gillnet gear. However, the incentive to shift effort will be highly dependent on: (1) the status of the target stock(s) in the area; and (2) competition for access to fishing grounds with vessels (e.g., scallop, trap/pot) that have pre-existing fishing access to the area. This is supported by the VTR effort data (Figure 64). Based

[62] This figure does not depict the boundaries of the Scallop FMP's Rotational Management Areas. The boundaries delineating the (prior) NLS Habitat and Groundfish Management Area in Figure 63Figure 63 are, in general, reflective of the area encompassed by the NLS Scallop Rotational Area. For details on the Scallop FMP's Rotational Management Area, see https://www.fisheries.noaa.gov/resource/map/atlantic-sea-scallop-managed-waters-fishing-year-2020.

on these maps, there is little difference in the distribution or magnitude of sink gillnet effort in SNE. There was no large shift of gillnet effort from within or outside of the region into the area formerly designated as the NLS Habitat and Groundfish Closure Areas. If a large shift in gillnet effort had occurred post-Amendment, it is likely that gillnet gear would be more concentrated in SNE, particularly in and around the former NLS Closure Areas. As provided in Figure 64, this did not occur. Based on this, the removal of the NLS Closure Areas provided little incentive to change gillnet effort in SNE. Although the effort post-Amendment (Figure 64B) captures a limited period of time, it is likely the pattern of fishing behavior and effort shown in the map is reflective of future effort in the area. Taking into consideration this information and the information provided in Figure 64, implementation of the Habitat Amendment resulted in small, localized shifts in gillnet effort in SNE; large shifts in effort from other regions outside of SNE were not apparent.

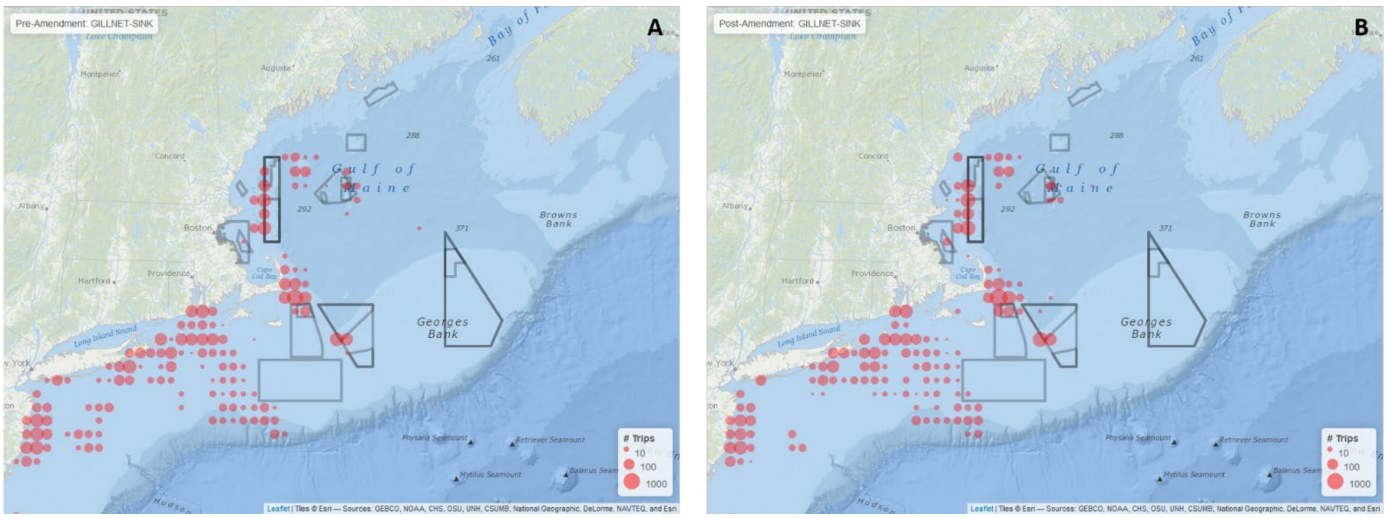

Figure 64: Sink gillnet VTRs pre-and post-Habitat Amendment. Figure A=Pre-Amendment (September 1, 2016-March 31, 2018) and Figure B=Post Amendment (April 1, 2018-October 31, 2019) (see Appendix 2)

Prior to the Amendment, while there was some bottom trawl effort along the Northwest corner of the Closure Area (Appendix 2), most effort appeared to be concentrated in waters off Rhode Island, southern Massachusetts (e.g., Nantucket Sound, waters south of Martha's Vineyard and Nantucket) and to waters west and south (i.e., along the continental shelf edge) of the NLS Closure Areas. While the Amendment's removal of the NLS Closure Areas has the potential to result in a shift in bottom trawl effort, any shift is likely to be minor relative to operating conditions prior to the Amendment. Specifically, changes in bottom trawl fishing effort are likely to be localized, with existing bottom trawl effort redistributing into the now "opened" area, resulting in a change in the distribution, but not quantity, of bottom trawl gear towed in SNE. However, the incentive to shift effort is highly dependent on: (1) the status of the target stock(s) in the area; and (2) competition for access to fishing grounds with vessels (e.g., scallop, trap/pot) that fished the area pre-Amendment. This is supported by the VTR data (Figure 65) which show little difference, pre- and post-Amendment, in the distribution or level of bottom trawl effort in SNE, specifically within the area formerly designated as the NLS Closure Areas. Based on this, the removal of the NLS Closure Areas provided little incentive to change bottom trawl effort in SNE. Although the post-Amendment effort (Figure 65B) captures a limited period of time, it is likely the pattern of fishing behavior and effort shown in the map is reflective of future effort in the area. Although a limited time was available for this analysis, it is long enough to detect even modest shifts in effect as described above for effort off Chatham. Taking into consideration this information and the information provided in Figure 65, implementation of the Habitat Amendment resulted in small, localized shifts in bottom trawl effort in SNE; however, large shifts in effort from other regions outside of SNE were not apparent.

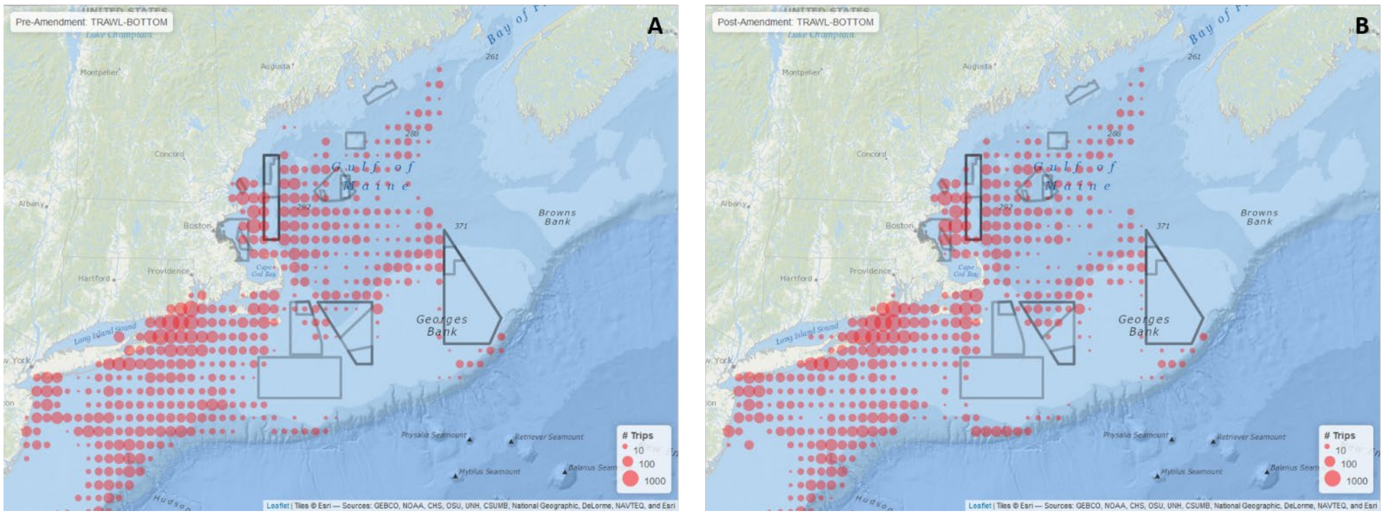

Figure 65: Bottom trawl VTRs pre-and post- Habitat Amendment. Figure A=Pre-Amendment (September 1, 2016-March 31, 2018) and Figure B=Post Amendment (April 1, 2018-October 31, 2019)

Pot/trap gear, the other gear type commonly found in SNE, has never been restricted in the area previously designated as the NLS Habitat and Groundfish Closure Areas. Trap/pot fisheries have operated throughout the region both inside and outside of the Closed Areas. Given that access to areas for this gear has not changed, the management measures implemented through the Amendment are not expected to provide incentive for the distribution or level (i.e., number of traps/trawls set) of trap/pot effort in this area of SNE to change relative pre-Amendment conditions. Specifically, as the fishing grounds for trap/pot fisheries are well established in SNE, post-Amendment, trap/pot fisheries are not expected to be forced to relocate from pre-existing fishing grounds as a result of other vessels (using other gear types) responding to the management measures implemented by the Amendment. Similarly, given the established fishing grounds of trap/pot fisheries in SNE, combined with the desire to avoid gear conflicts both within and outside of the fishery, any post-Amendment changes to habitat or groundfish management areas (i.e., opening or closing an area to bottom tending gear) are not expected to create incentive for large shifts in trap/pot effort within or around these affected areas, or in SNE overall. Should any shifts in pot trap effort occur post-Amendment, it is likely to be seen by those trap/pot vessels whose fishing grounds overlap with or are in close proximity to the management areas delineated or modified by the Amendment. Given this, any shifts in trap/pot effort are expected to be localized and not a result of a vessel attempting to gain access to an area that it previously had not fish.

In conclusion, small, localized changes in effort were seen for bottom trawl, scallop dredge, and gillnet gear, and risk to ESA-listed species from these changes will be described below. Fishing in SNE with all other gear types remained similar to pre-Amendment conditions; for these gear types, any changes in area fished, quantity of gear set/towed, and/or placement of fishing gear in SNE are not expected. There was also no detectable shift in effort into SNE from other surrounding sub-regions (e.g., GOM, Mid-Atlantic) for any gear type as a result of the management measures implemented under the Amendment.

7.8.1.10. Assessment of Risks to Protected Species

In the *Status of the Species* (see section 4), Table 42 identifies the ESA-listed species and critical habitat that occur in the action area and that may be adversely affected (e.g., there have been observed and documented interactions in the fisheries or with gear type(s) similar to those used in the fisheries) by the proposed action. Of the species identified in Table 42, ESA listed species of large whales, sea turtles, and Atlantic sturgeon are likely to occur in SNE and overlap with the fisheries operating in this region. Given the distribution and migration patterns of Atlantic salmon (see *Status of the Species*), it is extremely unlikely this species will occur and overlap with fisheries operating in SNE. In addition, as manta rays have only been observed as far north as New Jersey, this species is not expected to occur in SNE. Based on this information and the information provided in the *Status of the Species*, the following evaluation will only considered gear interaction risks to listed species of large whales, sea turtles, and Atlantic sturgeon. Taking into consideration this information, and our evaluation of fishing effort pre-and post-Amendment (see above), we then evaluate how these risks may have changed with the implementation of the Amendment.

Large Whales in SNE: Occurrence and Gear Interaction Risks
As provided in the *Status of the Species*, ESA-listed species of North Atlantic right, fin, sei, and sperm whales are likely to occur in SNE. North Atlantic right and fin whales, accounting for some seasonal variability, can consistently be found throughout this region (Hayes et al. 2019 Hayes et al. 2020, NOAA Right Whale Advisory Sighting System[63]; OBIS-SEAMAP[64]; WhaleMap[65]). For North Atlantic right whales, a review of recent opportunistic sighting, acoustic, and aerial survey data indicate a high use area in SNE, specifically, the region south of Martha's Vineyard and Nantucket Islands (Davis et al. 2017, Hayes et al. 2020, Hayes et al. 2019, Leiter et al. 2017), NOAA Right Whale Advisory Sighting System, WhaleMap). Sei and sperm whales commonly occur in the deep waters off of SNE, predominantly in areas along the shelf edge, shelf break, and/or in ocean basins between banks; however, depending on season, incursions onto the continental shelf of SNE do occur, primarily when there is suitable prey availability (Hayes et al. 2020), OBIS-SEAMAP).

Given the above information, North Atlantic right, fin, sei and sperm whales are likely to overlap with fisheries operating in SNE; however, not all fisheries operating in this region pose an entanglement risk to these species. The greatest entanglement risk to large whales is posed by fixed gear used in trap/pot or sink gillnet fisheries. Fin, sei, and North Atlantic right whales have died or been seriously injured from entanglement in fishing gear along the Gulf of Mexico Coast, U.S. East Coast, and Atlantic Canadian Provinces from 2012 to 2018; however, there have been no confirmed M/SI from entanglement in fishing gear for sperm whales since 2011[66] (Hayes et

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[63] https://fish.nefsc.noaa.gov/psb/surveys/MapperiframeWithText.htm
[64] http://seamap.env.duke.edu/
[65] https://whalemap.ocean.dal.ca/WhaleMap/
[66] A sperm whale was reported entangled in monkfish net on the Canadian Grand Banks in 2011, but was released alive and gear free Ledwell and Huntington 2012 as cited in Waring et al. 2015.

al. 2020, Hayes et al. 2019, Henry et al. 2020, Henry et al. 2019, Waring et al. 2015, Waring et al. 2014).

There have been no observed or documented interactions between large whales and other gear types fished in SNE, such as bottom trawl and scallop dredge (Henry et al. 2017, Henry et al. 2015, 2016, Henry et al. 2020, Henry et al. 2019), see also NMFS' Marine Mammal Stock Assessment Reports: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessment-reports-region). Based on this, interactions between these gear types and North Atlantic right, fin, sei, and sperm whales are extremely unlikely.

Sea turtles in SNE: Occurrence and Gear Interaction Risks
As described in the *Status of the Species*, sea turtles occur in continental shelf waters of SNE, arriving in this region between late spring and summer (May to June) and leave the area in the fall (approximately by the end of November). Gillnet, scallop dredge, and bottom trawl fisheries operating in SNE use gear types known to pose an interaction risk to sea turtles, and interactions with these gear types have been observed (NEFSC observer/sea sampling database, unpublished data). There have been no observed or documented interactions between leatherback sea turtles and scallop dredge gear in this sub-region (NEFSC observer/sea sampling database, unpublished data). Based on information provided by the GAR STDN (unpublished data), from 2009 through 2018, leatherback sea turtle have been reported entangled in vertical line gear, with multiple cases reported in SNE. In addition, based on data provided by the GAR STDN (unpublished data), hard-shelled sea turtles can become entangled in the vertical lines associated with trap/pot gear, with only one case reported in SNE.

Atlantic sturgeon in SNE: Occurrence and Gear Interaction Risks
As provided in *Status of the Species*, Atlantic sturgeon (adult and sub-adult), from any DPS, are likely to occur in SNE, primarily inshore of the 164-ft (50 m) depth contour. (Dunton et al. 2010, Laney et al. 2007, Stein et al. 2004a, b, Waldman et al. 2013, Wirgin et al. 2015b). However, excursions into deeper (e.g., 246 ft (75 m)) continental shelf waters have been documented (Collins and Smith 1997, Dunton et al. 2010, Erickson et al. 2011, Stein et al. 2004a, b, Timoshkin 1968). Of the gear types used in SNE, gillnet and bottom trawl gear pose the greatest interaction risk to this species. Numerous interactions between Atlantic sturgeon and gillnet or bottom trawl gear have been observed in the Northwest Atlantic, with many of these interactions observed in SNE (NEFSC observer/sea sampling database, unpublished data). Observed or reported interactions between Atlantic sturgeon and other gear types fished in SNE (i.e., trap/pot and scallop dredge) are rare to non-existent (NEFSC observer/sea sampling database, unpublished data); based on this, interactions between these gear types and Atlantic sturgeon are extremely unlikely.

Summary of Co-occurrence and Gears that Pose a Risk to Listed Species
Given the information above, ESA-listed species of large whales, sea turtles, and Atlantic sturgeon are at risk of interacting with several gear types fished in SNE, with some gear types posing more or less of an interaction risk than others. Table 80 provides a summary of those gear types that pose the greatest entanglement risk to listed species of large whales, sea turtles, and Atlantic sturgeon.

Table 80: Fishing gear types that pose the greatest entanglement risk to ESA-listed species of large whales, sea turtles, and Atlantic sturgeon.

Listed Species Group	Gear Types
Large whales	Sink Gillnet; Pot/Trap
Sea turtles	Sink Gillnet; Bottom Trawl; Scallop Dredge Pot/Trap
Atlantic sturgeon	Sink Gillnet; Bottom Trawl

7.8.2. *Assessment of Risk with the Implementation of the Habitat Amendment*

As described above, there is no evidence that the Habitat Amendment changed fishing behavior or effort for trap/pot gear fished in SNE. Relative to operating conditions in SNE prior to the Amendment, there is little to no change in the area fished or the quantity of fishing gear set or towed in SNE post-Amendment. Species occurrence and distribution may respond and adapt to changes in the marine environment (e.g., changes to prey availability, water temperature changes); however, the overall occurrence and broad scale distribution of these species in SNE has remained relatively consistent from pre-Amendment to post-Amendment. As the degree of overlap between listed species and trap/pot gear in SNE in future fishing years is expected to remain similar to the overlap prior to the Amendment, pre- or- post Amendment, the likelihood of an interaction occurring in any area of SNE is expected to be the same.

Post-Amendment fishing data did show some change in bottom trawl and gillnet fishing effort in SNE. Bottom trawl and gillnet fisheries in SNE overlapped with ESA-listed species prior to the Amendment, posing a level of interaction risk to these species. The changes in effort observed post-Amendment, therefore, have the potential to modify this pre-existing level of risk. Specifically, listed species potentially affected by any post-Amendment changes in bottom trawl effort in SNE are sea turtles and Atlantic sturgeon; for gillnet gear, the listed species potentially affected are sea turtles, Atlantic sturgeon, and large whales (Table 80). While there were localized shifts in bottom trawl and gillnet effort post-Amendment, overall bottom trawl and gillnet effort in SNE did not increase. Bottom trawl gear became more dispersed post Amendment. The overall distribution of gillnet gear remained relatively the same, with small (insignificant) shifts in or around pre-Amendment fishing grounds in SNE. Given that vessels are fishing the same general area as they had pre-Amendment, changes in bottom trawl or gillnet operations (i.e., increase the amount of gillnet or bottom trawl gear fished) or increases in the gear soak or tow duration due to implementation of the Amendment are not expected. In addition, the overall occurrence and broad scale distribution of these species in SNE has remained relatively consistent from pre-Amendment to post-Amendment. Given the above, the degree of overlap between listed species and gear in future fishing years is expected to remain similar to the overlap prior to the Amendment. While these species may move between areas of SNE, the interaction risk is relatively constant across the area.

Post-Amendment fishing data also showed a change in scallop dredge effort in SNE. Prior to the Amendment, scallop vessels fishing with scallop dredge gear posed a level of interaction risk to listed species that overlapped with the scallop fishery. The changes in effort observed post-Amendment, therefore, have the potential to modify this pre-existing level of risk. Specifically, listed species potentially affected by any post-Amendment changes in scallop dredge effort in SNE are sea turtles and Atlantic sturgeon, the only listed species known to interact with this gear type. Review of fishing data showed localized shifts in scallop dredge effort post Amendment; however, with this shift, there also appeared to be an increase in effort in the area formerly designated as the NLS Habitat Closure, and currently designated as the NLS Access Area under the Scallop FMP. Effort in this area is not unrestricted as accessibility is limited by the Scallop FMP's rotational management program. Given the duration of time this area was closed to the scallop fishery pre-Amendment, a healthy biomass of scallops became established in the former NLS Habitat Closed Area. Once opened, vessels redirected effort into the area to harvest these harvestable scallops. With a high biomass of harvestable scallops, there is likely to be a decrease in a scallop dredge vessel's area swept. A decrease in area swept equates to scallop dredge gear being present for less time in the water/on the bottom, which in turn, equates to a reduction in the potential for sea turtles or Atlantic sturgeon to interact with this gear type. In addition, as described above, the shift in effort post- Amendment was local; that is, pre-existing effort located outside the boundaries of the former NLS Habitat Closed Area, now shifted inside the former Closed Area. As a result, the observed shift is not expected to result in changes in encounter rates between listed species and gear, and therefore, sea turtle or Atlantic sturgeon bycatch in SNE are likely to remain similar pre- and post-Amendment. In addition, species occurrence and distribution may respond and adapt to changes in the marine environment (e.g., changes to prey availability, water temperature changes); however, the overall occurrence and broad scale distribution of these species in SNE has remained relatively consistent from pre-Amendment to post-Amendment. Given the above, the degree of overlap between listed species and gear in future fishing years is expected to remain similar to the overlap prior to the Amendment.

7.8.3. *EFH and Habitat Areas of Particular Concern*

The Magnuson-Stevens Act defines EFH as ''those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity.'' The Habitat Amendment designated EFH for monkfish, numerous species of skates, the Atlantic sea scallop, Atlantic herring, the deep sea red-crab, Atlantic salmon, and numerous species from the Northeast multispecies complex (NEFMC 2016b). EFH designations do not implement additional fishing restrictions on their own. The Council took a comprehensive approach to minimizing to the extent practicable the adverse effects of fishing EFH across all species and life stages, focusing on those habitats most vulnerable to fishing impacts. In designating EFH, per 50 CFR part 600 (subpart J), the Amendment described and identified habitats or habitat types determined to be EFH for each life stage of the managed species. This included providing information on the physical, biological, and chemical characteristics of the EFH and, if known, how these characteristics influence the use of EFH for the species/life stage. With this information, the Amendment identified the specific geographic location and extent of the EFH for each species.

The Amendment also identified multiple Habitat Areas of Particular Concern (HAPC). HAPCs highlight specific types or areas of habitat within EFH that are particularly vulnerable to human impacts (e.g., fishing and non-fishing impacts), and therefore, may need some manner of protection. However, the designation of a HAPC itself does not necessitate the implementation of specific management measures, such as gear restrictions, to protect the area. No management measures were implemented as part of the HAPC designations in the Amendment (NEFMC 2016b), although many of the habitat management areas described above cover HAPCs that were designated because of its vulnerability to fishing impacts.

As the designation of EFH and HAPCs in the GOM, GB, and SNE did not implement any management measures, these designation are not expected to have an effect on fisheries operating within these regions. As provided above, interaction risks with listed species are strongly associated with the quantity of gear in the water (e.g., number of vertical lines, gillnets, trawls), gear soak/tow duration, and the temporal and spatial overlap of the gear and protected species. Taking into consideration this and the information provided above, relative to operating conditions prior to the Amendment, the Habitat Amendment's designation of EFH or HAPC will not effect: (1) temporal and spatial overlap between gear and listed species of sea turtles, fish and whales; (2) quantity of gear set or towed, and/or, (3) gear soak or tow duration. Based on this, the designation of EFH or HAPC through the Habitat Amendment is not expected to result in new or elevated interaction risks to any listed species in the GOM, on GB, or in SNE.

7.8.3.1. Spawning Protection Areas

Prior to the Habitat Amendment, the following spawning protection areas existed in the GOM or on GB:

- GOM Cod Spawning Protection Area (i.e., Whaleback; Figure 66)[67]: From April 1 through June 30 of each year, this area was closed to all fishing and fishing vessels except for: (1) Vessels that did not have a federal Northeast Multispecies permit and are fishing exclusively in state waters; (2) charter and party or recreational vessels; and, (3) vessels fishing with exempted gears (e.g., pots and traps, purse seines, surfclam/quahog dredge gear, pelagic hook and line).
- GB Seasonal Spawning Closure Area (Figure 67)[68]: From May 1 to May 31, no fishing vessel or person on a fishing vessel could enter, fish, or be in the area except for vessels fishing: (1) with exempted gears (e.g., pots and traps, purse seines, mid-water trawl, surfclam/quahog dredge gear, pelagic hook and line); (2) with scallop dredge gear under a scallop DAS; (3) in the CA I Hook Gear Haddock Access Area; (4) under the restrictions and conditions of an approved NE Multispecies sector operations plan; and (5) under the provisions of a Northeast multispecies Handgear A or B permit.

[67] 76 FR 23042 (May 1, 2011) designated the GOM Cod Spawning Protection Area.
[68] 79 FR 21658 (April 24, 2000) designated the GB Seasonal Spawning Closure Area.

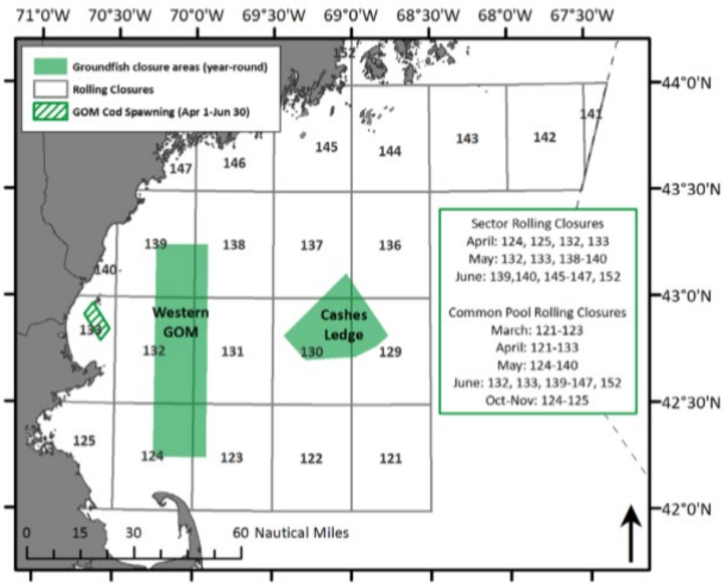

Figure 66: GOM Cod Spawning Protection Area

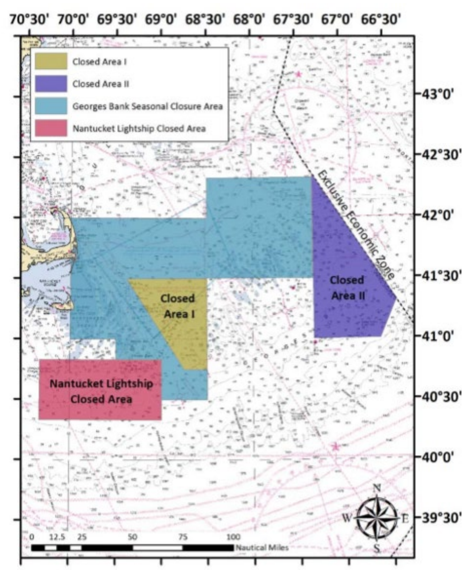

Figure 67: GB Seasonal Spawning Closure Area

Given the: (1) gear restrictions in the GOM or GB Spawning Protection/Closure Areas; (2) duration of time in which both Closed Areas have remained in place (i.e., ≥9 years); and, (3) implementation of sequential (later termed rolling) closures to protect GOM cod since 1998, with the most recent adjustment in 2015 (63 FR 15326, March 31, 1998; 80 FR 25110, May 1, 2015), fishing behavior and effort was well established in this area of the GOM and on GB prior to the Amendment. Fishing in or around the Closed Areas was predominately by vessels using bottom trawl, sink gillnet, scallop dredge, and/or trap/pot gear; other gear types fished in or around the Closed Areas included clam dredge, mid-water trawl, bottom longline, and purse seine.

Relative to pre-Amendment conditions in this area of the GOM or GB, the Amendment implemented the following management measures:

- Maintained the GOM Cod Spawning Protection Area (i.e., Whaleback). Gear restrictions and exemptions remain unchanged.
- Established the Winter Massachusetts Bay Spawning Closure. From November 1–January 31 of each year, the area is closed to all fishing vessels, with the same exemptions as those specified in the GOM Spawning Protection Area (i.e., Whaleback).
- Established the Spring Massachusetts Bay Spawning Protection Measure. From April 15–April 30), within statistical area 125, this area is closed to all vessels, except for vessels that are: (1) fishing in state waters that do not have a federal Northeast multispecies permit; (2) fishing with exempted gears; (3) in the midwater trawl and purse seine exempted fisheries; (4) fishing with scallop dredges on a scallop day-at-sea; (4) fishing in the scallop dredge exemption area; and, (5) charter, party, and recreational.
- Removed the GB Seasonal (May) Spawning Closure Area.

Prior to the Amendment, the GB Seasonal Spawning Closure restricted fishing access to the area during the month of May; however, there were exemptions to this rule as scallop dredge, purse seine, mid-water, trap/pot, and hydraulic clam dredge gears were permitted to operate in the area. In addition, vessels belonging to the NE Multispecies sector program or operating under the provisions of a Northeast multispecies Handgear A or B permit were exempted from the seasonal closure. The majority of fishing effort under the Northeast Multispecies FMP is by vessels enrolled in groundfish sectors. For example, during the NEFMC's April 2020 meeting, the groundfish working group provided a review of the groundfish catch share program. Review of the working group's report indicated that between fishing years 2016 and 2018, active sector vessels comprised the majority of the groundfish fleet (see Working Group Report: https://www.nefmc.org/library/april-2020-groundfish-report). Specifically, the report showed that between fishing years 2016 through 2018, there were between 190 to 215 active sector vessels, compared to 45 to 48 active common pool vessels. In addition, data from limited access groundfish vessels show that during each year from 2010-2015, more than 97 percent of annual groundfish revenue was earned by sector vessels and more than 80 percent of annual groundfish trips were taken by sector vessel (Murphy et al. 2018). Data also show that during each year from 2016-2019, more than 63 percent of active common pool vessels were issued Handgear A or Handgear B permits. During this period, more than 36 percent of common pool trips and more than 21 percent of groundfish landings were attributed to these vessels annually (permit and DMIS data as of 5/29/20; GARFO; run on June 17, 2020). Given the predominance of vessels enrolled in the sector program, few vessels belonging to the NE Multispecies FMP were restricted from fishing in this area prior to the Amendment. Therefore, the GB Seasonal Spawning Closure affected a relatively small number of vessels and likely had a limited overall impact on the distribution of fishing effort in the Georges Bank region.

Relative to operating conditions prior to the Amendment, there was little to no changes in the distribution or level of fishing effort in this area of GB, as a result of the Amendment's removal of the GB Seasonal Spawning Closure; this is supported by figures provided in Appendix 2. Specifically, vessels that were permitted to operate in the former Closed Area will continue to do so. For the limited number of vessels previously affected by the May Closure, there is no evidence that the removal of the closure created any incentive for effort to change or redistribute in a manner that differed from pre-Amendment operating conditions in the region (see Appendix 2). At most, these vessels will likely remain in the area instead of shifting effort to other regions during the month of May.

In the GOM, specifically the WGOM, the Amendment implemented two new Spawning Protection/Closure Areas, in addition to maintaining the Spawning Protection Area known as "Whaleback." The introduction of two additional spawning closures is likely to result in vessels that fished in these areas year-round shifting just outside of the closure's boundaries or to other waters of the WGOM. However, this is limited by the relatively small size of the newly designated spawning protection areas in relation to available fishing grounds in the GOM and the restricted duration of time operational management measures are in place. There is no evidence that the management measures implemented through the Amendment resulted in an overall change in fishing behavior or effort in this region of the GOM, relative to operating conditions prior to the Amendment (see Appendix 2). At most, relative to fishing behavior/effort in the GOM prior to the Amendment, changes in fishing behavior or effort, as result of the Amendment's implementation of additional Spawning Protection/Closure Areas in the GOM, were small and localized.

7.8.4. *Framework Adjustments and Monitoring*

The Habitat Amendment identified several administrative measures associated with the review and regulatory adjustment of habitat management measures outlined in the Amendment. Implementation of the Habitat Amendment resulted in the authorization of these measures. Specifically, the designation or removal of HMAs and changes to fishing restrictions within HMAs may be considered in a framework adjustment. Processes to evaluate the performance of habitat and spawning protection measures, as well as for the Council to identify and periodically revise research priorities to improve habitat and spawning area monitoring, were established through the Amendment.

Taking into consideration the above information, the administrative measures implemented through the Amendment are procedural and therefore, in and of themselves, will not cause the operation of the fisheries (e.g., effort, behavior) in the GOM, on GB, or in SNE to change relative to operating conditions prior to the Amendment. Given this, the implementation of these measures are not expected to result in direct or indirect effects to listed species.

7.8.5. *Conclusion: Overall Impacts of the Habitat Amendment to ESA-listed species*

Fisheries operating in the GOM, on GB, or in SNE pose an interaction risk to listed species of sea turtles, large whales, Atlantic sturgeon, and/or Atlantic salmon both pre- and post-Amendment. The level of risk is affected by: (1) the quantity of gear in the water (e.g., number of vertical lines, gillnets, trawls), (2) gear soak/tow duration, and (3) the temporal and spatial overlap of the gear and a protected species. Although the Habitat Amendment removed, established, or maintained Habitat Management Areas, EFH and HAPCs, and Spawning Protection/Closure Areas in the GOM, on GB, or in SNE, as provided in the analyses above, the measures implemented through the Amendment did not result in increased risk to ESA-listed species.

Based on these analyses, there were no large shifts in the magnitude or distribution of effort and no evidence that the management measures implemented through the Habitat Amendment caused new or elevated interaction risks to ESA-listed species. Albeit it to varying degrees, a pre-existing level of risk to listed species of sea turtles, whales and fish existed in the sub-regions. As provided above, interaction risks with listed species are strongly associated with the quantity of gear in the water (e.g., number of vertical lines, gillnets, trawls), gear soak/tow duration, and the temporal and spatial overlap of the gear and protected species. While minor shifts in effort and small increases were observed for some gears in some regions, these changes are not expected to increase the risk to protected species in any region for the reasons described above. Any shifts in effort observed were local and from areas with similar bycatch rates (i.e., the overlap of gear and species is similar). The small changes in effort that occurred from the implementation of the Habitat Amendment were restricted to changes within particular regions and for particular gears. As changes across regions were not observed, we would not expect the impact to protected species across all regions to differ from that which was assessed within a region. That is, all impacts to effort were localized. Based on these analyses, although the measures implemented through the Amendment are not expected to remove or reduce interaction risks to listed species, they are also not expected to result in new or increased interaction risks to these species relative to pre-existing conditions in the GOM, on GB, or in SNE. Given this, interactions risks to ESA-listed species of sea turtles, whales, and fish under post-Amendment operating conditions are not expected to differ from that which has been observed and considered by NMFS in its assessment of fishery interaction risks to these listed species (NMFS 2002a, 2012b, 2013b, 2014b) or in the analysis of effects above. Specifically, in the sections addressing the effects to listed species from gear interactions (sections 7.2 (large whales), 7.3 (sea turtles), 7.4 (Atlantic sturgeon), and 7.5 (Atlantic salmon), it was determined that the operation of the fisheries are likely to adversely affect ESA listed species.

In summary, after analyzing the relevant studies and data from nearly two seasons of post Habitat Amendment fishing activities, we have determined that the Amendment resulted in little to no change in fisheries behavior. Post Habitat Amendment data represent the best available science for determining the short term effects of this action as well as for predicting any potential future effect. We recognize that two seasons of data represents a limited study window; however, our analysis indicates that this time frame was sufficient to detect even modest shifts in effort (as we noted for the increased localized effort near Chatham). Because the measures implemented under the Habitat Amendment resulted in little to no change in the overall operation of the fisheries relative to pre-Amendment conditions, it is extremely unlikely that the number or severity of interactions to listed species will be different from what was estimated in these effects sections. Taking into consideration this and the information provided above, we have determined that effects to ESA-listed species following the implementation of the Habitat Amendment do not differ from the effects resulting from the authorization of the fisheries considered in this Opinion (see section 7) or the effects in underlying consultations on fisheries outside of this Opinion (e.g., Atlantic sea scallop).

8. 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.

This section attempts to identify the likely future changes and their impact on ESA-listed species and their critical habitats in the action area. This section is not meant to be a comprehensive socio-economic evaluation, but a brief outlook on future changes in the environment. Projections are based upon recognized organizations producing best available information and reasonable rough-trend estimates of change stemming from these data. However, all changes are based upon projections that are subject to error and alteration by complex economic and social interactions.

During this consultation, we searched for information on future state, tribal, local, or private (non-federal) actions reasonably certain to occur in the action area. We did determine that the implementation of the measures in the ALWTRP proposed rule is reasonably certain to reduce the risk of M/SI to right whales in state fisheries. As described in the *Environmental Baseline*, we estimate that an annual average of 3 right whale M/SI will be the result of entanglement with state fishing gear.

As described in the Proposed Action, NMFS is proposing federal regulatory measures to modify the ALWTRP. In addition to the proposed federal measures, modifications to the ALWTRP would include risk reduction measures implemented by the states of Maine and Massachusetts in exempted or state waters. In waters currently exempted from regulations under the ALWTRP, the Maine Department of Marine Resources will require weak insertions. Maine has already implemented gear marking requirements consistent with gear marking modifications proposed in the proposed rule. In 2021, Massachusetts Division of Marine Fisheries implemented a series of new regulations (322 CMR 12) to protect right whales. These measures expand upon the existing practices described in the *Environmental Baseline*. The new measures include: a February 1st through May 15th seasonal closure of all Massachusetts waters to trap gear fishing; a January 1st through May 15th closure of Cape Cod Bay and certain adjacent waters to gillnet gear; and a March 1st through April 30th speed limit for small vessels operating in Cape Cod Bay and certain adjacent waters. Each of these seasonal restrictions may be extended beyond their end date in response to the continued presence of right whales in Massachusetts waters. Massachusetts will also restrict buoy line diameters within state waters to restrain the introduction of stronger line into the fishery. As described in the DEIS for the ALWTRP proposed rule, the combined federal and state measures, which include the state measures described here, will achieve a 60 percent reduction in risk to right whales.

Using results from the DST (see section 7.2), we determined the proportion of reductions in M/SI due to the ALTWRP measures that would occur in state fisheries vs. federal fisheries. As described in section 7.2.1, an annual average of 7.57 right whale entanglements are expected to result in M/SI as a result of entanglement in U.S. trap/pot gear. The DST shows that 39.6 percent of the risk to right whales in the U.S. occurs in state waters. Therefore, we determined that an annual average of 3 right whale M/SI were the result of entanglement in gear used in the state fisheries. The measures implemented under the ALWTRP proposed rule will reduce risk of right whale M/SI entanglements in trap/pot gear in state waters by 31.5 percent. A 31.5 percent reduction of the annual estimate of total U.S. entanglements in trap/pot gear (7.57) results in a reduction of 2.39 M/SI entanglements in state waters. We subtract this reduction (2.39) from the total estimated M/SI entanglements in gear used in the state fisheries (3) resulting in 0.61 M/SI entanglements remaining in the U.S. state fisheries. Given that we are reasonably certain this risk reduction to right whales will occur in the state fisheries, we have determined that an annual average of 0.61 M/SI right whale entanglements will occur as a result of gear used in the state fisheries (Table 62).

Other than the U.S. state fisheries, we did not find any information about non-federal actions other than what has already been described in the *Environmental Baseline* (see section 5), most of which we expect will continue in the future. An increase in these activities could similarly increase their effect on ESA-listed species and, for some, an increase in the future is considered reasonably certain to occur. Given current trends in global population growth, threats associated with climate change, pollution, fisheries bycatch, aquaculture, vessel strikes and approaches, and sound are likely to continue to increase in the future, although any increase in effect may be somewhat countered by an increase in conservation and management activities. For the remaining activities and associated threats identified in the *Environmental Baseline* and *Climate Change* sections, and other unforeseen threats, the magnitude of increase and the significance of any anticipated effects remain unknown. The best scientific and commercial data available provide little specific information on any long-term effects of these potential sources of disturbance on ESA-listed species populations. Thus, with the exception of the risk reduction to right whale M/SI in state fisheries from measures implemented under the ALWTRP, this consultation assumes effects in the future would be similar to those in the past and, therefore, are reflected in the anticipated trends described in the *Status of the Species* (see section 4), *Environmental Baseline* (see section 5), and *Climate Change* (see section 6) sections.

9. INTEGRATION AND SYNTHESIS OF EFFECTS

The *Status of Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* sections of this Opinion discuss the natural and human-related factors that caused right, fin, sei, and sperm whales; Northwest Atlantic DPS of loggerhead, leatherback, Kemp's ridley, and North Atlantic DPS of green sea turtles; the five DPSs of Atlantic sturgeon; the GOM DPS of Atlantic salmon; and giant manta rays to become endangered or threatened and may continue to place those species at risk of extinction. "Jeopardize the continued existence of" means to engage in an action that reasonably 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). The present section of this Opinion applies that definition by examining the effects of the proposed action in the context of information presented in the *Status of the Species* (see section 4), *Environmental Baseline* (see section 5), *Climate Change* (see section 6) and *Cumulative Effects* (see section 8) sections to determine: (a) if the effects of the proposed action would be expected to reduce the reproduction, numbers, or distribution of the previously listed cetaceans, sea turtles, and fish, and (b) if any reduction in the reproduction, numbers, or distribution of these species causes an appreciable reduction in the species' likelihood of surviving and recovering in the wild.

In the 1998 NMFS/U.S. Fish and Wildlife Consultation Handbook, "survival" is defined as:

> For determination of jeopardy/adverse modification: the species' persistence as listed or as a recovery unit, beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment. Said another way, survival is the condition in which a species continues to exist into the future while retaining the potential for recovery. This condition is characterized by a species with a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, which exists in an environment providing all requirements for completion of the species' entire life cycle, including reproduction, sustenance, and shelter.

"Recovery" is defined as "[i]mprovement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in section 4(a)(1) of the Act."

The analytical process we undertake to make jeopardy determinations is described in regulation as:

> Add[ing] the effects of the action and cumulative effects to the environmental baseline and in light of the status of the species and critical habitat, formulate the Service's opinion as to whether the action is likely to jeopardize the continued existence of listed species or result in the destruction or adverse modification of critical habitat. (50 CFR 402.14(g))

Our task then, when making a jeopardy determination, is to consider the biological significance of proposed action's effects on ESA-listed species and to assess whether the proposed action appreciably reduces the survival or recovery of a listed species.

We evaluate this in the context of the recovery plans for each species. Recovery plans include criteria, which, when met, would result in downlisting (changing the listing from endangered to threatened) or in a determination that the species be removed from the List of Endangered and Threatened Wildlife. Recovery criteria can be viewed as targets, or values, by which progress toward achievement of recovery objectives can be measured. Recovery criteria may include such things as population numbers and sizes, management or elimination of threats by specific mechanisms, and specific habitat conditions. In newer recovery plans, recovery criteria are often framed in terms of population parameters (Demographic Recovery Criteria) and the five listing factors (Listing Factor Recovery Criteria). For some species, the plans have not been recently updated and do not include specific Demographic and Listing Factor Recovery Criteria. Regardless of whether these are included, we evaluate each species in the context of the criteria and objectives in its recovery plan.

This Opinion has identified in the Effects of the Proposed Action (section 7) that the proposed action may adversely affect right, fin, sei, and sperm whales as a result of entanglement in gear fished in the fisheries. No other effects to ESA-listed cetaceans are expected as a result of the activity. This Opinion has also identified that the proposed action may adversely affect Northwest Atlantic DPS of loggerhead, leatherback, Kemp's ridley, and North Atlantic DPS of green sea turtles; the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon; giant manta rays; and the GOM DPS of Atlantic salmon because of interaction with gear used in the fisheries and, for sea turtles, strikes from vessels used in the fisheries. No other effects to designate ESA-listed sea turtles, Atlantic sturgeon, giant manta rays, or Atlantic salmon are expected as a result of this activity. Nor do we expect adverse effects to any designated critical habitat. The discussion below provides NMFS' determinations of whether there is a reasonable expectation that right, fin, sei, and sperm whales; loggerhead, leatherback, Kemp's ridley, and green sea turtles; Atlantic sturgeon; Atlantic salmon; and giant manta rays will experience reductions in reproduction, numbers, or distribution in response to these effects, and whether any reductions in the reproduction, numbers, or distribution of these species can be expected to appreciably reduce the species' likelihood of surviving and recovering in the wild.

9.1. North Atlantic Right Whale

The North Atlantic right whale population faces a high risk of extinction. The population size is small enough for the death of any individuals to have measurable effects in the projections on its population status, trend, and dynamics. As described above, our task is to consider the biological significance of the proposed action's effects on North Atlantic right whales. We evaluate, based on the totality of the circumstances affecting the species and the best available scientific and commercial information, the nature and magnitude of the proposed action's effects, to determine whether such effects of the proposed action, while measureable, are consequential enough to appreciably reduce the species' likelihood of survival and recovery.

To evaluate whether the proposed is likely to jeopardize the continued existence of North Atlantic right whales, we qualitatively assessed the best available scientific and commercial data on right whales and developed a population projection model to quantitatively predict the female population trajectory over 50 years (Linden 2021). To determine if the proposed action is expected to appreciably reduce the likelihood of both the survival and the recovery of right whales in the wild, we must evaluate the effects of the fisheries as now proposed (section 7.2) considered in context of the *Status of the Species* (section 4), *Environmental Baseline* (section 5), *Climate Change* (section 6), and *Cumulative Effects* (section 8). Here, we review the information considered and provide the details of our analysis.

Information from the Status of the Species, Environmental Baseline, Climate Change, and Cumulative Effects Considered in the Analysis

As discussed in the *Status of the Species* section of this Opinion, North Atlantic right whales follow a general pattern of foraging at high latitudes (e.g., southern New England and Canadian waters) in the spring and summer months and calving at lower latitudes (i.e., off Florida) in the winter months. The North Atlantic right whale population has been declining since 2010 (Pace et al. 2017). Recent modeling efforts indicate that low female survival, a male biased sex ratio, and low calving success are contributing to the population's current decline (Pace et al. 2017). Using the methods in Pace et al. (2017), the most recent estimate is 368 (95 percent credible interval range of 356-378) individuals as of January 2019 (Pace 2021). The species has low genetic diversity, as would be expected based on its low abundance, and the species' resilience to future perturbations is expected to be very low (Hayes et al. 2018a). Furthermore, entanglement in fishing gear appears to have had substantial health and energetic costs that affect both survival and reproduction of right whales (van der Hoop et al. 2017a). Vessel strikes and entanglement of right whales in U.S. and Canadian waters continue to occur.

As described in the *Environmental Baseline* and *Climate Change* sections, ongoing effects in the action area (e.g., global climate change, decreased prey abundance, vessel strikes, and entanglements in U.S. state and federal fisheries) have contributed to concern for the species' persistence. Sublethal effects from entanglement cannot be separated out from other stressors (e.g., prey abundance, climate variation, reproductive state, vessel collisions) which co-occur and affect calving rates.

As described in the *Effects of the Proposed Action* section, the information documented for some right whale interaction cases contains evidence that allowed attribution of the event to a country and/or a specific cause (i.e., gear entanglement, vessel strike); other cases are of unknown origin and/or unknown cause. As described in section 7.2, gear analysis and sightings data allowed us to partition total M/SIs to the United States and Canada. For the cases attributed to the United States, we were able to further partition them between entanglements and vessel strikes. Data are not available to partition the total mortality occurring in Canada between entanglement and vessel strike. Current estimates based on the best available scientific and commercial data indicate that the federal fisheries in this Opinion will entangle, on average annually, a total of

9.14 percent of the North Atlantic right whale population. As described in the *Effects of the Proposed Action* and in more detail below, given the reductions in entanglements that will result from the implementation of the North Atlantic Right Whale Conservation Framework for Federal Fisheries in the Greater Atlantic Region (Framework), we expect that substantially less than 9.14 percent of the population will become entangled following the implementation of measures described in the Framework. However, we cannot quantify how much less at this time.

We estimated that, from 2010-2018, an annual average of 4.7 right whale M/SI (4.57 in trap/pot gear and 0.125 in gillnet gear) were the result of entanglements in gear used in the federal fisheries in this Opinion. The implementation of Phase 1 of the Framework (the proposed ALWTRP measures) in the American lobster and Jonah crab fisheries is expected to reduce M/SI of right whale entanglements in trap/pot gear in federal waters by 44 percent[69]. That is, after implementation of the current ALWTRP rulemaking, an annual average of 2.69 (=4.7-(0.44*4.57)) right whale M/SI entanglements are expected to occur in federal waters. In 2023, Phase 2 of the Framework is expected to reduce M/SI of right whale entanglement in other federal trap/pot fisheries and gillnet fisheries by 60 percent. After implementation of these measures, 2.61 (=2.69 – (0.6*0.125)), on average annually, right whale M/SI entanglements are expected to occur in federal waters. In 2025, Phase 3 of the Framework will be implemented and further reduce M/SI of right whale entanglement in federal waters by an additional 60 percent. After the implementation of the Phase 3 measures, an annual average of 1.04 (=2.61-(0.6*2.61)) right whale M/SI entanglements expected to occur in federal waters (Table 81). In 2030, Phase 4 of the Framework will further reduce M/SI of right whale entanglement in federal waters by an additional 87 percent. After the implementation of the Phase 4 measures, an annual average of 0.136 (=1.04-(0.87*1.04)) right whale M/SI entanglements are expected to occur in federal waters, which equals approximately one M/SI every 7 years (Table 81).

Table 81: Framework actions and associated reductions in M/SI

Action	M/SI reductions in federal fisheries	M/SI in federal fisheries prior to the action,:	M/SI in federal fisheries after the action,
Phase 1 (current TRT rulemaking Action)	Reduce M/SI in trap/pot gear by 44 percent	4.7 (4.57 in trap/pot and 0.125 in gillnet)	2.69 (2.56 in trap/pot and 0.125 in gillnet)
Phase 2	Reduce M/SI in gillnet gear by 60 percent	2.69 (2.56 in trap/pot and 0.125 in gillnet)	2.61 (2.56 in trap/pot and 0.05 in gillnet)
Phase 3	Reduce M/SI in fixed gear fisheries by 60 percent	2.61 (across the gillnet and trap/pot fisheries)	1.04 (across the gillnet and trap/pot fisheries)
Phase 4	Reduce M/SI in fixed gear fisheries by 87 percent	1.04 (across the gillnet and trap/pot fisheries)	0.136 (across the gillnet and trap/pot fisheries)

[69] The results of the DST show that 60.4 percent of the risk of M/SI from pot/trap gear to right whales in the U.S. occurs in federal waters. The DST also showed that of the 58.1 percent risk reduction in M/SI due to the ALWTRP proposed rule, 26.6 percent of that risk reduction occurs in federal waters. This results in a 44 percent reduction in risk of M/SI from pot/trap gear to right whales in federal waters.

As described above, each phase of the Framework reduces M/SI in federal fisheries by a fixed percentage. With these reductions, we anticipate that there will be, on average annually, 2.69 M/SI during Phase 1, 2.61 during Phase 2, 1.04 during Phase 3, and 0.136 during Phase 4. Given the length of each phase, we would anticipate a total of 8.07 M/SI in Phase 1, 5.22 in Phase 2, and 5.2 in Phase 3 due to the federal fisheries included in this Opinion. Phase 4 would reduce M/SI to 0.136 starting in year 11, resulting in 5.44 additional M/SI in years 11 through 50. Based on this, we estimate that 18.49 whales will die or be seriously injured in the 10 years before the Framework is fully implemented, and 5.44 in the following 40 years. This results in an estimated total of 23.93 entanglements resulting in M/SI in gear used by the federal fisheries over the course of 50 years.

As described in the *Cumulative Effects* section, we did not find any additional information, with the exception of measures to reduce risk from the U.S. state fisheries through the Massachusetts Division of Marine Fisheries regulations (322 CMR 12) and the ALWTRP proposed action, about future effects to right whales that would be reasonably certain to result from non-federal actions beyond what was described in the *Environmental Baseline*. Most of the effects described in the *Environmental Baseline* are expected to continue in the future. The implementation of the measures under the ALWTRP is reasonably certain to occur in state waters and is estimated to reduce the risk of M/SI to right whales in the lobster and Jonah crab fisheries by 31.5 percent. This reduction is expected to reduce the risk of the annual average right whale M/SI entanglements in trap/pot gear in state waters (currently estimated at 3) by 2.39, resulting in an estimated annual average of 0.61 right whale M/SI in the state fisheries after the implementation of the current ALWTRP rulemaking.

Jeopardy Assessment

In this assessment, we evaluate whether the level of fishery interactions during and following full implementation of the Framework allows the right whale population to maintain a status where the action is not jeopardizing the continued existence of the species, and the population maintains the ability to achieve recovery. We evaluate this by comparing how the population would fare with no impact from the proposed action (i.e., no entanglements in federal waters) to how the population would fare with anticipated impacts from the action (i.e., entanglements and M/SI in federal waters). Below is a description of the quantitative and qualitative information that we consider in our jeopardy analysis.

Population Projection Model

NMFS developed a population projection model to predict the female right whale population trajectory over 50 years (Linden 2021). As with any model, there is uncertainty in the population projections. Linden (2021) describes the methodology used for the population projections and explores a range of scenarios. The projections included in the Appendix underwent peer review by the Center for Independent Experts in 2020. As described in Linden (2021), the full posterior distributions of parameters from updated fitting of the Pace et al. (2017) model were used (as matrix inputs) in the projections. This means that uncertainty in the demographic parameter estimates was fully propagated. Also discussed by Linden (2021) is that the source of the reductions in M/SI is immaterial from a population perspective. However, it is important for management purposes and is an important caveat with the data used. The population projection model does not depict absolute population trends, but rather shows relative outcomes under various management strategies to allow for the comparison of potential future outcomes (D.

Linden, pers. comm.). As described above, we used the best available information, including peer-reviewed information when available, to apportion M/SI when the source (either country and/or cause) of the M/SI was unknown.

Fifty years was chosen as the appropriate timeframe as it provides sufficient time for any reductions in M/SI to be reflected in the trajectories and provides a long enough period to assess the long-term trend. The population consequences of different actions are difficult to distinguish when they are only allowed to manifest for a few years, while any environmental projections beyond 10-20 years are subject to large amounts of uncertainty. Climate change is one of the factors influencing future conditions. We believe that 50 years achieves this balance and periods longer than 50 years (e.g. 100 years) would introduce too much uncertainty to be confident in the trends. We believe using this period for the projections provides the best available information to determine the future population trend of female right whales.[70]

The Framework reduces M/SI in the federal fisheries over four-phases, allowing interactions to continue to occur at a higher level in the earlier phases of the Framework before attaining the final reductions to an annual average of 0.136 M/SI in the federal fisheries. The reductions in M/SI specified under the Framework are considered in the population projections using a per capita mortality rate, and the actual number of M/SI can change based on stochasticity in the projected population size in a given year and the random draw of actual deaths in a given year (see Linden 2021). The projections consider how the projected size and per capita rates interact across time. The reductions in mortalities due to the implementation of the Framework will contribute positively to the parameters included in the model as they will survive and contribute to the population. Given these interactions, the estimates of mortalities due to the federal fisheries described above differ from model predictions. As described in Appendix 3, Linden (2021) evaluated multiple hypothetical risk reduction scenarios. However, the scenarios used for the analysis in this Opinion examine the outcome of the mitigation measures as described in the Conservation Framework.

The projections use information on right whale survival and calving to forecast changes in population size and provide insight on future population growth. The initial female right whale population size used in the projections was 179, split among calves (3), juveniles (22), and mature adults (154). The starting population size of females (179) is an average of 2010-2019 to represent an approximation of the age structure. Each scenario evaluated a different level of reduction in human-caused mortalities. To examine the influence of anthropogenic mortality reduction on survival and recovery of the species, we ran 5,000 simulations for each risk reduction scenario. For each scenario, we calculated the median population trajectory and probability of decline. We summarize the projection model here; for detailed methods, see Appendix 3 (Linden 2021).

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[70] We projected the female population forward because females are the limiting factor for population growth given they contribute new individuals through reproduction. Anthropogenic threats or environmental conditions are not known to affect males and female right whales differently beyond reproductive dynamics, so future population dynamics are adequately captured by females alone. It is important to note that the projections are just one of several prongs in our analysis, we considered impacts to both males and females in our qualitative analysis.

As described above, the projections consider the effects of the proposed action in the context of information provided in the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* sections. The factors discussed in each of those sections have impacted, and continue to impact, the calving and survival of right whales and are included in the calving numbers and survival estimates used as inputs to the projections. As described in the *Effects of the Proposed Action*, the survival estimates are based on data available from 2010-2018. A total mortality estimate is not available for 2019; therefore, we do not consider the 2019 observed data in the survival estimates. Given that the starting population is based on 2010-2019 data, the projections include calving data from the same time period. Therefore, by using data from 2010-2019, the projections implicitly consider the effects of all sources of M/SI on right whales and consider impacts to calving rates due to sublethal effects from entanglement, prey availability, and other stressors.

Using the DST as the best available information, the ALWTRP action will reduce M/SI in U.S. waters by at least 58.1 percent across lobster and Jonah crab fisheries. Additionally, implementation of the Framework will reduce M/SI by at least 60 percent in gillnet and other trap/pot fisheries in 2023, reduce M/SI by at least an additional 60 percent in fixed gear in federal waters in 2025, and by an additional 87 percent in fixed gears in federal waters in 2030. As previously described, using estimates of right whale M/SI in U.S. waters, the DST allows us to quantify the number of right whale M/SI occurring in U.S. state and federal waters. Additionally, the DST allows us to quantify the relative risk reduction that will occur in federal and state waters in the United States due to the implementation of the current ALWTRP proposed rule. This represents the best available data for estimating reductions in risk from state and federal fisheries. We then developed scenarios which would reduce M/SI from entanglement in gear used in the U.S. state and federal fisheries. The projections simulate how the population trajectory may change under these scenarios.

To estimate the demographic rates, a state-space mark-recapture model was fit to North Atlantic right whale sightings data collected from 1990-2019 to generate posterior distributions for stage-based population sizes, deaths, and survival rates. The projections included stochastic simulations that re-sampled demographic rates from observed calving records and sighting histories of cataloged individuals to assess the influence that simple and per capita reductions in anthropogenic mortality might have on population trajectories. The projections used a subset of the estimated demographic rates to focus inferences in two ways: 1) only females were projected, given they are the primary driver of population growth dynamics; and 2) projections used rates from the 2010-2019 time period to capture the post-2010 ecological conditions that are considered most representative of current conditions and coincide with the recent population decline. Using these female calving (2010-2019) and survival rates (2010-2018), the approach predicts the future right whale female population.

Resolving Data Uncertainties for Model Inputs

As noted above, this population projection model is designed to compare potential future trajectories of the right whale population. Given the uncertainties and the model's general design, it is incapable of predicting the actual future population trend of right whales. When dealing with data uncertainties (e.g, a range of potential calving rates, or unquantified benefits from conservation measures), we utilized metrics representing the worst case scenario. Consequently, model outputs very likely overestimate the likelihood of a declining population.

For example, in 2017, the Government of Canada introduced measures designed to protect North Atlantic right whales from both the fishing and shipping industries (section 4.2.1). Following the implementation of mitigation measures, ten right whale mortalities were confirmed to have occurred in Canadian waters in 2019. Canada has modified their measures annually to reduce M/SI. In 2020 and 2021, they implemented additional measures. Given the limited time these measures have been in effect as well as annual changes to and the dynamic nature of the measures, there is no way to quantitatively or qualitatively assess the benefit at this time. Since we have determined that a quantitative prospective assessment of risk reduction from future mitigation measures in Canada is not feasible, we assumed a worst case scenario where the estimated number of right whale M/SIs due to vessel strike and entanglement that occurred in Canada between 2010 and 2018 will continue to occur in the future. Similarly, even though the United States has taken actions to reduce future vessel strikes, we assume that M/SI resulting from vessel strikes in U.S. waters continue to occur in the future at the same level as 2010-2018. Also, even though we anticipate a reduction in the percentage of entangled right whales through implementation of the Conservation Framework (these reductions cannot be quantified as they are confounded by other stressors (e.g., environmental factors)), the projection outputs do not consider any increase in the female right whale population trajectories due to a reduction in sublethal effects (i.e., ALWTRP proposed rule, any future risk reduction measures). Lastly, although calving rates were generally higher pre-2010, given the uncertainty of future calving rates, we used post-2010 calving rates in the population projections as they were considered most representative of current conditions.

While uncertainty surrounds the population trajectories for this species, the results of the projections coupled with qualitative assessments of other factors, described in detail below, represent the best available information to determine whether the effects of the proposed action appreciably reduce the likelihood of survival and recovery.

The results of the projections assessed in our evaluation include the median population trajectory, the probability of a declining female right whale population, and the estimated number of females in the future. The median population trajectories are used to calculate resulting growth rates, represented by a λ value. In population biology, λ is the finite rate of increase over some time interval (e.g., Year A to Year B); it is the ratio of the population size at the end (Year B) to the population size at the start (Year A). The following equation was used to calculate the geometric mean annual λ for the interval of length T: $(\text{Year B}/\text{Year A})^{(1/(T))}$. When λ is greater than 1, the population is growing; when λ is less than 1, the population is declining. The growth rate was calculated for the 50-year period, the Framework implementation period (through year 10), and following full implementation (year 11-50).

The projections predicted the future female right whale population trajectory of the proposed action, which includes the implementation of the Framework. To support our determination of whether the proposed action is appreciably reducing in the likelihood of the survival and recovery of right whales, we first compare the population trajectory with no impact from the proposed action (i.e., no entanglements in federal waters) to the population trajectory that includes the anticipated impacts from the action (i.e., entanglements in federal waters). Theoretical projections are provided in the figure below to illustrate how the likelihood of jeopardy is assessed (Figure 68). In conducting our jeopardy analysis, we evaluate the difference between the trajectories in the absence of the action (yellow line) to the trajectory with the action occurring (black line). The projections assume calving in future years is similar to calving from

2010 to 2019. We use the calculated reductions in M/SI from entanglements in U.S. waters to analyze how increases in survival change the trajectory. All other impacts (e.g., U.S. and Canadian vessel strikes, Canadian entanglements) affecting survival over the 9-year period, and calving over the 10-year period, were presumed to continue unchanged into the future.

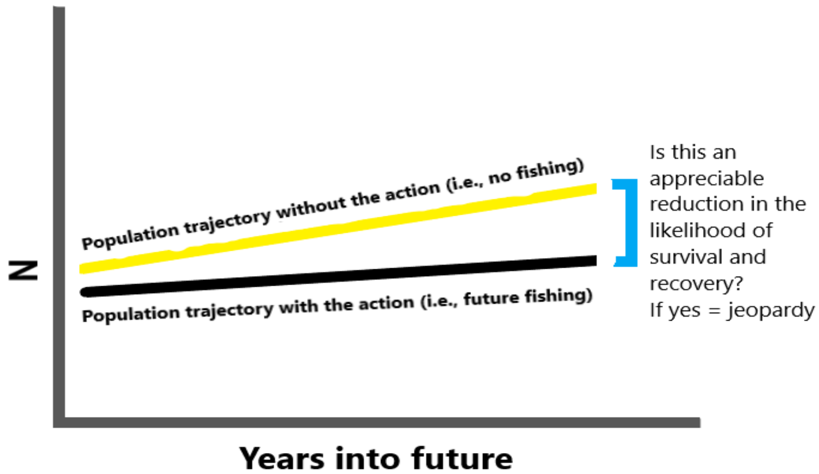

Figure 68: Theoretical population projections illustrating how each inform the jeopardy analysis and conclusion

Population Projection Results

<u>Scenario 1: Female population trajectory *without* anticipated impacts of the proposed action (no federal fisheries scenario)</u>

Scenario 1 evaluates how the female right whale population would fare in the future without M/SI caused by U.S. federal fisheries, while ongoing actions outside the scope of this Opinion continue and future non-federal actions that are reasonably certain to occur (i.e., risk reduction measures in state fisheries through the ALWTRP) are in place. With respect to the survival inputs, Scenario 1 includes all mortality sources except entanglements in the federal fisheries. That is, the scenario assumes no M/SI from federal fisheries, providing a trajectory to which we will compare to the trajectory of the proposed action. As described in section 7.2, the operation of the U.S. federal fisheries results in an annual average of 4.7 right whale M/SI due to entanglement. For Scenario 1, we assume that the 4.7 whales (average annual) that would have been seriously injured or have died in the federal fishery will survive.

Scenario 1 also considers reductions in M/SI in future years from actions outside the Opinion. As described in the *Cumulative Effects* section and above, the implementation of the ALWTRP proposed rule will reduce M/SI to right whales from entanglements in state waters by an annual average of 2.39 right whales. Therefore, Scenario 1 predicts the future population of female right whales if an annual average of 7.09 (= 4.7 in federal waters+2.39 in state waters) right whale M/SI ceased to occur. All other impacts contributing to the decline of the right whale population are considered to continue at the same rate experienced between 2010 and 2018.

The results of the Scenario 1 estimate the median right whale abundance will be 163 females in 10 years, which is a loss of 16 females over the first 10 years. In 50 years, the results of the Scenario 1 projection estimate the median right whale abundance to be 108 females, a loss of 71 females over the 50 years. This indicates that even in the absence of the U.S. federal fisheries, the female right whale population will decline (Figure 69), with a λ of 0.989742 (= (108/179)^(1/49)) over the 50 year time period. The projection shows 97.72 percent of the simulations resulted in a declining female right whale population 50 years into the future (Figure 70).

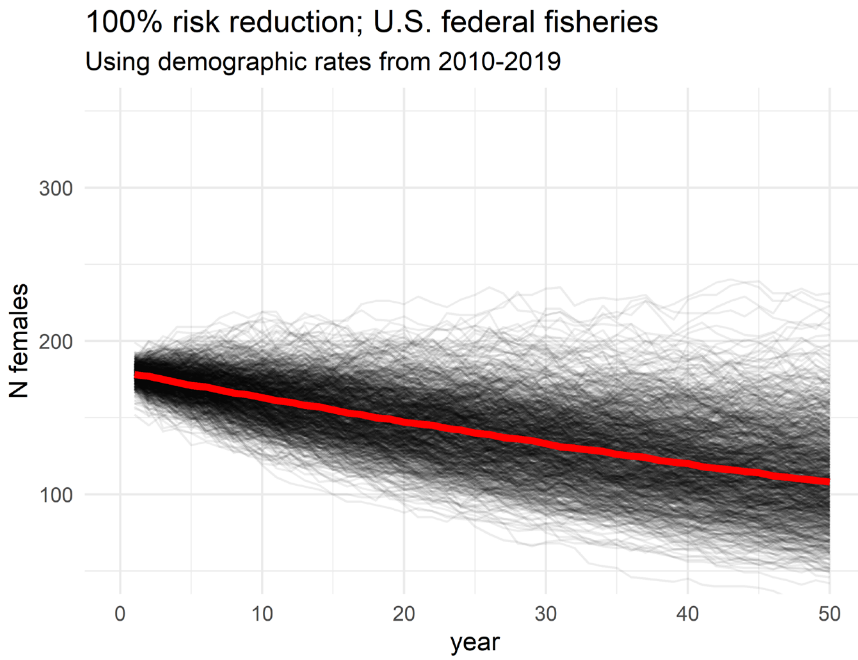

Figure 69: Population projections (n=5000) of North Atlantic right whale females using demographic rates from 2010–2019. Median population size and resulting growth rate (λ) in red. The risk reduction from no federal fisheries and the risk reduction from U.S. state waters with ALWTRP proposed measures in place.

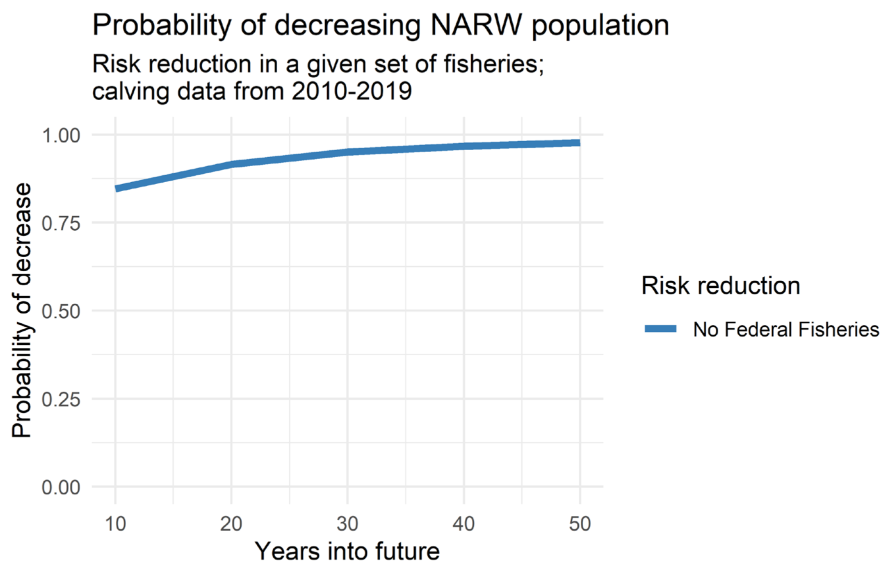

Figure 70: Probability of a decreasing NARW female population size using calving rates from 2010–2019 with no M/SI in the federal fisheries and the risk reductions in state waters with ALWTRP proposed measures in place.

Scenario 2: Female population trajectory *with* anticipated impacts of the proposed action (proposed action scenario)

Scenario 2 evaluates how the female right whale population would fare over the next 50 years with anticipated reductions in M/SI in U.S. federal fisheries with the implementation of the Framework. Phase 1 of the Framework (the current proposed ALWTRP rule) is assumed to be implemented at year 0, Phase 2 (60 percent reduction in M/SI in other federal trap/pot and gillnet fisheries) at year 3, Phase 3 (further 60 percent reduction in M/SI in federal fisheries) at year 5, and Phase 4 (further 87 percent reduction in M/SI in federal fisheries) at year 10. As with Scenario 1, ongoing actions outside the scope of this Opinion continue and future actions (i.e., risk reduction measures in state fisheries) that are reasonably certain to occur are in place.

As described above, we estimate that currently there are an annual average of 4.7 right whale M/SI due to entanglement in the federal fisheries in this Opinion. As described in the *Environmental Baseline*, we also determined that the operation of the U.S. state fisheries currently results in an annual average of 3 right whale M/SI due to entanglement in trap/pot gear. The implementation of Phase 1 of the Framework will reduce the annual average of M/SI in the federal fisheries from 4.7 to 2.69. Following the implementation of Phase 2 of the Framework, we anticipate that an annual average of 2.61 M/SI will continue to occur in the federal fisheries. Within 5 years, an additional 60 percent reduction in M/SI will be implemented under Phase 3 of the Framework. For the model, we assume that this implementation occurs at year 5, resulting in an annual average of 1.04 M/SI in federal fixed gear fisheries. Within 10 years, an additional 87 percent reduction in M/SI will be implemented under Phase 3 of the Framework. For the model, we assume that this implementation occurs at year 10, resulting in an annual average of 0.136 M/SI in the federal fixed gear fisheries from year 10 forward. Therefore, Scenario 2 predicts the future population of female right whales if the annual average right whale M/SI was reduced by 4.4 (reduced by 2.39 in state waters and 2.01 in federal waters) in years 0-3; 4.48 (reduced by 2.39 in state waters and 2.09 in federal waters) in years 3-5; 6.04 (reduced by 2.39 in state waters and 3.65 in federal waters) in years 6-10; and 6.95 (reduced by 2.39 in state waters and 4.56 in federal waters) in years 10-50. This results in an annual average of 0.61 (=3-2.39) right whale M/SIs in state trap/pot gear and 0.136 (=4.7-4.56) in federal fixed gear after the implementation of the Framework. All other impacts contributing to the decline of the right whale population are considered to be continuing at the same rate experienced between 2010 and 2018. As described above, there is a loss of females even in the absence of the federal fisheries. Therefore, the losses described below in the results of Scenario 2 are not solely due to the operation of the federal fisheries.

The results of the Scenario 2 (Figure 71) estimate the median right whale abundance to be 102 females in 50 years, which is a loss of 77 females over the 50 years. The λ over this period is 0.988588 ((102/179) ^ (1/49)). This indicates that with the operation of the U.S. federal fisheries after implementation of the Framework, the female population decreases. The projection shows 98.8 percent of the simulations resulted in a declining female right whale population 50 years into the future (Figure 72).

Given that the Framework is phased in under Scenario 2, it is important to evaluate the projection results before and after its full implementation. The results of the Scenario 2 projection estimate the median right whale abundance to be 157 females in 10 years, which is a loss of 22 females over the first 10 years. This indicates that the female population declines with the operation of the U.S. federal fisheries during the implementation of the first two phases of the Framework (Figure 71). The λ is 0.985535 (=(157/179) ^ (1/9)).

After the Framework is fully implemented (at year 10), the median right whale abundance is 157 females. (Table 82). The results of the Scenario 2 projection estimate the median right whale abundance to be 102 females in 50 years, which is a loss of 55 females over years 10-50. The λ is 0.989003 (=(102/157) ^ (1/39)) over years 10-50. This indicates that the female population declines more slowly with the operation of the U.S. federal fisheries after the implementation of the full Framework (Figure 71).

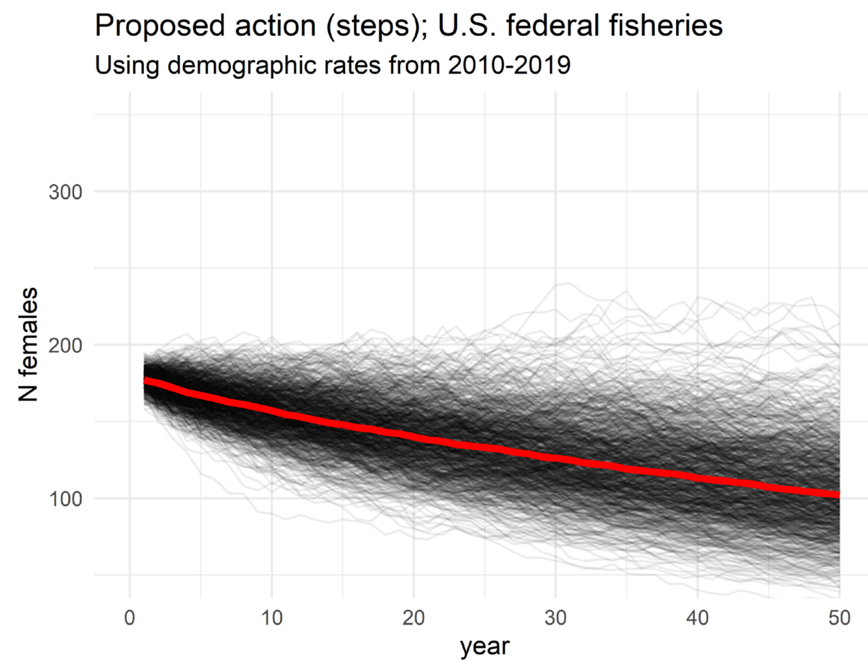

Figure 71: Population projections (n=5000) of North Atlantic right whale females using demographic rates from 2010–2019. Median population size in red. The risk reduction from federal fisheries is with the Framework in place, and the risk reduction in U.S. state waters is with ALWTRP proposed measures in place.

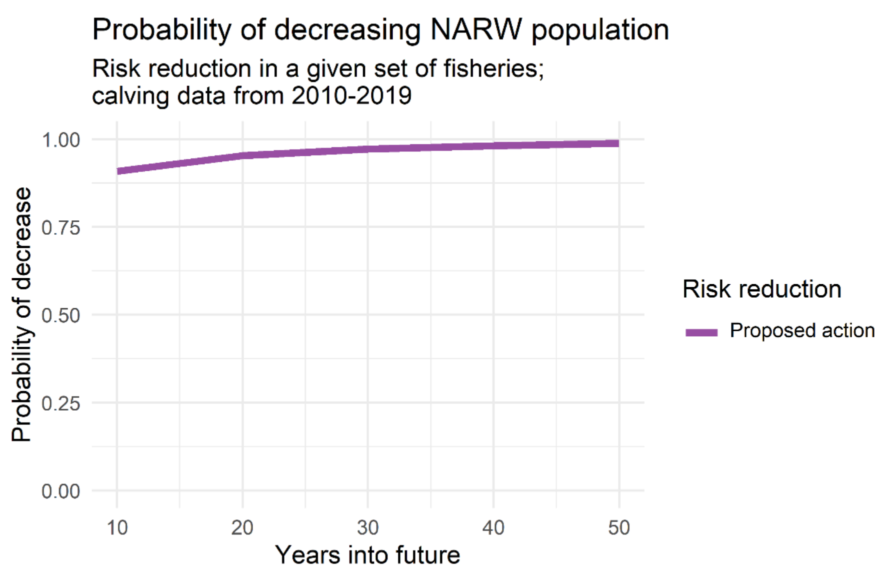

Figure 72: Probability of a decreasing NARW female population size using calving rates from 2010–2019. The risk reduction from federal fisheries is with the Framework in place. The risk reduction from U.S. state is with measures in ALWTRP proposed rule in place.

Assessment of the Population Projections

As part of our jeopardy analyses, we compared the population trajectory without the action (Scenario 1) to the trajectory with the action (Scenario 2) (Figure 73). This is indicative of the extent federal fisheries considered in the Opinion are contributing to the declining trajectory. We evaluated this in the context of whether the difference represents a reduction in the likelihood of survival and recovery of North Atlantic right whales that is biologically meaningful.

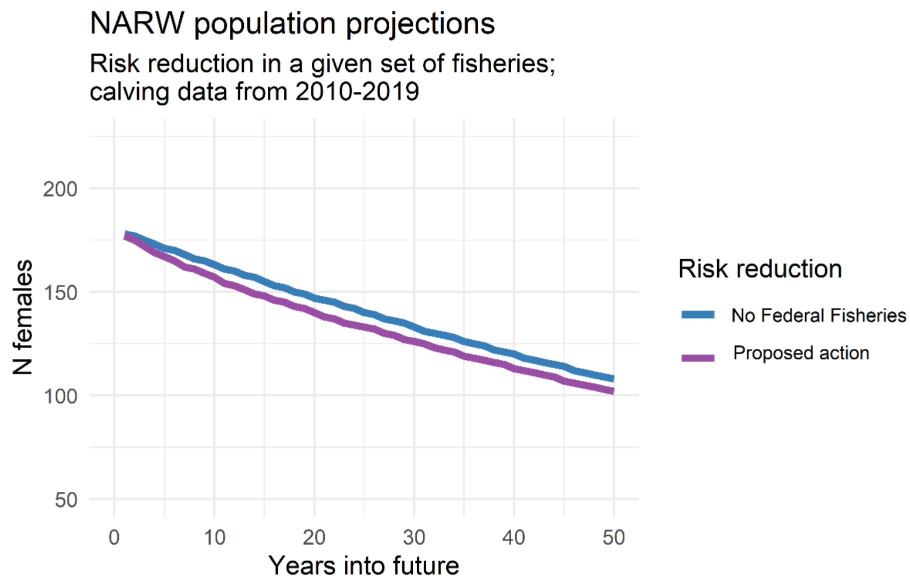

Figure 73: Median North Atlantic right whale female population size from population projections (N=5,000) using demographic rates from 2010–2019. The proposed action (in purple) includes the risk reduction from the Framework in federal waters and in U.S. state waters with ALWTRP proposed measure. The projection with no risk from federal fisheries (in blue) includes the risk reduction from the future ALWTRP reductions in state waters.

Table 82: Comparison of population projection results with proposed action and without proposed action.

	Geometric mean female population growth rate (λ)	**Percent of simulations showing a declining trend (%)**	**Number of females (initial estimate = 179)**	**Percent of population entangled (lethal or non-lethal) (%)**
Proposed action years 0-50	0.988588	98.8	102	Less than 9.14, but % unknown
Proposed action years 0-10	0.985535	N/A	157	9.14
Proposed action years 10-50	0.989003	N/A	102	Less than 9.14, but % unknown
No federal fisheries	0.989742	97.72	108	0

Entanglement mortalities affect successful reproduction by reducing the number of sexually mature individuals producing viable offspring. Although lifetime reproductive success can be highly variable, based on historic calving intervals (Pettis et al. 2017), a healthy female can produce up to 11 calves in 50 years. Small populations are inherently at risk of extinction, in part, because of the unequal reproductive success of individuals within the population, such that some individuals produce more offspring than others (Coulson et al. 2006). Random chance can also affect the sex ratio and genetic diversity of a small population, leading to lowered reproductive success of the population as a whole. The smaller the population, the more weight an individual's reproductive success has on the population's growth or decline and a greater chance that random variation will result in too few individuals to maintain the population (Coulson et al. 2006). Additionally, years with a small number of births can contribute to a declining trend or a halt in growth rate for multiple years indicating reproductive variability has a noticeable effect on species viability (Hayes et al. 2018a, Pace et al. 2017). However, the projections use the calving rates from 2010-2019 and assume these rates will continue into the future. This gives the benefit of the doubt to the species by using years with a low number of births.

The projections show, even in the absence of the federal fisheries, a declining right whale female population over the 50 year time period, with a 97.72 percent chance of a declining trend, and an estimated loss of 71 females (Table 82). With the implementation of the proposed action, including the Framework, the projections indicate that the female right whale population will decline initially and then continue to decline but at a slower rate that is comparable to the trajectory with no effects from the federal fisheries (Table 82). Under the proposed action, there is a 98.8 percent chance of a declining trend and an estimated loss of 77 females from the current female population (n=179) over the 50-year period. As there is a decrease of 71 whales over the 50 years in the absence of M/SI in federal fisheries, there is a difference of 6 females at year 50. The total loss of the females in both scenarios is due to the M/SI occurring from all sources, including M/SIs in the federal fisheries. As the population declines, the difference of 6 females remains over the next 40 years of the projections, fluctuating throughout the years between 6 and 8.

The population projections indicate that after 50 years there would be a difference of approximately 6 females between Scenario 1 (no federal fisheries) and Scenario 2 (proposed action). Assuming that females represent approximately 40 percent of the population, this results in a difference of approximately 15 (male and female) whales between the two trajectories. This differs from the estimate above that 23.93 whales will be seriously injured or die over 50 years (approximately 18.5 M/SI in the first 10 years) due to the operation of the fisheries and the implementation of the Framework. The projections indicate a lesser reduction than what was estimated from the data given that they use a per capita mortality rate and consider how the projected population size and per capita rates interact across time.

With the full implementation of the Framework, the reductions in the number of whales that die or are seriously injured contributes positively to the parameters included in the model as the whales that survive to contribute to the population. This is more reflective of what would be expected to occur with the implementation of the Framework as it accounts for these contributions to the population. Although the proposed action will likely reduce the number of individual right whales compared to the no federal fisheries scenario, the projections indicate the female population declines after the implementation of the Framework at a comparable rate to the no federal fisheries scenario. While M/SI associated with the proposed action occur at a higher rate during the first 10 years, the projections indicate that these M/SIs will not increase the population's rate of decline compared to the no federal fisheries scenario after the implementation of the full Framework.

Sublethal Effects Analysis
As described in the *Effects of the Proposed Action*, it is likely non-lethal entanglement in fishing gear may negatively affect the health or body condition of a whale. However, these effects cannot be separated out from the effects of multiple stressors (e.g., prey abundance, climate variation, reproductive state, exposure to harmful algal blooms, vessel collisions) that co-occur and, individually and cumulatively, can affect the health of animals, and, subsequently, the calving rate of the population. While we believe, based on the best available scientific and commercial data described below, some portion of the observed variability in right whale calving rates is due to the sublethal effects of entanglements, we cannot quantify the degree to which entanglements are affecting calving rates.

During the first 10 years of the proposed action, the operation of the federal fisheries is likely to contribute to decreased calving rates due to the sublethal effects. As described in the *Effects of the Proposed Action*, given that some of the risk reduction measures in Phase 1 and 2 are designed to reduce the severity of entanglements and not the likelihood, the federal fisheries are expected to entangle an annual average of 9.14 percent of the right whale population during the first 5 years. After the implementation of Phases 3 and 4, we expect that substantially less than 9.14 percent of the population will become entangled. However, we cannot quantify how much less without more information. Some animals will shed the gear themselves; others will continue to carry trailing gear, with some proportion of these being disentangled by trained responders. We anticipate that most of the right whales that do not die as a result of their injuries may experience varying levels of sublethal effects from the exposure to entanglement. These effects range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. These negative effects include increased energetic demands due to increased drag, foraging being impeded, and stress (Hayes et al. 2018a). Drag from fishing gear may also reduce the female reproductive energy budget, extending the time of gaining the energy needed for reproduction by months to years (van der Hoop et al. 2017a).

In order to achieve the risk reduction requirements in Phase 3 and 4 of the Framework, the federal fisheries will need to implement measures that will substantially reduce the co-occurrence of right whales and vertical lines. This reduction in co-occurrence is expected to not only reduce M/SI resulting from entanglements but also reduce entanglements that do not result in M/SI. As such, there would be fewer entanglements overall and, therefore, fewer sublethal effects to right whales. We also expect calving rates would likely improve following the implementation of the Framework (proposed action scenario) as sublethal effects will be reduced; however, the degree to which calving rates may change cannot be estimated. Similarly, we would expect that calving rates would increase in the absence of the fishery (no federal fisheries scenario). Although we currently cannot quantify the degree to which entanglements are affecting calving rates, average calving intervals have increased from 4 years in 2009 to 10 years in 2017, and this may be due, in some part, to stress from entanglement (Pettis et al. 2018). However, we believe that under current conditions, the entanglements that will occur due to the operation of the fisheries before the full implementation of the Framework will not reduce the calving rates beyond the levels observed from 2010-2019. After the full implementation of the Framework, we expect to see a decrease in the average calving interval and higher calving rates than the levels observed from 2010-2019. Given that this potential increase in calving is not considered in the population projections, both scenarios are conservative. That is, it is likely that both projections are an underestimate, and the right whale population would fare better than the trajectories (Figure 73) indicate if calving rates increase.

Genetics Analysis

Population growth is important because of the influence of demographic and individual heterogeneity on a population's long-term viability (Fagan and Holmes 2006). The larger the population size, the greater the buffer against stochastic events. As the population decreases further, the population may be at risk of becoming so small that the genetic make-up of the remaining individuals is not the same as the initial population. This is known as a genetic bottleneck and when this occurs, the species becomes less resilient which can increase risk of extinction (Hayes et al. 2018a). The proposed action is expected to reduce the female right whale population by 6 compared to the federal fisheries scenario. However, 6 females represents 3.35 percent of the current female population. The loss of 3.35 percent of the female population, when examined individually, is not be expected to be responsible for a genetic bottleneck in the population. The proposed action implements the Framework early enough for the population's trajectory to essentially match the rate of decline to the no federal fisheries scenario after the initial losses while the Framework is implemented. Similar population trajectories after year 10 indicate that the proposed action would not substantially contribute to the loss of genetic heterogeneity to the point of the population being at risk of a genetic bottleneck.

Quasi-Extinction

We also considered whether a quasi-extinction threshold could be used to help assess whether the proposed action is appreciably reducing the likelihood of survival and recovery of the species. There is no defined threshold for quasi-extinction of North Atlantic right whales. As described in the *Status of the Species*, the western population may have numbered fewer than 100 individuals in 1935, when international protection for right whales came into effect (Kenney et al. 1995). Additionally, right whale abundance was estimated to be 162 animals in 1980 and 270 animals in 1990, and abundance increased by approximately 2.8 percent per year from 1990 until 2011 (Pace et al. 2017). Although the current estimate of 368 individuals in the population is extremely low, these prior estimates suggest the population is capable of recovering at lower levels than the current estimate. Using the population projection model, we evaluated whether prior estimates could be used as a quasi-extinction threshold given that after reaching these levels, the right whale population was capable of achieving an increasing trajectory, showing that recovery from these numbers was possible with the environmental conditions and anthropogenic risks present during that period. The female population in 1980 was estimated to be 63 females (total population 162:63 females, 82 males, 17 unknown). We evaluated the likelihood that the populations would reach this level in 50 years. Under the proposed action, there is a 5.6 percent probability the population will reach this level in 50 years. Under the no federal fisheries scenario, there is a 3.5 percent probability the population will reach this level in 50 years. Therefore, the likelihood of reaching this level is similar between the two scenarios and is low. Our analysis suggests that with or without the proposed action, it is extremely unlikely that in 50 years, the female right whale population would decline to the level of the 1980 female population estimate. However, we determined that our analysis could not rely on this threshold as the level from which right whales would maintain their ability to recover given that the environmental conditions and anthropogenic risks have changed since that time.

Current Conditions

Current conditions that continue to act on the species, like the effect of Canadian fisheries and vessel strikes in the United States and Canada, puts this population at high risk of extinction, and as a result the continuing declining trend, with or without the action, is of particular concern. We ran the population projection model to assess the level of overall risk reduction needed to result in an increasing population trajectory. The results indicate that even with a very high level of risk reduction in the United States, the population trajectory will not increase if right whale mortalities continue to occur at current levels in Canadian waters.

We evaluated the population trajectory if, in addition to the proposed action, similar reductions in M/SI to those in the Framework are implemented to reduce M/SI from all sources in Canada. As we cannot partition out vessel strikes and fishery entanglements in Canada, we first calculated the percent reduction for all U.S. sources (state and federal fisheries, vessels strikes) at each phase. We then assumed that this percent reduction would be applied to all M/SI in Canada at the same time as the implementation of the phases in the Framework. We then projected the population trajectory with total M/SI in Canadian waters (from all sources) given these reductions in Canada and in the United States fisheries. The figure below (Figure 74) depicts the population trajectory if both the United States and Canada reduce mortalities at the same level. Under this scenario, the reductions would result in an increasing population trajectory, with a 37.96 percent probability of a declining trend, and an increase of 13 females over 50 years. All scenarios are expected to result in an increase in calving. This increase is not considered in the population projections; therefore, the three scenarios representing the implementation of measures to reduce M/SI are conservative. That is, it is likely that the projections underestimate the likelihood of an increasing right whale population and that the actual right whale population will likely fare better than the trajectories indicate.

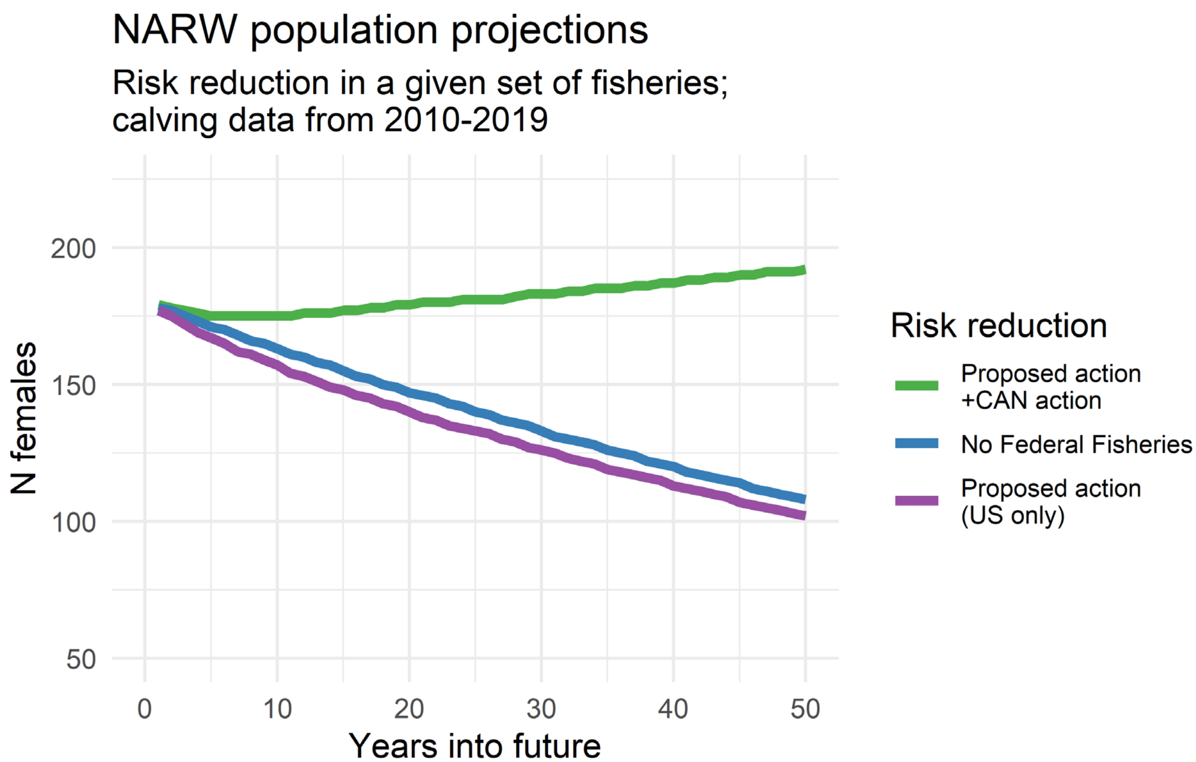

Figure 74: Population projections (n=5000) of North Atlantic right whale females using demographic rates from 2010–2019. The projections include the proposed action and hypothetical, equivalent measures in Canada. The 100 percent U.S. federal projection is with no risk from federal fisheries (in blue) and includes the risk reduction from the future ALWTRP reductions in state waters. The proposed action (U.S only) (in purple) includes risk reduction from federal fisheries with implementation of the ALWTP proposed measures/Framework and risk reduction in U.S. state waters with the ALWTRP proposed measures in place. The proposed action +CAN action (in green) includes risk reduction from federal fisheries with the ALWTRP proposed measures/Framework in place, risk reduction in U.S. state fisheries with ALWTRP proposed measures in place, and identical risk reduction percentages in Canada from all sources.

Determination

Based on our analysis, we expect that with the proposed action, the status and trend of the population of right whales would decline during the first 10 years and with the implementation of the Framework, continue to decline but at a rate comparable to the no federal fisheries scenario. The proposed action simulations show that a declining trend is 1.08 percent more likely compared to no federal fisheries scenario over the 50 year time period (Table 82), and results in 6 fewer females. The difference between these two scenarios is caused by M/SI during the first 10 years, and the rate of decline after year 10 in both scenarios is essentially the same. Our projections show that the probability of the species continuing to decline, with or without the proposed action, is extremely high (98.8 percent and 97.72 percent) respectively. Additionally, the projections indicate a difference of 6 females (15 right whales) between the proposed action and no federal fishery over 50 years, which is not expected to be responsible for a genetic bottleneck in the population. However, as discussed above, the model was designed to facilitate relative comparisons of potential futures under differing management regimes, and the projections were generated utilizing worst case assumptions for several key variables, so these projections should not be interpreted as an accurate predictor of the actual future right whale population.

The results of the projections and information on how non-lethal entanglements may affect calving rates represent the best available information to determine whether the proposed action is likely or not likely to jeopardize the continued existence of right whales. Given that the projections show a similar decreasing population trend for both the proposed action and no federal fisheries scenarios, and that we expect calving rates to increase at a similar rate in both scenarios, we believe the proposed action will not have biologically meaningful impacts on the overall reproduction, numbers and distribution of right whales in the wild.

Based on our analysis, the nature and magnitude of the proposed action's effects, when considered together with the species status and all other threats acting on it, would have inconsequential impacts on the species' overall reproduction, numbers and distribution in the wild. Given all of the available data, we conclude that right whale entanglements due to the operation of the federal fisheries will not result in an appreciable reduction in the likelihood of survival and recovery of North Atlantic right whales compared to the no federal fishery scenario. Therefore, we believe that the proposed action, including the implementation of the Framework, is not likely to jeopardize the continued existence of North Atlantic right whales.

Recovery Assessment
In addition to analyzing the effects of the action on survival of right whales, we are required to consider what impacts it will have on recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer warranted. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., "endangered"), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., "threatened") because of any of the following five listing factors: (1) the present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The goal of the 2005 revised Recovery Plan for North Atlantic Right Whale is to recover North Atlantic right whales to a level sufficient to warrant their removal from the List of Endangered and Threatened Wildlife and Plants under the ESA. The intermediate goal is to reclassify the species from endangered to threatened. The revised Recovery Plan states that North Atlantic right whales may be considered for reclassifying to threatened when all of the following have been met:

1. the population ecology (range, distribution, age structure, and gender ratios, etc.) and vital rates (age-specific survival, age-specific reproduction, and lifetime reproductive success) of right whales are indicative of an increasing population.
2. the population has increased for a period of 35 years at an average rate of increase equal to or greater than 2 percent per year.
3. none of the known threats to NARWs (summarized in the five listing factors in the recovery plan) are known to limit the population's growth rate.

All of these address the need for an increasing population growth rate. While the proposed action does not result in an increasing growth rate, the full implementation of the Framework is expected to further the objectives to obtain an increasing growth rate. We do not believe that the proposed action will impede progress on carrying out any aspect of the recovery plan or achieving the overall recovery strategy. Therefore, the effects of the action are not expected to appreciably impact the North Atlantic right whale population's ecology or vital rates. Over the 50 year time period, the federal fisheries are likely to have minimal impact on these aspects of the population. The removal of six female right whales during the first 10 years, followed by a comparable trend to the no federal fishery trajectory during the following 40 years is not expected to have consequential effects on the average right whale population trend.

The Recovery Plan also lists the objective that:

4. given current and projected threats and environmental conditions, the right whale population has no more than a 1 percent chance of quasi-extinction in 100 years.

Above, we determined that the mortality of North Atlantic right whales associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the species, and we do not expect the proposed action to have consequential effects on NARW population potential for recovery. Although the current population estimate for right whales is extremely low, the population estimate has been much lower in the past, suggesting that the population is capable of recovering at lower levels than the current estimate. Therefore, we believe that the proposed action will not increase the chances of quasi-extinction in 100 years.

Conclusion

In conclusion, we believe that the lethal and nonlethal takes of North Atlantic right whales associated with the proposed action that includes implementation of the Framework, when considered together with the species status and all other threats acting on it, are not expected to cause an appreciable reduction in the likelihood of both the survival and recovery of the species in the wild. The impacts from the continued authorization of the fisheries will not appreciably affect the population's persistence into the future or its potential for recovery.

9.2. Fin Whale

As described in the *Effects of the Proposed Action*, we anticipate the annual average of 1.89 fin whale entanglements, of which, 1.08 are expected to result in M/SI. No vessel strikes of fin whales are anticipated. Entanglement in fishing gear and vessel strikes, as described in the *Environmental Baseline*, may occur in the action area. As noted in the *Cumulative Effects* section of this Opinion, we have not identified any cumulative effects different from those considered in the *Status of the Species* and *Environmental Baseline* sections of this Opinion, inclusive of how those activities may contribute to climate change. As described in the *Climate Change* section, climate change may result in changes in the distribution or abundance of fin whales in the action area; however, we have not identified any different or exacerbated effects of the action in the context of anticipated climate change.

An estimated 1.89 entanglements (1.08 lethal) annually as a result of the proposed action may reduce the number of fin whales in the population, compared to their numbers in the absence of the proposed action. This would result in a reduction in future reproduction, assuming the individual was a female and would have survived otherwise to reproduce. If it was a male, the genetic contribution from that individual would be lost. Whether this reduction in numbers and reproduction would appreciably reduce the likelihood of survival of the fin whales depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

As described in further detail in the *Status of the Species*, of the three to seven stocks thought to occur in the North Atlantic Ocean, one occurs in U.S. waters, where NMFS' best estimate of abundance is 7,418 individuals (Hayes et al. 2020). According to the latest NMFS stock assessment report for fin whales in the Western North Atlantic, information is not available to conduct a trend analysis for this population (Hayes et al. 2020). Across the range, there are over 100,000 fin whales occurring primarily in the North Atlantic Ocean, North Pacific Ocean, and Southern Hemisphere. While uncertainty surrounds the population trend of this species, we must determine whether the takes under the proposed action are too high to allow survival and recovery of the species given its current status and uncertain population trajectory. We must evaluate whether the effects of the fishery as now proposed, considered in context of the environmental baseline and cumulative effects, are expected to appreciably reduce the likelihood of both the survival and the recovery of species in the wild. To try to answer this question, we examined the total population size relative to anticipated take levels, taking into account the period over which the take would occur.

As described in the *Effects of the Proposed Action*, although it is likely non-lethal entanglement in fishing gear may negatively affect the health or body condition of a whale, multiple stressors (e.g., prey abundance, climate variation, reproductive state, exposure to harmful algal blooms, vessel collisions) co-occur and, individually and cumulatively, can affect the health of animals, and, subsequently, the calving rate of the population. However, based on the best available scientific and commercial data, we believe sublethal effects of entanglement may contribute to a decrease in calving rates, but we cannot quantify the degree to which entanglements are affecting calving rates.

The operation of the federal fisheries may contribute to decreased calving rates due to the sublethal effects of entanglement. The federal fisheries are expected to entangle an annual average of 1.89 fin whales. Some animals will shed the gear themselves; others will continue to carry trailing gear. We anticipate that most of the fin whales that do not die as a result of their injuries may experience varying levels of sublethal effects from the exposure to entanglement. These effects range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. As described in section 7.2, non-lethal entanglements may negatively affect the health or body condition of a whale, increase energetic demands, reduce female's reproductive energy budget, impede foraging, and increase stress hormones (Hayes et al. 2018a, van der Hoop et al. 2017a). Although we currently cannot quantify the degree to which entanglements may affect calving rates, we expect that a portion of animals that survive entanglement may experience fitness level impacts that could lead to a decreased calving rate in the population. Even though some individual whales are expected to experience a reduction in fitness, we would not expect such impacts to have meaningful effects at the population level given the current status of the fin whale population that will be exposed. However, we believe the low number of entanglements that are anticipated will not reduce the calving rates of fin whales. For this reason, we do not anticipate that the sublethal effects to fin whales will result in changes in the number, distribution, or reproductive potential of fin whales in the North Atlantic.

The annual mortality of 1.08 fin whales from the proposed action would represent approximately 0.01 percent (=1.08/7418)*100) of the current estimate of 7,418 fin whales in the North Atlantic Ocean. This calculation does not account for any additional deaths of dependent calves that may result from mothers that were entangled and subsequently died. We did not have enough information on fin whale demographics within the U.S. and their entanglement rates to make assumptions regarding dependent calf deaths in the calculations. However, the estimated entanglement (see section 7.2) was based on calculations that used a conservative overestimate of apportionment of fin whale entanglements to the U.S. federal fisheries (e.g., unknown country of origin assumed to be U.S.). This conservative overestimate qualitatively provides some buffer to the impact of the action from the possible loss of a dependent calf. The M/SI of 1.08 fin whales (adult, juvenile, or adult/with calf) annually is very small and contributes only minimally to the overall mortality on the population. We believe that the resulting mortality of fin whales associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the species, and we expect the fin whale population to remain large enough and to retain the potential for recovery. We believe entanglement in fishing gear is still a threat, and efforts to reduce interactions are key to conservation of the species. The effects of the proposed action will most directly affect the overall size of the population, which we believe is currently sufficiently large to withstand this very low level of impact, and the proposed action will not cause the population to lose genetic heterogeneity, broad demographic representation, or successful reproduction, nor affect the species ability to meet its lifecycle requirements, including reproduction, sustenance, and shelter.

The proposed action is not likely to affect the recovery potential of fin whales. The 2010 Recovery Plan for fin whales included two criteria for consideration for reclassifying the species from endangered to threatened:

1. Given current and projected threats and environmental conditions, the fin whale population in each ocean basin in which it occurs (North Atlantic, North Pacific and Southern Hemisphere) satisfies the risk analysis standard for threatened status (has no more than a 1 percent chance of extinction in 100 years) and has at least 500 mature, reproductive individuals (consisting of at least 250 mature females and at least 250 mature males) in each ocean basin. Mature is defined as the number of individuals known, estimated, or inferred to be capable of reproduction. Any factors or circumstances that are thought to substantially contribute to a real risk of extinction that cannot be incorporated into a Population Viability Analysis will be carefully considered before downlisting takes place.
2. None of the known threats to fin whales are known to limit the continued growth of populations. Specifically, the factors in 4(a)(l) of the ESA are being or have been addressed: A) the present or threatened destruction, modification or curtailment of a species' habitat or range; B) overutilization for commercial, recreational or educational purposes; C) disease or predation; D) the inadequacy of existing regulatory mechanisms; and E) other natural or manmade factors.

The proposed action will not result in any condition that impacts the time it will take to reach these goals or the likelihood that these goals will be met given that the proposed action will not affect the trend of the species or prevent or delay it from achieving an increasing population or the species growth rate and will not affect the chance of extinction.

Based on this analysis, the proposed action is not likely to result in an appreciable reduction in the likelihood of survival and recovery of fin whales in the wild. These conclusions were made in consideration of the endangered status of fin whales, other stressors that individuals are exposed to within the action area as described in the *Environmental Baseline* and *Cumulative Effects*, and any anticipated effects of climate change on the abundance and distribution of fin whales in the action area.

9.3. Sei Whale

As described in the *Effects of the Proposed Action*, we anticipate the annual average of one sei whale entanglement, which may result in M/SI. No vessel strikes of sei whales are anticipated. Entanglement in fishing gear and vessel strikes as described in the *Environmental Baseline*, may occur in the action area. As noted in the *Cumulative Effects* section of this Opinion, we have not identified any cumulative effects different than those considered in the *Status of the Species* and *Environmental Baseline* sections of this Opinion, inclusive of how those activities may contribute to climate change. As described in the *Climate Change* section, climate change may result in changes in the distribution or abundance of sei whales in the action area; however, we have not identified any different or exacerbated effects of the action in the context of anticipated climate change.

An estimated one entanglement annually as a result of the proposed action may reduce the number of sei whales in the population, compared to their numbers in the absence of the proposed action. This would result in a reduction in future reproduction, assuming the individual was a female and would have survived otherwise to reproduce. If it was a male, the genetic contribution from that individual would be lost. Whether this reduction in numbers and reproduction would appreciably reduce the likelihood of survival of the fin whales depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

As described in the *Status of the Species*, the best abundance estimate for the Nova Scotia stock of sei whales is 6,292 animals, though the abundance survey from which this estimate was derived excluded waters off the Scotian Shelf, an area encompassing a large portion of the stock's range (Hayes et al. 2020). According to the latest NMFS stock assessment report for sei whales in the western North Atlantic, there are insufficient data to determine population trends for sei whales (Hayes et al. 2020). Across its range, it is estimated that there are over 50,000 sei whales. In the North Pacific, an abundance estimate for the entire North Pacific population of sei whales is not available. However, in the western North Pacific, it is estimated that there are 35,000 sei whales (Cooke 2018a). In the eastern North Pacific (considered east of longitude 180º), two stocks of sei whales occur in U.S. waters: Hawaii and Eastern North Pacific. Abundance estimates for the Hawaii stock are 391 sei whales (N_{min}=204), and for Eastern North Pacific stock, 519 sei whales (N_{min}=374) (Carretta et al. 2019a). In the Southern Hemisphere, recent abundance of sei whales is estimated at 9,800 to 12,000 whales.

While uncertainty surrounds the population trend of this species, we must determine whether the takes under the proposed action are too high to allow survival and recovery given the current status of the species and uncertain population trajectory. We must evaluate whether the effects of the fishery as now proposed, considered in context of the environmental baseline and cumulative effects, are expected to appreciably reduce the likelihood of both the survival and the recovery of species in the wild. To try to answer this question, we examined the total population size relative to anticipated take levels, taking into account the period over which the take would occur.

As described in the *Effects of the Proposed Action*, although it is likely non-lethal entanglement in fishing gear may negatively affect the health or body condition of a whale, multiple stressors (e.g., prey abundance, climate variation, reproductive state, exposure to harmful algal blooms, vessel collisions) co-occur and, individually and cumulatively, can affect the health of animals, and, subsequently, the calving rate of the population. However, based on the best available scientific and commercial data, we believe sublethal effects of entanglement may contribute to a decrease in calving rates, but we cannot quantify the degree to which entanglements are affecting calving rates.

The operation of the federal fisheries may contribute to decreased calving rates due to the sublethal effects of entanglement. The federal fisheries are expected to entangle an annual average of 1 sei whale. Some animals will shed the gear themselves; others will continue to carry trailing gear. We anticipate that most of the sei whales that do not die as a result of their injuries may experience varying levels of sublethal effects from the exposure to entanglement. These effects range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. As described in section 7.2, non-lethal entanglements may negatively affect the health or body condition of a whale, increase energetic demands, reduce female's reproductive energy budget, impede foraging, and increase stress hormones (Hayes et al. 2018a, van der Hoop et al. 2017a). Although we currently cannot quantify the degree to which entanglements may affect calving rates, we expect that a portion of animals that survive entanglement may experience fitness level impacts that could lead to a decreased calving rate in the population. Even though some individual whales are expected to experience a reduction in fitness, we would not expect such impacts to have meaningful effects at the population level given the current status of the sei whale population that will be exposed. However, we believe the low number of entanglements that are anticipated will not reduce the calving rates of sei whales. For this reason, we do not anticipate that the sublethal effects to sei whales will result in changes in the number, distribution, or reproductive potential of sei whales in the North Atlantic.

The annual mortality of 1 sei whale from the proposed action would represent approximately 0.01 percent (=1/6292)*100) of the current estimate of 6,292 sei whales in the North Atlantic Ocean. This calculation does not account for any additional deaths of dependent calves that may result from mothers that were entangled and subsequently died. We did not have enough information on sei whale demographics within the U.S. and their entanglement rates to make assumptions regarding dependent calf deaths in the calculations. However, the entanglement estimate (see 7.2.1.5) was based on calculations that used a conservative overestimate of apportionment of sei whale entanglements to the U.S. federal fisheries (e.g., likely undocumented events assumed to be U.S.). This conservative overestimate qualitatively provides some buffer to the impact of the action from the possible loss of a dependent calf. The M/SI of 1 sei whale (adult, juvenile, or adult/with calf) annually is very small and contributes only minimally to the overall mortality of the population. We believe that the resulting mortality of sei whales associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the species, and we expect the sei whale population to remain large enough and to retain the potential for recovery. We believe entanglement in fishing gear is still a threat, and efforts to reduce interactions are key to conservation of the species. The effects of the proposed action will most directly affect the overall size of the population, which we believe is currently sufficiently large to withstand this very low level of impact, and the proposed action will not cause the population to lose genetic heterogeneity, broad demographic representation, or successful reproduction, nor affect the species ability to meet its lifecycle requirements, including reproduction, sustenance, and shelter.

The proposed action is not likely to affect the recovery potential of sei whales. The 2011 Recovery Plan for sei whales included two criteria for consideration for reclassifying the species from endangered to threatened:

1. Given current and projected threats and environmental conditions, the sei whale population in each ocean basin in which it occurs (North Atlantic, North Pacific and Southern Hemisphere) satisfies the risk analysis standard for threatened status (has no more than a 1 percent chance of extinction in 100 years) and the global population has at least 1,500 mature, reproductive individuals (consisting of at least 250 mature females and at least 250 mature males in each ocean basin). Mature is defined as the number of individuals known, estimated, or inferred to be capable of reproduction. Any factors or circumstances that are thought to substantially contribute to a real risk of extinction that cannot be incorporated into a Population Viability Analysis will be carefully considered before downlisting takes place.
2. None of the known threats to sei whales are known to limit the continued growth of populations. Specifically, the factors in 4(a)(l) of the ESA are being or have been addressed: A) the present or threatened destruction, modification or curtailment of a species' habitat or range; B) overutilization for commercial, recreational or educational purposes; D) the inadequacy of existing regulatory mechanisms; and E) other natural or manmade factors (there are no criteria for Factor C, disease or predation).

The proposed action will not result in any condition that impacts the time it will take to reach these goals or the likelihood that these goals will be met. This is because the proposed action will not affect the trend of the species or prevent or delay it from achieving an increasing population or otherwise affect the number of individuals or the species growth rate and will not affect the chance of extinction.

Based on this analysis, the proposed action is not likely to result in an appreciable reduction in the likelihood of survival and recovery of sei whales in the wild. These conclusions were made in consideration of the endangered status of sei whales, other stressors that individuals are exposed to within the action area as described in the *Environmental Baseline* and *Cumulative Effects*, and any anticipated effects of climate change on the abundance and distribution of sei whales in the action area.

9.4. Sperm Whale

As described in the *Effects of the Proposed Action*, we anticipate the annual average of one sperm whale entanglement, which may result in M/SI. No vessel strikes of sperm whales are anticipated. Entanglement in fishing gear and vessel strikes as described in the *Environmental Baseline*, may occur in the action area. As noted in the *Cumulative Effects* section of this Opinion, we have not identified any cumulative effects different than those considered in the *Status of the Species* and *Environmental Baseline* sections of this Opinion, inclusive of how those activities may contribute to climate change. As described in the *Climate Change* section, climate change may result in changes in the distribution or abundance of sperm whales in the action area; however, we have not identified any different or exacerbated effects of the action in the context of anticipated climate change.

An estimated one entanglement annually as a result of the proposed action may reduce the number of sperm whales in the population, compared to their numbers in the absence of the proposed action. This would result in a reduction in future reproduction, assuming the individual was a female and would have survived otherwise to reproduce. If it was a male, the genetic contribution from that individual would be lost. Whether this reduction in numbers and reproduction would appreciably reduce the likelihood of survival of the sperm whales depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

As described in the *Status of the Species*, the best abundance estimate for the North Atlantic stock is 4,349 individuals (Hayes 2019). There are no reliable estimates for sperm whale abundance across the entire Atlantic Ocean, however, the most recent global sperm whale population estimate is 360,000 whales (Whitehead 2009). While uncertainty surrounds the population trend of this species, we must determine whether the takes under the proposed action are too high to allow survival and recovery given the current status of the species and uncertain population trajectory. We must evaluate whether the effects of the fishery as now proposed, considered in context of the environmental baseline and cumulative effects, are expected to appreciably reduce the likelihood of both the survival and the recovery of species in the wild. To try to answer this question, we examined the total population size relative to anticipated take levels, taking into account the period over which the take would occur.

As described in the *Effects of the Proposed Action*, although it is likely non-lethal entanglement in fishing gear may negatively affect the health or body condition of a whale, multiple stressors (e.g., prey abundance, climate variation, reproductive state, exposure to harmful algal blooms, vessel collisions) co-occur and, individually and cumulatively, can affect the health of animals, and, subsequently, the calving rate of the population. However, based on the best available scientific and commercial data, we believe sublethal effects of entanglement may contribute to a decrease in calving rates, but we cannot quantify the degree to which entanglements are affecting calving rates.

The operation of the federal fisheries may contribute to decreased calving rates due to the sublethal effects of entanglement. The federal fisheries are expected to entangle an annual average of 1 sperm whale. Some animals will shed the gear themselves; others will continue to carry trailing gear. We anticipate that most of the sperm whales that do not die as a result of their injuries may experience varying levels of sublethal effects from the exposure to entanglement. These effects range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. As described in section 7.2, non-lethal entanglements may negatively affect the health or body condition of a whale, increase energetic demands, reduce female's reproductive energy budget, impede foraging, and increase stress hormones (Hayes et al. 2018a, van der Hoop et al. 2017a). Although we currently cannot quantify the degree to which entanglements may affect calving rates, we expect that a portion of animals that survive entanglement may experience fitness level impacts that could lead to a decreased calving rate in the population. Even though some individual whales are expected to experience a reduction in fitness, we would not expect such impacts to have meaningful effects at the population level given the current status of the sperm whale population that will be exposed. However, we believe the low number of entanglements that are anticipated will not reduce the calving rates of sperm whales. For this reason, we do not anticipate that the sublethal effects to sperm whales will result in changes in the number, distribution, or reproductive potential of sperm whales in the North Atlantic.

The annual mortality of 1 sei whale from the proposed action would represent approximately 0.02 percent (=1/4349)*100) of the current estimate of 4,349 sperm whales in the North Atlantic Ocean. This calculation does not account for any additional deaths of dependent calves that may result from mothers that were entangled and subsequently died. We did not have enough information on sperm whale demographics within the United States and their entanglement rates to make assumptions regarding dependent calf deaths in the calculations. However, the entanglement (section 7.2) was based on calculations that used a conservative overestimate of apportionment of sperm whale entanglements to the U.S. federal fisheries (e.g., likely undocumented events assumed to be U.S.). This conservative overestimate qualitatively provides some buffer to the impact of the action from the possible loss of a dependent calf. The M/SI of 1 sperm whale (adult, juvenile, or adult/with calf) annually is very small and contributes only minimally to the overall mortality on the population. We believe that the resulting mortality of sperm whales associated with the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the species, and we expect the sperm whale population to remain large enough and to retain the potential for recovery. We believe entanglement in fishing gear is still a threat, and efforts to reduce interactions are key to conservation of the species. The effects of the proposed action will most directly affect the overall size of the population, which we believe is currently sufficiently large to withstand this very low level of impact, and the proposed action will not cause the population to lose genetic heterogeneity, broad demographic representation, or successful reproduction, nor affect the species ability to meet its lifecycle requirements, including reproduction, sustenance, and shelter.

The proposed action is not likely to affect the recovery potential of sperm whales. The 2010 Recovery Plan states that sperm whales may be considered for reclassifying to threatened when all of the following have been met:

1. Given current and projected threats and environmental conditions, the sperm whale population in each ocean basin in which it occurs (Atlantic Ocean/Mediterranean Sea, Pacific Ocean, and Indian Ocean) satisfies the risk analysis standard for threatened status (has no more than a 1 percent chance of extinction in 100 years) and the global population has at least 1,500 mature, reproductive individuals (consisting of at least 250 mature females and at least 250 mature males in each ocean basin). Mature is defined as the number of individuals known, estimated, or inferred to be capable of reproduction. Any factors or circumstances that are thought to substantially contribute to a real risk of extinction that cannot be incorporated into a Population Viability Analysis will be carefully considered before downlisting takes place.
2. None of the known threats to sperm whales is known to limit the continued growth of populations. Specifically, the factors in 4(a)(l) of the ESA are being or have been addressed: A) the present or threatened destruction, modification or curtailment of a species' habitat or range; B) overutilization for commercial, recreational or educational purposes; C) disease or predation; D) the inadequacy of existing regulatory mechanisms; and E) other natural or manmade factors.

The proposed action will not result in any condition that impacts the time it will take to reach these goals or the likelihood that these goals will be met. This is because the proposed action will not affect the trend of the species or prevent or delay it from achieving an increasing population or otherwise affect its growth rate and will not affect the chance of extinction.

Based on this analysis, the proposed action is not likely to result in an appreciable reduction in the likelihood of survival and recovery of sperm whales in the wild. These conclusions were made in consideration of the endangered status of sperm whales, other stressors that individuals are exposed to within the action area as described in the *Environmental Baseline* and *Cumulative Effects*, and any anticipated effects of climate change on the abundance and distribution of sperm whales in the action area.

9.5. Green Sea Turtle, North Atlantic DPS

The North Atlantic DPS of green sea turtles is listed as threatened under the ESA. As is the case with the other three sea turtle species addressed in this Opinion, North Atlantic DPS of green sea turtles face numerous threats on land and in the water that affect the survival of all age classes.

There are four regions that support high nesting concentrations in the North Atlantic DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo); United States (Florida), and Cuba. Using data from 48 nesting sites in the North Atlantic DPS, nester abundance which was estimated at 167,528 total nesters (Seminoff et al. 2015). The years used to generate the estimate varied by nesting site but were between 2005-2012. The largest nesting site (Tortuguero, Costa Rica) hosts 79 percent of the estimated nesting. It should be noted that not all female turtles nest in a given year (Seminoff et al. 2015). Nesting in the area has increased considerably since the 1970s, and nest count data from 1999-2003 suggested that 17,402-37,290 females nested there per year (Seminoff et al. 2015). In 2010, an estimated 180,310 nests were laid at Tortuguero, the highest level of green sea turtle nesting estimated since the start of nesting track surveys in 1971. This equated to somewhere between 30,052 and 64,396 nesters in 2010 (Seminoff et al. 2015). Nesting sites in Cuba, Mexico, and the United States were either stable or increasing (Seminoff et al. 2015). More recent data is available for the southeastern United States. Nest counts at Florida's core index beaches have ranged from less than 300 to almost 41,000 in 2019. The INBS is carried out on a subset of beaches surveyed during the SNBS and is designed to measure trends in nest numbers. The nest trend in Florida shows the typical biennial peaks in abundance and has been increasing (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/; Figure 37). The SNBS is broader but is not appropriate for evaluating trends. In 2019, approximately 53,000 green turtle nests were recorded in the SNBS (https://myfwc.com/research/wildlife/sea-turtles/nesting/). Seminoff et al. (2015) estimated total nester abundance for Florida at 8,426 turtles.

NMFS recognizes that the nest count data available for green sea turtles in the Atlantic indicates increased nesting at many sites. However, NMFS also recognizes that the nest count data, including data for green sea turtles in the Atlantic, only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females available to nest or the number of immature females that will reach maturity and nest in the future.

Green sea turtles have been observed to interact with both gillnet and bottom trawl gear used in the ten fisheries that are the focus of this Opinion. Based on information from Murray (2018, 2020), Linden (2020), and data from NEFOP, ASM, and the GAR STDN, we anticipate 42 green sea turtles will interact with gear utilized in the ten fisheries assessed in this Opinion every five years. Green sea turtles that interact with gear used in these fisheries (which for the purposes of this Opinion includes gillnet, bottom trawl, hook gear, and trap/pot gear only) are those that are captured or entangled in the gear. An estimated ten green sea turtles are expected to interact with gillnet gear every five years based on recent observer data from the NEFOP and ASM programs. An estimated 32 green sea turtles are expected to interact with bottom trawl gear every five years, based on the interaction rates in Murray (2020) and federal waters take apportionment (Linden 2020). No green sea turtles are expected to interact with trap/pot gear in the lobster, red crab, Jonah crab, black sea bass, and scup fisheries. An additional 15 sea turtles may interact with fishing vessels utilized in the ten fisheries every five years. For the purposes of assessing impacts to the North Atlantic DPS of green sea turtles, we assume all these interactions are with green sea turtles.

Based on the lengths of soak/tow times for gillnet and bottom trawl fisheries in the action area, captures of green sea turtles in these gears could result in serious injuries or mortalities due to forced submergence. Currently there are no regulatory controls on tow times in these bottom trawl fisheries and the only restriction on gillnet soak times is the 30-day limit under the ALWTRP regulations. However, TEDs are required in the mid-Atlantic summer flounder fishery to allow turtles to escape from trawl nets and to reduce bycatch related mortality. Serious injuries or mortalities could also occur as a result of entanglement in gillnet gear, which could hamper swimming, feeding, or surfacing behaviors and lead to asphyxiation or necrosis of body parts. Of the anticipated interactions, 78 percent (8) of the anticipated interactions in gillnet gear and 50 percent (16) of the interactions in bottom trawl gear are expected to lead to mortality in a 5-year period . In addition, 15 sea turtles of any species may be die from being struck by vessels operating in the fisheries over the 5-year period. While it is less likely that these will be green sea turtles, for assessing impacts on green sea turtles, we assume that all could be greens. Therefore, 39 of the 57 green sea turtles that interact with gear or vessels in these fisheries every five years are expected to die or sustain serious injuries leading to death or failure to reproduce. This results in the loss of 8 green sea turtles, on average, each year.

As described above, it is reasonable to expect that both benthic immature and sexually mature green sea turtles may be captured in gillnet and bottom trawl gear as a result of the operation of the fisheries. It is assumed that there is an equal chance of lethally capturing a male or female green sea turtle since available information suggests that both sexes occur in the action area. Lethal interactions would reduce the number of green sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same.

Lethal interactions would also result in a potential reduction in future reproduction, assuming some individuals would be females and would have otherwise survived to reproduce. For example, an adult female green sea turtle lays three clutches of eggs, on average (Seminoff et al. 2015), every two to years (Troëng and Chaloupka 2007, Witherington and Ehrhart 1989, Zurita et al. 1994). Green turtle clutches range from 108 eggs in Costa Rica to 136 eggs in Florida (Seminoff et al. 2015, Tiwari et al. 2006, Witherington and Ehrhart 1989). A small percentage of the eggs are expected to survive to sexual maturity. A lethal capture of a female green sea turtle in gillnet or bottom trawl gear would remove reproductive output from the species. The anticipated lethal interactions are expected to occur anywhere in the action area, and green sea turtles generally have large ranges in which they disperse. Thus, no reduction in the distribution of green sea turtles is expected from these interactions. Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

We believe the proposed actions are not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the green sea turtle. Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the U.S. populations of green sea turtles would not be appreciably affected. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals. Since the abundance trend information for green sea turtles is clearly increasing while takes have been occurring, we believe the lethal interactions attributed to the proposed actions will not have any measurable effect on that trend. In addition, 8 green sea turtle mortalities per year represents a very small fraction, < 0.1 percent (=(8/8426+30052)*100), of the overall population estimated from recent nester data in Florida (8,426) and Costa Rica (30,052). As described in the *Environmental Baseline*, although the Deepwater Horizon oil spill is expected to have resulted in adverse impacts to green sea turtles, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would have changed the species' status to an extent that the expected interactions from these fisheries would result in a detectable change in the population status of green sea turtles in the Atlantic. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed actions could be seen as reducing the likelihood of survival and recovery of the species.

As also described in the *Environmental Baseline*, regulatory actions have been taken to reduce anthropogenic effects to green sea turtles in the Atlantic. These include measures to reduce the number and severity of green sea turtle interactions in the U.S. South Atlantic and Gulf of Mexico shrimp, Mid-Atlantic large mesh gillnet, Mid-Atlantic sea scallop dredge, summer flounder trawl, and the Virginia pound net fisheries—all of which are causes of green sea turtle mortality in the Atlantic. Since most of these regulatory measures have been in place for several years now, it is likely that current nesting trends reflect the benefit of these measures to Atlantic green sea turtles. Therefore, the current nesting trends for green sea turtles in the Atlantic are likely to continue to improve as a result of the regulatory actions taken for these and other fisheries. There are no new known sources of mortality for green sea turtles in the Atlantic other than potential impacts from the Deepwater Horizon oil spill.

The recovery plan for Atlantic green sea turtles (NMFS and USFWS 1991a) lists the following recovery objectives which are relevant to the proposed actions in this Opinion, and must be met over a period of 25 continuous years:

1. The level of nesting in Florida has increased to an average of 5,000 nests per year for at least six years.
2. A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

Along the Atlantic coast of eastern central Florida, a mean of 5,055 nests were deposited each year from 2001 to 2005 (Meylan et al. 2006) and 10,377 each year from 2008 to 2012 (B. Witherington, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013, as cited in Seminoff et al. 2015). Nesting has increased substantially over the last 20 years and peaked in 2011 with 15,352 nests statewide (Chaloupka et al. 2007; B. Witherington, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013 as cited in in Seminoff et al. 2015). The status review estimated total nester abundance for Florida at 8,426 turtles (Seminoff et al. 2015). As described above, sea turtle nesting in Florida is continuing to increase. For the most recent 6-year period of SNBS data, there were 5,895 in 2014, 37,341 nests in 2015, 5,393 in 2016, 53,102 in 2017, 4,545 in 2018, and 53,011 in 2019 (see https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). Thus, this recovery criterion continues to be met.

Several actions are being taken to address the second objective; however, there are currently few studies, and no estimates, available that specifically address changes in abundance of individuals on foraging grounds. A study in the central region of the Indian River Lagoon (along the east coast of Florida) found a 661 percent increase in juvenile green sea turtle capture rates over a 24-year study period from 1982-2006 (Ehrhart et al. 2007). Wilcox et al. (1998) found a dramatic increase in the number of green sea turtles captured from the intake canal of the St. Lucie nuclear power plant on Hutchinson Island, Florida beginning in 1993. During a 16-year period from 1976-1993, green sea turtle captures averaged 24 per year. Green sea turtle catch rates for 1993, 1994, and 1995 were 745, 804, and 2,084 percent above the previous 16-year average annual catch rates (Wilcox et al. 1998). In a study of sea turtles incidentally caught in pound net gear fished in inshore waters of Long Island, New York, Morreale and Standora (2005) documented the capture of more than twice as many green sea turtles in 2003 and 2004 with less pound net gear fished, compared to the number of green sea turtles captured in pound net gear in the area during the 1990s. Yet other studies have found no difference in the abundance of green sea turtles on foraging grounds in the Atlantic (Bjorndal et al. 2005, Epperly et al. 2007). Given the clear increases in nesting, however, it is reasonably likely that numbers on foraging grounds have increased.

Based on the information provided above, the loss of 8 green sea turtles annually from the North Atlantic DPS as a result of the operation of the fisheries will not appreciably reduce the likelihood of survival for green sea turtles in the North Atlantic given that is not expected to measurably affect the increasing nesting trend in Florida, that the population size is relatively large, and that measures to reduce the number of North Atlantic DPS of green sea turtles that are injured and die (which should result in increases to the numbers of green sea turtles in the North Atlantic that would otherwise have not occurred in the absence of those regulatory measures) are in place. Given that the action is not expected to measurably affect the nesting trend, the operation of the fisheries will also not appreciably reduce the likelihood of recovery of green sea turtles in the North Atlantic DPS. The fisheries assessed in this Opinion have no adverse effects on green sea turtles that occur outside of the North Atlantic. Therefore, since the operation of the fisheries will not appreciably reduce the likelihood of survival or recovery of green sea turtles in the North Atlantic, the proposed actions will not appreciably reduce the likelihood of survival or recovery for the species.

9.6. Kemp's Ridley Sea Turtle

Kemp's ridley sea turtles are listed as a single species classified as endangered under the ESA. Kemp's ridleys occur in the Atlantic Ocean and Gulf of Mexico. The only major nesting site for Kemp's ridleys is a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963, NMFS and USFWS 2015, USFWS and NMFS 1992).

Nest count data provides the best available information on the number of adult females nesting each year. As is the case with other sea turtles species, nest count data must be interpreted with caution given that these estimates provide a minimum count of the number of nesting Kemp's ridley sea turtles. In addition, the estimates do not account for adult males or juveniles of either sex. Without information on the proportion of adult males to females and the age structure of the population, nest counts cannot be used to estimate the total population size (Meylan 1982, Ross 1996), letter to J. Lecky, NMFS Office of Protected Resources, from N. Thompson, NMFS Northeast Fisheries Science Center, December 4, 2007). Nevertheless, the nesting data does provide valuable information on the extent of Kemp's ridley nesting and the trend in the number of nests laid. It is the best proxy we have for estimating population changes.

Following a significant, unexplained one-year decline in 2010, Kemp's ridley sea turtle nests in Mexico reached a record high of 21,797 in 2012 (Gladys Porter Zoo nesting database, unpublished data). In 2013 and 2014, there was a second significant decline in Mexico nests, with only 16,385 and 11,279 nests recorded, respectively. In 2015, nesting in Mexico improved to 14,006 nests, and in 2016 overall numbers increased to 18,354 recorded nests. There was a record high nesting season in 2017, with 24,570 nests recorded (J. Pena, pers. comm. to NMFS SERO PRD, August 31, 2017 as cited in NMFS 2020c) and decreases observed in 2018 and again in 2019 (Figure 39). In 2019, there were 11,140 nests in Mexico. It is unknown whether this decline is related to resource fluctuation, natural population variability, effects of catastrophic events like the Deepwater Horizon oil spill affecting the nesting cohort, or some other factor. A small nesting population is also emerging in the United States, primarily in Texas. From 1980-1989, there were an average of 0.2 nests/year at Padre Island National Seashore (PAIS), rising to 3.4 nests/year from 1990-1999, 44 nests/year from 2000-2009, and 110 nests per year from 2010-2019. There was a record high of 353 nests in 2017 (NPS 2020). It is worth noting that nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015-2017 (NMFS 2020c) and decreases in nesting in 2018 and 2019 (NPS 2020).

Estimates of the adult female nesting population reached a low of approximately 250-300 in 1985 (NMFS and USFWS 2015, TEWG 2000). Gallaway et al. (2016) developed a stock assessment model for Kemp's ridley to evaluate the relative contributions of conservation efforts and other factors toward this species' recovery. Terminal population estimates for 2012 summed over ages 2 to 4, ages 2+, ages 5+, and ages 9+ suggest that the respective female population sizes were 78,043 (SD = 14,683), 152,357 (SD = 25,015), 74,314 (SD =10,460), and 28,113 (SD = 2,987) (Gallaway et al. 2016). Using the standard IUCN protocol for sea turtle assessments, the number of mature individuals was recently estimated at 22,341 (Wibbels and Bevan 2019). The calculation took into account the average annual nests from 2016-2018 (21,156), a clutch frequency of 2.5 per year, a remigration interval of 2 years, and a sex ratio of 3.17 females:1 male. Based on the data in their analysis, the assessment concluded the current population trend is unknown (Wibbels and Bevan 2019). However, some positive outlooks for the species include recent conservation actions, including the expanded TED requirements in the shrimp fishery (84 FR 70048, December 20, 2019) and a decrease in the amount of shrimping off the coast of Tamaulipas and in the Gulf of Mexico (NMFS and USFWS 2015).

Genetic variability in Kemp's ridley turtles is considered to be high, as measured by nuclear DNA analyses (i.e., microsatellites) (NMFS et al. 2011). If this holds true, then rapid increases in population over one or two generations would likely prevent any negative consequences in the genetic variability of the species (NMFS et al. 2011). Additional analysis of the mtDNA taken from samples of Kemp's ridley turtles at Padre Island, Texas, showed six distinct haplotypes, with one found at both Padre Island and Rancho Nuevo (Dutton et al. 2006).

Kemp's ridley sea turtles have been documented to interact with both gillnet and bottom trawl gear in the action area. The distribution of Kemp's ridleys overlaps seasonally with the use of these gears, and they are known to be captured in or entangled by gears used in several of the fisheries assessed in this Opinion, albeit at low levels. Based on information from Murray (2018; 2020), Linden (2020) and the GAR STDN, we anticipate 292 Kemp's ridley sea turtles will interact with gear utilized in the ten fisheries assessed in this Opinion every five years. Kemp's ridley sea turtles that interact with gear used in these fisheries (which for the purposes of this Opinion includes gillnet, bottom trawl, hook gear, and trap/pot gear only) are those that are captured or entangled in the gear. An estimated 239 Kemp's ridleys are expected to interact with gillnet gear every five years based on the interactions rates in Murray (2018) and federal waters take apportionment (Linden 2020). In addition, an estimated 53 Kemp's ridleys are expected to interact annually with bottom trawl gear, based on Murray (2020) and the federal waters take apportionment (Linden 2020). No Kemp's ridleys are expected to interact with trap/pot gear in the lobster, red crab, Jonah crab, black sea bass, and scup fisheries. An additional 15 sea turtles may interact with fishing vessels utilized in the ten fisheries every five years. While it is more likely these will be loggerhead or leatherback sea turtles, for assessing impacts on Kemp's ridley sea turtles, we assume that all could be Kemp's ridleys.

Of the anticipated interactions, 78 percent (187) of the anticipated interactions (239) in gillnet gear and 50 percent (27) of the interactions (53) in bottom trawl gear are expected to lead to mortality in a 5-year period. In addition, 15 sea turtles of any species may die from being struck by vessels operating in the fisheries over the 5-year period. Therefore, 229 of the 307 Kemp's ridley sea turtles that interact with gear or vessels in these fisheries every five years are expected to die or sustain serious injuries leading to death or failure to reproduce. This results in the loss of 46 Kemp's ridley sea turtles, on average, each year. Either male or female Kemp's ridleys may be captured/entangled in these fisheries since available information suggests that both sexes occur in the action area. All Kemp's ridleys interacting with these fisheries in the action area are expected to be immatures.

The proposed actions would reduce the species' population compared to the number that would have been present in the absence of the proposed actions, assuming all other variables remained the same. Using the estimate of mature animals (22,341) in Wibbels et al. (2019), the loss of 46 animals per year represents a small fraction, approximately 0.2 percent (=46/22,341*100) of the overall population. The proposed actions could also result in a potential reduction in future reproduction, assuming at least some of these individuals would be female and would have survived to reproduce in the future. The annual loss of adult females could preclude the production of thousands of eggs and hatchlings, of which a small percentage is expected to survive to sexual maturity. Thus, the death of any females that would otherwise have survived to sexual maturity would eliminate their contribution to future generations, and result in a reduction in sea turtle reproduction. The anticipated lethal interactions are expected to occur anywhere in the action area, and sea turtles generally have large ranges in which they disperse. Thus, no reduction in the distribution of Kemp's ridley sea turtles is expected from these fishery interactions. Whether the reductions in numbers and reproduction of Kemp's ridley sea turtles would appreciably reduce their likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In addition, the species' limited range and low global abundance makes it particularly vulnerable to new sources of mortality as well as demographic and environmental stochasticity, which are often difficult to predict with any certainty.

It is likely that the Kemp's ridley was the sea turtle species most affected by the Deepwater Horizon oil spill on a population level. In addition, the sea turtle strandings documented in 2010 and 2011 in Alabama, Louisiana, and Mississippi primarily involved Kemp's ridley sea turtles. Necropsy results indicated that mortality was caused by forced submergence, which is commonly associated with fishery interactions (77 FR 27413, May 10, 2012). As described in the *Environmental Baseline*, regulatory actions have been taken to reduce anthropogenic effects to Kemp's ridley sea turtles. These include measures implemented to reduce the number and severity of Kemp's ridley sea turtle interactions in the U.S. South Atlantic and Gulf of Mexico shrimp, mid-Atlantic large mesh gillnet, Mid-Atlantic summer flounder, Mid-Atlantic scallop dredge, and the Virginia pound net fisheries. In 2021, the expanded TED requirements in the shrimp trawl fishery will become effective, further reducing impacts to sea turtles.

There are no new known sources of mortality for Kemp's ridley sea turtles other than potential impacts from the Deepwater Horizon oil spill. Nevertheless, the effects on Kemp's ridley sea turtles from the proposed actions are not likely to appreciably reduce overall population numbers over time due to current population size, expected recruitment, and the implementation of additional conservation requirements in the shrimp trawl fishery, even in light of the adverse impacts expected to have occurred from the Deepwater Horizon oil spill.

It is important to remember that with significant inter-annual variation in nesting data, sea turtle population trends necessarily are measured over decades and the long-term trend line better reflects the population increase in Kemp's ridleys. With the recent nesting data, the population trend has become less clear. Nonetheless, data from 1990 to present continue to support that Kemp's ridley sea turtles have shown a generally increasing nesting trend. Even with reported biennial fluctuations in nesting numbers from Mexican beaches, all years since 2006 have reported over 10,000 nests per year, indicating an increasing population over the previous decades. We believe this long-term trend in nesting is likely evidence of a generally increasing population, as well as a population that is maintaining (and potentially increasing) its genetic diversity. These nesting data are indicative of a species with a high number of sexually mature individuals. Additionally, new measures have been implemented in the shrimp trawl fishery, which will further reduce impacts to the population. The loss of 46 Kemp's ridleys annually is not expected to change the trend in nesting, the distribution of, or the reproduction of Kemp's ridley sea turtles. Therefore, we do not believe the proposed action will cause an appreciable reduction in the likelihood of survival of this species in the wild.

The recovery plan for the Kemp's ridley sea turtle (NMFS et al. 2011) lists the following recovery objectives for downlisting that are relevant to the fisheries assessed in this Opinion:

1. Demographic: A population of at least 10,000 nesting females in a season (as measured by clutch frequency per female per season) distributed at the primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) in Mexico is attained. Methodology and capacity to implement and ensure accurate nesting female counts have been developed.
2. Listing factor: TED regulations, or other equally protective measures, are maintained and enforced in U.S. and Mexican trawl fisheries (e.g., shrimp, summer flounder, whelk) that are known to have an adverse impact on Kemp's ridleys in the Gulf of Mexico and Northwest Atlantic Ocean.

With respect to the demographic recovery objective, the nesting numbers in the most recent three years indicate there were 24,570 nests in 2017, 17,945 in 2018, and 11,090 in 2019 on the main nesting beaches in Mexico. Based on 2.5 clutches/female/season, these numbers represent approximately 9,828 (2017), 7,178 (2018), and 4,436 (2019) nesting females in each season. The number of nests reported annually from 2010 to 2014 declined overall; however, they rebounded in 2015 through 2017, and declined again in 2018 and 2019. Although there has been a substantial increase in the Kemp's ridley population within the last few decades, the number of nesting females is still below the number of 10,000 nesting females per season required for downlisting (NMFS and USFWS 2015). Since we concluded that the potential loss of Kemp's ridley sea turtles is not likely to have any detectable effect on nesting trends, we do not believe the proposed action will impede progress toward achieving this recovery objective. Nonlethal captures of these sea turtles would not affect the adult female nesting population or number of nests per nesting season. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of Kemp's ridley sea turtles' recovery in the wild.

In regards to the listing factor recovery criterion, the recovery plan states, "the highest priority needs for Kemp's ridley recovery are to maintain and strengthen the conservation efforts that have proven successful. In the water, successful conservation efforts include maintaining the use of TEDs in fisheries currently required to use them, expanding TED-use to all trawl fisheries of concern, and reducing mortality in gillnet fisheries. Adequate enforcement in both the terrestrial and marine environment also is also noted essential to meeting recovery goals" (NMFS et al. 2011). We are currently undertaking several of these initiatives, which should aid in the recovery of the species. The required use of TEDs in shrimp trawls in the United States under sea turtle conservation regulations and in Mexican waters has had dramatic effects on the recovery of Kemp's ridley sea turtles.

Based on the information provided above, the loss of 46 Kemp's ridley sea turtles annually in the fisheries will not appreciably reduce the likelihood of survival and recovery for Kemp's ridley sea turtles given the long term nesting trend, the population size, and ongoing and future measures (i.e., expanded TED regulations in the shrimp trawl fishery) that reduce the number of Kemp's ridley sea turtles that are injured and die.

9.7. Loggerhead Sea Turtle, NWA DPS

The Northwest Atlantic DPS of loggerhead sea turtles is listed as threatened under the ESA. Based on nesting data and population abundance and trends at the time, NMFS and USFWS determined in 2011 that the Northwest Atlantic DPS should be listed as threatened and not endangered based on (1) the large size of the nesting population, (2) the overall nesting population remains widespread, (3) the trend for the nesting population appears to be stabilizing, and (4) substantial conservation efforts are underway to address threats (76 FR 58868, September 22, 2011).

It takes decades for loggerhead sea turtles to reach maturity. Once they have reached maturity, females typically lay multiple clutches of eggs within a season, but do not typically lay eggs every season (NMFS and USFWS 2008). There are many natural and anthropogenic factors affecting the survival of loggerheads prior to their reaching maturity as well as for those adults who have reached maturity. As described in the *Status of the Species*, *Environmental Baseline*, *and Cumulative Effects* sections above, loggerhead sea turtles in the action area continue to be affected by multiple anthropogenic impacts including bycatch in commercial and recreational fisheries, habitat alteration, vessel interactions, hopper dredging, power plant intakes, and other factors that result in mortality of individuals at all life stages. Negative impacts causing death of various age classes occur both on land and in the water. Many actions have been taken to address known negative impacts to loggerhead sea turtles. However, others remain unaddressed, have not been sufficiently addressed, or have been addressed in some manner but whose success cannot be quantified.

As previously stated, there are five subpopulations of loggerhead sea turtles in the western North Atlantic (recognized as recovery units in the 2008 recovery plan for the species). These subpopulations show limited evidence of interbreeding. Recent assessments have evaluated the nesting trends for each recovery unit. It should be noted, and it is explained further below, that nesting trends are based on nest counts or nesting females. They do not include non-nesting adult females, adult males, or juvenile males or females in the population.

Ceriani and Meylan (2017) and Bolten et al. (2019) looked at trends by recovery unit. Information on nest counts is presented in the *Status of the Species*. Trends by recovery unit were variable. For the Northern Recovery Unit, nest counts at loggerhead nesting beaches in North Carolina, South Carolina, and Georgia declined at 1.9 percent annually from 1983 to 2005 (NMFS and USFWS 2008). More recently, the trend has been increasing. Ceriani and Meylan (2017) reported a 35 percent increase for this recovery unit from 2009 through 2013. A longer-term trend analysis based on data from 1983 to 2019 indicates that the annual rate of increase is 1.3 percent (Bolten et al. 2019).

Nest counts at index beaches in Peninsular Florida showed a significant decline in loggerhead nesting from 1989 to 2007, most likely attributed to mortality of oceanic-stage loggerheads caused by fisheries bycatch (Witherington et al. 2009). From 2009 through 2013, a 2 percent decrease for the Peninsular Florida Recovery Unit was reported (Ceriani and Meylan 2017). Using a longer time series from 1989-2018, there was no significant change in the number of annual nests (Bolten et al. 2019). It is important to recognize that an increase in the number of nests has been observed from 2007 to 2018 (Bolten et al. 2019). Using short-term trends in nesting abundance can be misleading, and trends should be considered in the context of one generation (50 years for loggerheads) (Bolten et al. 2019).

The Dry Tortugas Recovery Unit includes all islands west of Key West, Florida. A census on Key West from 1995 to 2004 (excluding 2002) estimated a mean of 246 nests per year, or about 60 nesting females (NMFS and USFWS 2008). No trend analysis is available because there was not an adequate time series to evaluate the Dry Tortugas Recovery Unit and there are gaps in the data prohibiting a robust analysis (Bolten et al. 2019, Ceriani et al. 2019, Ceriani and Meylan 2017).

Evaluation of long-term nesting trends for the Northern Gulf of Mexico Recovery Unit is difficult given changes to survey coverage. From 1995 to 2005, the recovery unit exhibited a significant declining trend (Conant et al. 2009, NMFS and USFWS 2008). In the 2009-2013 trend analysis by Ceriani and Meylan (2017), a 1 percent decrease for this recovery unit was reported, likely due to diminished nesting on beaches in Alabama, Mississippi, Louisiana, and Texas. More recently, nest numbers have increased (Bolten et al. 2019). A longer-term analysis from 1997-2018 found that there has been a non-significant increase of 1.7 percent (Bolten et al. 2019).

The majority of nesting in the Greater Caribbean Recovery Unit occurs on the Yucatán Peninsula, in Quintana Roo, Mexico, with 903 to 2,331 nests annually (Zurita et al. 2003). Other significant nesting sites are found throughout the Caribbean, including Cuba, with approximately 250 to 300 nests annually (Ehrhart et al. 2003), and over 100 nests annually in Cay Sal in the Bahamas (NMFS and USFWS 2008). In the trend analysis by Ceriani and Meylan (2017), a 53 percent increase for this Recovery Unit was reported from 2009 through 2013.

Estimates of the total loggerhead population in the Atlantic are not currently available. However, there is some information available for portions of the population. From 2004-2008, the loggerhead adult female population for the Northwest Atlantic ranged from 20,000 to 40,000 or more individuals (median 30,050), with a large range of uncertainty in total population size (NMFS SEFSC 2009). The estimate of Northwest Atlantic adult loggerhead females was considered conservative for several reasons. The number of nests used for the Northwest Atlantic was based primarily on U.S. nesting beaches. Thus, the results are a slight underestimate of total nests because of the inability to collect complete nest counts for many non-U.S. nesting beaches within the DPS. In estimating the current population size for adult nesting female loggerhead sea turtles, the report simplified the number of assumptions and reduced uncertainty by using the minimum total annual nest count (i.e., 48,252 nests) over the five years. This was a particularly conservative assumption considering how the number of nests and nesting females can vary widely from year to year (e.g., the 2008 nest count was 69,668 nests, which would have increased the adult female estimate proportionately to between 30,000 and 60,000). In addition, minimal assumptions were made about the distribution of remigration intervals and nests per female parameters, which are fairly robust and well known. A loggerhead population estimate using data from 2001-2010 estimated the loggerhead adult female population in the Northwest Atlantic at 38,334 individuals (SD =2,287) (Richards et al. 2011).

The AMAPPS surveys and sea turtle telemetry studies conducted along the U.S. Atlantic coast in the summer of 2010 provided preliminary regional abundance estimate of about 588,000 loggerheads along the U.S. Atlantic coast, with an inter-quartile range of 382,000-817,000 (NMFS 2011c). The estimate increases to approximately 801,000 (inter-quartile range of 521,000-1,111,000) when based on known loggerheads and a portion of unidentified sea turtle sightings (NMFS 2011c). Although there is much uncertainty in these population estimates, they provide some context for evaluating the size of the likely population of loggerheads in the Atlantic.

Although limited information is available on the genetic makeup of loggerheads in an area as extensive as the action area, it is likely that loggerheads interacting with these ten fisheries originate from several, if not all of the recovery units. Sea turtles from each of the five Northwest Atlantic nesting stocks have been documented in the action area. A genetic study on immature loggerheads captured in the Pamlico-Albemarle Estuarine Complex in North Carolina between 1995-1997 indicated that 80 percent of the juveniles and sub-adults utilizing this foraging habitat originated from the south Florida nesting stock, 12 percent from the northern nesting stock, 6 percent from the Yucatán nesting stock, and 2 percent from other rookeries (including the Florida Panhandle, Dry Tortugas, Brazil, Greece, and Turkey nesting stocks) (Bass et al. 2004). Similarly, genetic analysis of samples collected from loggerheads from Massachusetts to Florida found that all five western Atlantic loggerhead stocks were represented (Bowen et al. 2004). However, earlier studies indicated that only a few nesting stocks were represented along the U.S. Atlantic coast. Mixed stock analysis of a foraging aggregation of immature loggerhead sea turtles captured in coastal waters off Florida, found three stocks: south Florida (69 percent of the loggerheads sampled) respectively), northern (10 percent, respectively), and Mexico (20 percent) (Witzell et al. 2002). Similarly, analysis of stranded turtles from Virginia to Florida indicated that the turtles originated from three nesting areas: south Florida (59 percent), northern (25 percent), and Mexico (20 percent) (Rankin-Baransky et al. 2001).

More recently, Haas et al. (2008) used two approaches in identifying the contribution of each stock in the U.S. Atlantic sea scallop fishery bycatch: an equal contribution from each stock or a weighted contribution by rookery sizes. The sea scallop fishery generally operates in the same areas as the fisheries considered in this Opinion and; therefore, the results are applicable to these fisheries. When weighted by population size, 89 percent of the loggerheads captured in the U.S. Atlantic scallop fishery from 1996-2005 originated from the south Florida nesting stock, 4 percent were from the Mexican stock, 3 percent were from the northern (northeast Florida to North Carolina) stock, 1 percent were from the northwest Florida stock, and 0 percent were from the Dry Tortugas stock. The remaining 3 percent of loggerheads sampled were attributed to nesting stocks in Greece (Haas et al. 2008). Haas et al. (2008) noted that these results should be interpreted with caution given the small sample size and resulting difficulties in precisely assigning rookery contributions to a particular mixed population. A re-analysis of loggerhead genetics data by the Atlantic Loggerhead TEWG has found that it is unlikely that U.S. fishing fleets are interacting with the Mediterranean DPS (LaCasella et al. 2013). Given that updated, more refined analyses are ongoing and the occurrence of Mediterranean DPS juveniles in U.S. Atlantic waters is rare and uncertain, if occurring at all, it is unlikely that individuals from the Mediterranean DPS would be present in the action area (Memorandum from Patricia A. Kurkul, Regional Administrator, to the Record, November 29, 2011). As a result, those records are excluded from our analysis and are reapportioned to the five Northwest Atlantic stocks, which are expected to contribute to individuals in the action area. Note that when equal contributions of each stock were considered, Haas et al. (2008) found that the results varied from the weighted contributions but the south Florida nesting stock still contributed the majority of scallop fishery bycatch (63 percent).

These loggerhead nesting stocks in Haas et al. (2008) do not share the exact delineations of the recovery units identified in the 2008 recovery plan. However, the PFRU encompasses the south Florida stock, the NRU is roughly equivalent to the northern nesting stock, the northwest Florida stock is included in the NGMRU, the Mexico stock is included in the GCRU, and the DTRU encompasses the Dry Tortugas stock. The available genetic analyses indicate the majority of bycatch in Northeast and Mid-Atlantic waters comes from the PFRU with smaller contributions from the other recovery units in the Northwest Atlantic DPS. However, the exact percentages of fisheries bycatch from specific nesting beaches and recovery units are not available at this time and may be variable from year to year. As a result, we are relying on the genetic analysis weighted by population size presented in Haas et al. (2008), which is the most recent and one of the most comprehensive (in terms of the area from which samples were acquired) of the loggerhead genetics studies. The best available information indicates that the proportion of the interactions from each recovery unit is consistent with the relative sizes of the recovery units.

Based on information from Murray (2018; 2020), (2020), and the GAR STDN, we anticipate 1,995 loggerhead sea turtles from the Northwest Atlantic DPS will interact with gear utilized in the ten fisheries assessed in this Opinion every five years. Loggerhead sea turtles that interact with gear used in these fisheries (which for the purposes of this Opinion includes gillnet, bottom trawl, hook gear, and trap/pot gear only) are those that are captured or entangled in the gear. An estimated 1,036 loggerheads are expected to interact with gillnet gear every five years based on the interaction rates in Murray (2018) and federal waters take apportionment (Linden 2020). In addition, an estimated 954 loggerheads are expected to interact every five years with bottom trawl gear, based on Murray (2020) and the federal waters take apportionment (Linden 2020). Five loggerheads are expected to interact with trap/pot gear in the lobster, red crab, Jonah crab, black sea bass, and scup fisheries every five years. An additional 15 loggerhead sea turtles may interact with fishing vessels utilized in the ten fisheries every five years. For the purposes of evaluating effects to the Northwest Atlantic loggerhead DPS, we are considering that all 15 will be loggerhead sea turtles.

Of these, 78 percent (808) of the interactions (1,036) in gillnet gear, 50 percent (477) of the interactions (954) in bottom trawl gear, and 64 percent (4) of the interactions (5) in trap/pot are expected to lead to mortality every five years. In addition, 15 sea turtles are estimated to be struck by vessels operating in the fisheries in this Opinion every five years. These interactions may be lethal. Therefore, 1,304 of the 2,010 loggerheads that interact with gear or vessels in these fisheries every five years are expected to die or sustain serious injuries leading to death or failure to reproduce. This results in the loss of 261 loggerhead turtles, on average, each year.

The vast majority of the 261 loggerheads mortalities anticipated, on average, annually (i.e., 1,304 mortalities over five years) due to the ten fisheries assessed in this Opinion are likely to originate from the PFRU, with the remainder originating from the NRU, GCRU, NGMRU, and DTRU. Using the mean percent contributions in Haas et al. (2008) and then reapportioning the extra 3 percent of turtles that had been attributed to nesting stocks in Greece, we expect that 237 of the loggerhead mortalities will be from the PFRU, 9 from the NRU, 11 from the GCRU, 3 from the NGMRU, and 1 from the DTRU. Therefore, we conclude that none of the recovery units will be disproportionately impacted by interactions in these fisheries. Thus, genetic heterogeneity should be maintained in the species.

The lethal removal of 1,304 loggerhead sea turtles from the Northwest Atlantic DPS every five years (again, on average 261 per year) will reduce the number of loggerhead sea turtles compared to the number that would have been present in the absence of the proposed actions (assuming all other variables remained the same). These lethal interactions would also result in a future reduction in reproduction due to lost reproductive potential, as some of these individuals would be females who would have reproduced in the future, thus eliminating each female individual's contribution to future generations. For example, an adult female loggerhead sea turtle in the Northwest Atlantic DPS can lay three or four clutches of eggs every two to four years, with 100 to 126 eggs per clutch (NMFS and USFWS 2008). The annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. A reduction in the distribution of loggerhead sea turtles is not expected from lethal interactions attributed to the proposed actions. Because all the potential interactions are expected to occur at random throughout the action area and loggerheads generally have large ranges in which they disperse, the distribution of loggerhead sea turtles in the action area is expected to be unaffected.

Whether the reductions in the Northwest Atlantic DPS of loggerhead numbers and reproduction attributed to the proposed actions would appreciably reduce the likelihood of survival for loggerheads depends on what effect these reductions in numbers and reproduction have on overall population sizes and trends. That is, whether the estimated reductions, when viewed within the context of the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* are to such an extent that adverse effects on population dynamics are appreciable. Loggerhead sea turtles are a slow growing, late-maturing species. Because of their longevity, loggerheads require high survival rates throughout their life to maintain a population (Conant et al. 2009). In other words, late-maturing species are less tolerant of high rates of anthropogenic mortality. Conant et al. (2009) concluded that loggerhead natural growth rates are low, natural survival needs to be high, and even low (1-10 percent) to moderate (10-20 percent) mortality can drive the population into decline. Because recruitment to the adult population is slow, population modeling studies suggest even small increased mortality rates in adults and sub-adults could substantially impact population numbers and viability (Chaloupka and Musick 1997, Crouse et al. 1987, Crowder et al. 1994, Heppell et al. 2005).

Actions have been taken to reduce anthropogenic impacts to loggerhead sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of juveniles and adults in various fisheries and other marine activities. Conant et al. (2009) concluded that the results of their models (i.e., predicted continued declines) are largely driven by mortality of juvenile and adult loggerheads from fishery bycatch that occurs throughout the Northwest Atlantic. While significant progress has been made to reduce bycatch in some fisheries in certain parts of the loggerhead's range, and the results of new nesting trend analyses may indicate the positive effects of those efforts, notable fisheries bycatch persists. The question we are left with for this analysis is whether the effects of the proposed actions appreciably reduce survival and recovery, given the current status of the species and predicted population trajectories, as well as the many natural and human-caused impacts on sea turtles. We may not see the long-term effects of the Deepwater Horizon oil release event and climate change on the population status and trends of loggerheads for several years to come.

As described in the *Status of the Species*, we consider that the Deepwater Horizon oil release had an adverse impact on loggerhead sea turtles, and resulted in mortalities, along with unknown lingering impacts outside the action area resulting from nest relocations, non-lethal exposure, and foraging resource impacts. However, there is no information to indicate that a significant population-level impact has occurred that would have changed the species' status to an extent that the expected interactions from the fisheries assessed in this Opinion would result in a detectable change in the population status of the Northwest Atlantic DPS of loggerhead turtles. This is especially true given the size of the population and that, unlike Kemp's ridleys, the

Northwest Atlantic DPS of loggerheads is proportionally much less dependent on Gulf of Mexico.

It is possible that the Deepwater Horizon oil release reduced the survival rate of all age classes to varying degrees and may continue to do so for some undetermined time. However, there is no information at this time that it has, or should be expected to have, substantially altered the long-term survival rates in a manner that would significantly change the population dynamics compared to the conservative estimates used in this Opinion. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed actions would reduce the likelihood of survival of the species.

We have determined that the effects on loggerhead sea turtles associated with the proposed actions are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the Northwest Atlantic loggerhead DPS, even in light of the impacts of the Deepwater Horizon oil release and climate change. Over the proposed action, we expect the Northwest Atlantic DPS of adult females to remain large (tens or hundreds of thousands of individuals) and to retain the potential for recovery, as explained below. While the effects of the proposed actions will most directly affect the overall size of the population, the annual take represents a very small fraction, approximately 0.7 percent (=261/38,334*100) of the overall female population estimated by Richards et al. (2011). The mortality estimate includes both juveniles and adults while the population estimate is only female adults so this is a conservative estimate. We expect that the population will remain large for several decades to come. The action is not expected to reduce the genetic heterogeneity, broad demographic representation, or successful reproduction of the population, nor affect loggerheads' ability to meet their life cycle requirements, including reproduction, sustenance, and shelter.

In the recovery plan for loggerheads, the nesting beach Demographic Recovery Criteria are specific to recovery units. This criteria for nests and nesting females were based on a time frame of one generation for U.S. loggerheads, defined in the recovery plan as 50 years. To be considered for delisting, each recovery unit will have recovered to a viable level and will have increased for at least one generation. The rate of increase used for each recovery unit was dependent upon the level of vulnerability of the recovery unit. The minimum statistical level of detection (based on annual variability in nest counts over a generation time of 50 years) of 1 percent per year was used for the PFRU, the least vulnerable recovery unit. A higher rate of increase of 3 percent per year was used for the NGMRU and DTRU, the most vulnerable recovery units. A rate of increase of 2 percent per year was used for the NRU, a moderately vulnerable recovery unit (NMFS and USFWS 2008).

A fundamental problem with restricting population analyses to nesting beach surveys is that they may not reflect changes in the non-nesting population. This is because of the long time to maturity and the relatively small proportion of females that are reproducing on a nesting beach. A decrease in oceanic juvenile or neritic juvenile survival rates may be masked by the natural variability in nesting female numbers and the slow response of adult abundance to changes in recruitment to the adult population (Chaloupka and Limpus 2001). In light of this, two additional Demographic Criteria were developed to ensure a more representative measure of population status was achieved. These criteria are not delineated by recovery unit because individuals from the recovery units mix in the marine environment; therefore, they are applicable to all recovery units. The first of these additional Demographic Criteria assesses trends in abundance on foraging grounds, and the other assesses age-specific trends in strandings relative to age-specific trends in abundance on foraging grounds. For the foraging grounds, a network of index in-water sites, both oceanic and neritic, distributed across the foraging range must be established and monitored to measure abundance. Recovery can be achieved if there is statistical confidence (95 percent) that a composite estimate of relative abundance from these sites is increasing for at least one generation. For trends in strandings relative to in-water abundance, recovery can be achieved if stranding trends are not increasing at a rate greater than the trends in in-water relative abundance for similar age classes for at least one generation. Recovery criteria must be met for all recovery units in order for the species to be de-listed (NMFS and USFWS 2008).

Assuming some or all loggerhead mortalities through interactions with these fisheries are females, the loss of female loggerhead sea turtles as a result of the proposed actions is expected to reduce the reproduction of loggerheads in the Northwest Atlantic DPS compared to the reproductive output of Northwest Atlantic DPS of loggerheads in the absence of the proposed actions. In addition to being linked to survival, these losses are relevant to the Demographic Recovery Criteria for nests and nesting females. As described in the *Status of the Species*, nesting trends for each of the loggerhead sea turtle recovery units in the Northwest Atlantic Ocean DPS are variable. Overall, short-term trends have shown increases, however, over the long-term the DPS is considered stable.

Assuming half the loggerheads interacting with the fisheries in this Opinion are females and interactions are with adults (a worst case scenario with respect to the reproductive value to the population), the loggerhead mortalities from the fisheries would remove 0.34 percent of the DPS (131 out of the estimated 38,334 adult female loggerheads in the Northwest Atlantic from Richards et al. (2011)). In general, while the loss of a certain number of individuals from a species may have an appreciable reduction on the numbers, reproduction, and distribution of the species, this is likely to occur only when there are very few individuals in a population, the individuals occur in a very limited geographic range, or the species has extremely low levels of genetic diversity. This situation is not likely in the case of the Northwest Atlantic DPS of loggerheads because the species is widely geographically distributed, it is not known to have low levels of genetic diversity, and there are tens to hundreds of thousands of individuals (and possibly more) in the DPS.

In determining whether the operation of the ten fisheries would reduce appreciably the likelihood of survival and recovery of loggerhead sea turtles, NMFS also considered the PVA for loggerhead sea turtles based on the impacts of the Atlantic sea scallop fishery (Merrick and Haas 2008). We recognize that this PVA was published in 2008 and new information has become available since its publication. However, this is the most recently available PVA and does provide information to consider in our analysis. This information is considered with the information above to assess the impacts of the fisheries on the Northwest Atlantic DPS of loggerhead sea turtles.

The Atlantic sea scallop PVA estimated quasi-extinction (the point at which so few animals remain that the species/population will inevitably become extinct) likelihoods under conditions with and without fishery effects (Merrick and Haas 2008). Since the PVA was count-based, the only relatively complete and available population time series at the time (nesting beach counts for 1998-2005) was used for the analysis. As such, the analysis focused on the viability of the adult females and did not model the viability of the entire loggerhead population (Merrick and Haas 2008). The PVA is described in detail in Merrick and Haas (2008) (Appendix 4).

The PVA established a baseline using the rate of change of the adult female population (which implicitly included the mortalities from these ten fisheries up to that time), and the 2005 count of adult females estimated from all beaches in the Southeast United States based on an extrapolation from nest counts (Merrick and Haas 2008). The rate of change was then adjusted by adding back the scallop fishery interactions (converted to adult female equivalents) and re-running the PVA. The results of these two analyses were then compared. The authors concluded that both the baseline and adjusted baseline (adding back the scallop fishery interactions) had quasi-extinction probabilities of zero (0) at 25, 50, and 75 years, and a probability of 1 percent at 100 years.

Although the PVA uses data from 1989-2005, and models different effects of the scallop and other Atlantic fisheries on loggerheads than what may occur presently, it is still informative for consideration in this Opinion. The PVA analysis done for the 2008 Atlantic sea scallop biological opinion (NMFS 2008a). and our comparison of its results to the current status and trends of the Northwest Atlantic loggerhead DPS (in light of effects from these fisheries, other baseline activities, and climate change) supports the conclusion that operation of the ten fisheries will neither affect the number of nests and nesting females (Demographic Criteria #1) nor the trends in abundance on foraging grounds (Demographic Criteria #2) to the point where there is an appreciable reduction in the species' likelihood of recovery. Based on the rate of change of the adult female population, the PVA determined that there was only a 1 percent chance that loggerheads in the Atlantic could become quasi-extinct within 100 years either with or without scallop fishery interactions. Again, it should be reiterated that the effects of baseline takes in other fisheries, including those assessed in this Opinion, were built into the assumptions underlying the 2008 PVA model. In addition, the Murray (2018) and Murray (2020) reports as well as data from the Sea Turtle Injury Working Group evidence that the current level of bycatch and mortality in the ten fisheries addressed in this Opinion are less than they were in 2008 when the original PVA was run.

Even amidst ongoing threats to the species such as fishery mortality and climate change, the potential average loss of 1,304 loggerheads from the Atlantic over the next five years (and in future 5-year periods) is not likely to result in any appreciable decline to the Northwest Atlantic DPS. This is due to: (1) the large size of the current nesting population, (2) the fact that the overall nesting population remains widespread, (3) the trend for the nesting population appears to be stabilizing, and short-term trends in some recovery units are increasing, since the time period considered during the PVA, and (4) substantial conservation efforts have been implemented and are underway to address threats.

9.8. Leatherback Sea Turtle

Leatherback sea turtles are listed as endangered under the ESA. Leatherbacks are widely distributed throughout the oceans of the world and are found in waters of the Atlantic, Pacific, and Indian Oceans, the Caribbean Sea, Mediterranean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback nesting occurs on beaches of the Atlantic, Pacific, and Indian Oceans as well as in the Caribbean (NMFS and USFWS 2013). Leatherbacks face a multitude of threats that can cause death prior to and after reaching maturity. Some activities resulting in leatherback mortality have been addressed.

Dutton et al. (2013) evaluated the stock structure of leatherbacks in the Atlantic. Samples from eight nesting sites in the Atlantic and one in the southwest Indian Ocean identified seven management units in the Atlantic and revealed fine scale genetic differentiation among neighboring populations. The mtDNA analysis failed to find significant differentiation between Florida and Costa Rica or between Trinidad and French Guiana/Suriname (Dutton et al. 2013). In 2020, seven leatherback populations that met the discreteness and significance criteria of DPSs were identified (NMFS and USFWS 2020). These include the Northwest Atlantic, Southwest Atlantic, Southeast Atlantic, Southwest Indian, Northeast Indian, West Pacific, and East Pacific. The population found within the action is area is the Northwest Atlantic Atlantic DPS (Figure 44). While NMFS and USFWS concluded that seven populations met the criteria for DPSs, the species continues to be listed at the global level (85 FR 48332, August 10, 2020). Therefore, this analysis considers the range-wide status.

The most recent published assessment, the leatherback status review, estimated that the total index of nesting female abundance for the Northwest Atlantic DPS is 20,659 females (NMFS and USFWS 2020). This abundance estimate is similar to other estimates. The TEWG estimate approximately 18,700 (range 10,000 to 31,000) adult females using nesting data from 2004 and 2005 (TEWG 2007). The IUCN Red List assessment for the NW Atlantic Ocean subpopulation estimated 20,000 mature individuals (male and female) and approximately 23,000 nests per year (data through 2017) with high inter-annual variability in annual nest counts within and across nesting sites (Northwest Atlantic Leatherback Working Group 2019). The estimate in the status review is higher than the estimate for the IUCN Red List assessment, likely due to a different remigration interval, which has been increasing in recent years (NMFS and USFWS 2020). For this analysis, we found that the status review estimate of 20,659 nesting females represents the best available scientific information given that it uses the most comprehensive and recent demographic trends and nesting data.

Previous assessments of leatherbacks concluded that the Northwest Atlantic population was stable or increasing (TEWG 2007, Tiwari et al. 2013b). However, as described in the *Status of the Species,* more recent analyses indicate that the overall trends are negative (NMFS and USFWS 2020, Northwest Atlantic Leatherback Working Group 2018, 2019). At the stock level, the Working Group evaluated the NW Atlantic – Guianas-Trinidad, Florida, Northern Caribbean, and the Western Caribbean stocks. The NW Atlantic – Guianas-Trinidad stock is the largest stock and declined significantly across all periods evaluated, which was attributed to an exponential decline in abundance at Awala-Yalimapo, French Guiana as well as declines in Guyana; Suriname; Cayenne, French Guiana; and Matura, Trinidad. Declines in Awala-Yalimapo were attributed, in part, due to beach erosion and a loss of nesting habitat (Northwest Atlantic Leatherback Working Group 2018). The Florida stock increased significantly over the long-term, but declined from 2008-2017 (Northwest Atlantic Leatherback Working Group 2018). Slight increases in nesting were seen in 2018 and 2019, however, nest counts remain low compared to 2008-2015 (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). The Northern Caribbean and Western Caribbean stocks have also declined. The Working Group report also includes trends at the site-level, which varied depending on the site and time period, but were generally negative especially in the recent period.

Similarly, the leatherback status review concluded that the Northwest Atlantic DPS exhibits decreasing nest trends at nesting aggregations with the greatest indices of nesting female abundance. Though some nesting aggregations indicated increasing trends, most of the largest ones are declining. This trend is considered to be representative of the DPS (NMFS and USFWS 2020). Data also indicated that the Southwest Atlantic DPS is declining (NMFS and USFWS 2020).

Populations in the Pacific have shown dramatic declines at many nesting sites (Mazaris et al. 2017, Santidrián Tomillo et al. 2017, Santidrián Tomillo et al. 2007, Sarti Martínez et al. 2007, Tapilatu et al. 2013). The IUCN Red List assessment estimated the number of total mature individuals (males and females) at Jamursba-Medi and Wermon beaches to be 1,438 turtles (Tiwari et al. 2013a). More recently, the leatherback status review estimated the total index of nesting female abundance of the West Pacific DPS at 1,277 females for the West Pacific DPS and 755 females for the East Pacific DPS (NMFS and USFWS 2020). The East Pacific DPS has exhibited a decreasing trend since monitoring began with a 97.4 percent decline since the 1980s or 1990s, depending on nesting beach (Wallace et al. 2013). Population abundance in the Indian Ocean is difficult to assess due to lack of data and inconsistent reporting. Most recently, the 2020 status review estimated that the total index of nesting female abundance for the SW Indian DPS is 149 females and that the DPS is exhibiting a slight decreasing nest trend (NMFS and USFWS 2020). While data on nesting in the Northeast Indian Ocean DPS is limited, the DPS is estimated at 109 females. This DPS has exhibited a drastic population decline with extirpation of the largest nesting aggregation in Malaysia (NMFS and USFWS 2020).

There have been several documented captures of leatherback sea turtles in gillnet, bottom trawl, and trap/pot gear utilized by the fisheries in the action area. Leatherback interactions with the fisheries are likely to continue given that the distribution of leatherbacks overlaps with areas where the gears are fished. Based on information from Murray (2018), Murray (2020), Linden (2020), and the GAR STDN, we anticipate 142 leatherback sea turtles will interact with gear utilized in the ten fisheries assessed in this Opinion every five years. Leatherback sea turtles that interact with gear used in these fisheries (which for the purposes of this Opinion includes gillnet, bottom trawl, hook gear, and trap/pot gear only) are those that are captured or entangled in the gear. An estimated 52 leatherbacks are expected to interact with gillnet gear every five years based on the interaction rates in Murray (2018) and federal waters take apportionment (Linden 2020). In addition, an estimated 40 leatherbacks are expected to interact annually with bottom trawl gear, based on Murray (2020) and the federal waters take apportionment (Linden 2020). Also, 50 leatherbacks are expected to interact with trap/pot gear in the federal lobster, red crab, Jonah crab, black sea bass, and scup fisheries every five years. An additional 15 sea turtles may interact with fishing vessels utilized in the ten fisheries every five years from any sea turtle species. For assessing impacts on leatherback sea turtles, we assume that all of these animals could be leatherbacks.

Of these, 78 percent (41) of the interactions (52) in gillnet gear, 50 percent (20) of the interactions (40) in bottom trawl gear, and 64 percent (32) of interactions (50) in trap/pot gear are expected to lead to mortality every five years. As described above, we anticipate that 15 sea turtles of any species may be struck by vessels operating in the fisheries over the 5-year period and that these interactions could be lethal. Therefore, 108 of the 157 leatherback sea turtles that interact with gear or vessels in these fisheries every five years are expected to die or sustain serious injuries leading to death or failure to reproduce. This results in the loss of approximately 22 leatherback sea turtles, on average, each year.

Captures and/or entanglements of leatherback sea turtles in gillnet, bottom trawl, and trap/pot gear could result in death due to forced submergence, given that there are no regulatory controls on tow/soak times in these fisheries other than the 30-day maximum soak period for fixed gear under the ALWTRP. Given that leatherbacks forage within the water column rather than on the bottom, interactions with bottom trawl gear are expected to occur when the gear is traveling through the water column versus on the bottom. Interactions between leatherbacks and gillnet and trap/pot gear are expected to occur in the net panels of gillnet gear and in the vertical lines.

The lethal removal of 108 leatherback sea turtles every five years (on average, approximately 22 per year) will reduce the number of leatherback sea turtles as compared to the number that would have been present in the absence of the proposed actions (assuming all other variables remained the same). The lethal interactions could also result in a potential reduction in future reproduction, assuming one or more of these individuals would be female and otherwise survived to reproduce in the future. A leatherback sea turtle will lay multiple nests (clutches) each year. In the Northwest Atlantic DPS, eggs per clutch is 82 for the western Atlantic, and clutch frequency averages 5.5 nests per year (NMFS and USFWS 2020). Therefore, an adult female leatherback sea turtle can produces hundreds of eggs per nesting season. Although a significant portion of the eggs can be infertile (NMFS and USFWS 2020), the annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. Thus, the death of any female leatherbacks that would have otherwise survived to reproduce would eliminate the individual's and its future offspring's contribution to future generations. The anticipated lethal interactions are expected to occur anywhere in the action area. Given that these sea turtles generally have large ranges in which they disperse, no reduction in the distribution of leatherback sea turtles is expected from the proposed actions. Whether the estimated reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction have relative to current population sizes and trends.

We believe the proposed actions are not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of leatherback sea turtles in the wild. Approximately 0.1 percent (=22/20,659*100) of the population is anticipated to die annually through the proposed action. Both juvenile and adult leatherbacks interact with gears used in these fisheries. It should be noted that the abundance estimate is for nesting females only (i.e., does not include adult males or earlier life stages such as juveniles); therefore, the percent of the population that dies due to the proposed action is expected to be less than the percentage estimated here. Although the anticipated mortalities would result in a reduction in absolute population numbers, it is not likely this reduction would appreciably reduce the likelihood of survival of this species. If the hatchling survival rate to maturity is greater than the mortality in the population, the loss of breeding individuals would be replaced through recruitment of new breeding individuals from successful reproduction of sea turtles unaffected by the proposed actions. Considering the number of lethal interactions relative to the population size, we believe the proposed actions are not likely to have an appreciable effect on overall population trends. In addition, the proposed actions are expected to control those impacts by maintaining effort levels consistent with or lower than those that have occurred in previous years.

Fisheries bycatch has been identified as a threat to the Northwest Atlantic DPS of leatherback sea turtles. The Leatherback Working Group noted that leatherback entanglements in vertical line fisheries (e.g., pot gear targeting crab, lobster, conch, fish) in continental shelf waters off New England, USA, and Nova Scotia, Canada, were a potential mortality sink that require continued monitoring and bycatch reduction efforts. However, the majority of the documented fisheries bycatch and mortality has occurred in fisheries outside of the fisheries considered in this Opinion. Across the range of the DPS, thousands of mature individuals are lost annually due to gillnet bycatch (especially off nesting beaches). In particular, studies estimate that well over 1,000 leatherback turtles die annually due to drift and bottom-set gillnets off Trinidad (Lum 2006, NMFS and USFWS 2020). Longline bycatch is also considered to be a widespread threat to the DPS, likely resulting in the loss of thousands of individuals annually

As explained in the *Environmental Baseline*, although no direct leatherback impacts (i.e., oiled sea turtles or nests) from the Deepwater Horizon oil spill in the northern Gulf of Mexico were observed, some impacts from that event may be expected. However, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would change the species' status to an extent that the expected interactions from these fisheries would result in a detectable change in the population status of leatherback sea turtles. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed actions could be seen as reducing the likelihood of survival and recovery of the species.

As described in the *Environmental Baseline*, regulatory actions have been taken to reduce anthropogenic effects to Atlantic leatherbacks. These include measures to reduce the number and severity of leatherback interactions in the U.S. Atlantic longline fisheries and the U.S. South Atlantic and Gulf of Mexico shrimp fisheries. Reducing the number of leatherback sea turtles injuries and mortalities from these activities is expected to increase the number of Atlantic leatherbacks and increase leatherback reproduction in the Atlantic. Since most of these regulatory measures have been in place for several years now, it is likely that current nesting trends reflect the benefit of these measures to Atlantic leatherback sea turtles. There are no new known sources of mortality for leatherback sea turtles in the Atlantic other than potential impacts from the Deepwater Horizon oil spill.

Based on the information provided above, the loss of 22 leatherback sea turtles annually in the Atlantic due to the fisheries will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic given the relatively large population size and measures taken to reduce the number of Atlantic leatherback sea turtles that are injured or die in the Atlantic Ocean. The fisheries have no effects on leatherback sea turtles that occur outside of the Atlantic Ocean. Given that the operation of the fisheries will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic Ocean, it will not appreciably reduce the likelihood of survival of the species.

The recovery plan for Atlantic leatherback sea turtles (NMFS and USFWS 1992) lists the following recovery objective, which is relevant to the proposed actions in this Opinion:

- The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico; St. Croix, U.S. Virgin Islands; and along the east coast of Florida.

We believe the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. Since we concluded that the potential loss of leatherback sea turtles is not likely to have any detectable effect on nesting trends, we do not believe the proposed action will impede progress toward achieving this recovery objective. Nonlethal captures of these sea turtles would not affect the adult female nesting population or number of nests per nesting season. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. Since the fisheries have no effects on leatherback sea turtles that occur outside of the Atlantic, their operation will not appreciably reduce the likelihood of recovery for the species.

9.9. Atlantic Sturgeon

Whether the reduction in numbers and reproduction from the loss of Atlantic sturgeon resulting from the proposed actions would appreciably reduce the species likelihood of survival and recovery depends on how the changes in numbers and reproduction would affect the population's growth rate, and whether the growth rate would allow the species to recover. For the population of each DPS to remain stable, a certain amount of spawning must occur within each DPS to offset deaths within each population. Two ways to measure spawning production are spawning stock biomass per recruit (SSB/R) and eggs per recruit (EPR). The EPR_{max} refers to the maximum number of eggs produced by a female Atlantic sturgeon over the course of its lifetime assuming no fishing mortality (or other sources of anthropogenic mortality, as well). Similarly, SSB/R_{max} is the expected contribution a female Atlantic sturgeon would make to the total weight of the fish in a stock that are old enough to spawn during its lifetime over the course of its lifetime, assuming no fishing mortality. In both cases, as fishing mortality increases, the expected lifetime production of a female decreases from the theoretical maximum (i.e., SSB/R_{max} or EPR_{max}) due to an increased probability the animal will be caught and, therefore, unable to achieve its maximum potential (Boreman 1997). Since the EPR_{max} or SSB/R_{max} for each individual within a population is the same, it is appropriate to talk about these parameters not only for individuals but for populations as well.

Maintaining a SSB/R of at least 20 percent of SSB/R_{max} has been suggested as a level that allows a population to remain stable (i.e., retain the capacity for survival) (Goodyear 1993). Maintaining a SSB/R of at least 50 percent of SSB/R_{max} has been suggested an appropriate target for rebuilding (i.e., recovery) (Boreman et al. 1984). Boreman (1997) indicates that since stock biomass and egg production are typically linearly correlated it is appropriate to apply the 20 percent (Goodyear 1993) and 50 percent (Boreman 1997) thresholds directly to EPR estimates.

Boreman (1997) reported adult female Atlantic sturgeon in the Hudson River could have likely sustained a fishing mortality rate of 14 percent and still retained enough spawners for the population to remain stable (i.e., maintain an EPR of at least 20 percent of EPR_{max}). Additionally, Boreman (1997) suggested a fishing mortality rate of 5 percent corresponds to maintaining an EPR of at least 50 percent of EPR_{max} (Boreman 1997). Boreman (1997) estimates were calculated using preliminary data provided by Kahnle who subsequently worked on analyses (ASMFC 1998b, Kahnle et al. 2007) which calculated $EPR_{50\%} = 0.03$ using updated and more complete information. ASMFC (2007), ASSRT (2007), and Kahnle et al. (2007) all used $F=0.03= EPR_{50\%}$ as the maximum fishing mortality rate for maintaining and recovering populations of Atlantic sturgeon. We will also use this as a metric for analyzing impacts to Atlantic sturgeon.

These fishing mortality rates are specific to adult female spawners. Since estimates of fishing mortality rates that would equal 50 percent of EPR_{max} are not available for any of the five Atlantic sturgeon DPS, the information on the Hudson River is the best available. While we have some limited information on male to female ratios for the Hudson River (Erickson et al. 2011, Kahnle et al. 2007, Pekovitch 1979), we do not know the current sex ratio for adult or sub-adult sturgeon for any of the five Atlantic sturgeon DPSs. In the absence of this information, we chose to evaluate our anticipated takes of all adults against these female-specific fishing mortality rates because we believe doing so is conservative toward the species.

We have considered the best available information to determine the DPSs of origin for lethal interactions. Using the genetic mixed stock analysis from Kazyak et al. (2020), we have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: Gulf of Maine 8.7 percent; NYB 71.4 percent; Chesapeake Bay 10.7 percent; Carolina 2.6 percent; South Atlantic 5.6 percent; and Canada 1.0 percent. Given these percentages, we expect 15 of the annual Atlantic sturgeon mortalities from the ten federal fisheries in this Opinion will originate from Gulf of Maine DPS; 118 from the New York Bight DPS, 17 from the Chesapeake Bay DPS, 4 from the Carolina DPS; and 9 from the South Atlantic DPS. That equates to 75 Gulf of Maine, 590 New York Bight, 85 Chesapeake Bay, 20 Carolina, and 45 South Atlantic DPS fish will die as a result of the fisheries every five years.

9.9.1. *Gulf of Maine DPS*

The Gulf of Maine DPS is listed as threatened, and while Atlantic sturgeon occur in several rivers of the Gulf of Maine region, recent spawning has only been physically documented in the Kennebec River. However, spawning is suspected to occur in the Androscoggin, Piscataqua, and Merrimack Rivers. There is currently no census of the number of Atlantic sturgeon in any river nor is any currently available for the entire DPS. NMFS use of the NEAMAP data indicates that the estimated ocean population of Gulf of Maine DPS Atlantic sturgeon sub-adults and adults is 7,455 individuals. Gulf of Maine origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. While there are some indications that the status of the Gulf of Maine DPS may be improving, there is currently not enough information to establish a trend for any life stage or for the DPS as a whole. The ASMFC stock assessment concluded that the abundance of the Gulf of Maine DPS is "depleted" relative to historical levels. The assessment also concluded that there was a 51 percent probability that the abundance of the Gulf of Maine DPS has increased since implementation of the 1998 fishing moratorium, but there was a 74 percent probability that mortality for the Gulf of Maine DPS exceeds the mortality threshold used for the assessment (ASMFC 2017).

The proposed action may result in an average of 123 Atlantic sturgeon takes from the Gulf of Maine DPS annually. We estimated those takes will likely result in the mortality of 15 Atlantic annually (or 75 every five years). The ASMFC stock assessment concluded that the abundance of the Gulf of Maine DPS is "depleted" relative to historical levels. The assessment also concluded that there was a 51 percent probability that the abundance of the Gulf of Maine DPS has increased since implementation of the 1998 fishing moratorium, but there was a 74 percent probability that mortality for the Gulf of Maine DPS exceeds the mortality threshold used for the assessment (ASMFC 2017).

Annually, we anticipate that an average of 15 individuals (i.e., adults and/or sub-adults) from the Gulf of Maine DPS may be lethally taken by the proposed action. The opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of one individual from the Gulf of Maine DPS annually (NMFS 2012b). The opinion for the Southeastern U.S. shrimp trawl fishery provides for an average of one lethal take from the Gulf of Maine DPS annually (NMFS 2014d). The opinion for the Atlantic shark fisheries managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of 3 (rounded up from 2.7) individuals from the Gulf of Maine DPS annually (NMFS 2020c). Collectively, we anticipate that 20 individual Atlantic sturgeon, or 0.27 percent (=20/7455*100) of the adult/sub-adult population from the Gulf of Maine DPS may be removed annually because of federal fisheries. This 0.27 percent is below the estimated 3 percent federal fishing mortality rate we believe the population could likely withstand and still maintain 50 percent of EPR_{max}. In other words, the fishing mortality from these fisheries alone would likely not result in less than 50 percent of EPR_{max} for the Gulf of Maine DPS.

The proposed action may result in the annual average removal of 15 Atlantic sturgeon that would have been reproductive adults from the Gulf of Maine DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the Gulf of Maine DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where Gulf of Maine DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by Gulf of Maine DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by Gulf of Maine DPS sub-adults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of 15 Gulf of Maine DPS Atlantic sturgeon will not appreciably reduce the likelihood of survival (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment) and recovery of the Gulf of Maine DPS. The action will not affect Gulf of Maine DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

As a recovery plan has not yet been drafted for Atlantic sturgeon, we evaluated the five listing factors. Recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The proposed action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of Gulf of Maine DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of Gulf of Maine DPS Atlantic sturgeon. The proposed action will not utilize Gulf of Maine DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the average annual mortality of 15 Gulf of Maine DPS Atlantic sturgeon; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the Gulf of Maine DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of Gulf of Maine DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the Gulf of Maine DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the average annual mortality of 15 Gulf of Maine DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

9.9.2. *New York Bight DPS*

The New York Bight DPS is listed as endangered, and while Atlantic sturgeon occur in several rivers in the New York Bight, recent spawning has only been physically documented in the Hudson and Delaware Rivers. The essential physical features necessary to support spawning and recruitment are also present in the in the Connecticut and Housatonic Rivers (82 FR 39160; August 17, 2017). However, there is no current evidence that spawning is occurring nor studies underway to investigate whether spawning is occurring in those rivers, aside from one recent study which found young South Atlantic DPS fish in the Connecticut River, which was unexpected to the researchers (Savoy et al. 2017). Based on existing data, we expect any New York Bight DPS Atlantic sturgeon in the action area to originate from the Hudson or Delaware River.

There are no abundance estimates for the entire New York Bight DPS or for the entirety of either the Hudson River or Delaware River spawning populations. There are, however, some estimates for specific life stages (e.g., natal juvenile abundance, spawning run abundance, and effective population size). Using side scan sonar technology in conjunction with detections of previously tagged Atlantic sturgeon, Kazyak et al. (2020) estimated the 2014 Hudson River spawning run size to be 466 sturgeon (95% CI = 310-745). Based on genetic analyses of two different life stages, subadults and natal juveniles, effective population size for the Hudson River spawning population has been estimated to be 198 (95% CI=171.7-230.7; O'Leary et al. 2014) and 156 (95% CI=138.3-176.1) (Waldman et al. 2019) while estimates for the Delaware River spawning population from the same studies were 108.7 (95% CI=74.7-186.1) (O'Leary et al. 2014) and 40 (95% CI=34.7-46.2) (Waldman et al. 2019). The difference in effective population size for the Hudson and Delaware River spawning populations across both studies support that the Hudson

River spawning population is the more robust of the two spawning groups. This conclusion is further supported by genetic analyses that demonstrated Atlantic sturgeon originating from the Hudson River spawning population were more prevalent in mixed aggregations than sturgeon originating from the Delaware River spawning population, even when sampling occurred in areas and at times that targeted for adults belonging to the Delaware River spawning population (Wirgin et al. 2015a, Wirgin et al. 2015b). Waldman et al.'s calculations of maximum effective population size, and comparison of these to four other spawning populations outside of the New York Bight DPS further supports our previous conclusion that the Hudson River spawning population is more robust than the Delaware River spawning population and is likely the most robust of all of the U.S. Atlantic sturgeon spawning populations.

For this Opinion, we have estimated adult and sub-adult abundance of the New York Bight DPS based on available information for the genetic composition and the estimated abundance of Atlantic sturgeon in marine waters (Damon-Randall et al. 2013, Kocik et al. 2013). We concluded that sub-adult and adult abundance of the New York Bight DPS was 34,566 sturgeon based upon the NEAMAP data. This number encompasses many age classes since sub-adults can be as young as two years old when they first enter the marine environment, and adults can live to approximately 60 years old (Hilton et al. 2016). For example, a study of Atlantic sturgeon captured in the geographic New York Bight determined that 742 of the Atlantic sturgeon captured represented 21 estimated age classes and that, individually, the sturgeon ranged in age from 2 to 35 years old (Dunton et al. 2016).

The 2017 ASMFC stock assessment determined that abundance of the New York Bight DPS is "depleted" relative to historical levels (ASMFC 2017). However, the assessment also determined there is a relatively high probability (75 percent) that the New York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 31 percent probability that mortality for the New York Bight DPS exceeds the mortality threshold used for the assessment (ASMFC 2017).

The proposed action may result in an average of 1,004 Atlantic sturgeon takes from the New York Bight DPS annually. Annually, we anticipate that 118 individuals from the New York Bight DPS may be lethally taken by the proposed action. The opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of one individual from the New York Bight DPS annually (NMFS 2012b). The opinion for the Southeastern U.S. shrimp trawl fishery provides for an average lethal takes of three individuals from the New York Bight DPS annually (NMFS 2014d). The opinion for the fisheries managed under the Consolidated HMS FMP (excluding pelagic longline) provides incidental take coverage for an average lethal take of 12 individuals from the New York Bight DPS annually (NMFS 2020c). Together, we anticipate a total of 134 Atlantic sturgeon from the New York Bight DPS may be removed annually because of federal fisheries, or 0.39 percent (=134/34,566*100) of the sub-adult/adult population in the New York Bight DPS (i.e., 34,566, based on NMFS use of the NEMAP data). This 0.39 percent is below the estimated 3 percent federal fishing mortality rate we believe the population could likely withstand and still maintain 50 percent of EPR_{max}. In other words, the fishing mortality from these fisheries alone would likely not result in less than 50 percent of EPR_{max} for the New York Bight DPS.

The proposed action may result in the anticipated annual average removal of 118 Atlantic sturgeon that would have been reproductive adults from the New York Bight DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the New York Bight DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where New York Bight DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by New York Bight DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by New York Bight DPS sub-adults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of 118 New York Bight DPS Atlantic sturgeon will not appreciably reduce the likelihood of survival (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment) and recovery of the New York Bight DPS. The action will not affect New York Bight DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

As described for the Gulf of Maine DPS, we evaluated the five listing factors as a recovery plan has not been drafted. The proposed action is not expected to modify, curtail or destroy the range of the species since it will result in a small reduction in the number of New York Bight DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of New York Bight DPS Atlantic sturgeon. The proposed action will not utilize New York Bight DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of 118, on average annually, New York Bight DPS Atlantic sturgeon; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the New York Bight DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of New York Bight DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the New York Bight DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the mortality of an annual average of 118 New York Bight DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

9.9.3. ***Chesapeake Bay DPS***

The Chesapeake Bay DPS is listed as endangered, and while Atlantic sturgeon occur and may potentially spawn in several rivers of the Chesapeake Bay. There is evidence of spawning in the James River; Pamunkey River, a tributary of the York River; and Marshyhope Creek, a tributary of the Nanticoke River (Balazik and Musick 2015, Hager et al. 2014, Kahn et al. 2014, NMFS 2017b). In addition, detections of acoustically-tagged adult Atlantic sturgeon in the Mattaponi and Rappahannock Rivers at the time when spawning occurs in others rivers, and historical evidence for these as well as the Potomac River supports the likelihood of Atlantic sturgeon spawning populations in the Mattaponi, Rappahannock, and Potomac rivers (NMFS 2017b).

Chesapeake Bay origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. There is currently no census nor enough information to establish a trend for any life stage, for the James River spawning population, or for the DPS as a whole, although the NEAMAP data indicates that the estimated ocean population of Chesapeake Bay DPS Atlantic sturgeon is 8,811 sub-adult and adult individuals. The 2017 ASMFC stock assessment determined that abundance of the Chesapeake Bay DPS is "depleted" relative to historical levels (ASMFC 2017). The assessment also determined there is a relatively low probability (36 percent) that abundance of the Chesapeake Bay DPS has increased since the implementation of the 1998 fishing moratorium, and a 30 percent probability that mortality for the Chesapeake Bay DPS exceeds the mortality threshold used for the assessment (ASMFC 2017).

The proposed action may result in an annual average of 151 Atlantic sturgeon takes from the Chesapeake Bay DPS. Annually, we anticipate that an average of 17 individuals from the Chesapeake Bay DPS may be lethally taken by the proposed action. The opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of one individual from the Chesapeake Bay DPS annually (NMFS 2012b). The opinion for the Southeastern U.S. shrimp trawl fishery provides for an average of one lethal take of an individual from the Chesapeake Bay DPS annually (NMFS 2014d). The opinion for fisheries (excluding pelagic longline) managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of three individuals from the Chesapeake Bay DPS annually (NMFS 2020c). Together, we anticipate that a total of 22 Atlantic sturgeon from the Chesapeake Bay DPS may be removed annually because of federal fisheries, or 0.25 percent (=22/8,811*100) of the adult/sub-adult population in the Chesapeake Bay DPS. This 0.25 percent is below the estimated 3 percent federal fishing mortality rate we believe the population could likely withstand and still maintain 50 percent of EPR_{max}. In other words, the fishing mortality from these fisheries alone would likely not result in less than 50 percent of EPR_{max} for the Chesapeake Bay DPS.

The proposed action may result in the anticipated annual average removal of 17 Atlantic sturgeon that would have been reproductive adults from the Chesapeake Bay DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the Chesapeake Bay DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where Chesapeake Bay DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by Chesapeake Bay DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by Chesapeake Bay DPS sub-adults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of 17 Chesapeake Bay DPS Atlantic sturgeon will not appreciably reduce the likelihood of survival (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment) and recovery of the of the Chesapeake Bay DPS. The action will not affect Chesapeake Bay DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

As described for the Gulf of Maine DPS, we evaluated the five listing factors as a recovery plan has not been drafted. The proposed action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of Chesapeake Bay DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of Chesapeake Bay DPS Atlantic sturgeon. The proposed action will not utilize Chesapeake Bay DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of an annual average 17 Chesapeake Bay DPS Atlantic sturgeon; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the Chesapeake Bay DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of Chesapeake Bay DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the Chesapeake Bay DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the average annual mortality of 17 Chesapeake Bay DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

9.9.4. *Carolina DPS*

The Carolina DPS is listed as endangered and consists of Atlantic sturgeon originating from at least five rivers where spawning is still thought to occur. Carolina DPS origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. There is currently no census of the number of Atlantic sturgeon in any river nor is any currently available for the entire DPS, although the NEAMAP data indicates that the estimated ocean population of Carolina DPS Atlantic sturgeon, sub-adults and adults, is 1,356 individuals. The 2017 ASMFC stock assessment determined that abundance of the Carolina DPS is "depleted" relative to historical levels (ASMFC 2017). The assessment also determined there is a relatively high probability (67 percent) that abundance of the Carolina DPS has increased since the implementation of the 1998 fishing moratorium, and a 75 percent probability that mortality for the Carolina DPS exceeds the mortality threshold used for the assessment (ASMFC 2017).

The proposed action may result in 36 Atlantic sturgeon takes from the Carolina DPS annually. Annually, we anticipate that four individuals from the Carolina DPS may be lethally taken by the proposed action. The opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of one individual from the Carolina DPS annually (NMFS 2012b). The opinion for the Southeastern U.S. shrimp trawl fishery provides for an average of one lethal take of an individual from the Carolina DPS annually (NMFS 2014d). The opinion for HMS fisheries (excluding pelagic longline) managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of two (rounded up from 1.7) individuals from the Carolina DPS annually (NMFS 2020c). Together, we anticipate that a total of 8 Atlantic sturgeon from the Carolina DPS may be removed annually because of federal fisheries, or 0.59 percent (=8/1,356*100) of the subadult and adult population in the Carolina DPS. This 0.59 percent is below the estimated 3 percent federal fishing mortality rate we believe the population could likely withstand and still maintain 50 percent of EPR_{max}.

The proposed action may result in the anticipated annual average removal of four Atlantic sturgeon that would have been reproductive adults from the Carolina DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the Carolina DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where Carolina DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by Carolina DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by Carolina DPS sub-adults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of four Carolina DPS Atlantic sturgeon will not appreciably reduce the likelihood of survival (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment) and recovery of the of the Carolina DPS. The action will not affect Carolina DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

As described for the Gulf of Maine DPS, we evaluated the five listing factors as a recovery plan has not been drafted. The proposed action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of Carolina DPS

Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of Carolina DPS Atlantic sturgeon. The proposed action will not utilize Carolina DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of no more than 4 Carolina DPS Atlantic sturgeon, on average annually; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the Carolina DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of Carolina DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the Carolina DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the average annual mortality of four Carolina DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

9.9.5. *South Atlantic DPS*

The South Atlantic DPS is listed as endangered and consists of Atlantic sturgeon originating from at least six rivers where spawning is still thought to occur. In 2004 and 2005, there were an estimated 343 adults spawning in the Altamaha River, Georgia (Schueller and Peterson 2006). This represents a percentage of the total adult population for the Altamaha River. Males spawn every 1-5 years and females spawn every 2-5 years; thus, the total Altamaha River adult population, assuming a 2:1 ratio of males to females as seen in the Hudson River, could range from 457- 1,715. Spawning occurs in at least five other rivers in this DPS. Therefore, the number of Atlantic sturgeon in the Altamaha River population is only a portion of the total DPS. No census of the number of Atlantic sturgeon in any of the other spawning rivers or for the DPS as a whole is available. However, the NEAMAP data indicates that the estimated ocean population of South Atlantic DPS Atlantic sturgeon sub-adults and adults is 14,911 individuals.

The 2017 ASMFC stock assessment determined that abundance of the South Atlantic DPS is "depleted" relative to historical levels (ASMFC 2017). Due to a lack of suitable indices, the assessment was unable to determine the probability that the abundance of the South Atlantic DPS has increased since the implementation of the 1998 fishing moratorium. However, it was determined that there is a 40 percent probability that mortality for the South Atlantic DPS exceeds the mortality threshold used for the assessment (ASMFC 2017).

The proposed action may result in 79 Atlantic sturgeon takes from the South Atlantic DPS annually. Annually, we anticipate that an average of nine individuals from the South Atlantic DPS may be lethally taken by the proposed action. The opinion for the Atlantic sea scallop trawl fishery provides for an average lethal take of one adult or sub-adult sturgeon from the South Atlantic DPS annually (NMFS 2012b). The opinion for the Southeastern U.S. shrimp trawl fishery provides for an average of two lethal takes adult or sub-adult sturgeon from the South

Atlantic DPS annually (NMFS 2014d). The opinion on fisheries (excluding pelagic longline) managed under the Consolidated HMS FMP provides incidental take coverage for an average lethal take of 7 (rounded up from 6.3 individuals) from the South Atlantic DPS annually (NMFS 2020c). Together, we anticipate that 19 Atlantic sturgeon from the South Atlantic DPS may be removed annually because of federal fisheries, or 0.13 percent (=19/14,911*100) of the adult and sub-adult population in the South Atlantic DPS. This 0.13 percent is below the estimated 3 percent federal fishing mortality rate we believe the population could likely withstand and still maintain 50 percent of EPR_{max}. In other words, the fishing mortality from these fisheries alone would likely not result in less than 50 percent of EPR_{max} for the South Atlantic DPS.

The proposed action may result in the anticipated annual average removal of nine Atlantic sturgeon that would have been reproductive adults from the South Atlantic DPS, which would reduce the reproductive potential of the DPS. The reproductive potential of the South Atlantic DPS will not be affected in any way other than through a reduction in numbers of individuals. Reproductive potential of other captured and released individuals is not expected to be affected in any way. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. The proposed action will also not affect the spawning grounds within the rivers where South Atlantic DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds used by South Atlantic DPS fish. The proposed action is not likely to reduce distribution because the action will not impede Atlantic sturgeon from accessing any seasonal concentration areas, including foraging areas that may be used by South Atlantic DPS sub-adults or adults. Further, the action is not expected to reduce the river-by-river distribution of Atlantic sturgeon. Any effects to distribution will be minor and temporary.

Based on the information provided above, the annual average death of nine South Atlantic DPS Atlantic sturgeon will not appreciably reduce the likelihood of survival (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment) and recovery of the of the South Atlantic DPS . The action will not affect South Atlantic DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population to persist, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring.

Recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The proposed action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of South Atlantic DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of South

Atlantic DPS Atlantic sturgeon. The proposed action will not utilize South Atlantic DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The proposed action is likely to result in the capture and injury of Atlantic sturgeon and the mortality of no more than nine South Atlantic DPS Atlantic sturgeon, on average annually; as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the South Atlantic DPS. As the reduction in numbers and future reproduction is not significant, the loss of these individuals is not likely to change the status of South Atlantic DPS Atlantic sturgeon. The effects of the proposed action will not likely delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause the mortality of a significant percentage of the species as a whole and this mortality is not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the South Atlantic DPS can be brought to the point at which they are no longer listed as threatened.

Based on the analysis presented herein, the proposed action, resulting in the average annual mortality of nine South Atlantic DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

9.10. GOM DPS Atlantic Salmon

Atlantic salmon have been observed to interact with both gillnet and bottom trawl gear used in the ten fisheries that are the focus of this Opinion. Based on recent NEFOP and ASM data from 2010-2019, as well as historic observer program data dating back to 1989, we anticipate two interactions with either bottom trawl or gillnet gear every five years as a result of the operation of these fisheries, both of which may be lethal. Due to the sizes of the fish involved and the locations of these past captures, future interactions may be with either post-smolt or adult Atlantic salmon and all are anticipated to be with ESA-listed GOM DPS fish.

The marine life history of Atlantic salmon of U.S. origin is not as well understood as the freshwater phase. Atlantic salmon of U.S. origin are highly migratory, undertaking long marine migrations from their natal rivers to the Northwest Atlantic Ocean, where they are distributed seasonally over much of the region. The marine phase starts with the completion of smoltification and migration through the estuary of the natal river. Part of the migratory pattern of post-smolts and adults overlaps with the action area at times when the ten fisheries are active.

To determine if the proposed actions will jeopardize the GOM DPS of Atlantic salmon, we must conduct an analysis of the effects of the proposed actions on the likelihood of the species' survival and recovery. The 2019 recovery plan for Atlantic salmon (USFWS and NMFS 2019) incorporates an approach termed Recovery Planning and Implementation, which focuses on the three statutory requirements in the ESA, including site-specific recovery actions; objective, measurable criteria for delisting; and time and cost estimates to achieve recovery and intermediate steps. The 2019 recovery plan projects four phases of recovery over a 75-year timeframe to achieve delisting of the GOM DPS of Atlantic salmon. The four phases are:

- Phase 1: The first recovery phase focuses on identifying the threats to the species and characterizing the habitat needs of the species necessary for their recovery.

- Phase 2: The second recovery phase focuses on ensuring the persistence (survival) of the GOM DPS through the use of the conservation hatcheries while abating imminent threats to the continued existence of the DPS. Phase 2 focuses on freshwater habitat used by Atlantic salmon for spawning, rearing, and upstream and downstream migration; it also emphasizes research on threats within the marine environment.
- Phase 3: The third phase of recovery will focus on increasing the abundance, distribution, and productivity of naturally reared Atlantic salmon. It will involve transitioning from dependence on the conservation hatcheries to wild smolt production.
- Phase 4: In Phase 4, the GOM DPS of Atlantic salmon is recovered and delisting occurs. The GOM DPS will be considered recovered once: a) 2,000 wild adults return to each Salmon Habitat Recovery Unit (SHRU), for a DPS-wide total of at least 6,000 wild adults; b) each SHRU has a population growth rate of greater than 1.0 in the 10-year period preceding delisting, and, at the time of delisting, the DPS demonstrates self-sustaining persistence; and c) sufficient suitable spawning and rearing habitat for the offspring of the 6,000 wild adults is accessible and distributed throughout the designated Atlantic salmon critical habitat, with at least 30,000 accessible and suitable HUs in each SHRU, located according to the known migratory patterns of returning wild adult salmon.

We are presently in Phase 2 of the recovery program (ensuring the survival of the GOM DPS through the use of the conservation hatcheries while abating imminent threats to the continued existence of the DPS). As indicated in the 2019 recovery plan for Atlantic salmon, the U.S. FWS and NMFS do not have plans to transition from dependence on conservation hatcheries to wild fish production in the foreseeable future. Therefore, for purposes of our survival analysis, we assume hatchery supplementation will continue in all three SHRUs over the course of the proposed actions. We also expect that as passage improves in certain Gulf of Maine rivers, it may become a higher priority for stocking. The hatchery program, sponsored by the U.S. FWS, has been in place for over 100 years and because we do not have any information to the contrary, we expect it will continue over the duration of the proposed actions. The importance of continuation of the hatchery program is recognized in the 2019 recovery plan and continuation of the hatchery and stocking efforts are an integral part of the recovery strategy.

As detailed in the 2019 recovery plan, in order for the listing status of Atlantic salmon to change, each of the three relevant biological criteria (abundance, productivity, and habitat) must be met in two (downlisting) or three (delisting) of the recovery units. The biological criteria for reclassifying (downlisting) the GOM DPS of Atlantic salmon from endangered status to threatened status are:

- ***Abundance:*** The DPS has total annual returns of at least 1,500 adults originating from wild origin, or hatchery stocked eggs, fry or parr spawning in the wild, with at least 2 of the 3 SHRUs having a minimum annual escapement of 500 naturally reared adults.
- ***Productivity:*** Among the SHRUs that have met or exceeded the abundance criterion, the population has a positive mean growth rate greater than 1.0 in the 10-year (two-generation) period preceding reclassification.
- ***Habitat:*** In each of the SHRUs where the abundance and productivity criterion have been met, there is a minimum of 7,500 units of accessible and suitable spawning and rearing habitats capable of supporting the offspring of 1,500 naturally reared adults.

The biological criteria for removing Atlantic salmon from the endangered species list are:

- ***Abundance*****:** The DPS has a self-sustaining annual escapement of at least 2,000 wild origin adults in each SHRU, for a DPS-wide total of at least 6,000 wild adults.
- ***Productivity:*** Each SHRU has a positive mean population growth rate of greater than 1.0 in the 10-year (two-generation) period preceding delisting. In addition, at the time of delisting, the DPS demonstrates self-sustaining persistence, whereby the total wild population in each SHRU has less than a 50-percent probability of falling below 500 adult wild spawners in the next 15 years based on PVA projections.
- ***Habitat:*** Sufficient suitable spawning and rearing habitat for the offspring of the 6,000 wild adults is accessible and distributed throughout the designated Atlantic salmon critical habitat, with at least 30,000 accessible and suitable Habitat Units in each SHRU, located according to the known migratory patterns of returning wild.

In 2019, 1,528 pre-spawn salmon returned to the GOM DPS (includes wild, naturally-reared, and hatchery raised salmon). Of those, 15 percent returned to the Downeast Coastal SHRU, 79 percent returned to the Penobscot Bay SHRU, and 6 percent returned to the Merrymeeting Bay SHRU. The abundance of returning salmon was more than 20 percent higher than the 10-year average, and the proportion of the run that was naturally reared (24 percent) was higher than what has been seen on average over the last decade (16 percent) (Kircheis et al. 2020). Regardless, the abundance of wild and naturally reared returns remain well below what is needed for either reclassification or delisting (USFWS and NMFS 2019). Based upon the 2019 return percentages summarized above, we expect that 79 percent of future mortalities from the ten fisheries will be salmon from the Penobscot Bay SHRU while the other 21 percent of mortalities will be from the other two SHRUs. Over a 10-year period, that would roughly equate to three mortalities of GOM DPS salmon from the larger Penobscot Bay SHRU and one mortality from the smaller Downeast Coastal or Merrymeeting Bay SHRUs.

The mean 10-year population growth rate for the GOM DPS as a whole in 2019 was 1.12, making it the eighth consecutive year where that threshold rate has exceeded 1.0. However, the reclassification and delisting productivity criteria require that each SHRU sustain a population growth rate of more than 1.0, in addition to meeting the relevant abundance criteria. In 2019, the 1.0 threshold was exceeded at both the Merrymeeting Bay (1.84) and Penobscot Bay (1.08) SHRUs, but was not met at the Downeast Coastal SHRU (0.99) (USFWS and NMFS 2019).

In 2019, a minimum of 31 connectivity projects were conducted that improved access to 108 stream miles of rivers in the GOM DPS. These projects do not necessarily lead to gains that can be counted towards the habitat recovery criteria, as many of them are upstream of barriers that have not yet been deemed accessible themselves. However, the most notable project in 2019, the breaching of the Head Tide Dam on the Sheepscot River, restored access to 2,363 habitat units in the Merrymeeting Bay SHRU, which have been added to the total accessible habitat units under the recovery criteria. It should be noted that the number of projects reported in the 2019 SHRU reports are likely an underestimate of the number of projects actually conducted. As of 2019, all three SHRUs have achieved the reclassification (downlisting) goal of at least 7,500 accessible habitat units. However, none of the SHRUs have yet to achieve the delisting goal of 30,000 accessible habitat units (USFWS and NMFS 2019).

The jeopardy analysis below makes a conclusion regarding the survival and recovery of the GOM DPS of Atlantic salmon as a whole, and not just survival and recovery of the species in the action area. Therefore, in the survival and recovery portions of this analysis, we consider how the consequences to individual salmon that were identified in the *Effects of the Proposed Actions* section of this Opinion will affect the marine population of Atlantic salmon, how the consequences to the marine population will affect the Downeast Coastal, Penobscot Bay, and Merrymeeting Bay SHRUs, and then finally, how the consequences to the three SHRUs are likely to affect the survival and recovery of the GOM DPS as a whole. As highlighted in the 2019 recovery plan, the survival and recovery of all three SHRUs is necessary for attainment of the delisting criteria and recovery of the GOM DPS.

Survival Analysis
When considering how a proposed action is likely to affect the survival of a species, we consider effects to reproduction, numbers, and distribution. The number of returning adult Atlantic salmon to the Downeast Coastal, Penobscot Bay, and Merrymeeting Bay SHRUs is a measure of both the reproduction and numbers of the species. We consider the ability of pre-spawn Atlantic salmon to access high quality spawning and rearing habitat in the six major Downeast Rivers (i.e., Dennys, East Machias, Machias, Pleasant, Narraguagus, and Union), the Penobscot River, and the Kennebec and Androscoggin Rivers as a measure of distribution. Below, we analyze whether the proposed actions will reduce the numbers, reproduction, or distribution of the GOM DPS of Atlantic salmon in the action area and the three SHRUs to a point that appreciably reduces the species' likelihood of survival in the wild.

Two lethal interactions in the ten fisheries every five years would reduce the number of GOM DPS Atlantic salmon compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal interactions would also result in a potential reduction in future reproduction, assuming some individuals would be females and would have otherwise survived to reproduce. For example, an adult 2SW female Atlantic salmon can produce a total of 1,500 to 1,800 eggs per kilogram of body weight, yielding an average of 7,500 eggs (Baum and Meister 1971), of which a small percentage are expected to survive to sexual maturity. A lethal capture of an adult female GOM DPS Atlantic salmon in gillnet or bottom trawl gear would likely remove this level of reproductive output from the species. Over a 10-year period, three adult females could be removed from the Penobscot Bay SHRU and one adult female from either the Downeast Coastal or Merrymeeting Bay SHRU. The anticipated lethal interactions could occur anywhere in the action area, but are most likely to occur in the Gulf of Maine or on Georges Bank. Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The most recent data available on the population trend of Atlantic salmon indicate that their abundance within the range of the GOM DPS has been generally declining since the 1800s (Fay et al. 2006). Contemporary estimates of abundance for the entire GOM DPS have rarely exceeded 5,000 individuals in any given year since 1967 (Fay et al. 2006), and appear to have stabilized at very low levels since 2000. After a period of slow population growth between the 1970s and the early 1980s, adult returns of salmon in the GOM DPS peaked around 1985 and declined through the 1990s and early 2000s. The population growth observed in the 1970s is likely attributable to favorable marine survival and increases in hatchery capacity, particularly from the Green Lake National Fish Hatchery that was constructed in 1974. Marine survival remained relatively high throughout the 1980s, and salmon populations in the GOM DPS remained relatively stable until the early 1990s. In the early 1990s, marine survival rates decreased, leading to the declining trend in adult abundance observed throughout 1990s and early 2000s. An increase in the abundance of returning adult salmon was observed between 2008 and 2011, but returns then dropped significantly after 2011. The last couple of years have been relatively good years for returns (higher than the 10-year average), but have not been close to what was observed in 2011.

Adult returns for the GOM DPS remain well below conservation spawning escapement (CSE) goals that are widely used (ICES 2005) to describe the status of individual Atlantic salmon populations. When CSE goals are met, Atlantic salmon populations are generally self-sustaining. When CSE goals are not met (i.e., less than 100 percent), populations are not reaching full potential; and this can be indicative of a population decline. For all GOM DPS rivers in Maine, current Atlantic salmon populations (including hatchery contributions) are well below CSE levels required to sustain themselves (Fay et al. 2006), which is further indication of their poor population status.

The observed declines in Atlantic salmon suggests that the combined impacts from ongoing activities described in the *Environmental Baseline*, *Cumulative Effects*, and the *Status of Listed Species* (including those activities that occur outside of the action area of this Opinion) are continuing to cause the population to deteriorate. However, we believe the proposed actions are not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the GOM DPS Atlantic salmon. For the population to remain stable, Atlantic salmon must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals from successful reproduction of Atlantic salmon that were not seriously injured or did not die in the fisheries. While the abundance trend information for Atlantic salmon is either stable or declining, we believe the very small numbers of lethal interactions attributed to the proposed actions will not have any measurable effect on that trend for three reasons. First, the loss of individual Atlantic salmon due to the proposed actions is not expected to impact the genetic heterogeneity of the three SHRUs or the species as a whole because the fisheries are widespread throughout the Gulf of Maine and Georges Bank and not likely to disproportionately capture individuals from the smaller, more vulnerable Merrymeeting Bay and Downeast Coastal SHRUs compared to the larger Penobscot Bay SHRU. Second, although we assume that the Atlantic salmon captures could be of high reproductive value adult females from the GOM DPS, it is possible that they could also be from non-listed Canadian, Saco, Merrimack, or Connecticut River stocks or involve adult male or post-smolt life stages with less reproductive value. Finally, the already existing salmon hatchery programs throughout the range of the GOM DPS should be able to replace the small amount of individuals lost from the DPS due to the ten fisheries over time (as long as they continue to operate, biological recovery criteria continue to be met, and freshwater and marine survival do not get significantly worse).

In summary, the proposed actions are anticipated to result in a small decrease in the numbers and reproduction of Atlantic salmon in the action area and the DPS as a whole, compared to current conditions. When compared to a future scenario without the proposed actions (i.e., no fishing activities under the ten fisheries), the proposed action would reduce the potential numbers and reproductive potential (through a reduction in numbers) of Atlantic salmon in the North Atlantic Ocean, but would have a negligible impact on the species' distribution. Based on the analysis provided above, the potential loss of two Atlantic salmon post-smolts or adults every five years will not reduce the likelihood of survival of the GOM DPS of Atlantic salmon.

The *Status of the Species* section and four-phased approach summarized above generally describe the actions needed for recovery of the GOM DPS. Although commercial and recreational fisheries are identified in the recovery plan as threats to the GOM DPS, the fisheries included in this Opinion are not identified among those threats, which include the directed West Greenland fishery as well as directed and subsistence fisheries in Newfoundland and Labrador, Canada. Improving the survival of Atlantic salmon in the marine environment is also an important part of meeting the objective of GOM DPS Atlantic salmon recovery (USFWS and NMFS 2019). The average return estimate for all GOM DPS Atlantic salmon from 2010-2019 is 1,247 fish (Kircheis et al. 2020). The number of interactions (0.4) annually is less than 0.04 percent (=0.4/1,247*100) of the returning population. Given that we determined above that this small number of lethal interactions will not affect the population trends, there is no indication that bycatch in the ten fisheries assessed here are considered a threat to Atlantic salmon recovery.

As mentioned in the survival analysis above, the proposed actions will not affect Atlantic salmon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring and it will not result in effects to the environment that would prevent Atlantic salmon from completing their entire life cycle, including reproduction, sustenance, and shelter.

Despite the threats faced by individual Atlantic salmon inside and outside of the action area, the proposed action will not increase the vulnerability of individual Atlantic salmon to these threats, and exposure to ongoing threats will not increase susceptibility to effects related to the proposed actions. While we are not able to predict with precision how climate change will impact GOM DPS Atlantic salmon in the action area or how the species will adapt to climate change-related environmental impacts, we do not expect the proposed actions to contribute to climate related effects in the action area. We have considered the effects of the proposed actions in light of cumulative effects explained above, including climate change, and have concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change. Therefore, we believe that the capture and subsequent loss of two GOM DPS Atlantic salmon every five years as a result of the operation of the fisheries will not reduce the likelihood of recovery the GOM DPS of Atlantic salmon. Based on the analysis presented herein, the proposed actions are not likely to appreciably reduce the survival and recovery of the species.

9.11. Giant Manta Ray

As described in the *Status of the Species*, giant manta rays can be found worldwide and are listed as threatened throughout its range. There are no current and accurate abundance estimates available, as the species tends to be only sporadically observed. There is no population growth rate available for the giant manta ray, however, the best available data indicate that the species has suffered population declines of significant magnitude (up to 95 percent in some places) in the Indo-Pacific and Eastern Pacific portion of its range. These declines are largely based on trends in landings and market data, diver sightings, and anecdotal observations. The observed declines in giant manta rays suggest that the combined impacts from ongoing activities described in the *Environmental Baseline*, *Cumulative Effects*, and the *Status of the Species* (including those activities that occur outside of the action area of this Opinion) are continuing to cause the population to deteriorate. Giant manta rays are at risk throughout a significant portion of their range, due in large part to the observed declines in the Indo-Pacific. However, larger subpopulations of the species still exist, including off Mozambique, Ecuador, and potentially Thailand. In areas where the species is not subject to fishing, populations may be stable. However, given the migratory nature of this species, population declines in waters where the manta rays are protected have also been observed but are attributed to overfishing of the species in adjacent areas within its large home range.

As described above, it is unlikely that overutilization as a result of bycatch mortality is a significant threat to giant manta rays in the Atlantic Ocean (83 FR 2916; January 22, 2018). However, information is severely lacking on both population sizes and distribution of the giant manta ray as well as current catch and fishing effort on the species throughout this portion of its range. The species is not considered to be at high risk in the Atlantic; however, if the species was hypothetically extirpated within the Indo-Pacific and eastern Pacific portion of the range, only the potentially small and fragmented Atlantic populations would remain. The demographic risks associated with small and fragmented populations, and discussed in the proposed rule 82 FR 3694, January 12, 2017), such as demographic stochasticity, dispensation, and inability to adapt to environmental changes, would become significantly greater threats to the species as a whole, and coupled with the species' inherent vulnerability to depletion, indicate that even low levels of mortality could cause drastic declines in the population.

As described in the *Environmental Baseline* and *Climate Change* sections, effects from U.S. fishing have resulted in interactions with giant manta rays and large-scale impacts that affect ocean temperatures, currents, and potentially food chain dynamics, may pose a threat to this species. However, given the migratory behavior of the giant manta ray and tolerance to both tropical and temperate waters, these animals likely have the ability to shift their range or distribution to remain in an environment conducive to their physiological and ecological needs, providing the species with resilience to these effects.

As described in the *Effect of the Action* section, based on NEFOP data, we anticipate that 4 interactions will occur every five years. One lethal take is expected to occur in either gillnet or trawl gear over the 10-year period of this Opinion.

The non-lethal interactions are not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Manta rays appear to be able to heal from wounds very quickly (McGregor et al. 2019). A study of *Manta alfredi* in Indonesia found that while the long-term impacts of sublethal effects from fishing interactions is unknown, there are some observations to suggest that reproductively is not significantly impaired. Observations of pregnant individuals with single cephalic fin amputations suggest that manta rays retain their reproductive fitness even with these sub-lethal injuries. Further, the proportion of pregnant manta rays with injuries was not significantly lower than the overall proportion of females with injuries, suggesting that injured manta rays are not substantially impaired reproductively (Germanov et al. 2019). Given this information, we anticipate no reductions in reproduction or numbers of this species. Since these captures may occur throughout the action area and would be released within the general area where caught, no change in the distribution of this species is anticipated. Therefore, we believe the non-lethal take of an average of 0.8 giant manta rays per year will not result in population level impacts nor will it change their distribution.

Lethal interactions would reduce the number of giant manta rays, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal interactions would also result in a potential reduction in future reproduction, assuming the individual is female and would have otherwise survived to reproduce. A lethal capture of an adult female giant manta ray in gillnet or trawl gear would remove this level of reproductive output from the species. The anticipated lethal interaction is expected to occur anywhere in the action area south of Long Island, New York. Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

Available abundance data is represented by records of more than 90 individuals (Kendall 2010 as cited in Miller and Klimovich 2017) and over 500 individuals (F. Young, pers. comm. 2017 as cited in Miller and Klimovich 2017) observed off the east coast of Florida. If we assume the population present in the action area includes at least 500 individuals, one lethal take would represent 0.2 percent (=1/500*100) of this subpopulation. This one interaction, however, would be expected over the 10-years period.

For the population to remain stable, giant manta rays must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the survival rate to maturity is greater than the mortality rate of the population, the loss of a breeding individual would be exceeded through recruitment of new breeding individuals from successful reproduction of giant manta rays that did not die as a result of the fisheries. While the abundance trend information for giant manta ray is declining, and possibly stable in areas where it is not subject to fishing, we believe that one lethal interaction attributed to the proposed actions over a 10-year period will not have any measurable effect on that trend. Thus, we believe the impact of the fisheries on giant manta rays is sufficiently small.

Since giant manta rays were recently listed, a recovery plan for them is not yet available. However, the first step in recovering a species is to reduce identified threats; only by alleviating threats can lasting recovery be achieved. The Final Listing Rule (83 FR 2916, January 22, 2018) noted that overall, current management measures that are in place for fishermen under U.S. jurisdiction appear to directly and indirectly contribute to the infrequency of interactions between U.S. fishing activities and the threatened giant manta ray. As such, NMFS does not believe these activities are contributing significantly to the identified threats of overutilization and inadequate regulatory measures and did not find that developing regulations under section 4(d) to prohibit some or all of these activities is necessary and advisable for the conservation of the species (considering the U.S. interaction with the species is negligible and its moderate risk of extinction is primarily a result of threats from foreign fishing activities). Any conservation actions for the giant manta ray that would bring it to the point that the measures of the ESA are no longer necessary will ultimately need to be implemented by foreign nations.

The proposed action is not likely to impede giant manta rays from continuing to survive and will not impede the process of restoring the ecosystems that affect giant manta rays. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the giant manta ray’s ability to meet its lifecycle requirements and to retain the potential for recovery. While a preliminary study suggests that the species may exist as isolated subpopulations (Stewart et al. 2016a) available tracking information indicates that manta rays are pelagic and migratory and can likely travel large distances to reproduce (Clark 2010, Miller and Klimovich 2017). A study off the Yucatan Peninsula found shared haplotypes across Atlantic, Pacific, and Indian Ocean basins, indicative a highly migratory species, with little impedance to dispersal and therefore, gene flow (Hacohen-Domené et al. 2017). The conservation biology "50/500" rule-of-thumb suggests that the effective population size (Ne; the number of reproducing individuals in a population) in the short term should not be <50 individuals in order to avoid inbreeding depression and demographic stochasticity (Franklin 1980, Harmon and Braude 2010). In the long-term, Ne should not be < 500 in order to decrease the impact of genetic drift and potential loss of genetic variation that will prevent the population from adapting to environmental changes (Franklin 1980, Harmon and Braude 2010). Taking into consideration this information and given the size of the population (at least 500 individuals) and the low number of takes estimated in the fisheries (<1 on average annually), we conclude the authorization of the fisheries will not result in the appreciably reduction of the giant manta population such that the rates of dispersal and in turn, gene flow, are altered in manner that reduces the populations ability to respond to stochastic environmental and demographic events. Given this, we expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. Based on the evidence available, we conclude that the incidental take of four giant manta rays every five years and the mortality of one giant manta ray over a 10-year period would not be expected to appreciably reduce the threatened giant manta ray's likelihood of survival and recovery in the wild.

10. INCIDENTAL TAKE STATEMENT (INCLUDING RPMS, T&CS, AND TAKE MONITORING PROTOCOL)

10.1. 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, unless a special exemption has been granted. Take is defined as "to harass, harm, pursue, hunt, shoot, capture, or collect, or to attempt to engage in any such conduct." Harm is further defined by NMFS to include any act which actually kills or injures fish or wildlife. Such an act may 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. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. "Otherwise lawful activities" are those actions that meet all state and federal legal requirements except for the prohibition against taking in ESA section 9 (51 FR 19936, June 3, 1986), which would include any state endangered species laws or regulations. Section 9(g) makes it unlawful for any person "to attempt to commit, solicit another to commit, or cause to be committed, any offense defined [in the ESA]" (16 U.S.C. 1538(g)). A "person" is defined in part as any entity subject to the jurisdiction of the United States, including an individual, corporation, officer, employee, department or instrument of the Federal government (see 16 U.S.C. 1532(13)). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not the purpose of carrying out an otherwise lawful activity is not considered to be prohibited under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement (ITS). In issuing ITSs, NMFS takes no position on whether an action is an "otherwise lawful activity."

The prohibitions against incidental take are currently in effect for endangered large whales, sea turtles, all five listed DPSs of Atlantic sturgeon, and the GOM DPS of Atlantic salmon. When a proposed federal action is found to be consistent with section 7(a)(2) of the ESA, section 7(b)(4) of the ESA requires NMFS or the USFWS to issue a statement specifying the impact of incidental taking, if any. It also states that reasonable and prudent measures (RPMs) necessary to minimize impacts of any incidental take be provided along with implementing terms and conditions. The measures described below are non-discretionary and must, therefore, be undertaken in order for the exemption in section 7(o)(2) to apply. Failure to implement the terms and conditions through enforceable measures may result in a lapse of the protective coverage section of 7(o)(2).

An ITS is not required to provide protective coverage for the giant manta ray because there are no take prohibitions under ESA section 4(d) for these species. Consistent with the decision in *Center for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012), however, this ITS is included to serve as a check on the no-jeopardy conclusion by providing a reinitiation trigger if the level of take analyzed in the Opinion is exceeded.

NMFS is including an incidental take exemption for non-lethal take of North Atlantic right, fin, sei, and sperm whales. At this time, we are authorizing zero lethal take of these whales because the lethal incidental take of ESA-listed whales has not been authorized under section 101(a)(5) of the MMPA. Following the issuance of such authorizations, NMFS may amend this Opinion to adjust lethal incidental take allowance for these species, as appropriate. NMFS recognizes that further efforts are necessary to reduce interactions between authorized federal fisheries and large whales in order to achieve the MMPA's goal of insignificant levels of incidental mortality and serious injury of marine mammals approaching a zero mortality and serious injury rate, taking into consideration the economics of the fishing industry, the availability of existing technology, and existing state or regional fishery management plans. NMFS continues to work toward this zero mortality goal of the MMPA through the means identified in the pertinent subsections of section 5.4 including continued development and implementation of the ALWTRP with the collaboration of the ALWTRT. Although NMFS has concluded that with implementation of the Framework, the ten fisheries are not likely to jeopardize the continued existence of the species in the wild by appreciably reducing the likelihood of survival and recovery of right, fin, sei, and sperm whales, the need for further efforts among stakeholders to reduce large whale/fishery interactions and achieve the zero mortality goal of the MMPA is not diminished by these no-jeopardy conclusions.

NMFS anticipates the following incidental takes of large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays may occur in the future because of the proposed action. The level of takes occurring annually is variable and influenced by sea temperatures, species abundances, fishing effort, and other factors that are difficult to predict. Because of this variability, it is unlikely that all species evaluated in this Opinion will be consistently impacted year after year. For example, some years may have no observed or otherwise documented interactions and, thus, no estimated take will occur. As a result, monitoring fisheries using 1-year estimated take levels is largely impractical. For these reasons, and based on our experience monitoring fisheries, we believe a 5-year time period is appropriate for meaningful monitoring of take with respect to the ITS. Table 83 displays the annual average take of these species over five years. In the case of North Atlantic right whales, take is specified as an annual percentage of the total population, as noted in the table.

Table 83: Average annual take over a 5-year period

	Total Take	**Lethal Take**[71]
Large Whales		
North Atlantic right whale	9.14% of population	0
Fin whale	1.89	0
Sei whale	1	0
Sperm whale	1	0
Sea Turtles		
Green, North Atlantic DPS	Gillnet: 2 Trawl: 6.4	Gillnet: 1.6 Trawl: 3.2
Kemp's ridley	Gillnet: 47.8 Trawl: 10.6	Gillnet: 37.4 Trawl: 5.4
Loggerhead, NWA DPS	Gillnet: 207.2 Trawl: 190.8 Pot/trap: 1	Gillnet: 161.6 Trawl: 95.4 Pot/trap: 0.8
Leatherback	Gillnet: 10.4 Trawl: 8 Pot/trap: 10	Gillnet: 8.2 Trawl: 4 Pot/trap: 6.4
Any combination of turtle species	Vessel strike: 3	Vessel strike: 3
ESA-listed Fish		
Atlantic sturgeon, Gulf of Maine DPS	Gillnet: 55 Trawl: 68	Gillnet: 11 Trawl: 4
Atlantic sturgeon, New York Bight DPS	Gillnet: 448 Trawl: 556	Gillnet: 90 Trawl: 28
Atlantic sturgeon, Chesapeake Bay DPS	Gillnet: 68 Trawl: 83	Gillnet: 13 Trawl: 4
Atlantic sturgeon, Carolina DPS	Gillnet: 16 Trawl: 20	Gillnet: 3 Trawl: 1
Atlantic sturgeon, South Atlantic DPS	Gillnet: 35 Trawl: 44	Gillnet: 7 Trawl: 2
Atlantic salmon	Gillnet and trawl combined: 0.4	Gillnet and trawl combined: 0.4
Giant manta ray	Gillnet and trawl combined: 0.8	Gillnet and trawl combined: 0.1 (1 every 10 years)

[71] For species with zero authorized lethal take, that take is exceeded and reinitiation is required if the proposed action results in a single lethal take or a prorated M/SI is assigned to the federal fishery.

10.2. Reasonable and Prudent Measures

NMFS has determined that the following Reasonable and Prudent Measures (RPMs) and Terms and Conditions (T&Cs) are necessary or appropriate to minimize impacts of the incidental take on large whales, sea turtles, the five DPSs of Atlantic sturgeon, the GOM DPS of Atlantic salmon, and giant manta rays in the ten fisheries assessed in this Opinion (Table 84). In order to be exempt from the prohibitions of section 9 of the ESA and regulations issued pursuant to section 4(d), NMFS must comply with the following T&Cs, which implement the RPMs. These T&Cs are non-discretionary.

Table 84: RPMs, Terms and Conditions, and justifications

Reasonable and Prudent Measures (RPMs)	**Terms and Conditions (T&Cs)**	
RPM 1: GEAR RESEARCH: • NMFS must continue to work with the fishing industry and partners to promote, fund, conduct, and/or review research on gear modifications to reduce incidental takes, and the severity of interactions that do occur, of ESA-listed species. • Since fishing characteristics and behavior vary between fisheries, NMFS must annually assess research to better characterize the fisheries covered in this Opinion and the nature of their interactions. • NMFS must continue to share gear research results and tools with Canadian partners to assist them in lowering the number and severity of large whale entanglements in their waters.	• NMFS must develop and evaluate gear research priorities and information needs for ESA-listed species included in the ITS annually. • NMFS must develop a "Roadmap to Ropeless Fishing" within one year of the publication of the Opinion. The Roadmap will identify the research and technology needs related to ropeless fishing, including how these needs will be met. The Roadmap will include consideration of economic, safety, operational, and enforcement aspects of ropeless technology. • NMFS must continue to investigate both new and existing modifications to gillnet, bottom trawl, and trap/pot gear and their effects on large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays. • NMFS must continue to support whale scarring research to estimate the number and severity of entanglements. • NMFS will convene a working group to review all the available information on Atlantic sturgeon bycatch in the federal large gillnet (>=7 inches stretched) mesh fisheries. Within **one year** of publication of this Opinion, the working group will develop an action plan to reduce Atlantic sturgeon bycatch in these fisheries by 2024.	RPM 1 and the accompanying Term and Condition specifies the need for continued gear research and evaluation, as well as further investigation and implementation of results to aid in bycatch reduction of large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays observed captured or entangled in gillnet, bottom trawl, and trap/pot fishing gear. This is essential for reducing the level and severity of incidental take associated with the fishing industry while maintaining sustainable fishing practices. Improving knowledge on dynamic fisheries and updating protocols and modifying current practices, when paired with updated information on where interactions are most likely to occur, are essential for the long-term reduction of impacts on large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays.

Reasonable and Prudent Measures (RPMs)	Terms and Conditions (T&Cs)	Justifications for RPMs and T&Cs
RPM 2: ECOLOGICAL STUDIES: • NMFS must continue to review available data to determine whether there are areas or conditions within the action area where large whale, sea turtle, Atlantic sturgeon, Atlantic salmon, and giant manta ray interactions with fishing gears used in the ten fisheries are more likely to occur.	• NMFS must continue to review all data available on the observed/documented take of large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays in these fisheries and other suitable information (e.g., data on observed interactions with other fisheries, species distribution information, or fishery surveys in the area where the fisheries operate) to assess whether there is sufficient information to undertake any additional analysis to attempt to identify correlations with environmental conditions or other drivers of incidental take within some or all of the action area. • If such analysis is deemed appropriate, within a reasonable amount of time after completing the review, NMFS must take appropriate action to reduce large whale, sea turtle, Atlantic sturgeon, Atlantic salmon, and giant manta ray interactions and/or their impacts.	RPM 2 and the accompanying Terms and Condition specify the importance of using current data already available to reduce the incidental bycatch and increase survivability of large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays. Temporal and spatial data can provide insight on where these interactions are most likely to occur, and can be paired with modifications to fishing practices to minimize the respective incidental capture and mortality of large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays.

Reasonable and Prudent Measures (RPMs)	Terms and Conditions (T&Cs)	Justifications for RPMs and T&Cs
<u>RPM 3: HANDLING</u> • NMFS must ensure that any bycaught or entangled large whale, sea turtle, Atlantic sturgeon, Atlantic salmon, and giant manta ray is handled in such a way as to minimize stress to the animal and increase its survival rate. • For sea turtles in a comatose or lethargic state, NMFS requires that they must be retained on board, handled, resuscitated, and released according to the established procedures, as practicable and in consideration of best practices for safe vessel and fishing operations.	• NMFS requires that vessel operators follow the sea turtle handling and resuscitation requirements at 50 CFR 223.206. Operators must bring comatose sea turtles aboard and perform resuscitation according to the regulations. If an observer is present, observer protocols will be followed, including bringing fresh dead animals to shore when feasible. • NMFS must distribute information to permit holders in the ten fisheries specifying handling and/or resuscitation requirements they must undertake for any caught or entangled sea turtles, Atlantic sturgeon, Atlantic salmon, or giant manta rays. For large whales, NMFS must provide permit holders with the contact information for the appropriate disentanglement response networks. • As new information becomes available, NMFS must update its protected species handling and release protocols. • Fishermen within these ten fisheries are authorized through this Opinion to disentangle sea turtles according to the STDN Disentanglement Guidelines. This authorization extends to sea turtles captured in the individual fishermen's gear as well as gear used in the federal fishery for which the vessel holds a permit as long as that fishery is covered in this Opinion. GARFO PRD will provide disentanglement guidelines to vessels. • NMFS must continue to require that disentanglement responders collect detailed, consistent information on the gear involved in entanglements and submit all information on the gear to NMFS.	RPM #3 and the accompanying Terms and Condition describe the importance of specific handling and resuscitation requirements in order to increase survivorship of large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays. Minimizing the stress of an animal that was involved in an incidental take increases its chances of survival post-release. By creating protocols that can be easily accessed by fishermen who may encounter these species in their respective fishery, NMFS and the industry can reduce the severity of the interaction and minimize potential long-term impacts on the animal, such as stress-related complications, and mortality.

Reasonable and Prudent Measures (RPMs)	Terms and Conditions (T&Cs)	Justifications for RPMs and T&Cs
<u>RPM 4: MONITORING 1</u>: • NMFS must ensure that monitoring and reporting of any large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays struck by vessels or encountered in gear used in the ten fisheries: (1) detects any adverse effects such as serious injury or mortality; (2) detects whether the anticipated level of take has occurred or been exceeded; and (3) includes the collection of necessary biological and life history data from individual encounters (e.g., species ID, date, location, size measurements, genetic information, photos/video).	• NMFS must continue to monitor the ten fisheries in order to document and report incidental bycatch and entanglement of large whales, sea turtles, Atlantic sturgeon, Atlantic salmon, and giant manta rays. • NMFS must continue to compile an annual omnibus report of observed large whale, sea turtle, Atlantic sturgeon, Atlantic salmon, and giant manta ray takes in New England and Mid-Atlantic fisheries, including trips where species from these ten FMPs are landed by May 1st each year. • NMFS must continue to produce updated bycatch estimates for sea turtles and Atlantic sturgeon in gillnet and trawl gear within the action area when sufficient information and an adequate sample size of data is available (this has typically been done on a five-year cycle). • Observers must continue to tag and take tissue samples (under their ESA section 10 permit) from incidentally captured sea turtles. The NEFSC will be the clearinghouse for any genetic samples of sea turtles taken by observers. • Observers must also take genetic samples (i.e., fin clips or scales) of all incidentally captured Atlantic sturgeon and Atlantic salmon according to the current observer protocols. Samples will be sent to the appropriate NMFS line office or research partner for analysis. • NMFS must conduct outreach to fishermen on reducing vessel strikes of sea turtles. Fishermen must immediately report any vessel strike to GARFO's Marine Animal Hotline directly or through the USCG. • Within one year, NMFS must specify mandatory harvester reporting requirements (e.g., vessel trip reports) for the American lobster and Jonah crab fisheries in federal waters.	RPM 4 and the accompanying Terms and Condition highlight the importance of monitoring effort in fisheries to predict what impacts these changes could have. With many different factors impacting a fishery, effort can fluctuate annually. NMFS must track vessel strikes and bycatch of protected species in these fisheries in order to identify potential impacts to protected species and make adjustments to management, as necessary.

Reasonable and Prudent Measures (RPMs)	Terms and Conditions (T&Cs)	Justifications for RPMs and T&Cs
RPM 5: MONITORING 2: • NMFS must continue efforts to better investigate and understand cryptic mortality of large whales and post-release mortality of sea turtles in gillnet, bottom trawl, and trap/pot gear used in the ten fisheries and the factors affecting these rates over time. • NMFS must continue efforts to understand the degree to which North Atlantic right whale cryptic mortality and mortality not attributed to source (i.e. vessel strike or entanglement) or country (i.e., United States or Canada) is a result of the operation of interactions in the federal fisheries.	• NMFS must continue to annually evaluate observed takes of large whales and sea turtles using the post-interaction mortality criteria for these species. NMFS has defined criteria for estimating post-interaction mortality in various fisheries using available scientific studies in conjunction with veterinary and other expert opinion, primarily based on animal behavior and the presence and severity of injuries. NMFS will continue apply this criteria to data collected by observers onboard commercial fishing vessels or by personnel specifically trained and permitted to disentangle and release large whales and sea turtles. • The action plan for sturgeon developed under RPM 1 must include an evaluation of information available on post-release mortality, identification of data needed to better assess impacts, and a plan, including timeframes, for obtaining and using this information to evaluate impacts. • NMFS must continue to evaluate all interactions with large whales to determine if gear marking or other gear characteristics can be used to identify the source of the interaction and to determine whether there is new information not considered in this Opinion on apportioning interactions to source. • The Final EIS for the amendments to the ALWTRP will describe, and NMFS will annually report on, monitoring and enforcement of the ALWTRP.	RPM 5 and the accompanying Terms and Condition specify the need for close monitoring of cryptic and post-interaction mortality rates for large whales and sea turtles and what factors affect these rates. Using the post-interaction mortality criteria created by experts in the field, NMFS can use the best available data annually to detect trends in these interactions, survival rates, and if there are modifications that can be made to the fishery, the handling and release of large whales and sea turtles, and disentanglement training to increase chances of survival post-interaction.

Reasonable and Prudent Measures (RPMs)	Terms and Conditions (T&Cs)	Justifications for RPMs and T&Cs
RPM 6: POPULATION ASSESSMENTS • NMFS must continue efforts to develop population evaluation assessment tools.	• NMFS must continue to support the development of the NEIT Population Evaluation Tool and NEFSC loggerhead PVA and other population assessment tools. • NMFS must use the best available tools, as appropriate, to assess the North Atlantic right whale extinction risk, current threats, and progress towards achieving the goals of the Conservation Framework, during the evaluation periods.	RPM 6 and the accompanying Term and Condition specify the need for supporting the development of tools needed to assess and monitor the fisheries impacts on ESA-listed species' populations. Population assessment tools will also improve NMFS' ability to monitor the implementation of the Conservation Framework.

10.3. Monitoring Protocols

10.3.1. *Large Whale Monitoring*

NMFS will continue to monitor levels of large whale entanglement in the ten fisheries. Each year, NMFS will evaluate the most recent annual scarring report published in the New England Aquarium's catalog report for right whales, as well as any other available information for all species, to determine the number of total entanglements that occurred each year following the publication of this Opinion. For right whales, the annual entanglement rate will be specified as the annual percentage of the total population determined to have been entangled each year. We will use a 5-year annual running average to determine if the ITS has been exceeded.

Serious injury determinations and stock assessment reports have been used as the principal means to estimate the large whale entanglements resulting in M/SI in the ten fisheries and to monitor M/SI levels. NMFS has developed a monitoring strategy for the ALWTRP and will produce an annual report stating the most up-to-date M/SI five year rolling average. To provide the most up-to-date rolling average possible, the five-year average will consist of the most recently available year's data from the Marine Animal Incident Database averaged with the previous four years of data obtained from the U.S. Atlantic and Gulf of Mexico Marine Mammal SARs. Analyzing the data in this way will reduce the two-year lag associated with using SAR estimates alone by one year. For the purposes of monitoring large whale takes, NMFS will use the same methodology used in the *Effects of the Proposed Action* to apportion entanglements that are not confirmed to fishery.

For the purposes of monitoring large whale entanglements and M/SI, NMFS will use the scarring reports, serious injury determination reports, SARs, and the ALWTRP monitoring reports to collect entanglement information. NMFS will re-examine interactions and M/SI annually in the ten fisheries. Using these data, NMFS will determine whether incidental take has been exceeded. In addition, as described in the Framework, we will evaluate the requirements of the Framework and information on the population status and environmental baseline. (i.e., changes to calving rates, risk reductions in Canada, risk reductions in U.S. state fisheries, or vessel-strike reductions in U.S. waters) in 2025-2026.

10.3.2. *Sea Turtle Monitoring*

NMFS must continue to monitor levels of sea turtle bycatch in the ten fisheries. Fisheries observer data, and their incorporation into statistical models (specifically, ratio estimator models as described in Murray (2018, 2020)) are being used as the principal means to estimate sea turtle bycatch rates in the ten fisheries and to monitor incidental take levels. At present, and due to reasons explained below, the NEFSC produces statistically robust sea turtle bycatch estimates for gillnet and bottom trawl gear on five-year rotational cycles. During those individual cycles, observer data by gear type is analyzed over 1-2 years and monitored over the following 3-4 years. NMFS must continue to use fisheries observer data and the NEFSC-produced bycatch estimates to monitor sea turtle bycatch in gillnet and bottom trawl gear that is authorized by the ten FMPs, though the role of observers and use of fishery dependent data will differ for each gear type. Entanglement reports have been used as the principal means to estimate sea turtle bycatch in the trap/pot fisheries and to monitor incidental take levels. NMFS must continue to use entanglement reports as well as available observer data to monitor sea turtle bycatch in trap/pot gear authorized by the FMPs.

Gillnet and bottom trawl gear

For the purposes of monitoring this ITS for the gillnet and bottom trawl components of the ten fisheries, we will continue to use records from the fisheries observer program as the primary means of collecting incidental take information. For sea turtles, the take estimates described in this Opinion were generated using a statistical model that is not feasible to conduct on an annual basis due to the data needs; length of time to develop, review, and finalize the estimates; and methodology, as explained below. In monitoring take, NMFS will evaluate the observer data and bycatch estimates.

In its discussion of sea turtle bycatch estimation, Murray (2009b) explains that "to directly compare future levels of loggerhead bycatch to the average annual estimates and [95 percent] confidence intervals [CIs] reported in this paper, these future estimates would also need to be 5-year averages." This necessity is reiterated in the Warden (2011b) trawl bycatch analysis for loggerhead sea turtles, which states that "if these interaction estimates are updated approximately every five years, then future levels of loggerhead interactions can be evaluated by comparing the average annual estimates and CIs reported in this paper to the future average annual estimates and CIs." Therefore, for the following reasons, we will continue to implement a five-year monitoring framework for sea turtles rather than an annual one:

- As mentioned throughout the Opinion, observed sea turtle interactions are rare, and we often need to pool data across years to have enough data to produce a robust, model-based estimate of total interactions. We need at least ten observations per parameter in the model. Thus, even with a very simple model, we usually require 20-30 observed bycatch events. It is uncommon to have this many observed sea turtle interactions in a single year, as documented in previous bycatch estimates. Subsequently, when we pool data over five years to report an annual average, we need another five years to compare averages, as explained above.
- It normally takes a year to process, clean, and analyze data for a valid bycatch estimate, for one gear type. With current resources, it is neither reasonable nor possible to estimate bycatch annually across multiple gear types.
- Annual estimates are unlikely to change considerably such that they affect the population assessments. On page 35 of Warden et al. (2015), the authors state that "when the population is large compared to the incidental mortality, frequent (e.g., annual) monitoring is not likely to produce results that are substantially different from the previous assessment. Less frequent but more comprehensive assessments, which explicitly address uncertainty, may provide more reliable information."

Although we collect raw data on the number of observed sea turtle takes in gillnet and bottom trawl fisheries as they are documented and verified (usually on a time lag of at least three months per the NEFOP's data quality control and assurance procedures), we cannot produce reliable short-term take estimates using them because observed sea turtle takes are rare events, dependent on a wide range of both human and natural factors that vary greatly over short time periods (i.e., less than a year). Examples of human factors include variation in the number of vessels fishing, time spent fishing, percent observer coverage, regulatory regimes, market forces, etc. Natural factors include changes in oceanographic conditions such as water temperature, distribution of prey, weather conditions, shifting distributions and abundance of sea turtles, etc. Typically, the number of takes observed in a short time period (i.e., one year), when considered with the factors identified above, means that the observed takes cannot be extrapolated to estimate the total number of takes with good precision. Nor do the raw data provide a large enough sample size to identify any exceedances of the incidental take level. For all of the foregoing reasons, we will rely on the statistical methods used in Murray (2018) and Murray (2020), which we have determined represent the best available scientific and commercial data for sea turtle bycatch estimation, to re-estimate takes in the gillnet and bottom trawl fisheries assessed in this Opinion approximately every five years.

With respect to green sea turtles in gillnet gear, we do not have a recent five-year bycatch estimate due to so few recorded interactions. Thus, the raw annual numbers of observed takes are the best available scientific and commercial data, and reviewing those numbers is the only available method for monitoring the incidental take levels in gillnet gear. Thus, we will continue to rely on such data for monitoring incidental takes of that species until a five-year estimate is available. Given the annual variability in take, the take levels are set as 5-year running averages (total for any 5-year period) and not for static 5-year periods (i.e., 2021-2025, 2022-2026, 2023-2027 and so on, as opposed to 2021-2025, 2026-2030, 2031-2035, etc.). This approach will allow us to reduce the likelihood of requiring reinitiation unnecessarily because of inherent variability in incidental take levels, but still allow for an accurate assessment of how the proposed action is performing versus our expectations. The 5-year running average will be calculated annually.

This two-pronged methodology for monitoring sea turtle incidental takes in gillnet and bottom trawl gear is consistent with the conceptual framework described in Haas (2010), in which a low level metric such as raw counts (simple to estimate, but less informative) could be used for monitoring incidental take on the short term (i.e., annually) and a higher level metric such as a bycatch estimate (difficult to estimate, more informative) could be used for monitoring incidental take over a longer (i.e., five year) time frame. For all four species of sea turtles, no other monitoring alternatives exist for gillnet and bottom trawl fisheries that are feasible on a shorter term than the 5-year period required to produce an updated bycatch estimate.

Pot/trap gear

For the purposes of monitoring the ITS in regards to sea turtles that are known to be entangled in trap/pot gear, NMFS will continue to use STDN data as the primary means of collecting incidental take information. NMFS will assess takes annually in the lobster, Jonah crab, red crab, scup, and black sea bass fisheries using all available and up-to-date STDN entanglement data. NMFS will use the same methodology used in the *Effects of the Proposed Action* to apportion entanglements that are not confirmed to fishery. Using these data, NMFS will determine if the five-year rolling average incidental take level in this Opinion has been met or exceeded.

10.3.3. ***Atlantic Sturgeon Monitoring***

NMFS must monitor levels of Atlantic sturgeon bycatch in the ten fisheries. Fisheries observer data, and their incorporation into statistical models (specifically, generalized linear models (ASMFC 2017, Miller and Shepard 2011)), has been used as the principal means to estimate Atlantic sturgeon bycatch in gillnet and bottom trawl fisheries and will be used to monitor incidental take levels in gear authorized by the FMPs for the ten fisheries.

For the purposes of monitoring this ITS for the gillnet and bottom trawl components of the ten fisheries, we will continue to use fisheries observer data as the primary means of collecting incidental take information. As the estimates depend on incidental take rate information over a several year period, re-examination after one year is not likely to produce any noticeable change in the take rate. For these reasons, we will re-estimate incidental takes in the ten fisheries every five years using appropriate statistical methods. For the five Atlantic sturgeon DPSs, we will use all available information (e.g., observed incidental takes, changes in fishing effort, etc.) to monitor the fishery.

10.3.4. *Atlantic Salmon and Giant Manta Ray Monitoring*

NMFS must monitor levels of Atlantic salmon and giant manta ray bycatch in the ten fisheries. Observer coverage has been used as the principal means to estimate Atlantic salmon and giant manta ray bycatch in U.S. Northeast and Mid-Atlantic gillnet and bottom trawl fisheries and will be used to monitor incidental take levels in gear that is authorized by the FMPs for the ten fisheries. Given the annual variability in take, the take levels are set as 5-year running average (total for any 5-year period) and not for static 5-year periods (i.e., 2021-2025, 2022-2026, 2023-2027 and so on, as opposed to 2021-2025, 2026-2030, 2031-2035, etc.). This approach will allow us to reduce the likelihood of requiring reinitiation unnecessarily because of inherent variability in incidental take levels, but still allow for an accurate assessment of how the proposed action is performing versus our expectations. The 5-year running average will be calculated annually.

For the purposes of monitoring this ITS for the gillnet and bottom trawl components of the ten fisheries, we will continue to use observer coverage as the primary means of collecting incidental take information. For the Atlantic salmon and giant manta rays, we will use all available information (e.g., observed takes, changes in fishing effort, etc.) to assess if the annual incidental take level in this Opinion has been exceeded.

10.4. Conservation Recommendations

In addition to section 7(a)(2), which requires agencies to ensure that proposed actions are not likely jeopardize the continued existence of listed species, section 7(a)(1) of the ESA places a responsibility on all federal agencies to utilize their authorities in furtherance of the purposes of the ESA by carrying out programs for the conservation of endangered and threatened species. Conservation Recommendations are discretionary activities designed to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. The following additional measures are recommended regarding incidental take and ESA-listed species conservation:

1. NMFS should continue to follow its established protocols for bringing to shore any sea turtle incidentally taken in fishing gears used in the ten fisheries that is freshly dead, that dies on the vessel shortly after the gear is retrieved, or dies following attempts at resuscitation in accordance with the regulations. The protocols include steps to be taken to ensure that the carcass can be safely and properly stored on the vessel and properly transferred to appropriate personnel for examination. The protocols also identify the purpose for examining the carcass and the samples to be collected. Port samplers and observers should also be trained in the protocols for notification of the appropriate personnel in the event that a vessel comes into port with a sea turtle carcass.
2. NMFS should develop guidance for fishing practices that minimize bycatch of giant manta rays, including handling and release procedures using different gears, and produce education and outreach materials about safe handling and release.

3. NMFS should develop standardized guidelines for fisheries data collection (e.g., species identification, sizing, tissue samples, and reproductive status) and monitoring (e.g., landings, discards, fishing effort, and gear types) for giant manta rays. NMFS should collect data or fund research to estimate post-release mortality across various sizes and gear types.
4. NMFS should also review its policies/protocols for the processing of genetic samples to determine what can be done to improve the efficiency and speed for obtaining results of genetic samples taken from all incidentally taken Atlantic sturgeon and Atlantic salmon.
5. NMFS should work with states to minimize take and its impacts in state permitted activities and encourage the states to seek authorization for incidental take that is otherwise unavoidable.
6. NMFS should support studies and stock assessments on seasonal ESA-listed species distribution and abundance in the action area, behavioral studies to improve our understanding of ESA-listed species interactions with fishing gear, and foraging studies including prey abundance/distribution studies (which may influence distribution), as well as studies and analysis necessary to develop population estimates for ESA-listed species.
7. NMFS should continue to undertake and support vessel and aerial surveys, passive acoustic monitoring, and the Sightings Advisory System to better understand the distribution and habitat use of ESA-listed species to inform assessments of co-occurrence.
8. NMFS will continue to conduct outreach to fishermen on reducing risks to protected species due to incidental bycatch and vessel strikes. NMFS should continue to develop and implement measures to reduce the risk of vessel strikes of large whales. NMFS has conducted a review of the vessel strike reduction measures, including an assessment of the effectiveness of mandatory vessel speed restrictions, as it pertains to right whale management. NMFS is currently evaluating the need for future action or potential modifications to the vessel strike reduction efforts to enhance protection of right whales.
9. NMFS should encourage all commercial and recreational fishermen to report sightings of large whales, especially right whales, to the following hotline numbers: **866-755-6622** (from Maine to Virginia) and **877-WHALE-HELP** (from North Carolina to Florida).
10. NMFS should increase its monitoring and surveillance of North Atlantic right whales to identify areas of predicted co-occurrence between whales and fishing gear and develop methods to validate the forecasts of co-occurrence.
11. NMFS should continue to undertake and support disentanglement activities, in coordination with the states, other members of the disentanglement and stranding network, and with Canada.
12. NMFS should continue to create education and outreach material to communicate conservation messages for ESA-listed species, including new materials for giant manta rays, through social media, websites, magazines, and print to federal agencies, local communities, and non-governmental organizations (NGOs).

13. NMFS should explore the methods and feasibility for authorizing fishermen to tag incidentally-captured Atlantic sturgeon with a Passive Integrated Transponder (PIT) tag.
14. NMFS should explore how to provide PIT tag readers to fishermen when and where their fishing efforts overlap with expected sturgeon aggregations areas.

11. REINITIATING CONSULTATION

This concludes formal consultation on the a of the fisheries operating under the eight federal (Atlantic Bluefish, Atlantic Deep-Sea Red Crab, Mackerel/Squid/Butterfish, Monkfish, Northeast Multispecies, Northeast Skate Complex, Spiny Dogfish, and Summer Flounder/Scup/Black Sea Bass) and two interstate fishery management plans (American Lobster and Jonah Crab) and on the Implementation of the New England Fisheries Management Council's Omnibus Essential Fish Habitat Amendment 2. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In the event that the amount or extent of take is exceeded, NMFS, GARFO must immediately request reinitiation of formal consultation.

This Biological Opinion considers Phase 1 of the Conservation Framework to be the measures that were proposed in the Proposed Rule of December 31, 2020 to amend the Atlantic Large Whale Take Reduction Plan. Because this Biological Opinion is being issued prior to the publication of the final rule, it is possible that the measures we implement in that final rule may differ somewhat from those specified in the proposed rule based on our consideration of comments received on the proposed rule and additional analyses conducted during development of the rule. Any substantive deviations from the proposed rule not previously considered in the Biological Opinion may trigger re-initiation of this consultation, but any measures that differ from the proposed rule that are determined to provide equal or greater conservation-value as compared to the measures in the proposed rule, would not trigger reinitiation.

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13. APPENDIX A: NORTH ATLANTIC RIGHT WHALE CONSERVATION FRAMEWORK FOR FEDERAL FISHERIES IN THE GREATER ATLANTIC REGION

North Atlantic Right Whale Conservation Framework for Federal Fisheries in the Greater Atlantic Region

Purpose

During the development of an Endangered Species Act (ESA) section 7 consultation on the authorization of federal fisheries in the Greater Atlantic Region, we identified the need to implement measures to further reduce entanglement of North Atlantic right whales (hereafter "right whales") to meet the mandates of the ESA. As described below, the Conservation Framework includes the measures proposed in a December 2020 rulemaking to modify the Atlantic Large Whale Take Reduction Plan (ALWTRP) and three additional phases. This Conservation Framework outlines NMFS' commitment to implement measures that are necessary for the recovery of right whales, while providing a phased approach and some flexibility to the fishing industry. NMFS is currently conducting an ESA section 7 consultation on the authorization of eight federal fisheries managed under the Magnuson-Stevens Act and two interstate fishery management plans under the Atlantic Coastal Act and the implementation of the New England Fisheries Management Council's Omnibus Essential Fish Habitat Amendment 2 (Batched Fisheries Opinion). The Batched Fisheries Opinion includes fisheries managed under the American lobster, Atlantic bluefish, Atlantic deep-sea red crab, Jonah crab, monkfish, Northeast multispecies, Northeast skate complex, spiny dogfish, Atlantic squid/mackerel/butterfish, and summer flounder/scup/black sea bass fishery management plans. It is our intent that these measures be considered as part of the proposed action in the consultation on the Batched Fisheries. The Conservation Framework includes fixed gear fisheries authorized under the respective fishery management plans included in the Batched Fisheries Opinion. The Conservation Framework does not specify particular measures but identifies the level of reductions in mortalities and serious injuries (M/SI) that NMFS is committed to achieve in order to meet its ESA mandates. Although we believe the Conservation Framework targets can be met through gear and operational measures, NMFS has the authority[72] to implement other measures (e.g., partial/complete closures) to reduce risk and will exercise that authority if needed.

Background

North Atlantic right whales, one of the world's most endangered large whale species, are protected under the Marine Mammal Protection Act (MMPA) and the ESA. While these two laws have different objectives, they work together to protect and recover North Atlantic right whales, restoring stocks to sustainable levels. NMFS has developed this North Atlantic Right Whale Conservation Framework to further reduce M/SI due to entanglements in federal fisheries to meet the mandates of the ESA, while recognizing the important role that the MMPA take reduction program goals and ongoing actions have in reducing mortalities and serious injuries in U.S. commercial fisheries and recovering the North Atlantic right whale species.

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[72] Marine Mammal Protection Act (16 U.S.C. 1361 et seq.), the Endangered Species Act (16 U.S.C. 1531 et seq.), the Magnuson-Stevens Fishery Conservation and Management Act (16 U.S.C. 1801 et seq.) and other statutes, as appropriate.

Primary threats to the species include climate change, entanglement in fishing gear, and vessel strikes. The fisheries included in this Conservation Framework are fixed gear fisheries in federal waters managed by NMFS' Greater Atlantic Regional Fisheries Office (GARFO) under the Magnuson-Stevens Fishery Conservation and Management Act and the Atlantic Coastal Fisheries Cooperative Management Act. Some of these fisheries are also regulated under the ALWTRP regulations issued pursuant to section 118 of the MMPA. Under the ESA, the impacts of federally-authorized activities are considered under the consultation requirements of section 7 of the ESA. In developing the ESA section 7 Biological Opinion on these fisheries, entanglements in the federal fisheries listed above were estimated to seriously injure or result in the death of an average of approximately five right whales each year[73]. This Conservation Framework outlines ongoing and planned actions to reduce M/SI of right whales incidental to these fisheries under the MMPA and the further reductions needed to meet the mandates of the ESA.

MMPA: Atlantic Large Whale Take Reduction Program Ongoing and Planned Activities

When incidental mortality or serious injury of marine mammals from commercial fishing exceeds a stock's "potential biological removal" (PBR) level, the MMPA directs NMFS to convene a take reduction team made up of stakeholders from the fishing industry, fishery management councils and commissions, state and federal resource management agencies, the scientific community, and conservation organizations to consider the best available information and develop recommended modifications to commercial fishery operations to reduce M/SI to below a stock's PBR. NMFS considers these recommendations in implementing regulatory and non-regulatory measures under a take reduction plan. The MMPA specifies that the goal of a take reduction plan shall be to reduce M/SI incidental to commercial fishing to below a stock's PBR. First implemented in 1997, the ALWTRP has been modified several times to reduce the risk of mortality and serious injury of large whales incidentally taken in commercial gillnet and trap/pot fisheries. The most recent final rule was published in May 2015 (80 FR 30367, May 28, 2015).

Because of the declining population and the persistent incidental entanglements resulting in M/SI above the stock's PBR, ALWTRP modifications have, and continue to be, directed primarily at reducing the risk of commercial fisheries on right whales. In late 2017, the evidence of a declining population exacerbated by high right whale mortalities caused NMFS to refocus the Atlantic Large Whale Take Reduction Team (ALWTRT) on new modifications to the ALWTRP. NMFS has proposed measures to reduce M/SI (85 FR 86878, December 31, 2020) and is planning to reconvene the ALWTRT to consider additional measures.

In the current proposed rule, NMFS has proposed modifications to the ALWTRP that focus on the Northeast Region lobster and Jonah crab trap/pot fisheries. In developing this action, NMFS estimated that to reduce M/SI to below PBR for right whales, entanglement risk across U.S.

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[73] For information on how these estimates, which include an estimate of observed/unknown cause and unobserved (i.e., cryptic) mortality resulting from entanglement in the fisheries, were calculated, see section 7.2 of the Biological Opinion on the Continued Implementation of Management Measures for the American Lobster, Atlantic Bluefish, Atlantic Deep-Sea Red Crab, Mackerel/Squid/Butterfish, Monkfish, Northeast Multispecies, Northeast Skate Complex, Spiny Dogfish, Summer Flounder/Scup/Black Sea Bass, and Jonah Crab Fisheries.

fisheries (state and federal) needs to be reduced by 60 to 80 percent. However, given additional sources of uncertainty in the 80-percent target, as well as the challenges achieving such a target without substantial economic impacts to the fishery, the ALWTRT focused on recommendations to achieve the lower 60-percent target. Therefore, under the ALWTRP, NMFS has proposed measures to reduce M/SI in the American lobster and Jonah crab pot/trap fisheries in both state and federal waters[74] by an estimated 60 percent. For a full description of how these targets were determined, the ALWTRT discussions, and the proposed measures, see the Draft Environmental Impact Statement for Amending the Atlantic Large Whale Take Reduction Plan: Risk Reduction Rule. The measures proposed under the ALWTRT are included as the first phase in this Conservation Framework.

In 2021, the ALWTRT will be asked to recommend modifications to the ALWTRP to address risk in the remaining fixed gear fisheries that use buoy lines, including other trap/pot fisheries and gillnet fisheries coastwide. The ALWTRT has begun discussing risk reduction considerations for late summer scoping. NMFS will consider how any changes to the ALWTRP under this future action contribute to meeting the goals of the Conservation Framework.

As described above, the MMPA and ESA work together to protect and recover right whales. While recommendations from the ALWTRT inform the development of measures integrated into the ALWTRP and associated regulations to meet the mandates of the MMPA, they also contribute to progress towards the ESA goals described below.

*ESA: Section 7 Consultation of Federal Fisheries Managemen*t

Under the ESA, the consultation considers the impacts of the federal fisheries (i.e., federally-permitted vessels operating in federal waters) on ESA-listed species. The implementation of the proposed modifications to the ALWTRP related to the Northeast Region lobster and Jonah crab trap/pot fisheries in federal waters is expected to reduce M/SI in the American lobster and Jonah crab pot/trap fisheries by approximately 60 percent[75]. Once the ALWTRP measures are implemented, NMFS estimates that, without further action, the federal fisheries are anticipated to result in the death of approximately an annual average of 2.69 right whales (27 right whales over a 10-year period).

We recognize that the fishing industry has implemented all the required mitigation measures since 1997. However, data suggest that mortalities and serious injuries of right whales continued at higher rates than are sustainable even with the measures implemented under the Take Reduction Plan. As a result of climate change and exposure to mortality in unregulated areas, the persistent deaths and injuries in U.S. fisheries cannot be sustained by the reduced North Atlantic right whale population. As the population of right whales continues to decline, we must

[74] The area include in the ALWTRP proposed rule is north of 40°00’ N latitude and east of 71° 51.5’ W longitude.

[75] It should be noted that the ALWTRP rulemaking includes both state and federal waters. The proposed measures across the state and federal waters are designed to achieve at least a 60 percent reduction in M/SI. For the purposes of this analysis, we do not apply a “credit” for measures that were previously implemented in the Massachusetts Restricted Area. Without this, the proposed rule achieves 58.1% reduction; 26.6% of that reduction is expected to occur in federal waters. Reduced impacts in state waters would contribute to an improved baseline. The impacts to the baseline from these measures are considered in the Biological Opinion.

acknowledge that previous efforts have not reduced entanglements to the degree needed to satisfy ESA and MMPA requirements, and additional efforts are necessary to recover this critically endangered species.

Our analyses indicate that further reductions in entanglements and M/SI in the federal fisheries under this Conservation Framework are needed to ensure the fisheries will not appreciably reduce the likelihood of the survival and recovery of the species as required by the ESA. To determine the extent to which additional measures are needed, we used qualitative and quantitative analyses. We developed a population projection model to predict the population trajectory over 50 years (Linden, 2021)[76]. We recognize that the fisheries are likely to be modified in the next 10 years; however, there is no information available at this time to predict how any future modifications will change the operation of the fisheries. While these changes cannot be considered in the projections now, we have developed and are committing to the comprehensive adaptive management approach and schedule described below so that as new information becomes available, the changes can be considered in the future. The adaptive management approach will also consider changes to calving rates and reductions of M/SI from other non-fishery sources (e.g., vessel strikes).

Using the population projections, we compared the trajectory of the female population after implementing the proposed ALWTRP measures to the trajectory projected if the remaining M/SI in the federal fisheries was further reduced by 25, 50, 75, or 95 percent. With no further reduction in M/SI, our analyses indicate the federal fisheries are impacting the survival and recovery of right whales. We also concluded that reductions below 95 percent were insufficient to meet the ESA mandates as survival and recovery would still be appreciably reduced due to the federal fisheries that would continue to occur, albeit at a lower level. We further refined our analysis and determined that M/SI in the federal fisheries needs to be reduced to 0.136 on average annually[77], within 10 years under a phased implementation (see below), to ensure that the fisheries will not appreciably reduce the likelihood of survival and recovery of the species. Unless M/SI from other sources (i.e., U.S. vessel strikes, non-federal U.S. fisheries, Canadian fisheries and vessel strikes (see below)) are reduced and/or calving rates increase, this level of reduction in M/SI in the federal fisheries is necessary to ensure the goals of the ESA, namely survival and recovery of the species, are met.

Therefore, through this Conservation Framework, we are committing to use our authorities to implement measures to further reduce entanglements and M/SI in federal fisheries, reducing M/SI from an annual average of 2.69 after the implementation of the proposed rule to no more than 0.136. The reduction in entanglements is also expected to reduce sublethal effects that may affect the health and reproductive output of right whales. These reductions will be phased in over the 10-year period (2021-2030). The Conservation Framework describes the targets to be achieved and the dates by which they must be implemented to ensure the Framework's goals are

[76] Linden, D. 2021. Population projections of North Atlantic right whales under varying human-caused mortality risk and future uncertainty National Marine Fisheries Service, Greater Atlantic Region, Gloucester, MA.

[77] Note that the numbers included here differ slightly from the numbers included in the draft Conservation Framework. This is a result of updates to the data considered and new runs of the population projections using the updated information.

achieved. At this time, the Conservation Framework does not specify the measures that will be implemented. When developing measures at each phase, we will be able to consider gear innovations, ALWTRT recommendations, fishing and shipping changes, and evidence of impacts of U.S. and Canadian right whale conservation.

M/SI in Canadian waters

In the Biological Opinion, we estimate the total M/SI of right whales across their full range. The estimated mortality was then partitioned between the United States and Canada following the methods used for M/SI in fisheries to develop the ALWTRT target. These methods were peer reviewed, and while the reviewers did not come to consensus on accuracy, they considered the approach reasonable. We estimate that, on average, approximately 21 right whales die or are seriously injured annually under current conditions. Of these, 11.10 are estimated to occur in Canadian waters and 10.02 in U.S. waters. In Canada, right whales are protected under the Species at Risk Act and the Fisheries Act. As described in the Opinion, the population projections demonstrate that action is needed in both countries to turn the population trajectory positive. Since 2017, the Government of Canada has implemented measures to protect North Atlantic right whales from impacts from both the fishing and shipping industries. Canada has modified their measures annually to reduce M/SI. Given the limited time these measures have been in effect as well as annual changes to and the dynamic nature of the measures, at this time, we have no way to accurately assess the benefit to right whales from Canada's recent measures. As such, in our current analysis, we are not able to quantify the level of risk reduction in Canada and include it in the analysis. However, we assert that the measures taken by Canada are and will continue to benefit right whales, and as part of our evaluation of new data and measures (see table below), we will periodically consider whether it is possible to attribute a benefit from Canadian measures in our analysis. Until this benefit can be assessed, this Conservation Framework takes a conservative approach that considers the retrospective recent serious injury and mortality rates and plans as if the Canadian measures are not benefitting the right whale population. As more information becomes available on risk reduction in Canadian waters and from other U.S. sources (e.g., vessel strikes), the Conservation Framework may be modified to reduce the degree to which additional measures are needed while ensuring that the fisheries in the Framework are not appreciably reducing survival and recovery of the species.

Adaptive Management within the Conservation Framework

This Conservation Framework is designed to increase the likelihood of not only survival but also successful recovery of right whales, as required by the ESA. To accomplish this, the Conservation Framework recognizes and addresses many sources of uncertainty. Conservative assumptions are made about future conditions, including environmental conditions, threats, and the species' response to management actions in the United States and Canada. We recognize that there are efforts to reduce M/SI from other sources, uncertainty associated with available data, and changing environmental conditions. To maintain the maximum likelihood of recovery success over time, this Conservation Framework is adaptive and allows for revisions as additional information becomes available or should any of the assumptions require revisions. Adaptive management, that is, adjusting management as management results, needs, and other events become better understood, provides a systematic means of addressing uncertainties and is an important component of this Conservation Framework. A primary tenet of adaptive management is to evaluate the efficacy of management actions. Therefore, the Conservation

Framework includes a comprehensive evaluation mid-way through implementation to determine whether the target reductions in M/SI currently specified for the final five years of the Framework need to be fully implemented. During the evaluation period, we will assess the U.S. and Canadian risk reduction measures, the population status, and calving and survival rates to determine the extent to which additional measures are needed. The Conservation Framework currently assumes no changes to the species' status or reductions in M/SI from other sources, with the exception of actions in state waters from the proposed ALWTRP rule or related state measures. If reductions in M/SI from sources other than the federal fisheries or improvements to the species status are identified during the evaluation, we will revisit this assumption to determine whether it is necessary for all elements of the Conservation Framework to be fully implemented to achieve its conservation goals.

Conservation Framework Actions

The Conservation Framework actions include the current ALWTRP rulemaking and anticipates three additional rulemakings over the next ten years. We will conduct evaluations at defined periods and adapt the Conservation Framework as appropriate. At year five, we will comprehensively evaluate whether and to what extent the fourth and final rulemaking needs to be implemented.

Phase	Year	Conservation Framework Action Description
	Annually	Provide updates, as appropriate, on the implementation of the Framework to the New England and Mid-Atlantic Fishery Management Councils, Atlantic States Marine Fisheries Commission, and ALWTRT.
1	2021	NMFS implements the MMPA ALWTRP rulemaking focused on 60% reduction in right whale M/SI incidental to the American lobster and Jonah crab trap/pot fisheries. In federal waters, this action reduces M/SI from entanglement, on average annually, to 2.69. Implementation for certain measures will begin in 2021; others will be phased over time.
2	2023	NMFS implements rulemaking to reduce M/SI in federal gillnet and other pot/trap (i.e., other than lobster and Jonah crab fisheries included in Phase 1) fisheries by 60%, reducing M/SI from entanglement, on average annually, to 2.61. As described above, the ALWTRT will convene in 2021 to recommend modifications to the ALWTRP to address risk in the remaining fixed gear fisheries. This phase will consider how any changes to the ALWTRP contribute to achieving the target reduction under this Framework.
Evaluation	2023-2024	NMFS evaluates any updated or new data on right whale population and threats to assess progress towards achieving the conservation goals of this Framework. At this time, we will also assess measures taken by Canada to address serious injury and mortality in Canadian waters.

Phase	Year	Conservation Framework Action Description
3	2025	NMFS implements rulemaking to further reduce M/SI by 60% in all federal fixed gear fisheries, reducing M/SI from entanglement, on average annually, to 1.04.
Evaluation	2025-2026	NMFS evaluates measures implemented in the 2025 action as well as new data on the right whale population and threats to assess progress towards achieving the conservation goals of this Framework. Based on the results of this evaluation, NMFS will determine the degree to which additional measures are needed to ensure the fisheries are not appreciably reducing the likelihood of survival and recovery. As described above, if actions outside the federal fisheries reduce risk to right whales by 0.5 M/SI on average annually (1 whale every two years), the M/SI reduction requirement in Phase 4 will be reduced from 87 to 39 percent. If M/SI from other sources is reduced by greater than one M/SI on average annually, we will evaluate whether further action in the federal fisheries is needed.
4	2030	In accordance with the goals identified in the 2025-2026 evaluation, NMFS implements regulations to further reduce M/SI (up to 87%) in fixed gear fisheries. With an 87% reduction, M/SI will be reduced to 0.136.

Evaluation of Reductions by 2030 Needed to Achieve Conservation Framework Goals

NMFS will evaluate population metrics and threats including, but not limited to:

1. Population status.
2. Population distribution and habitat use.
3. Calving and survival rates.
4. Entanglements in U.S. state, U.S. federal, and Canadian commercial fisheries.
5. Changes to the federal fisheries (e.g., changes in co-occurrence due to shifts in areas the fishery operates or changes in effort).
6. Vessel strikes in U.S. and Canadian waters.
7. Apportionment of M/SI (including cryptic mortality) to federal fisheries and other sources, including M/SI in Canada, and between vessel strikes and entanglement.

In 2025-2026, we will re-run the population projections to assess the female population trajectory given any new information. These population projections will help inform the level of further reductions in M/SI that will be needed to achieve the conservation goals of the Conservation Framework and to ensure the federal fisheries are not appreciably reducing the likelihood of survival and recovery. According to the current analysis, a reduction in M/SI in U.S. commercial fisheries of up to 87 percent would be required. That M/SI reduction may be reduced from the 87 percent target to a target of 39 percent if an action outside the federal fisheries reduces risk to right whales by 0.5 M/SI on average annually (1 whale every two years).

It is possible that population-wide risk reduction measures or population growth will reach a level at which further action in the federal fisheries is not needed. If M/SI from other sources is reduced by greater than one M/SI on average annually, we will evaluate whether further action in the federal fisheries is needed and, if so, at what level.[78]

Development of Measures - Engaging and Coordinating With Partners

As described above, this Conservation Framework specifies targets rather than particular measures to be implemented. We are committed to working with our partners on the implementation of measures to meet the goals of the Conservation Framework. Examples of potential conservation measures may include, but are not limited to, measures such as further buoy line reduction by increasing traps per trawl, further weakening of vertical buoy lines, converting to bottom-stowed vertical lines with remote retrieval devices (referred to as 'ropeless' fishing), targeted seasonal restricted areas closed to buoy lines, broad buoy line restrictions, and managing the number of vertical lines through a buoy line allocation program.

NMFS will consider input from the New England and Mid-Atlantic Fishery Management Councils and the Atlantic States Marine Fisheries Commission in developing and implementing mitigation measures under this Conservation Framework. We anticipate that the ALWTRT will be convened at least annually to evaluate incidental entanglement mortality and serious injury, right whale population status, gear monitoring, gear research, and compliance, as required by the MMPA. Any ALWTRT recommendations and associated MMPA rulemaking will be considered. Additionally, at the ALWTRT meetings, as appropriate, we will provide updates on the implementation of the Conservation Framework. Team members' individual input received during these updates, along with other new information, will be considered when developing mitigation measures to meet the objectives of the Conservation Framework. We are committed to implementing this Conservation Framework to further reduce M/SI in the federal fisheries to meet the mandates of the ESA and plan to work closely with our partners throughout the process. We will consider all input received by stakeholders in developing and implementing measures to reach the conservation targets by the dates specified in this Conservation Framework.

Other Sources of Mortality

While this Conservation Framework is specific to the federal fisheries in the Greater Atlantic Region, NMFS and our partners are also working to address other sources of M/SI in the United States and in Canada, as described below.

U.S. Commercial Fisheries in State Waters

We continue to work with states and the ALWTRT to reduce M/SI of large whales, not just right whales, incidentally captured in both state and federal fisheries. The current ALWTRP rulemaking includes measures that apply in state (as well as federal) lobster and crab trap/pot fisheries. Additionally, to obtain authorization for incidentally taking ESA-listed marine mammals, such as right whales, states must apply for an ESA section 10(a)(1)(B) incidental take

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[78]We understand that any changes to the Framework may require reinitiation of the Batched Fisheries Opinion under section 7 of the ESA.

permit for state fisheries. Those applications must include conservation plans that specify the anticipated impact of the state fisheries on the species and its habitat, measures to monitor, minimize, and mitigate such impacts, as well as other information. The Commonwealth of Massachusetts has indicated they are preparing an application, and other New England states have expressed interest or reached out for information on this process. Regardless of whether states apply for ESA section 10 incidental take permits, as noted previously, the ALWTRT will continue its work to identify take reduction measures for state fisheries as part of the MMPA take reduction process. In addition to the efforts of NMFS and the ALWTRT, states such as the Commonwealth of Massachusetts have enacted their own measures (e.g., time/area closures) that expand upon the ALWTRP measures.

U.S. Vessel Strikes

NMFS has implemented a number of measures to reduce the risk of vessel strikes to right whales in U.S. Atlantic waters. These include mandatory speed restrictions for most vessels greater than 65 feet in length transiting through designated Seasonal Management Areas, vessel routing measures to reduce the co-occurrence of vessels and whales, and the establishment of Dynamic Management Areas and Right Whale Slow Zones where vessels are requested to either slow down or avoid areas where aggregations of right whales have recently been detected. Additionally, NMFS maintains a longstanding 500-yard minimum approach distance for right whales to prevent accidental strikes and regularly reaches out to mariners through the USCG Mandatory Ship Reporting system, port meetings, and other avenues to educate vessel operators about speed restrictions and alert them to the presence of right whale aggregations.

However, vessel strike remains a threat to right whales in U.S. (and Canadian, see below) waters. In early 2021, NMFS released an assessment of the vessel strike reduction measures, including an assessment of the effectiveness of mandatory vessel speed restrictions, as it pertains to right whale management. NMFS collected comments on that assessment and, considering these public comments, is currently evaluating the need for future action or potential modifications to the vessel strike reduction efforts to enhance protection of right whales.

In addition to NMFS' efforts, the Commonwealth of Massachusetts has enacted its own measures in the Cape Cod Bay area, mandating a 10-knot speed limit for most vessels less than 65 feet in length during March and April when right whales commonly aggregate in the Bay.

Entanglements and Vessel Strikes in Canadian Waters

NMFS remains committed to working with Canada through various bilateral fora including the U.S.-Canada bilateral working group to focus on the cross-boundary conservation and protection of right whales. Specifically, NMFS continues to regularly engage with Fisheries and Oceans Canada and Transport Canada at both the senior leadership and staff levels to share information and explore opportunities for collaboration on transboundary resource management issues. This includes the efforts of the bilateral right whale working group to identify jointly data and management gaps that are impeding recovery of right whales in both Canada and the United States. NMFS is committed to working with Canada through the MMPA Import Provisions process to evaluate whether applicable Canadian fisheries have regulatory programs that are comparable in effectiveness to the regulatory program governing U.S. fisheries for protecting marine mammals, including right whales.[79] These bilateral efforts are important to achieving the United States and Canada's shared goals of conserving and restoring this species.

Each year, the Government of Canada implements a number of measures to address entanglements and vessel strikes involving right whales in Canadian waters. The measures that Fisheries and Oceans Canada has implemented currently include, but are not limited to, a new season-long closure area protocol in the Gulf of St. Lawrence, expansion of the dynamic fishery closure areas into the Bay of Fundy, and mandatory gear markings for all fixed gear fisheries in eastern Canada. Transport Canada's measures currently include, but are not limited to, a variety of mandatory (static and dynamic) and voluntary vessel speed restriction zones in the Gulf of St. Lawrence between April and November for most vessels greater than 13 meters (43 feet) in length. Canada has continued to modify their mitigation measures annually in response to interactions and information on right whale distribution and movements. However, entanglement and vessel strikes continue to be a threat to right whales in Canadian waters within and possibly beyond the areas under management. It is also not possible currently to quantify the level of reduction in M/SI that is achieved through existing Canadian management measures. It is important that evaluation of the effectiveness of Canadian's measures must be accomplished in the near future in order to fully assess the impacts of the measures on the overall survival and recovery of right whales.

Conclusion

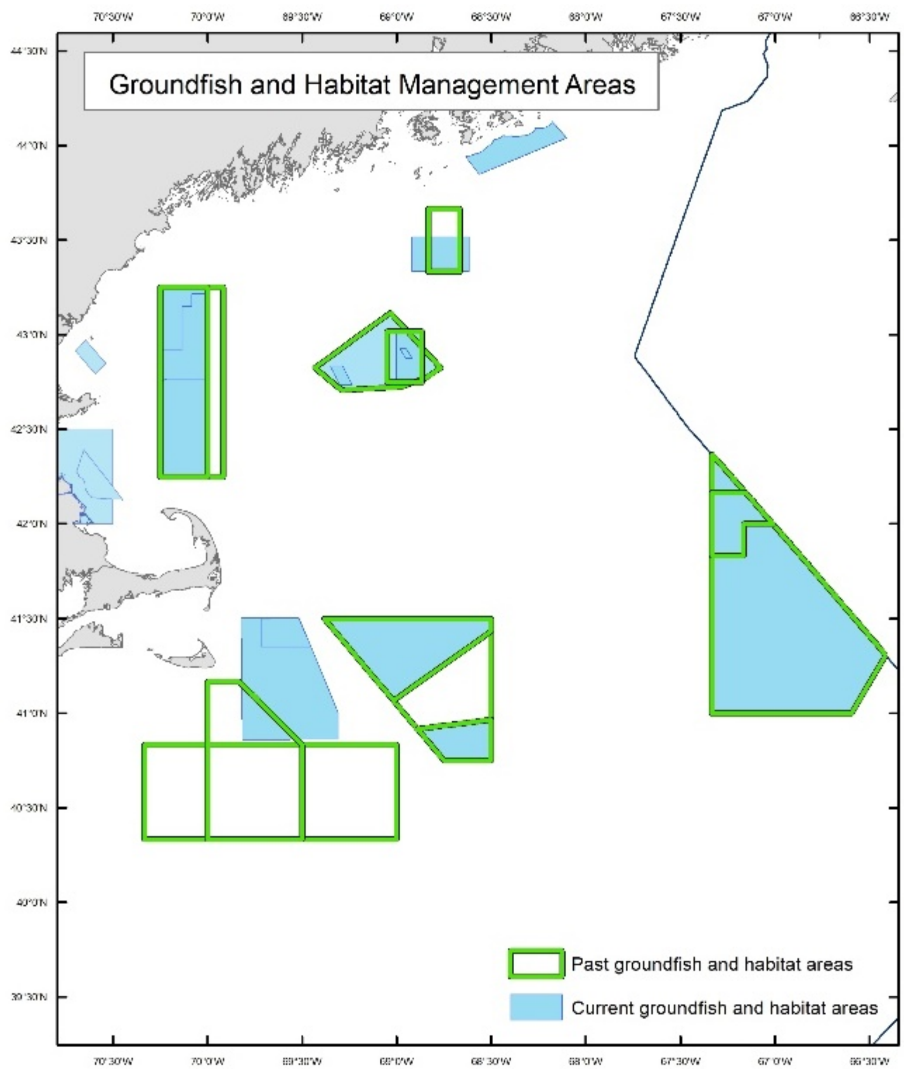
Significant efforts to recover North Atlantic right whales are currently underway and planned throughout the species' range. This Conservation Framework provides an additional commitment by NMFS GARFO to further efforts in federal waters to reduce mortalities and serious injuries due to entanglement in the fisheries managed by GARFO. Protecting and conserving this critically endangered species is especially important given the reduced rate of calving, the rapid decline in the population, and the evidence of a continued high rate of mortality. To ensure the species' recovery, the United States and Canada must introduce additional efforts to reduce right whale mortalities and serious injuries. NMFS remains committed to recovering right whales and is continuing to work to reduce mortalities and serious injuries.

[79] It is likely that the MMPA Import Provisions will be increasingly important, as environmental changes are likely to continue to shift lobster and right whale distribution into Canadian waters.

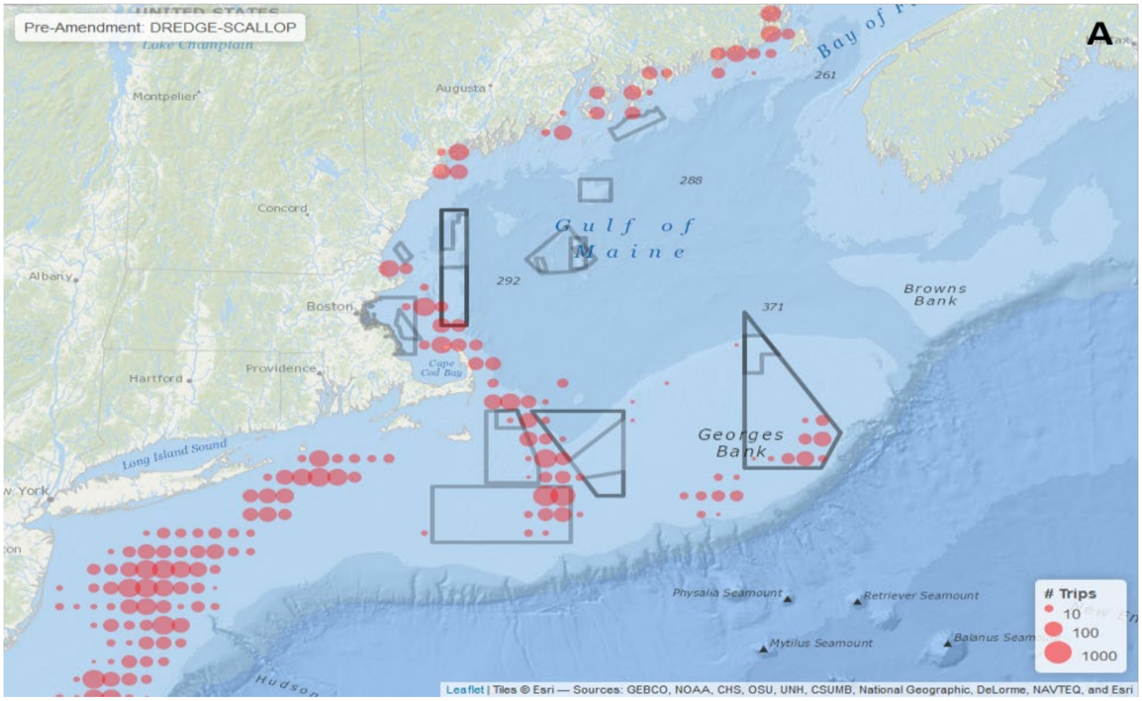
14. APPENDIX B: SCALLOP DREDGE, HYDRAULIC CLAM DREDGE, BOTTOM TRAWL, SINK GILLNET, BOTTOM LONGLINE, MID-WATER TRAWL, AND PURSE SEINE VTR DATA PRE-AND POST- HABITAT OMNIBUS AMENDMENT.

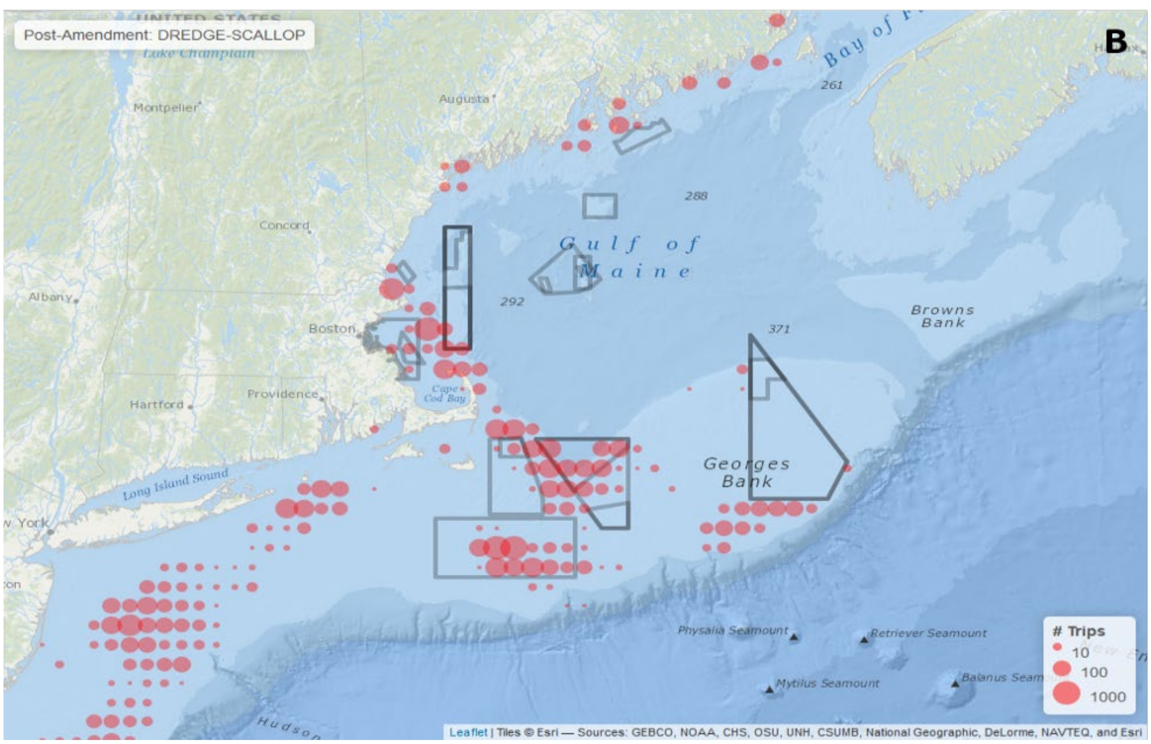
Maps of scallop dredge, hydraulic clam dredge, bottom trawl, sink gillnet, bottom longline, mid-water trawl, and purse seine VTR data Pre-and Post- Habitat Omnibus Amendment. Figure A=Pre-Amendment (September 1, 2016-March 31, 2018) and Figure B=Post Amendment (April 1, 2018-October 31, 2019). VTR data and maps were provided by GARFOs APSD. Refer to the *Reference Map* to identify areas delineated as groundfish and Habitat closures pre-and post-Amendment. Each map by gear type includes all pre- and post-Amendment areas.

Reference Map: Habitat and Groundfish Closure Areas Pre- and Post- Habitat Amendment

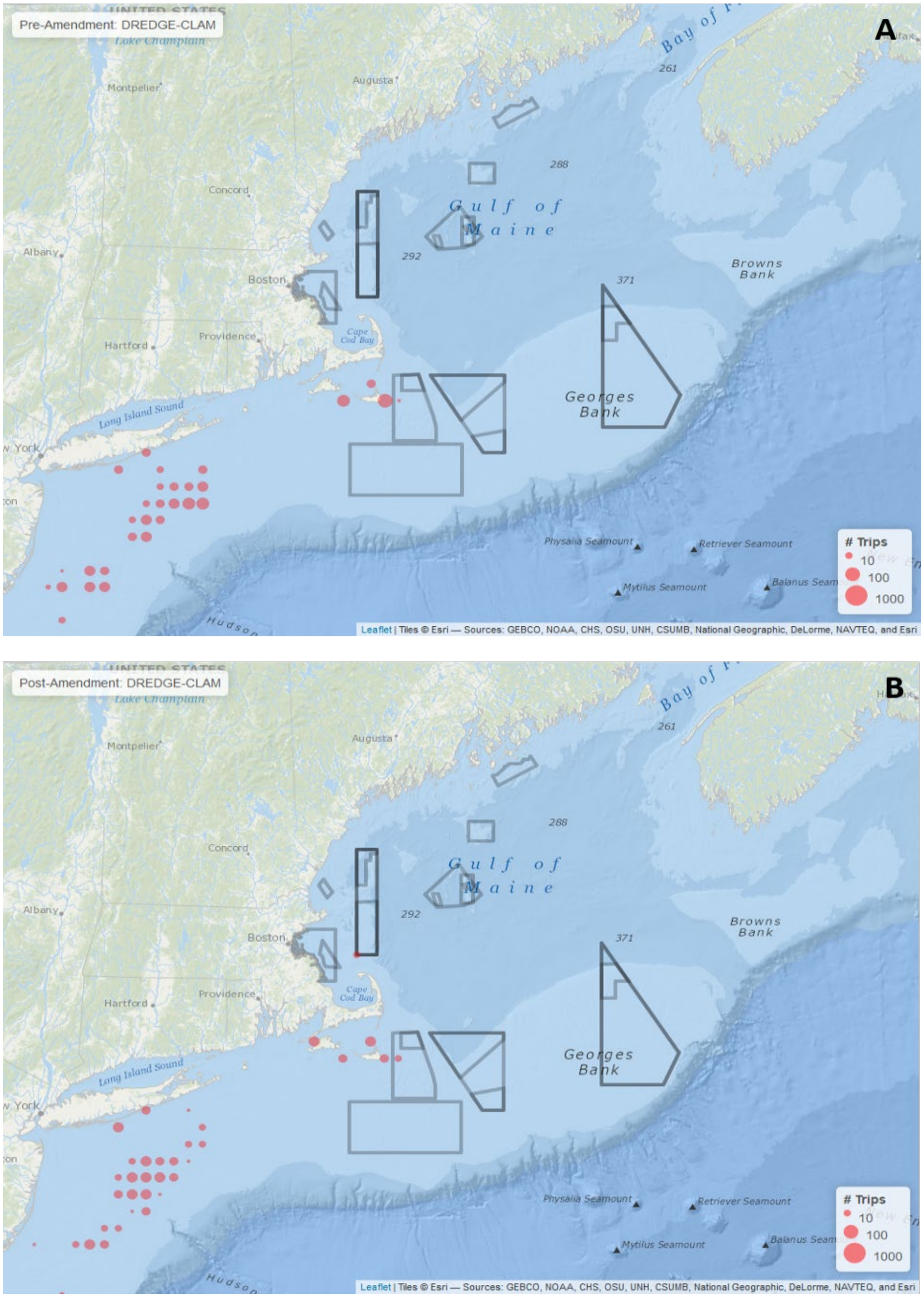


Scallop Dredge

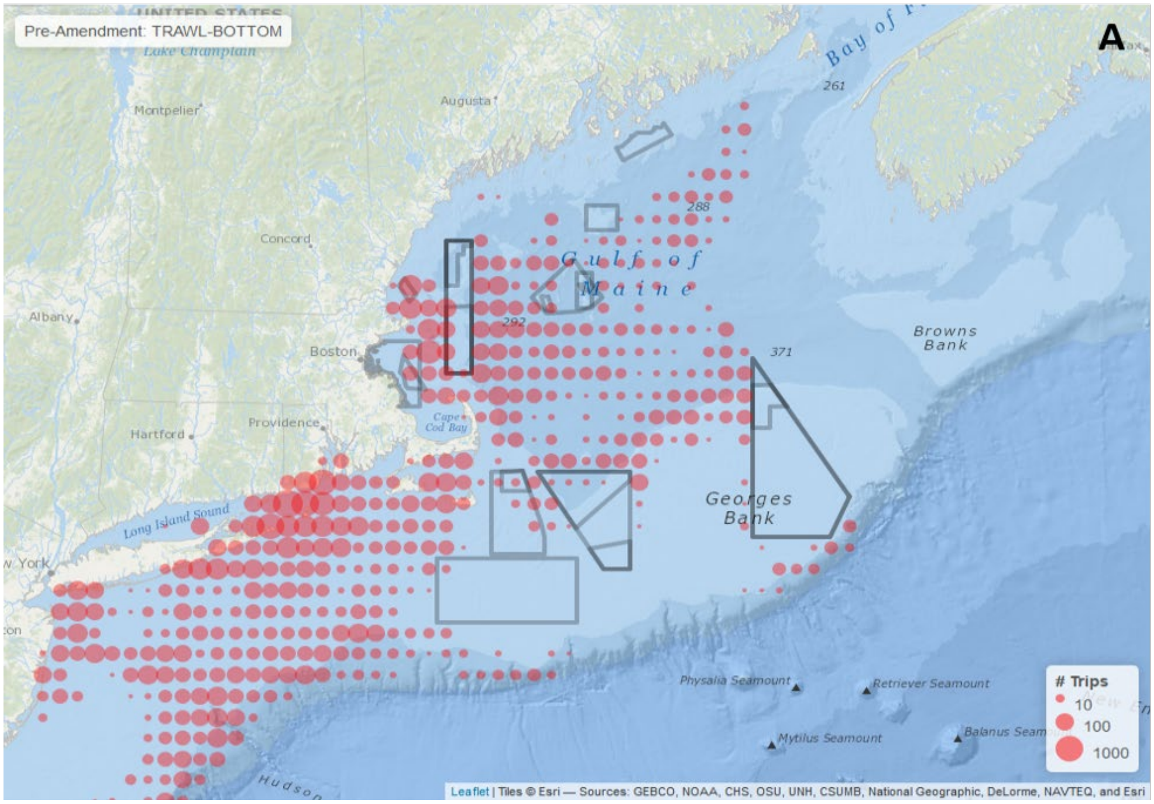


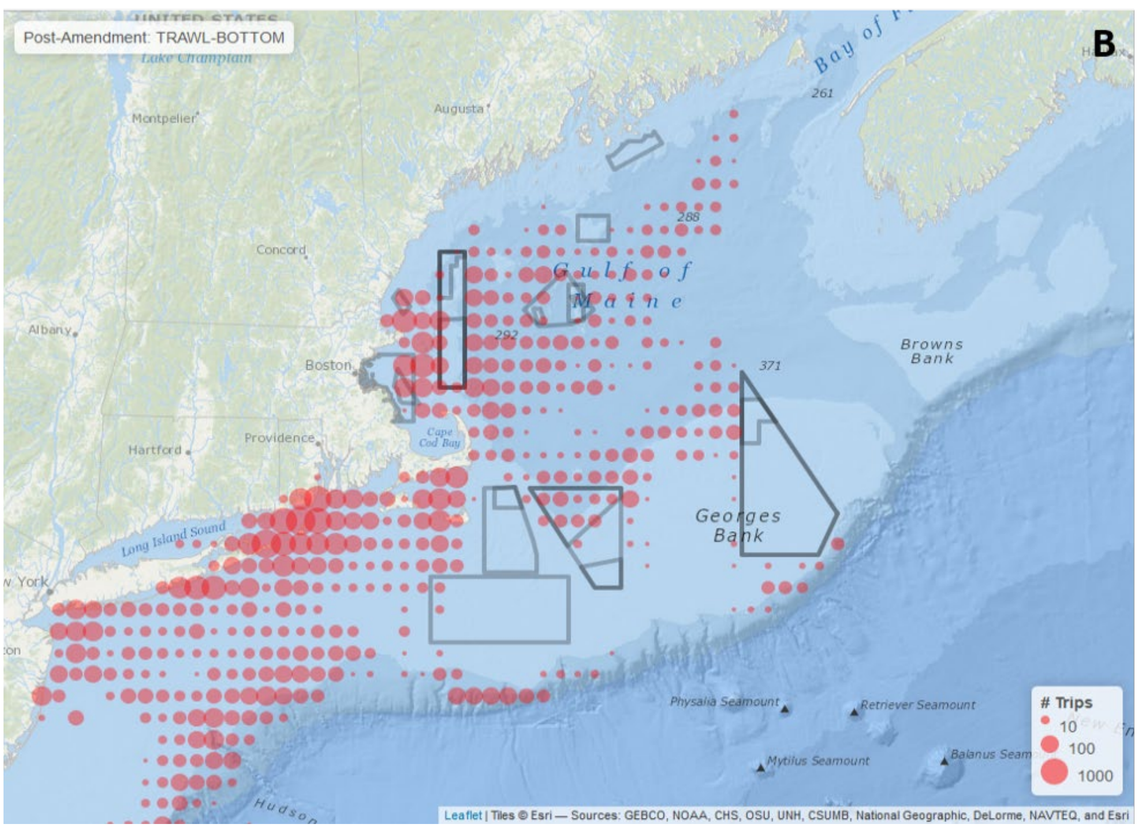


Hydraulic Clam Dredge

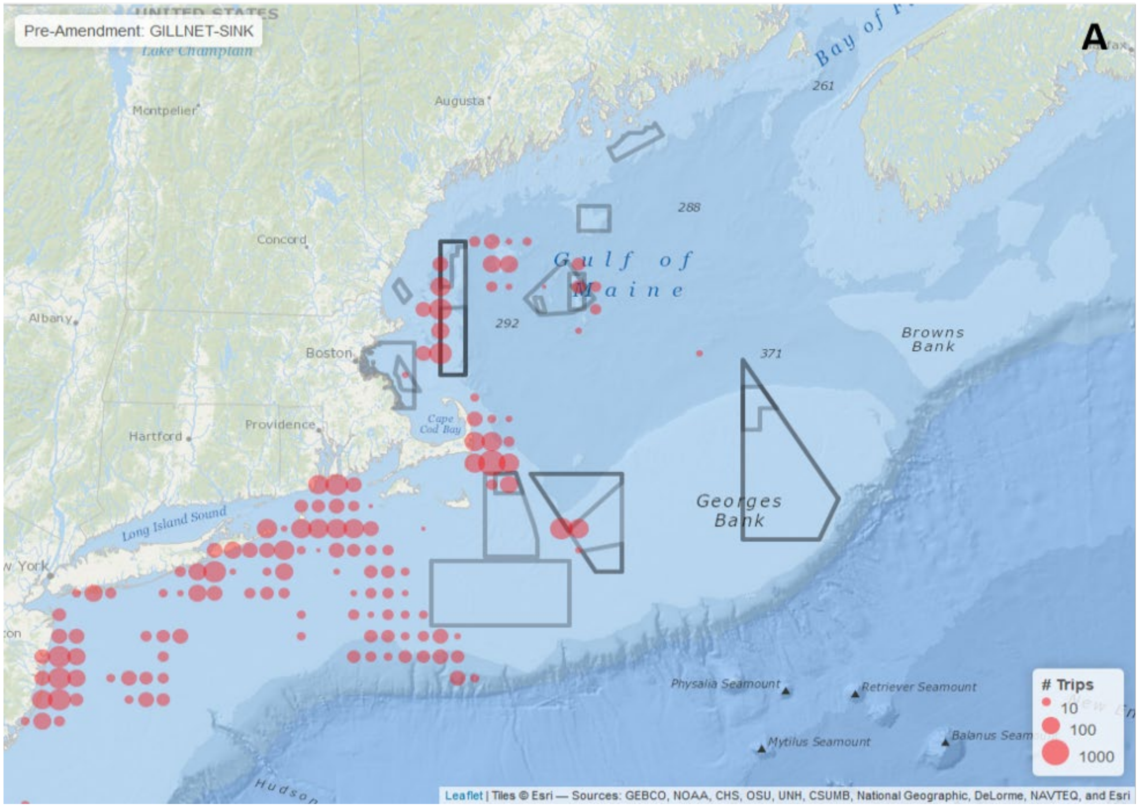


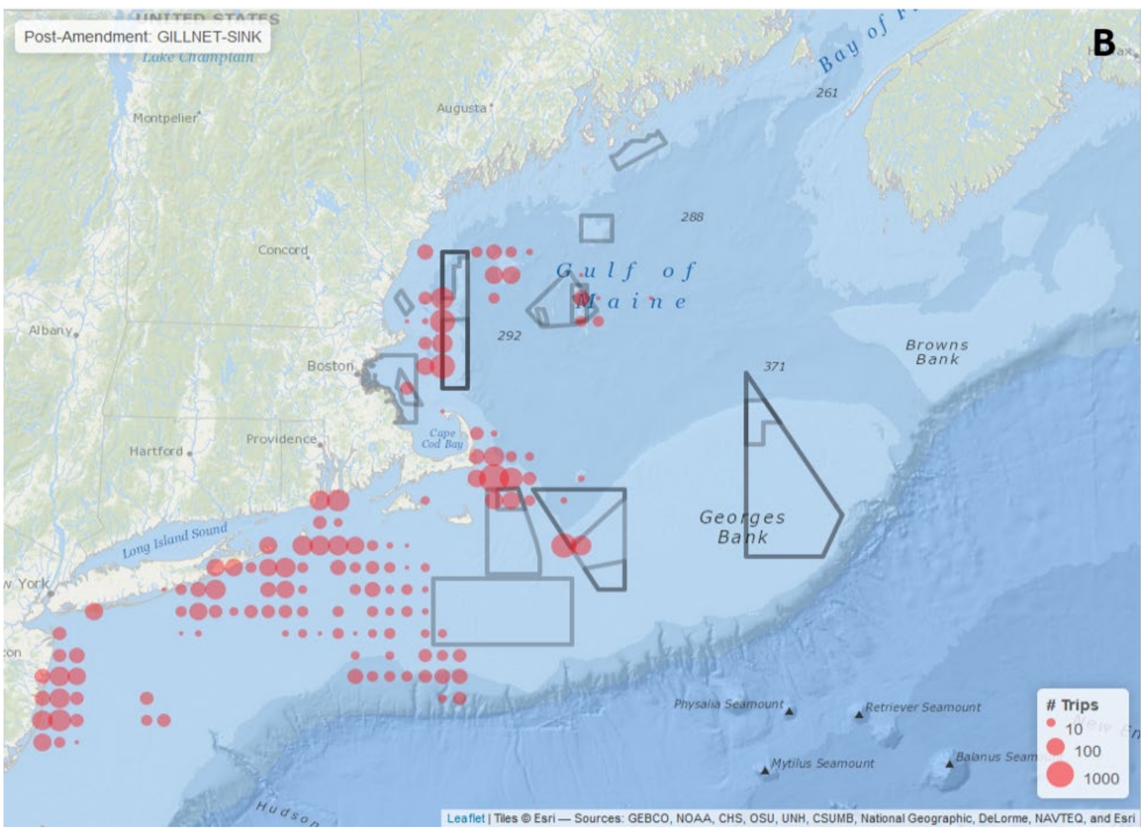
Bottom Trawl



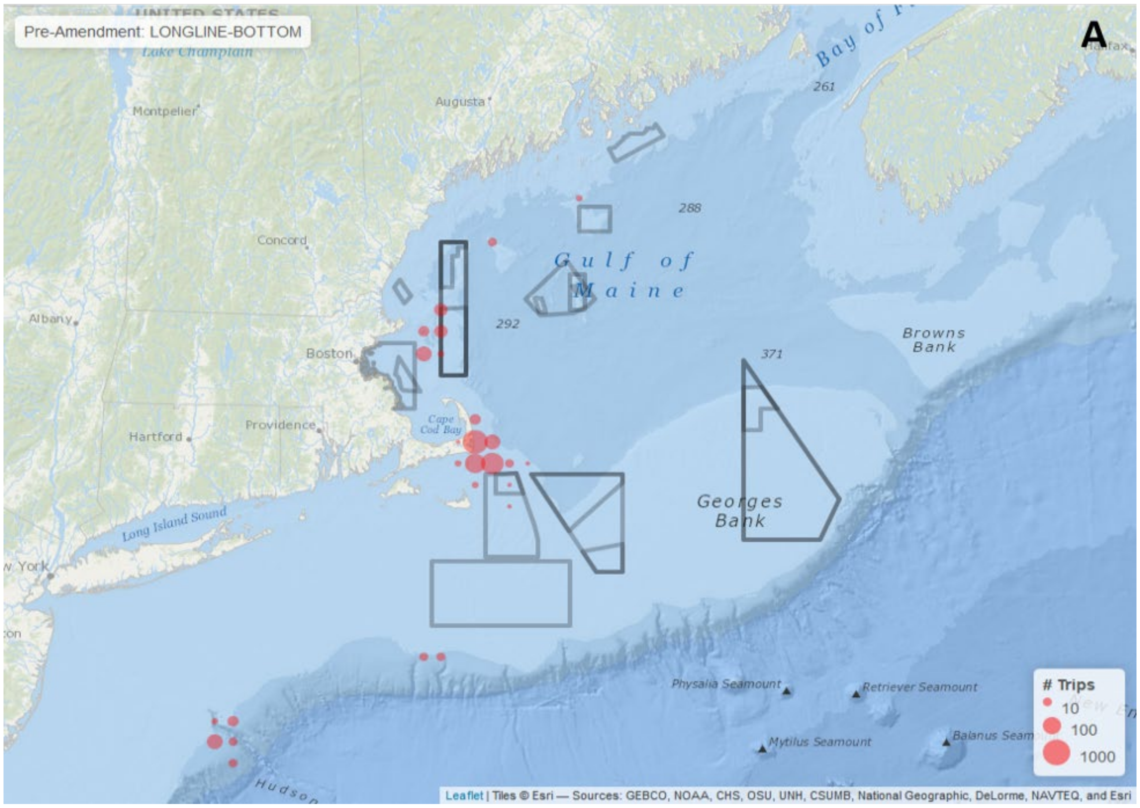


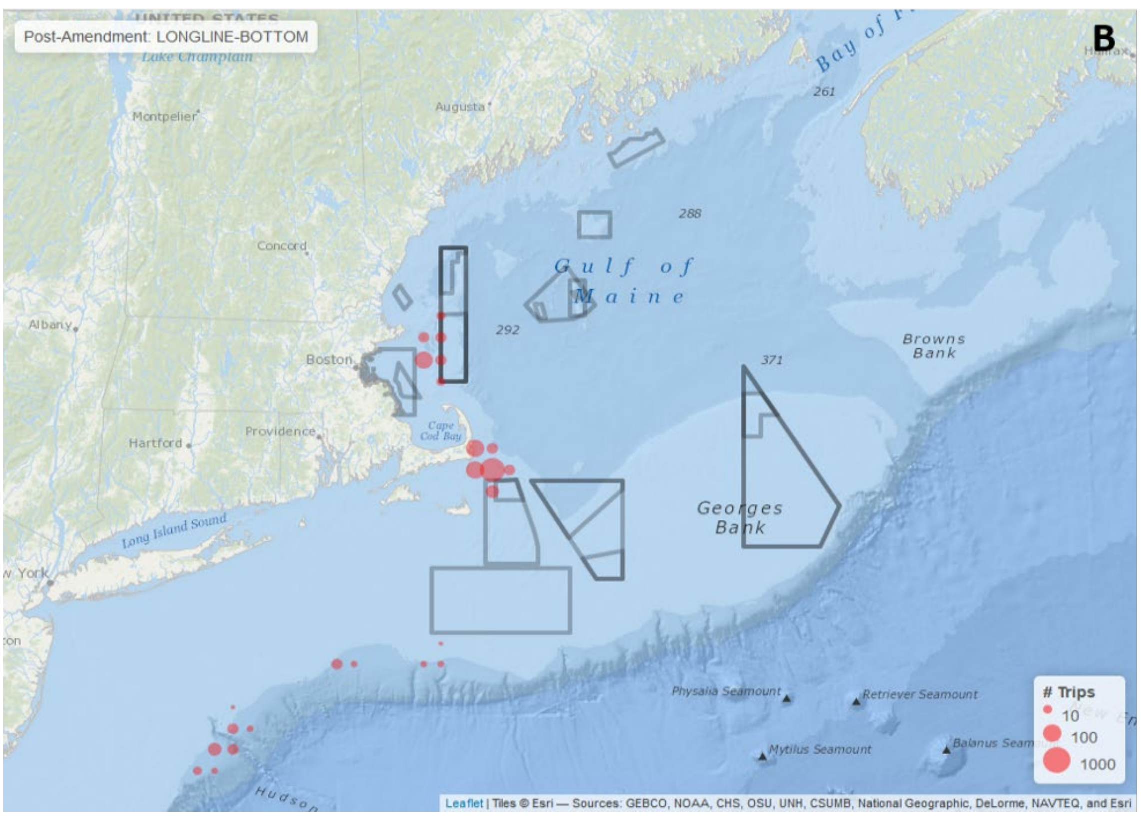
Sink Gillnet



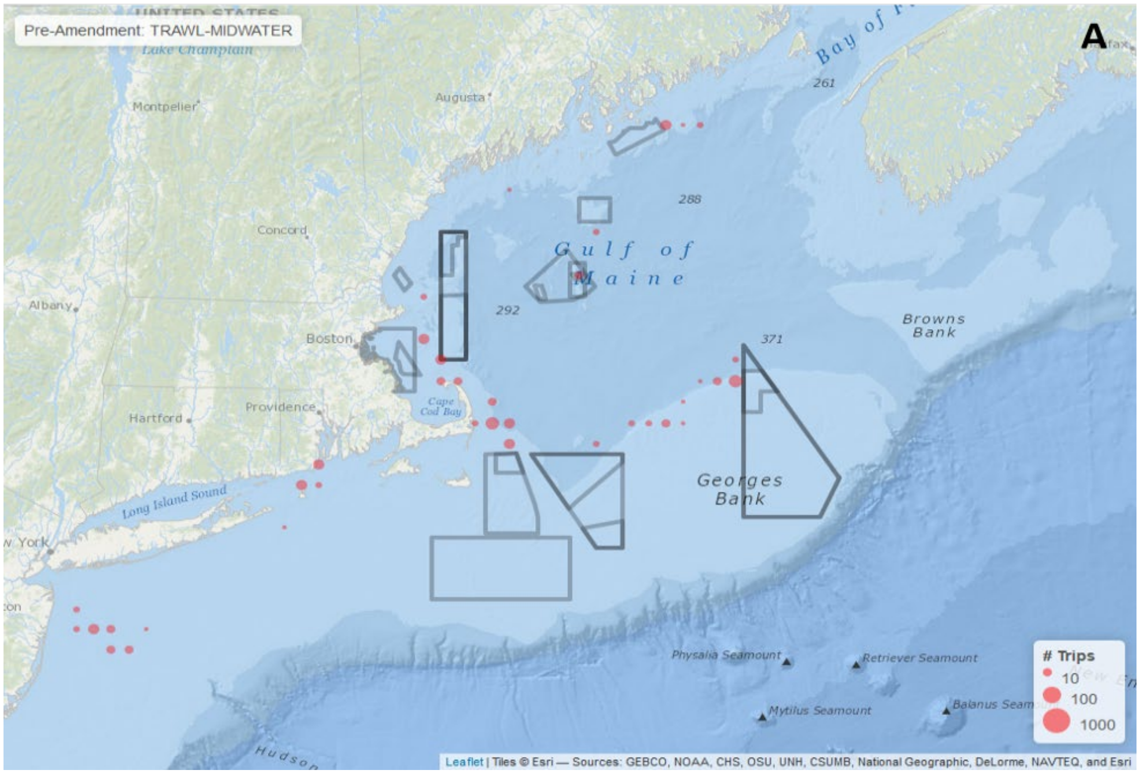


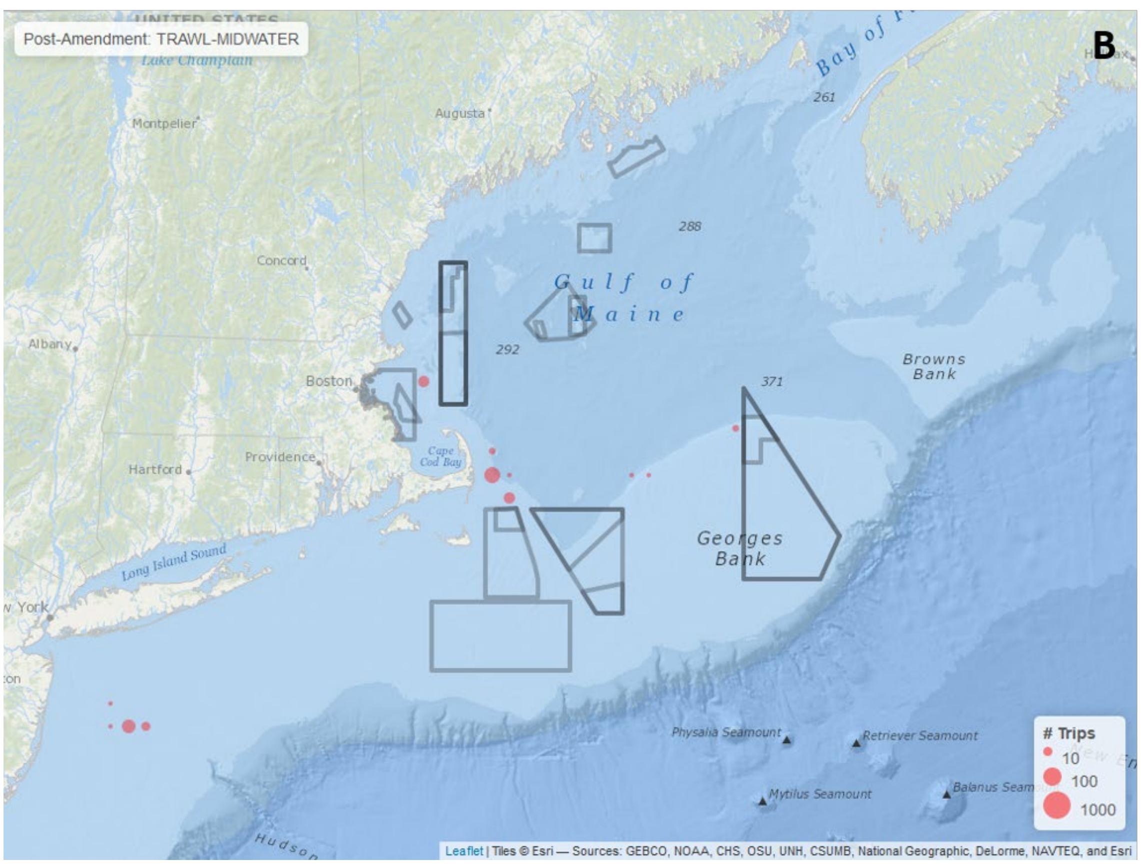
Bottom Longline



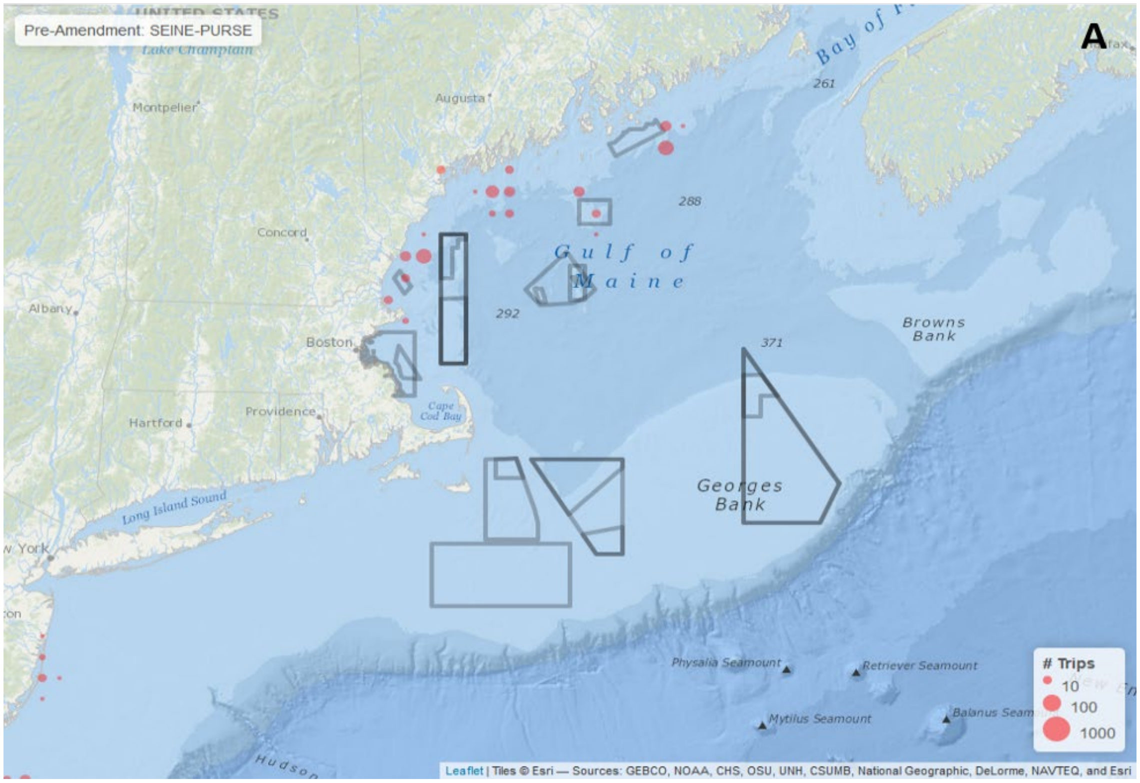


Mid-Water Trawl





Purse Seine


Pre-Amendment: SEINE-PURSE
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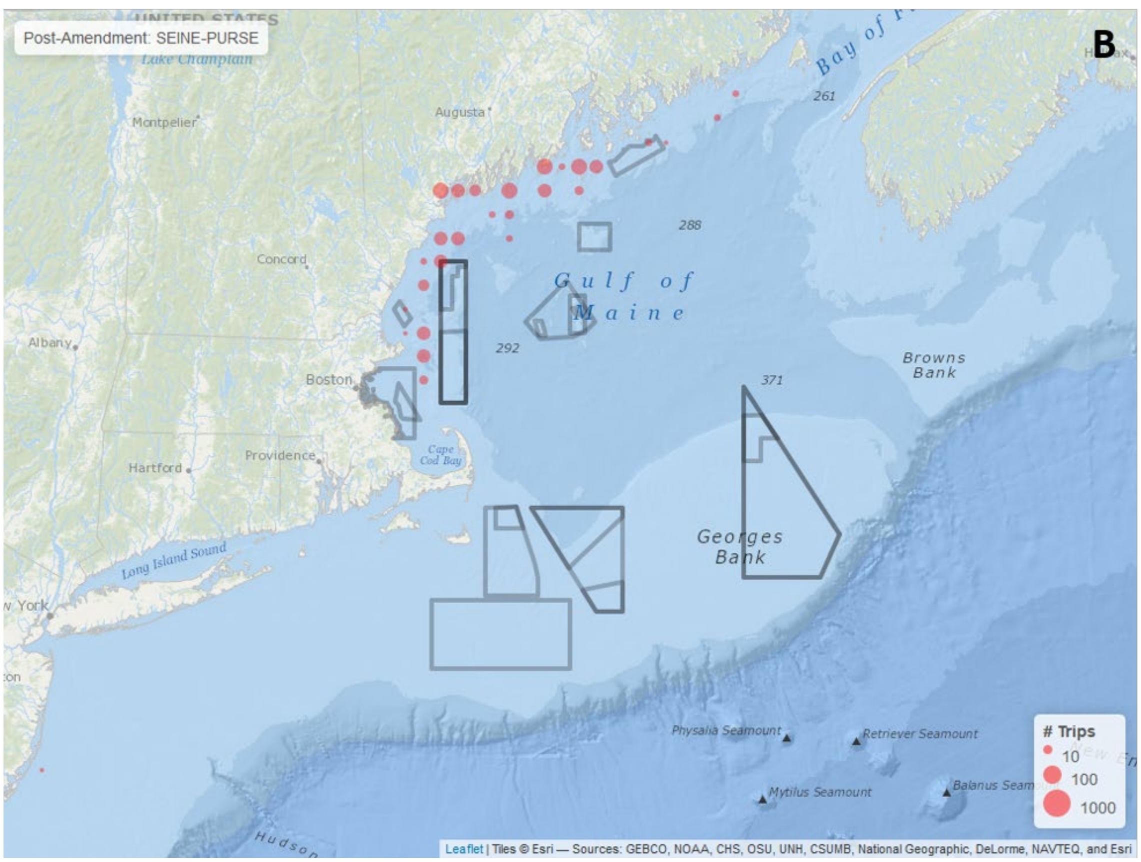
Leaflet | Tiles © Esri — Sources: GEBCO, NOAA, CHS, OSU, UNH, CSUMB, National Geographic, DeLorme, NAVTEQ, and Esri


Post-Amendment: SEINE-PURSE
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Leaflet | Tiles © Esri — Sources: GEBCO, NOAA, CHS, OSU, UNH, CSUMB, National Geographic, DeLorme, NAVTEQ, and Esri

15. APPENDIX C: POPULATION PROJECTIONS OF NORTH ATLANTIC RIGHT WHALES UNDER VARYING HUMAN-CAUSED MORTALITY RISK AND FUTURE UNCERTAINTY

Population projections of North Atlantic right whales under varying human-caused mortality risk and future uncertainty

Daniel W. Linden, NOAA/NMFS/GARFO

6 January 2021

Summary: The population decline of North Atlantic right whales (NARW) over the past decade has been due to high rates of human-caused mortality and low rates of reproduction. We used population projection models to explore how current demographic rates are limiting NARW recovery and whether reductions in human-caused mortality could mitigate a continued population decline into the future. We found that even a 100% reduction in risk from entanglement in the U.S. pot/trap fishery is not likely to prevent future declines given the current low rates of reproduction. A 50% total reduction in human-caused mortality across both the United States and Canada resulted in a positive trajectory on average, requiring ∼7 fewer NARW mortalities per year to achieve. Despite extensive monitoring, there remain key uncertainties regarding NARW population dynamics that may hinder conservation efforts.

Introduction

The North Atlantic right whale (*Eubaleana glacialis*; NARW) is listed as endangered under the Endangered Species Act and its recovery appears to be limited by human-caused mortality (Corkeron et al. 2018). Additional pressures on the population include environmental changes in prey availability and the resulting effects on calving rates (Meyer-Gutbrod and Greene 2018; Sorochan et al. 2019), which when combined with the high mortality rates have lead to a declining population size since 2010 (Pace, Corkeron, and Kraus 2017). Human causes of right whale mortality in the United States and Canada include entanglement in fishing gear (e.g., vertical lines from pots and traps) and vessel strike. Here, we used population projection models (Caswell 2001) to illustrate how human-caused mortality and uncertainty in reproduction lead to varying predictions about the long-term persistence of NARW.

Population projection models (or matrix population models) use information on the demographic parameters of a population (e.g., survival and reproduction) to forecast changes in population size and provide insights on future population growth (Caswell 2001). The application of population projections to NARW served as a motivating example for this modeling framework (Fujiwara and Caswell 2001), and two recent examples of NARW population projection models have relied on updated sightings data to provide contemporary context (Corkeron et al. 2018; Meyer-Gutbrod and Greene 2018). The components of a population projection include: 1) an age- or stage-structured population model (Fig. 1, as used in Corkeron et al. (2018)) that maps how individuals move between classes (ages or stages) over time; 2) the estimated rates of such movement between classes according to empirical data or hypothetical scenarios; and 3) the initial number of individuals in each class, from which a projection propagates.

Corkeron et al. (2018) approximated the growth potential of NARW in comparison to several populations of southern right whale (*Eubalaena australis*), with the goal of determining whether intrinsic factors or human-caused impacts were constraining the NARW population. Corkeron et al. (2018) used the highest estimated NARW survival probability (for adult females) and an optimal calving rate for their projection, with the initial number of individuals based on a total estimate from 1990 spread among classes (calves, juveniles, adults) according to a stable age distribution. Their projection model was applied in a retrospective manner by simulating the outcomes that may have occurred under optimal conditions for NARW population growth from 1990 to 2015. In contrast, Meyer-Gutbrod and Greene (2018) used estimates from their own fitting of a capture-recapture model to NARW sightings data (1980–2012) for their projections, with a slightly different stage-structured model (depicting deaths and reproductive states explicitly) and initial numbers based on the population estimates from 2012. Their analysis also included a comparison of calving rates during three periods (1980s, 1990s, 2000s) to quantify environmental uncertainty and the addition of increasing adult female NARW mortalities to understand human-caused risks (Meyer-Gutbrod and Greene 2018). In each of these recent examples, the NARW demographic parameters were chosen to represent either current understanding of the population or hypothetical changes to the environment (e.g., resources, risk) to examine the resulting population trajectories.

Important environmental changes within critical NARW habitat have occurred since the time period used for the Meyer-Gutbrod and Greene (2018) projections (Record et al. 2019). In 2010, there was a regime shift in seasonal sea surface temperature in the Gulf of Maine and Georges Bank, followed in 2012 by a shift in the Gulf of Maine zooplankton community (Morse et al. 2017). Changes in zooplankton productivity may be a critical pathway for climate to impact higher trophic levels of the Northeast continental shelf (Morse et al. 2017). The regime shift in 2010 coincided with a noticeable shift in NARW distribution and habitat use (Davis et al. 2017), and a trending decline in population size (Pace, Corkeron, and Kraus 2017). Climate-driven changes in the Gulf of Maine have shifted the seasonal patterns for essential NARW prey (*Calanus finmarchicus*), which likely caused NARW to shift their distribution in search of adequate prey availability (Davis et al. 2017; Morse et al. 2017; Record et al. 2019). There is no information available to suggest when a shift in the zooplankton community may occur again.

We built population projections using the most recent NARW sightings data (2010–2018), encapsulating the post-2010 ecological conditions that coincide with recent population declines. We projected the population forward under our current state of knowledge regarding NARW demographic parameters, along with several risk-reduction scenarios that reduced adult mortality through hypothetical mitigation measures (e.g., restricted fishing effort). Our projections indicate that recent low rates of calving and high rates of human-caused mortality will inhibit NARW population persistence in the absence of mitigation.

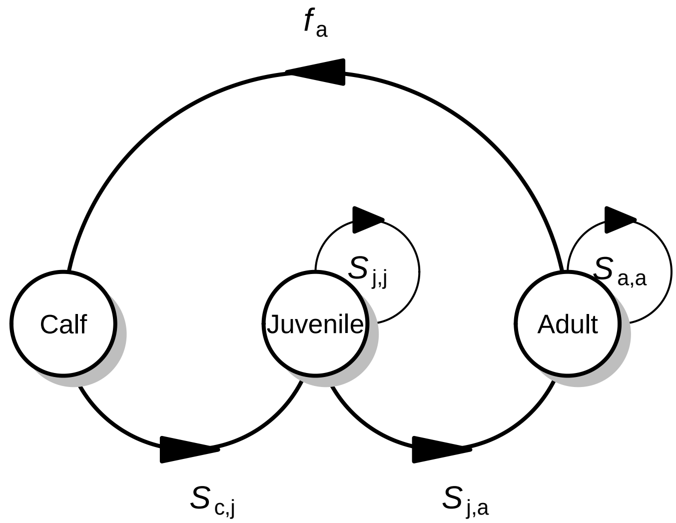

Figure 1: Simple stage-structured demographic model of female North Atlantic right whales, as used in Corkeron et al. 2018.

Methods

Stage-structured population model

Following Corkeron et al. (2018), we used a simple stage-structured population model (Fig. 1) with 3 stages (calf, juvenile, adult) and a focus on females. We assume individual females move between or stay within stages across time as defined by survival transition rates (S), and new individuals (calves) arise according to the fertility rate (f_{a}) of adults. The survival transitions include a single-year survival probability for a given stage and a stage duration dictating how many years an individual must survive in a given stage before maturing into the next stage. The transition rates are calculated by combining survival probabilities and stage durations using some algebra (Crouse, Crowder, and Caswell 1987). We used stage durations of 1 year for calves and 4 years for juveniles (to accommodate age classes in Pace, Corkeron, and Kraus (2017)), and we assumed a maximum longevity of 69 years for adults (Kraus and Rolland 2007).

The stage-structured population model is translated into a projection matrix by summing a matrix $\mathbf{T}$ of transitions and a matrix $\mathbf{F}$ of fertilities ($\mathbf{A} = \mathbf{T} + \mathbf{F}$), with each resulting matrix element quantifying how the individuals in a given stage (represented by rows) are converted into individuals of the same or different stage (represented by columns) (Caswell 2001). The matrix algebra is then, $\mathbf{n}_{t+1} = \mathbf{A}\mathbf{n}_t$, where $\mathbf{n}_t$ is the vector of individuals within each stage at time t, which converts to the vector of individuals at time $t+1$ according to the projection matrix $\mathbf{A}$. Here, our projection matrix was as follows:

$$\mathbf{A} = \begin{bmatrix} 0 & 0 & f_{\mathrm{a}} \\ S_{\mathrm{c,j}} & S_{\mathrm{j,j}} & 0 \\ 0 & S_{\mathrm{j,a}} & S_{\mathrm{a,a}} \end{bmatrix}$$

Demographic parameter estimates

We used the NARW sightings data collected during 1990–2018 to fit the Pace, Corkeron, and Kraus (2017) state-space mark-recapture model and generate posterior distributions for stage-based population sizes. Sightings data are collected by a large-scale research collaboration known as the North Atlantic Right Whale Consortium (https://www.narwc.org/) which maintains an extensive photograph catalog (Hamilton, Knowlton, and Marx 2007). Our approach to fitting the state-space model was fully described in Pace, Corkeron, and Kraus (2017) and is not repeated here. Survival probabilities varied as a fixed effect by sex and age, and randomly by year, while capture probabilities varied by sex and randomly by year and individual. One significant change from Pace, Corkeron, and Kraus (2017) was the addition of a "post-2010" difference in average survival probability that is currently incorporated into the stock assessment (*Richard Pace, NMFS, personal communication*).

The state-space model was fit using Markov chain Monte Carlo (MCMC) methods, allowing the full posterior distributions of any structural or derived parameters to be extracted. The main parameter estimates of interest were the female population sizes for each of 6 age classes (0, 1, 2, 3, 4, 5+), assuming calves = 0, juveniles = 1–4, and adults = 5+. We also extracted the posterior distributions for total female deaths by age class, allowing for a calculation of realized survival as follows:

$$S_t^k = 1 - (D_{t+1}^k / N_t^k)$$

Here, the realized survival S_t^k for age class k in year t was the inverse of the mortality rate, the ratio of dead individuals D_{t+1}^k in year $t+1$ to live individuals N_t^k in year t. The age class of an individual's last year alive was the age class to which its death was attributed. The state-space model easily generates posterior estimates for expected survival, specified as ϕ in Pace, Corkeron, and Kraus (2017), without the need for derived calculations of live and dead individuals. These estimates of expected survival were used for calves and juveniles due to small sample sizes. The derived calculations of realized survival were used for adult females to enable manual manipulation of deaths for scenario projections (*see* Risk reduction scenarios).

The fertility rates, or calving rates, were calculated using the ratio of observed calf counts to the estimated population size $N_t^{k=5+}$ of adult (age = 5+) females in a given year t. The extensive survey efforts on the breeding grounds allow for the assumption that observed calf counts are essentially a census of NARW calf production (Kraus and Rolland 2007). Since our population projections were focused on females, we also assumed that half of calves were female, reducing the effective fertility rates by 50%.

The demographic parameters for our projections were based on the state-space model estimates for the 2010–2018 time period. Thus, while the full time series was used to fit the model, the post-2010 parameter estimates were considered most representative of current conditions and used in the population projections. While Pace, Corkeron, and Kraus (2017) indicated little annual variation in NARW survival probability with no apparent trend, the fertility rates have undergone large fluctuations on near-decadal time steps (Meyer-Gutbrod and

Greene 2018). We assessed the influence of using a larger time series (1990–2018) to quantify uncertainty in fertility rates (Supplement 4).

Risk reduction scenarios

The U.S. Atlantic Large Whale Take Reduction Team has recommended mitigation measures for northeast U.S. pot/trap fisheries aimed at reducing NARW mortalities. To examine the outcome of hypothetical mitigation measures, we constructed scenarios that modified the survival probability estimates. We defined two types of mitigation: 1) reduction in mortality due to entanglement in U.S. pot/trap fisheries; and 2) reduction in mortality due to any human-related causes in Canada. Our first set of scenarios considered mitigation due to #1 while the second set considered mitigation due to both #1 and #2. We did not consider cause of mortality in Canada due to insufficient information on whale deaths and uncertainty regarding potential mitigation measures.

We calculated the modified survival estimates by partitioning total mortalities to the United States and Canada, and then quantified U.S. mortalities that were presumed to be pot/trap gear entanglement. The information documented for some observed NARW mortalities contained useful evidence that allowed attribution to a country, and further attribution to a specific cause (e.g., gear entanglement), but some observed mortalities were of unknown cause and origin. In addition, depending on year, the total estimated mortalities could contain a large number of unobserved (or "cryptic") mortalities. We relied on determinations by the Northeast Fisheries Science Center (NEFSC) and the Greater Atlantic Regional Fisheries Office (GARFO) to attribute country of origin and specific causes for those mortalities without observations. Mortalities with unknown origin were split 1:1 between the countries[1], while 80% of U.S. mortalities with unknown cause, observed or unobserved, were attributed to pot/trap gear entanglement (based on observed ratios[2]). The total estimated human-caused mortalities for 2010–2018 were therefore assumed to be 75 in Canada and 50 for pot/trap gear in the United States[3]. Note, the cryptic mortalities were calculated as the difference between the median estimate of total deaths (from the state-space model) and the count of observed mortalities.

For each set of scenarios, the total adult female deaths $D^{k=5+}_{2010-2018}$ were reduced by some percentage (0 to 100%) to reflect different magnitudes of mitigation, either by the United States or both countries (Table 1). We assumed that a given scenario reduced mortalities equally for males and females so mortality reductions were halved to be female specific. Once the total count of "saved females" was calculated for a scenario, the count was distributed at random across the 9 years according to a multinomial distribution, with probabilities determined by the relative number of adult female deaths in each year. For each iteration

[1]While the population spends more time in U.S. waters, this split in risk is supported by analysis of recovered entangling gear. The gear causing entanglement in Canada has also been found to be more lethal.

[2]During 2010–2018 there were ~12 serious injury/mortality (SI/M) events confirmed to be vessel strikes and 47 SI/M confirmed to be entanglement in the United States.

[3]Other human-caused mortalities in the United States were estimated to be ~14.6 SI/M for vessel strikes and 1.25 SI/M for non-pot/trap gear. These estimates include confirmed and presumed counts, with fractions reflecting how serious injuries are scored by the agency.

Table 1: Mortality reduction scenarios explored in the NARW population projections. The mortalities listed represent the total losses due to pot/trap gear entanglement in the U.S. and all human-related causes in Canada after some hypothetical mitigation measures. The 0% reduction represents the current mortality count (i.e., status quo) for the relevant causes during 2010–2018.

	NARW mortalities (2010–2018)		
Reduction (%)	U.S. (pot/trap)	Canada (all)	Total
0	50	75.0	125.0
10	45	67.5	112.5
20	40	60.0	100.0
30	35	52.5	87.5
40	30	45.0	75.0
50	25	37.5	62.5
60	20	30.0	50.0
70	15	22.5	37.5
80	10	15.0	25.0
90	5	7.5	12.5
100	0	0.0	0.0

of the posterior distribution from the state-space model, the random assortment of saved females was subtracted from $D^{k=5+}_{2010-2018}$ and new values for $S^{k=5+}_{2010-2018}$ were calculated. This procedure effectively lowered the mortality rates (or raised the survival rates) across years. For example, if $N_t = 100$ and $D_{t+1} = 10$, a 50% reduction would *on average* reduce D_{t+1} by 5 and result in the survival rate S_t increasing from 0.90 to 0.95. In instances where an equal or greater number of adult females were "saved" than were predicted to have died, given stochasticity in the random multinomial, we set $S^{k=5+}_t$ to 0.9999 to prevent matrix calculation problems. This conservative choice was meant to counteract both the assumption of an even split in mortality reductions between sexes and the restricted application to adults (5+). Note, the mortality reduction scenarios did not reduce the cumulative deaths from 2010–2018, they simply removed mortality events from individual years based on the total deaths in the 9-yr period.

Population projection analysis and simulations

We evaluated our projection framework to ensure a reasonable representation of the system before exploring future projections. First, we ensured that the population projection matrix model was both irreducible and ergodic (Stott et al. 2010), using mean values of the demographic parameters estimated for 1990–2018. We calculated eigenvalue elasticities (Caswell 2001) to examine how proportional changes in a projection matrix element influenced the asymptotic population growth rate (i.e., the dominant eigenvalue of the projection matrix) over the period. To further illustrate this influence we calculated population growth rates across observed ranges of values for fertility and adult female survival, changing the values of one parameter while altering the other. Finally, as an approximate validation for the projection framework we fit retrospective projections using baseline demographic estimates from 1991–2009 to compare with the actual estimates of population size during that period. We excluded the 2010–2018 period in this validation recognizing that the hypothesized regime shift in 2010 would not be well represented in the absence of an explicit model structure for the change.

We used the `popbio` package (Stubben and Milligan 2007) in `R` (R Core Team 2019) to simulate population projections with stochasticity, representing uncertainty in future NARW population dynamics. The stochasticity was manifested in two ways: 1) parameter value assignment in a given time step, representing environmental stochasticity, and 2) random number generation (i.e., births and deaths) given the assigned parameter value, representing demographic stochasticity. The median population size and demographic stage distribution for 2010–2018 according to the state-space model estimates was used as a starting population size.

Each projection spanned 50 time steps (years), and we ran 1,000 stochastic projections for a given reduction scenario. The parameter values for the projection matrix at each time step within a simulation were drawn from the available state-space model estimates for 2010–2018. For the survival transition rates, the full posterior distributions of $S^k_{2010-2018}$ were used to draw a single value and calculate the appropriate transition rate for each stage. For the fertility rates, 1 of 9 values was drawn with equal probability from the calving rate estimates for 2010–2018.

We used the 1,000 stochastic projections to calculate a median population trajectory and probability of decline for a given scenario. The latter was simply the proportion of projections that resulted in a decreased population size at 10-year intervals.

Results

State-space model estimates

Our fit of the state-space mark-recapture model to the NARW sightings data from 1990–2018 indicated convergence and matched closely with the 1990-2015 estimates in Pace, Corkeron, and Kraus (2017) (Supplement 1). A closer look at 2010–2018 (Fig. 2) suggests that a decrease in the juvenile age class has been responsible for the overall decline of females reported by Pace, Corkeron, and Kraus (2017). The low fertility rate in recent years (Fig. 3) suggests that decreases in juveniles are likely due to a lack of new calves that would otherwise replace maturing females. The relatively higher estimates of adult female mortalities since 2015 (Fig. 4) matches the decline in expected survival (Fig. 5).

Table 2: Elasticities of asymptotic population growth rate to projection matrix transitions, using mean NARW demographic parameter estimates for females from 1990–2018.

	calf	juv	adult
calf	0.000	0.000	0.037
juv	0.037	0.100	0.000
adult	0.000	0.037	0.788

Population projections

The elasticities of population growth rate in Table 2 indicated that adult female survival had the greatest potential to affect growth rate, according to the mean demographic parameters from 1990–2018 as specified in the projection matrix. The resulting population growth across the ranges and combination of adult female survival and calf rates illustrate how each contributes to positive or negative growth (Fig. 6). Positive growth was apparent before the 2010 regime shift ("A"; Fig. 6), with negative growth after ("B"; Fig. 6).

Stochastic projections of the retrospective time series (1991–2009) indicated that a random sampling of demographic parameter values was adequate to capture the population change over time (Fig. 7). In the absence of explicit knowledge about temporal mechanisms (e.g., annual prey availability), fluctuations of higher (year = 2009) or lower (year = 2000) growth were not reflected by the average trend.

The initial NARW female population size used in the future projections (beyond 2018) was 183, split among calves (4), juveniles (22), and adults (157). Several individual population projections are available for both the U.S. pot/gear reduction scenarios (Supplement 2) and the combined U.S./Canada reduction scenarios (Supplement 3). As an example, the status quo (0% risk reduction) scenario in Fig. S1 indicates a population decline under current rates of survival and fertility.

The average population trajectory for the U.S. pot/gear reduction scenarios indicated a decline for NARW adult females across all scenarios including that under 100% reduction (Fig. 8). Given the large amount of uncertainty across the n=1000 projections for any single scenario (see Fig. S1), even with 100% reduction the probability of a decreasing NARW population in 50 years was >0.37 (Fig. 9).

With a combination of risk reductions for U.S. pot/trap gear and all human-related causes in Canada, the average population trajectory for scenarios with >40% reduction indicated an increasing trend (Fig. 10). The probabilities of a decreasing NARW population in 50 years were <0.04 for all scenarios reducing risk by $\geqslant$ 70% (Fig. 11).

Discussion

The growth of any wild population will be limited by either survival or reproduction (or both), and the difficulty for conservation and management is understanding the context of those limitations and how they interact across time and space. Previous population projections of NARW have suggested that human-caused mortality, specifically that of females, is a limiting factor for the species (Corkeron et al. 2018; Meyer-Gutbrod and Greene 2018). The Meyer-Gutbrod and Greene (2018) projections suggested that the NARW population could recover if the demographic rates observed during 1980–2012 continued, though their conclusions had to be tempered by awareness of the unusual mortality event in 2017 that occurred while their study was in the process of being published. Using population estimates from 2010–2018, our projections indicate that current rates of survival and fertility will lead to a further decline in the absence of effective mitigation (Fig. S1).

This analysis demonstrates that both the United States and Canada must implement measures to mitigate total NARW mortalities across their range to achieve a positive population trajectory. Our projections suggest that hypothetical mitigation involving risk reduction in the U.S. pot/trap fishery alone is unlikely to prevent further population declines even with 100% reduction (Fig. 8). With 100% reduction in entanglement mortalities in the United States, the probability that the NARW population still declines in spite of such efforts is relatively high (>50%; Fig. 9). This result indicates that the NARW population is likely to decline if human-caused mortalities in Canada continue at current rates, regardless of efforts in the United States. With combined efforts by both countries to reduce mortality, a positive NARW population trajectory appears to be more achievable (Figs. 10-11).

A major caveat to the conclusions about targeted mitigation measures is the uncertainty in apportioning mortalities due to entanglement (or any cause) to fisheries effort in waters managed by the United States vs. Canada. The 1:1 ratio was considered a "reasonable assumption" based on a collection of evidence that suggests no better strategy to apportionment (*see* CIE reviews[4]). Our projections could test the sensitivity of the apportioning value used here with additional scenarios (e.g., 3:1 or 1:3), though this exercise would simply introduce more uncertainty in the future projections and not change conclusions about the number of "whales saved" that would be necessary to improve population trajectories. A positive average trajectory is possible at 50% total risk reduction (Fig. 10), which translates to 62.5 whales during 2010–2018 (Table 1) or ~7 whales/year saved from a human-caused mortality event. From a population perspective, the source of the mitigation measures is immaterial. For management purposes, this uncertainty could be important and should be a target for improved information.

Similar to Meyer-Gutbrod and Greene (2018), our conclusions regarding one influential demographic parameter are contingent on the dynamics of another. Using calving/fertility rates from 1990–2018, the average population trajectory increases under all scenarios (Supplement 4). Declines in the availability of *Calanus* spp. (the main prey of NARW) since 2010, coincid-

[4] Center for Independent Experts (CIE) peer review reports from December 2019 for the "North Atlantic right whale DST review": https://www.st.nmfs.noaa.gov/science-quality-assurance/cie-peer-reviews/cie-review-2019

ing with a shift to warming ocean temperatures in the Gulf of Maine (Davis et al. 2017; Morse et al. 2017; Record et al. 2019; Sorochan et al. 2019), together suggests that environmental conditions may no longer approach the cycles of prey availability experienced prior to 2010 (Meyer-Gutbrod and Greene 2018). For these reasons, we caution any interpretation of the population projections using the 1990–2018 calving rates given they are unlikely to be representative of fertility rates going forward, though we acknowledge the importance of such uncertainty.

Finally, the population model does not consider the relationship between entanglement injuries and calving probability (Pettis et al. 2017). It is possible that mitigation measures aimed at reducing the risk of entanglement mortality would also reduce sub-lethal entanglements to reproductive adult females that may be partially responsible for suppressed calving rates in recent years. Thus, the scenarios represented here may be underestimating the benefits of risk reduction to the population by focusing only on mortality. More complex individual-based models may be necessary to explore how retrospective patterns of entanglement and reproduction could manifest into the future under various conditions.

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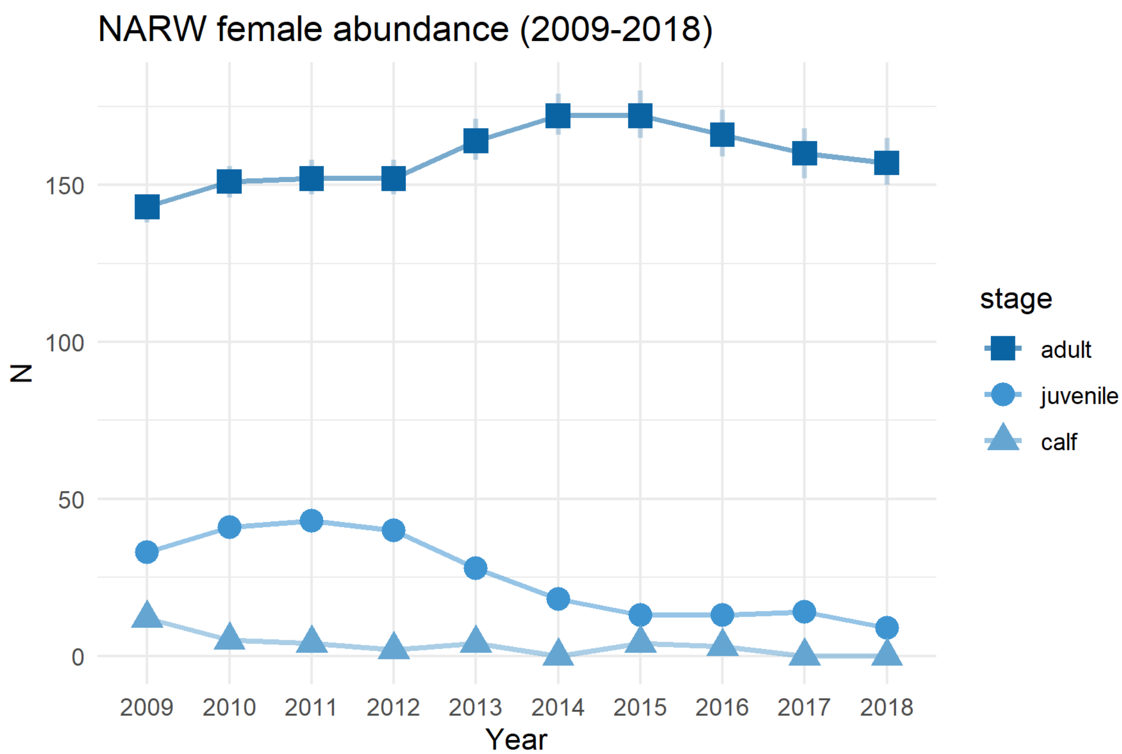

Figure 2: Abundance of NARW females during 2009–2018, as estimated by the state-space mark-recapture model of sightings data. Stages included adult (age = 5+), juvenile (ages = 1–4), and calf (age = 0). Median estimates with 95% credible intervals included. Note that estimates from 2009–2017 were used in calculating realized survival probabilities.

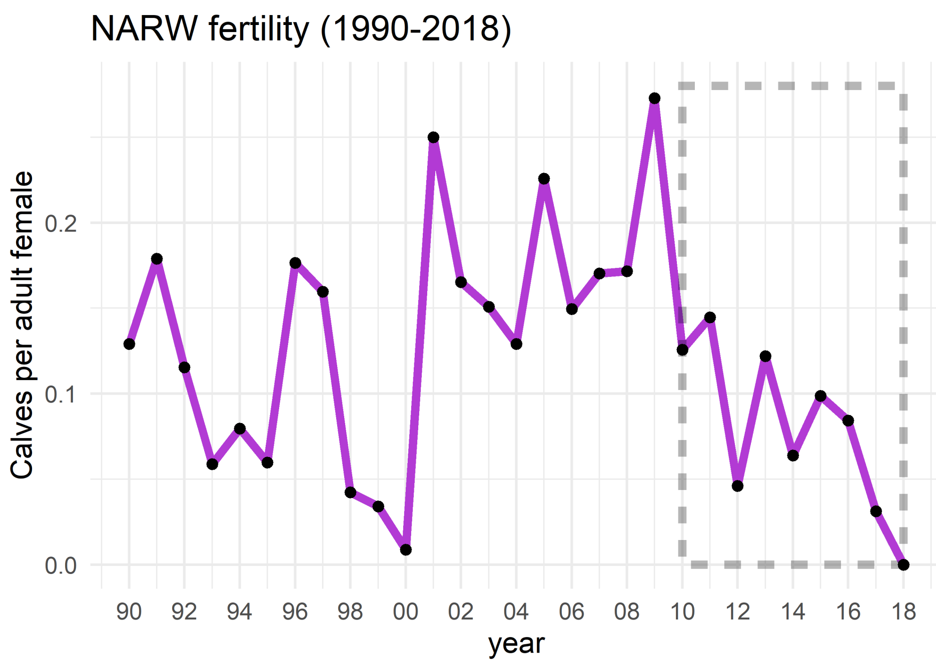

Figure 3: Fertility rate of NARW adult females during 1990–2018. Rates were calculated as the ratio of observed calves to median estimates of adult females (age 5+) from the state-space mark-recapture model. Gray box indicates time period used for main projections.

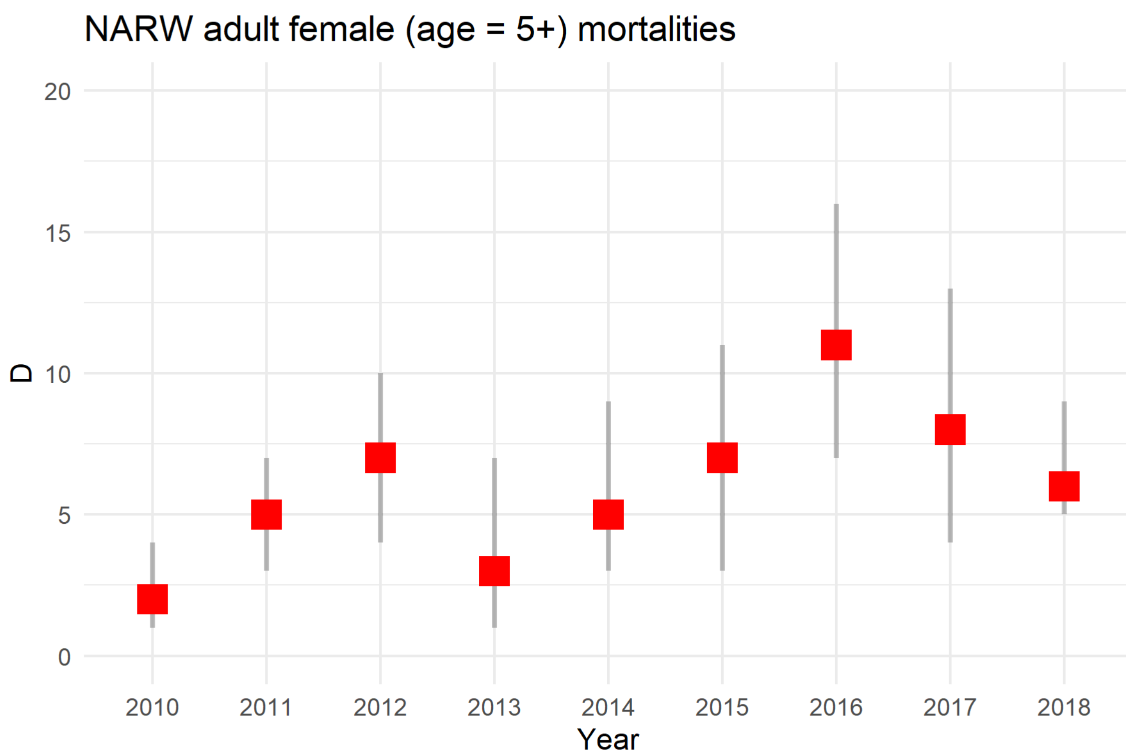

Figure 4: Mortalities of NARW adult females during 2010–2018, as estimated by the state-space mark-recapture model of sightings data. Median estimates with 95% credible intervals included.

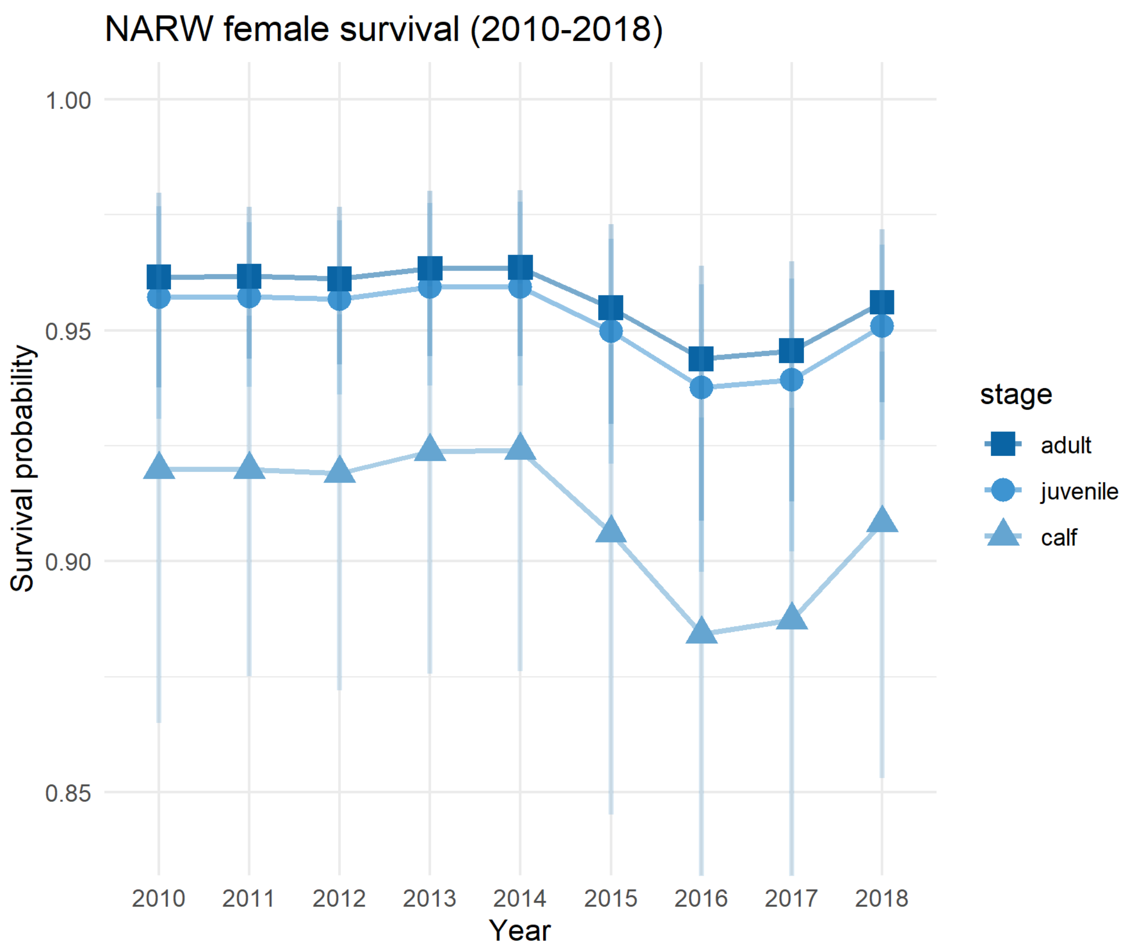

Figure 5: Expected survival probability of NARW adult females during 2010–2018, as estimated by the state-space mark-recapture model of sightings data. Median estimates with 95% credible intervals included. Estimate for a given year (t) represents the probability of surviving from year $t-1$ to t.

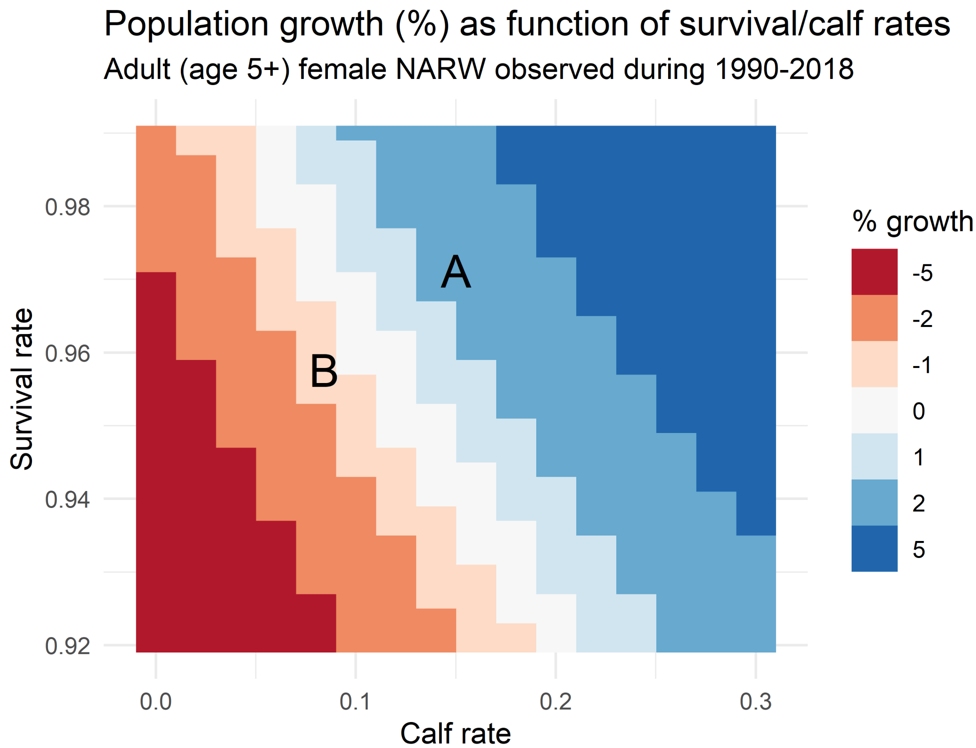

Figure 6: Percent NARW population growth as a function of survival rate and calf rates during 1990–2018. Average values for the pre-2010 (A) and post-2010 (B) regime shift are illustrated.

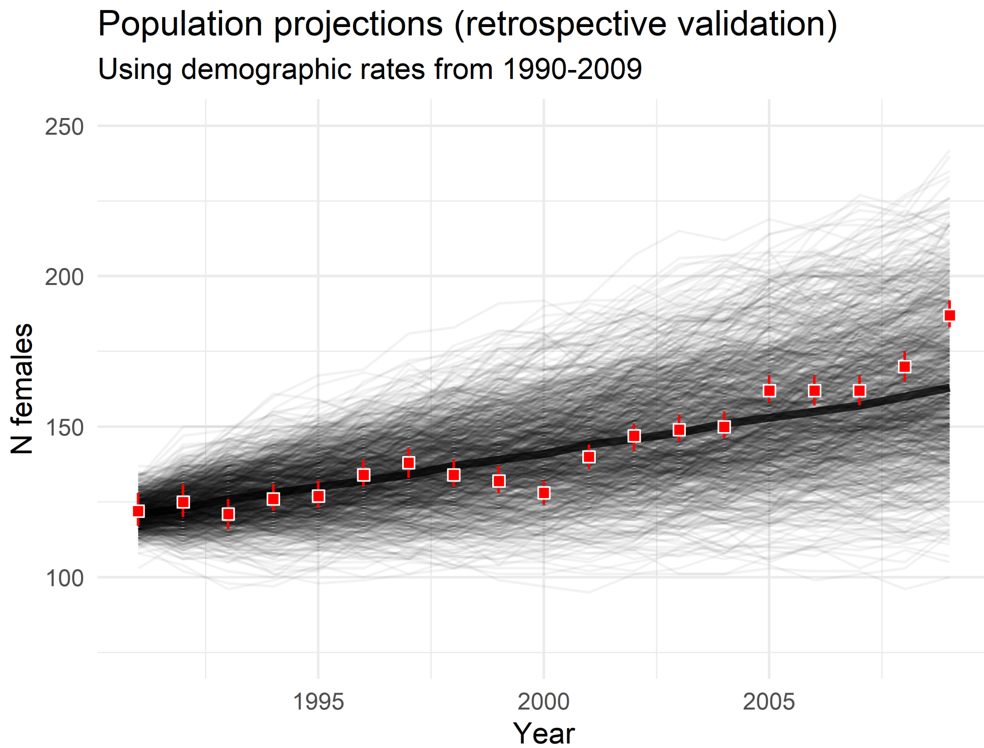

Figure 7: Population projections (n=1,000) of NARW females using demographic rates from 1990–2009, with median trend shaded darker. The red squares represent median population estimates from the Pace et al. 2017 state space model, with 95% credible interals.

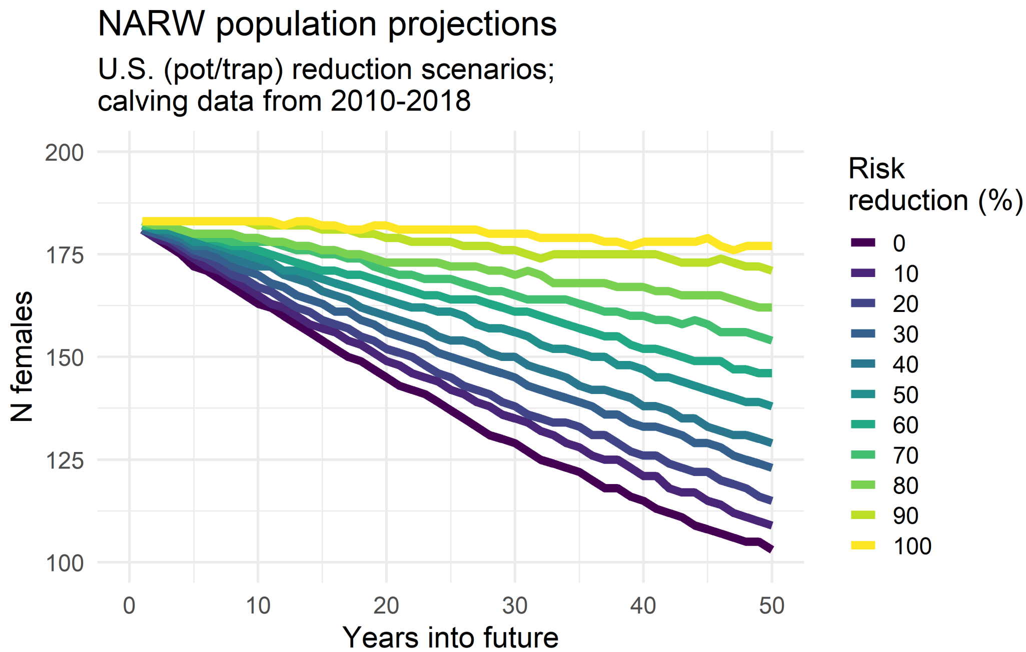

Figure 8: Average trajectory of NARW female population size for each risk reduction scenario using demographic rates from 2010–2018. Each scenario represented a reduction in mortality from U.S. pot/trap gear.

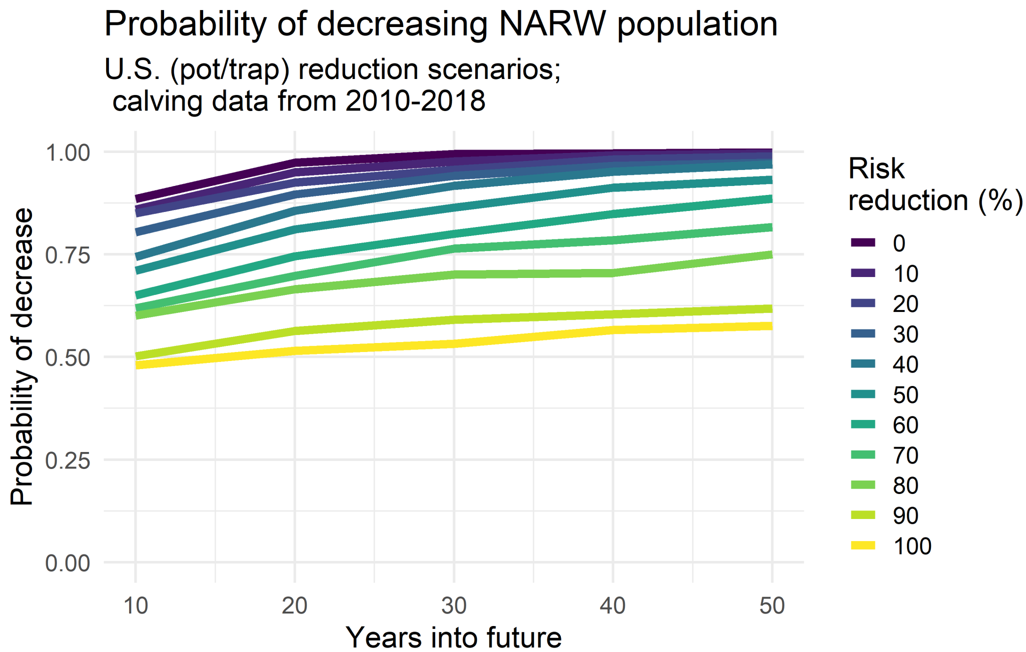

Figure 9: Probability of a decreasing NARW female population size for each risk reduction scenario using demographic rates from 2010–2018. Each scenario represented a reduction in mortality from U.S. pot/trap gear.

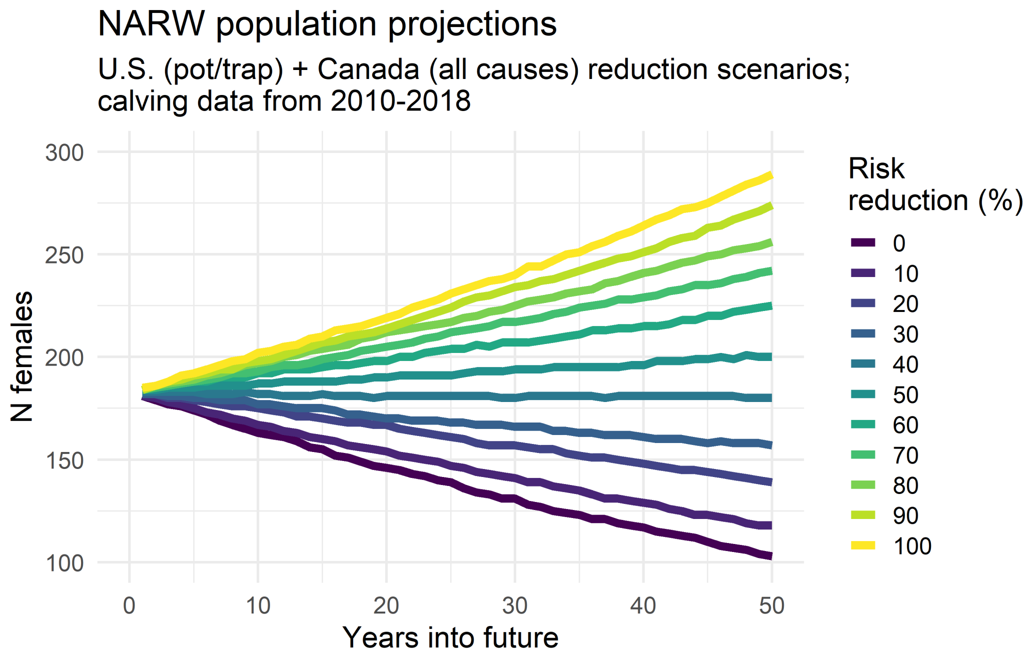

Figure 10: Average trajectory of NARW female population size for each risk reduction scenario using demographic rates from 2010–2018. Each scenario represented a reduction in mortality from U.S. pot/trap gear and all human-related causes in Canada.

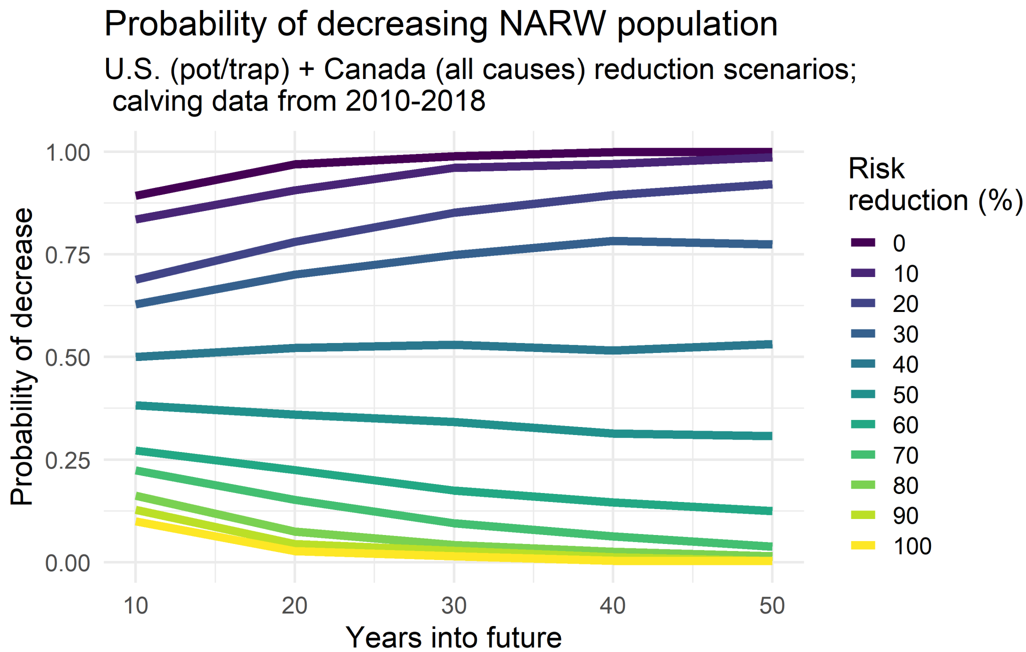

Figure 11: Probability of a decreasing NARW female population size for each risk reduction scenario using demographic rates from 2010–2018. Each scenario represented a reduction in mortality from U.S. pot/trap gear and all human-related causes in Canada.

Supplement 1 - Parameter estimates from the state-space mark-recapture model

Table S1: Medians and 95% credible intervals for major structural parameters of the state-space mark-recapture model for NARW sightings during 1990–2018. Parameter names match those of Pace et al. 2017 and include: the logit-scale parameters of the survival model with an intercept (BetaSex[1]), the effect of adult (age = 5+) females (BetaSex[2]), the linear effect of age (BetaAge) for non-adults, and the effect of the hypothesized regime shift after 2010 (BetaRegime); the annual deviations in survival probability across time (eta[t]); and the population estimates for adult females (NF[6,t]).

	Median	Lower 95%	Upper 95%
BetaSex[1]	2.7797	2.4112	3.1873
BetaSex[2]	-0.5570	-0.8409	-0.2781
BetaAge	0.2690	0.1813	0.3534
BetaRegime	-0.4645	-0.8446	-0.1080
eta[1]	0.0000	0.0000	0.0000
eta[2]	-0.0975	-0.7078	0.3991
eta[3]	0.0796	-0.3928	0.7365
eta[4]	-0.3046	-1.0198	0.1406
eta[5]	0.0298	-0.4704	0.6239
eta[6]	0.0894	-0.3590	0.7201
eta[7]	-0.0125	-0.5378	0.5067
eta[8]	-0.1051	-0.6768	0.3343
eta[9]	0.1026	-0.3290	0.7357
eta[10]	-0.1222	-0.6897	0.3125
eta[11]	-0.0911	-0.6424	0.3598
eta[12]	-0.0010	-0.4935	0.5150
eta[13]	-0.0825	-0.6066	0.3462
eta[14]	-0.0887	-0.6169	0.3471
eta[15]	0.0280	-0.4282	0.5382
eta[16]	0.1801	-0.2305	0.8350
eta[17]	-0.0975	-0.6015	0.3143
eta[18]	-0.0752	-0.5748	0.3449
eta[19]	0.1700	-0.2321	0.7790
eta[20]	0.2080	-0.1825	0.8412
eta[21]	0.1101	-0.2898	0.6593
eta[22]	0.1117	-0.2702	0.6193
eta[23]	0.0990	-0.3112	0.6254
eta[24]	0.1611	-0.2467	0.7753
eta[25]	0.1674	-0.2400	0.7825
eta[26]	-0.0393	-0.5288	0.4530

eta[27]	-0.2681	-0.8624	0.1438
eta[28]	-0.2370	-0.7924	0.1738
eta[29]	-0.0172	-0.4625	0.4191

	Median	Lower 95%	Upper 95%
NF[6,1]	93	88	98
NF[6,2]	95	91	101
NF[6,3]	104	100	109
NF[6,4]	102	98	107
NF[6,5]	113	109	117
NF[6,6]	117	113	122
NF[6,7]	119	115	124
NF[6,8]	119	114	124
NF[6,9]	118	113	123
NF[6,10]	117	113	121
NF[6,11]	114	110	118
NF[6,12]	124	120	128
NF[6,13]	127	123	131
NF[6,14]	126	122	131
NF[6,15]	124	119	128
NF[6,16]	124	120	129
NF[6,17]	127	122	132
NF[6,18]	135	130	140
NF[6,19]	134	130	139
NF[6,20]	143	138	147
NF[6,21]	151	146	156
NF[6,22]	152	147	158
NF[6,23]	152	147	158
NF[6,24]	164	158	171
NF[6,25]	172	166	179
NF[6,26]	172	165	180
NF[6,27]	166	159	174
NF[6,28]	160	152	168
NF[6,29]	157	150	165

Supplement 2 - Population projections for the U.S. pot/gear reduction scenarios

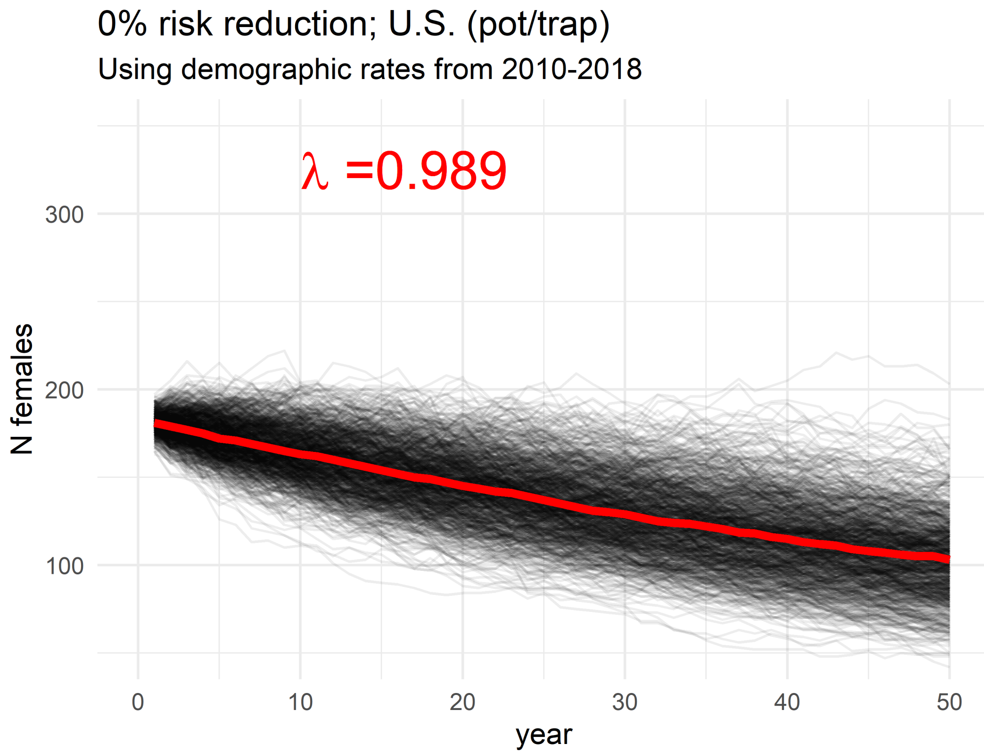

Figure S1: Population projections (n=1,000) of NARW females using demographic rates from 2010–2018. Median population size and resulting growth rate (λ) in red. The risk reduction from U.S. pot/trap gear was 0% (status quo).

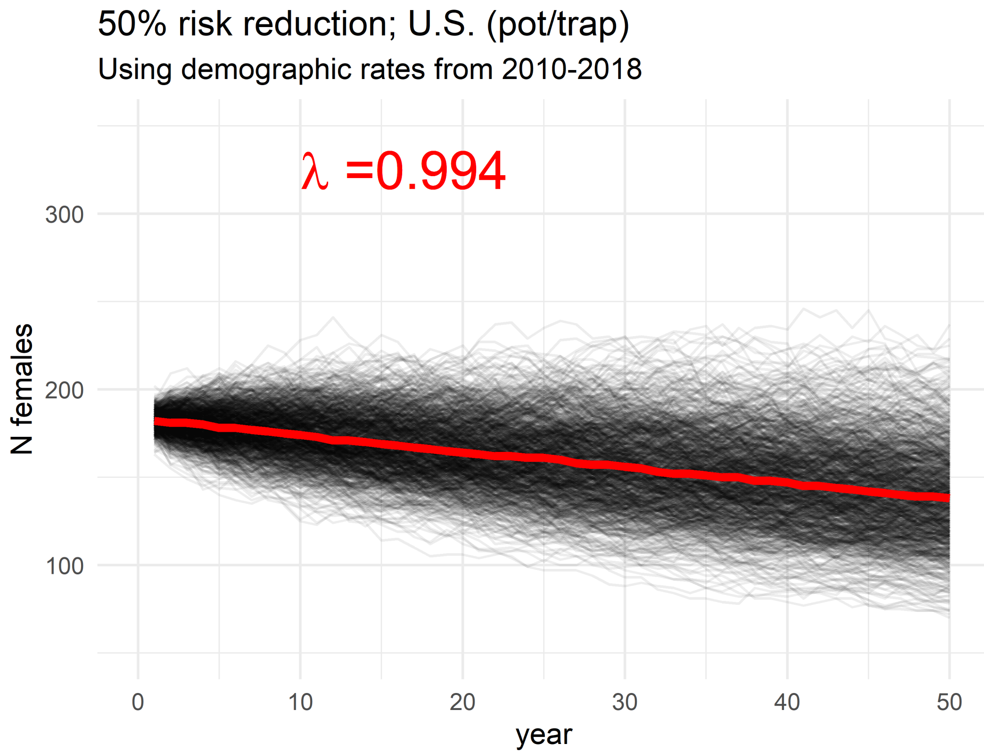

Figure S2: Population projections (n=1,000) of NARW females using demographic rates from 2010–2018. Median population size and resulting growth rate (λ) in red. The risk reduction from U.S. pot/trap gear was 50%.

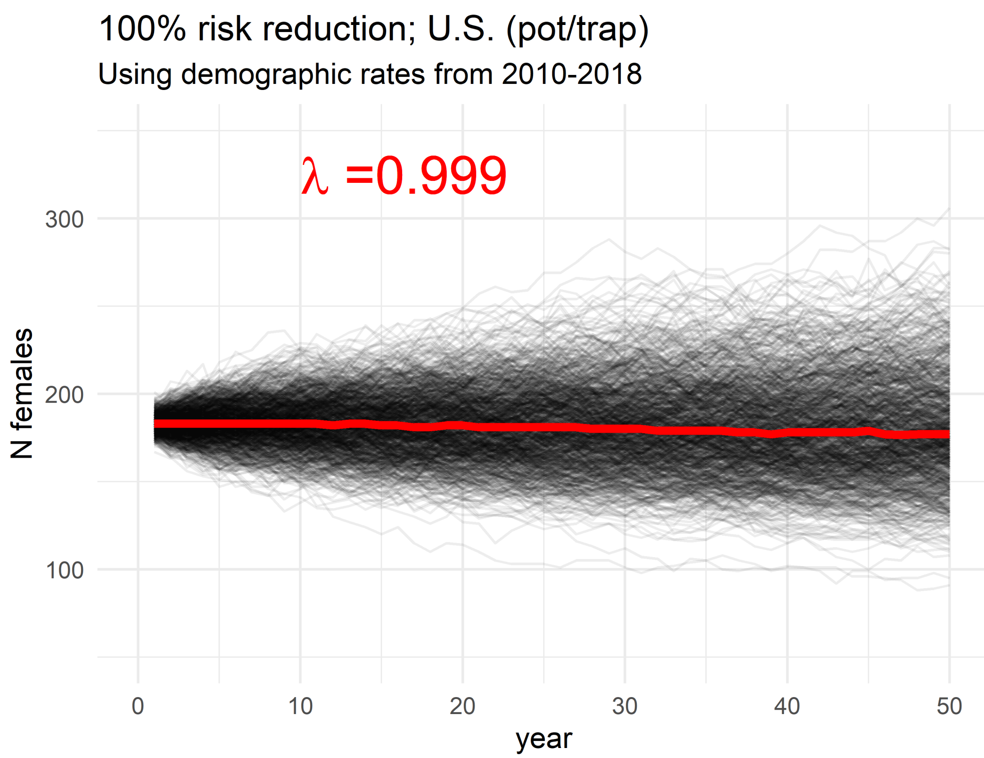

Figure S3: Population projections (n=1,000) of NARW females using demographic rates from 2010–2018. Median population size and resulting growth rate (λ) in red. The risk reduction from U.S. pot/trap gear was 100%.

Supplement 3 - Population projections for the combined U.S./Canada reduction scenarios

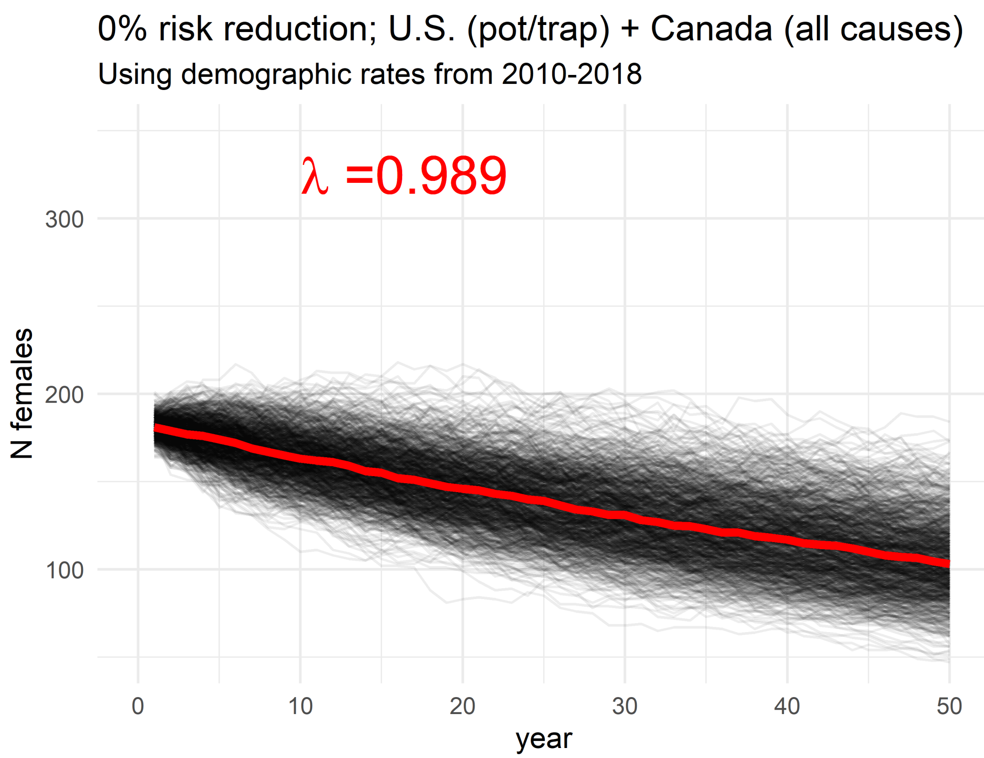

Figure S4: Population projections (n=1,000) of NARW females using demographic rates from 2010–2018. Median population size and resulting growth rate (λ) in red. The risk reduction from U.S. pot/trap gear and all human-related causes in Canada was 0% (status quo).

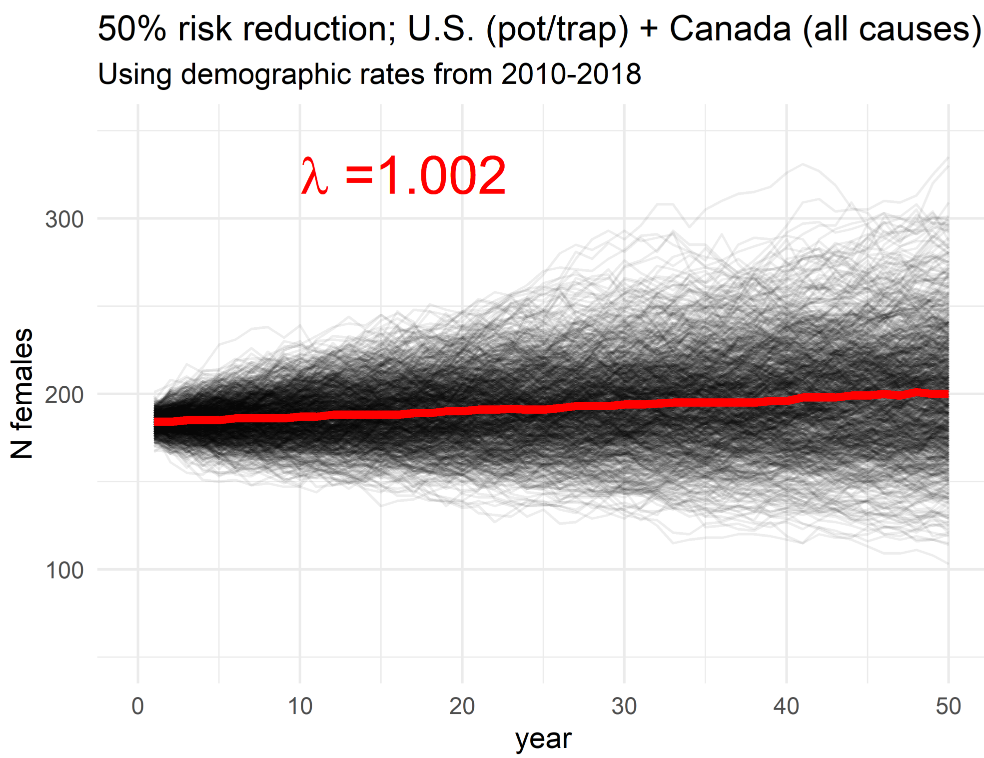

Figure S5: Population projections (n=1,000) of NARW females using demographic rates from 2010–2018. Median population size and resulting growth rate (λ) in red. The risk reduction from U.S. pot/trap gear and all human-related causes in Canada was 50%.

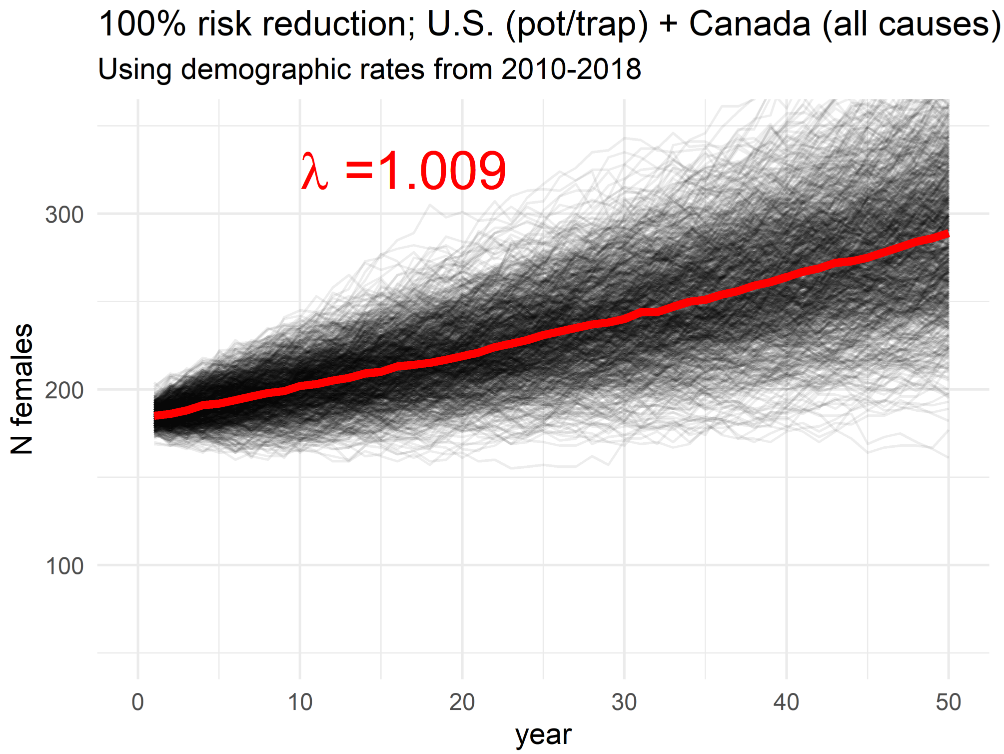

Figure S6: Population projections (n=1,000) of NARW females using demographic rates from 2010–2018. Median population size and resulting growth rate (λ) in red. The risk reduction from U.S. pot/trap gear and all human-related causes in Canada was 100%.

Supplement 4 - Population projections using fertility rates from 1990–2018

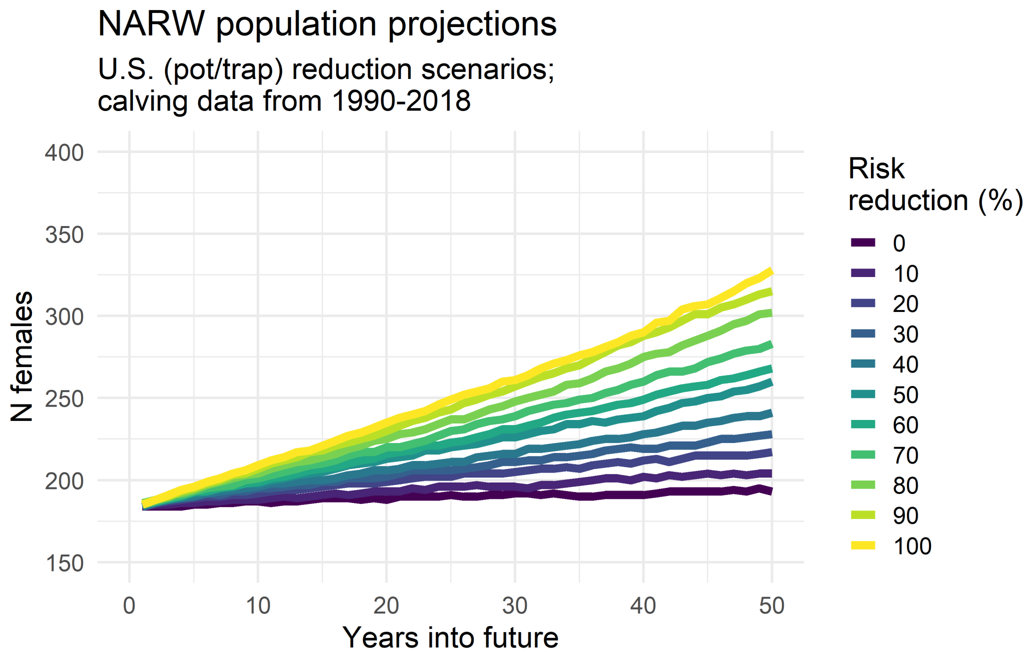

Figure S7: Average trajectory of NARW female population size for each risk reduction scenario using calving rates from 1990–2018. Each scenario represented a reduction in mortality from U.S. pot/trap gear.

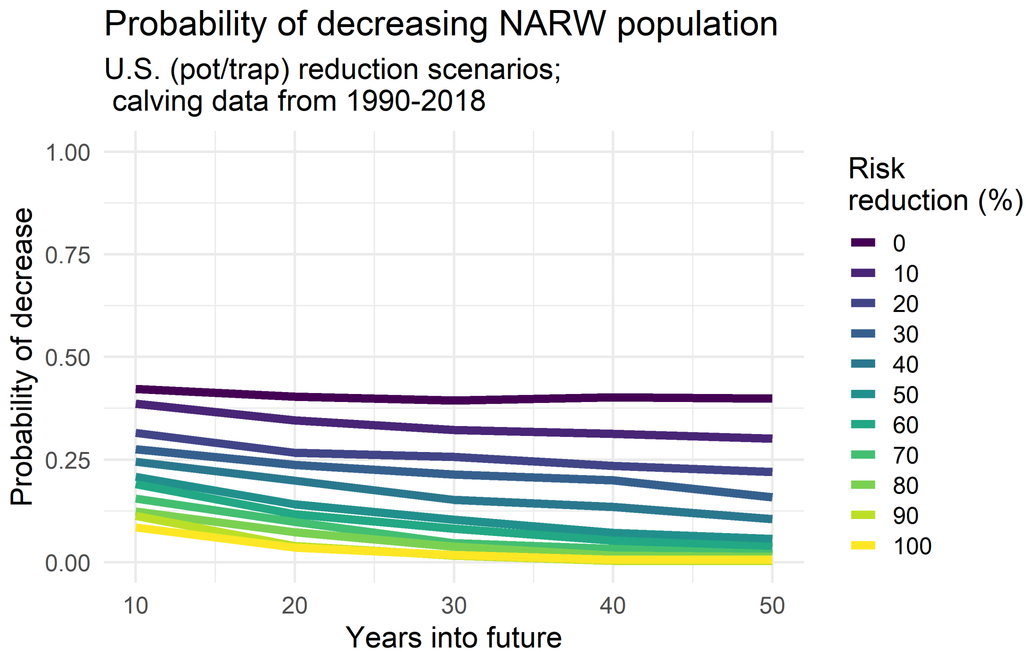

Figure S8: Probability of a decreasing NARW female population size for each risk reduction scenario using calving rates from 1990–2018. Each scenario represented a reduction in mortality from U.S. pot/trap gear.

Supplement 5 - R code for population projections

```
# Script to fit a population projection for NARW
#
# Data use MCMC output from Pace et al. 2017 state
# space models re-run for 1990-2018.  Follows similar
# approach to projections in Corkeron et al. 2018.
#
# https://onlinelibrary.wiley.com/doi/full/10.1002/ece3.3406
# https://royalsocietypublishing.org/doi/10.1098/rsos.180892
#
# D.W.Linden (daniel.linden@noaa.gov)
# 05 December 2020

library(ggplot2)
library(plyr)
library(tidyverse)
library(popbio)

# calf data (1990-2018)
dat.calf <- data.frame(
  year = 1990:2018,
  calves = c(12, 17, 12,  6,  9,  7, 21, 19,  5,  4,
              1, 31, 21, 19, 16, 28, 19, 23, 23, 39,
             19, 22,  7, 20, 11, 17, 14,  5,  0),
  #median estimates of adult NF (5+)
  NF = c( 93,  95, 104, 102, 113, 117, 119, 119, 118, 117,
         114, 124, 127, 126, 124, 124, 127, 135, 134, 143,
         151, 152, 152, 164, 172, 172, 166, 160, 157)
)
dat.calf$birth_rate <- dat.calf$calves/dat.calf$NF

# N live adult females 2009-2017 (100 samples of posterior for each year)
NaF <- structure(
  c(142, 141, 143, 145, 138, 146, 140, 138, 142, 144,
    141, 141, 148, 142, 145, 143, 138, 142, 141, 139, 142, 142, 145,
    141, 143, 141, 144, 144, 143, 145, 141, 141, 143, 143, 144, 142,
    145, 144, 144, 142, 140, 140, 144, 144, 145, 142, 145, 142, 137,
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    163, 162, 161, 164, 161, 162, 160, 168, 162, 160, 158, 163, 157,
    153, 166, 161, 155, 161, 156, 171, 162, 159, 158, 161, 160, 157,
    158, 164, 158, 158, 163, 158), .Dim = c(100L, 9L))


# N dead adult females 2010-2018 (100 samples of posterior for each year)
NdaF <- structure(
  c(3, 4, 3, 2, 3, 3, 3, 2, 2, 2, 2, 3, 2, 1, 3, 3, 3,
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    6, 8, 5, 5, 6, 5, 6, 6, 5, 7, 9, 5, 6, 8, 11, 6, 8, 8, 7, 6,
    5, 5), .Dim = c(100L, 9L))



#---------- Mortality reductions -----------#
reductions <- seq(0,1,by=0.1)

# US mortalities (due to entanglement) 2010-2018
base_morts_US <- 50
# number of "saved" individuals
saved_morts_US <- base_morts_US * reductions
# half of saves are female
saved_morts_US <- saved_morts_US/2

# Canada mortalities (all causes) 2010-2018
base_morts_CAN <- 75
# number of "saved" individuals
saved_morts_CAN <- base_morts_CAN * reductions
# half of saves are female
saved_morts_CAN <- saved_morts_CAN/2

# total saved mortalities
scenarios <- data.frame(
```

```r
  reduction = reductions,
  US_saves = round(saved_morts_US),
  US_Canada_saves = round(saved_morts_US+saved_morts_CAN)
)
n.scenarios <- nrow(scenarios)

fishery.name <- c("US_pot&trap","US&Can_specific")

# "current" age structure (calf, juv, adult)
nzero <- c(4,22,157)

reps <- 20 # number of trajectories/iterations (should be >1,000)
tmax <- 50 # length of the trajectories (years)

##-----------------------------------------------##
## start loops across fisheries & starting years
##-----------------------------------------------##

# U.S. (pots) only, calf rates starting in 2010
fishery <- 1; start <- 2010

#for (fishery in 1:2){             #US vs. US+Canada
#for (start in c(1990,2010)){

# size of posterior distribution
n.iter <- dim(NaF)[1]; n.yrs <- dim(NaF)[2]
# array of realized survival rates (for adult females)
SaF_array <- array(NA,dim=c(n.iter,n.yrs,n.scenarios))

for(scenario in 1:n.scenarios){
    saved <- as.vector(
      t(rmultinom(n.iter,
                  size = scenarios[scenario,fishery+1],
                  # probs are distributed according total deaths in year
                  prob =  apply(NdaF,2,median)/sum(apply(NdaF,2,median)))))
    # remove saved whales from realized deaths
    SaF_array[,,scenario] <- 1-((NdaF-saved)/(NaF))
  }
SaF_array[SaF_array >= 1] <- 0.9999


# save simulation objects
scen.df <- data.frame(fishery=NA,scenario=NA,start=NA,year=NA,N=NA)
tdf.list <- list()
```

```r
##------------------------------------------------##
## loop through scenarios
##------------------------------------------------##
sim.start <- Sys.time()
for(scenario in 1:n.scenarios){

totalpop <- matrix(0, tmax, reps) # initialize matrix to store trajectories

##------------------------------------------------##
## loop through trajectories
##------------------------------------------------##
for(j in 1:reps){

# starting age structure
n.cja <- nzero

print(paste0("scen ",scenario,"; rep ",j))

##------------------------------------------------##
## iterate across years (50) within trajectory
##------------------------------------------------##
for(i in 1:tmax){

# calf survival  (~0.908 for 2010-2018)
S.c <- plogis(rnorm(1,mean=2.3359,sd=0.3457))

# juvenile survival (~0.951 for 2010-2018)
S.j <- plogis(rnorm(1,mean=3.0079,sd=0.3094))

# adult female survival
#S.a <- plogis(rnorm(1,mean=3.1214,sd=0.3097))
S.a <- unlist(SaF_array[,,scenario])[
  sample(1:length(unlist(SaF_array[,,scenario])),size=1)]

#juvenile state duration 4 years in Pace et al. 2017
d.j <- 4

#taking adult female longevity at 69 given oldest known
d.a <- 59+4

#Pi={[1-(p^(d-1))]/[1-(p^d)]}*p
Pj<-((1-(S.j^(d.j-1)))/(1-(S.j^d.j)))*S.j
Pa<-((1-(S.a^(d.a-1)))/(1-(S.a^d.a)))*S.a
```

```
#Gi=[p^d*(1-p)]/[1-(p^d)]
Gj<-((S.j^d.j)*(1-S.j))/(1-(S.j^d.j))

ppm <- matrix(0,3,3)
ppm[2,1]<-S.c
ppm[2,2]<-Pj  #S_j,j
ppm[3,2]<-Gj  #S_j,a
ppm[3,3]<-Pa  #S_a,a

# calves per year average (for start:2018)
calf_rates <- unlist(dat.calf %>% filter(year %in% c(start:2018)) %>%
                     select(birth_rate))
calf_rate_i <- calf_rates[sample(1:length(calf_rates),size=1)]
# half of calves are F (1/2)
ppm[1,3]<- 1/2 * calf_rate_i

stages<-c('calf','immat','adlt')
ppm<-matrix(ppm,nrow=3,byrow=FALSE,
            dimnames = list(stages,stages))
ppm<-round(ppm,5)

#SURVIVAL transitions only
survm<-splitA(ppm)$T
#fertilities
fertm<-splitA(ppm)$F

##########################################
#run a stochastic projection
n.cja <- multiresultm(n.cja,survm,fertm)
# i = year, j = sim
totalpop[i,j]<-sum(n.cja)
  }
}

tdf <- as.data.frame(totalpop)
names(tdf) <- paste0("iter",1:reps)

tdf$year <- 1:tmax
tdf_long <- tdf %>% pivot_longer(cols=starts_with("iter"),
                                 names_to="iteration",
                                 names_prefix="iter",values_to="estimate")

scen.df <- rbind(scen.df,
                 data.frame(
```

```r
                    fishery=fishery.name[fishery],
                    scenario=scenarios$reduction[scenario],
                    start=start,
                    year=1:50,
                    N=apply(tdf,1,median)))

tdf.list[[scenario]] <- tdf_long
}
print(Sys.time()-sim.start)

# save the simulated projections
save(tdf.list,scen.df,file=paste0("NARW_50yr_project_",
     fishery.name[fishery],"_reduction_calving",start,"-2018.Rdata"))

#} #end starting yr loop
#} #end fishery loop



##-------------------------------------------------##
## Plotting results
##-------------------------------------------------##

# U.S. (pots) only, calf rates starting in 2010
fishery <- 1; start <- 2010

#for (fishery in 1:2){             #US vs. US+Canada
#for (start in c(1990,2010)){

load(file=paste0("NARW_50yr_project_",fishery.name[fishery],
                 "_reduction_calving",start,"-2018.Rdata"))

lambdas <-
  ((scen.df %>% filter(year==50) %>% group_by(scenario))$N/
  (scen.df %>% filter(year==1) %>% group_by(scenario))$N)^(1/50)

for(scenario in 1:n.scenarios){

tdf.list[[scenario]] %>%
  ggplot(aes(x=year,y=estimate,group=iteration)) +
  geom_line(alpha=0.07,col="black") + theme_minimal() +
  geom_line(data=data.frame(
    year=1:tmax,
    estimate=as.vector(tdf.list[[scenario]] %>%
```

```
                          group_by(year) %>%
                          summarise(estimate=median(estimate)))[,2],
                             iteration=0),
             aes(x=year,y=estimate),col="red",lwd=1.5) +
  coord_cartesian(xlim=c(0,50),ylim=c(50,350))+
  annotate(geom = "text",x=10,y=325, size=7,color="red",hjust=0,
            label= substitute(paste(lambda, "=", v),
                              list(v=round(lambdas[scenario],3)))
  )+
  labs(title=paste0(scenarios$reduction[scenario]*100,"% reduction",
                    c("; U.S. only","; U.S. + Canada")[fishery]),
       y="N adult females",
       subtitle=paste0("Using demographic rates from ",start,"-2018"))
ggsave(filename=paste0("NARW_proj_",start,"-2018_",fishery.name[fishery],
                       "_",scenarios$reduction[scenario],".png"),
       dpi=600,width=5.25,height=4)
}


##-----------------------------------------------##
# average projections across all scenarios

scen.df$scenario2 <- as.factor(scen.df$scenario*100)

scen.df %>% filter(!is.na(start)) %>%
  ggplot(aes(x=year,y=N,color=scenario2)) +
  geom_line(lwd=1.5,alpha=1) + theme_minimal() +
  scale_color_viridis_d()+
  coord_cartesian(xlim=c(0,50),
                  ylim=data.frame(
                    y1990=c(150,400),
                    y2010=c(100,300))[,match(start,c(1990,2010))]) +
  labs(color="Risk reduction (%)",title="NARW population projections",
       subtitle=paste0(
         c("U.S. only", "U.S. + Canada")[fishery],
         " reduction scenarios;\n calving data from ",start,"-2018"),
       x="Years into future",y="N adult females")
ggsave(filename=paste0("NARW_proj_",start,"-2018_",
                       fishery.name[fishery],"_ALLscenarios.png"),
       width=5.5,height=3.5,dpi=600)


##-----------------------------------------------##
# probabilities of population decline
```

```r
names(tdf.list) <- scenarios$reduction*100
tdf.df <- ldply(tdf.list,.id="Reduction")

tdf_10yr <- tdf.df %>% filter (year %in% seq(10,50,by=10)) %>%
  mutate(decrease = as.numeric(estimate < sum(nzero))) %>%
  group_by(Reduction,year) %>% summarise(prop_decrease=mean(decrease))

tdf_10yr %>%
  ggplot(aes(x=year,y=prop_decrease,color=factor(Reduction)))+
  geom_line(lwd=1.5,alpha=1) + theme_minimal() +
  scale_color_viridis_d()+
  coord_cartesian(xlim=c(10,50),ylim=c(0,1)) +
  labs(color="Risk reduction (%)",
       title="Probability of decreasing NARW population",
       subtitle=paste0(
         c("U.S. only", "U.S. + Canada ")[fishery],
         " reduction scenarios;\n calving data from ",start,"-2018"),
       y="Probability of decrease",x="Years into future")
ggsave(filename=paste0("NARW_proj_",start,"-2018_",
                       fishery.name[fishery],
                       "_ALLscenarios","_declineprob.png"),
       width=5.5,height=3.5,dpi=600)

#} #end starting yr loop
#} #end fishery loop
```

16. APPENDIX D: ANALYSIS OF ATLANTIC SEA SCALLOP FISHERY IMPACTS ON THE NORTH ATLANTIC POPULATION OF LOGGERHEAD SEA TURTLES

Analysis of Atlantic Sea Scallop (*Placopecten magellanicus*) Fishery Impacts on the North Atlantic Population of Loggerhead Sea Turtles (*Caretta caretta*)

U. S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, MA
February 2008

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Analysis of Atlantic Sea Scallop (*Placopecten magellanicus*) Fishery Impacts on the North Atlantic Population of Loggerhead Sea Turtles (*Caretta caretta*)

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ABSTRACT

An estimated 619 loggerhead turtles of various age and sex classes were taken annually during 1989-2005 in all components of the US Atlantic sea scallop (*Placopecten magellanicus*) fishery. We provide here a quantitative assessment of the potential for these takes to jeopardize the continued existence of the US Atlantic Ocean population of loggerhead sea turtles (*Caretta caretta*). A population viability analysis (PVA) was used to estimate quasi-extinction likelihoods under conditions with and without fishery effects. This PVA used US index nesting beach data for 1989-2005 to estimate the loggerhead population trend μ (mean growth rate) and variance σ^2. The starting population (N_0) for the exercise was the sum of nesting females estimated from the 2005 nest count in the North Carolina to Florida area. The base model (with fishery bycatch) was developed by using estimates of μ (-0.022), σ^2 (0.012), N_0 (34,881) and a quasi-extinction threshold of 250 adult females. Quasi-extinction likelihoods were bootstrapped (1000 iterations) under baseline conditions to derive confidence intervals. The μ for each bootstrap iteration was drawn from a normally distributed random sampling of μ values lying within the 95% confidence interval around the original μ. The model was then rerun with the estimated annual fishery mortality of adult females (102 turtles) added back into the population, thus changing the trend (μ = -0.019, σ^2 = 0.012, and N_0 = 34,881). Results of the two models were similar; the quasi-extinction probabilities were zero at 25, 50, and 75 years, and 0.01 at 100 years for both analyses. Median times to quasi-extinction were 207 years versus 240 years, and the number of bootstrap simulations with extinction probabilities greater than 0.05 in 100 years was 258 and 178, respectively. These results suggest that the annual take of loggerhead sea turtles in the US fisheries for Atlantic sea scallops, though detectable, does not significantly change the calculated risk of extinction of the population of adult female Western North Atlantic loggerheads over the next 100 years.

INTRODUCTION

Loggerhead sea turtles (*Caretta caretta*) are incidentally captured in US dredge and trawl fisheries for Atlantic sea scallops (*Placopecten magellanicus*) in the US Mid-Atlantic region. Increased federal observer coverage of these fisheries allowed the National Marine Fisheries Service (NMFS) to estimate the annual bycatch of loggerhead turtles in the fisheries through 2005 (Murray 2004a, 2004b, 2005, 2007). Recent observer reports document takes through 2007. As loggerhead turtles are a threatened species under the US Endangered Species Act (ESA), NMFS, under Section 7 of the ESA, must ensure that continuation of the sea scallop fisheries is not likely to jeopardize the continued existence of the species.

Impacts of US fisheries (e.g., Atlantic sea scallop, Mid-Atlantic bottom trawl, pelagic longline, and Gulf of Mexico/Southern Atlantic commercial shrimp) on the western North Atlantic loggerhead sea turtle population have been analyzed by Southeast Fisheries Science Center (SEFSC) staff and the loggerhead sea Turtle Expert Working Group (TEWG 1998, 2000; SEFSC 2001; Epperly et al. 2002). However, reduced loggerhead nesting on southeastern US beaches suggests these analyses require updating. The TEWG is currently working on a reanalysis, but the limited data available on current population parameters (e.g., stage specific survival) suggest that the previous demographic models may be difficult to revise.

We provide here an alternative quantitative approach to the assessment of the risk the US Atlantic sea scallop fisheries have of jeopardizing the continued existence of the western North Atlantic Ocean populations of loggerhead sea turtles. This approach is simpler than previously used for western North Atlantic (WNA) loggerheads and is similar to that used by Snover (2005) in her analysis of the impact of the Western Pacific Pelagics Fisheries on several Pacific sea turtle species. We use a population viability analysis (PVA) to estimate quasi-extinction likelihoods under conditions with and without fishery effects. The PVA is count-based (Dennis et al. 1991; Morris et al. 1999; Holmes 2001; Morris and Doak 2002; Snover 2005) which will allow the use of the only relatively complete and available population time series—index nesting beach[1] counts for 1989-2005. As such, the analyses focus on the viability of the adult female portion of the population and should not be considered to model viability of the entire population.

We first present the PVA results under baseline conditions by using the rate of change of the adult female population (which implicitly includes the mortalities from the scallop and other fisheries) and the 2005 count of adult females estimated from all beaches in the Southeast based on an extrapolation from nest counts. We then adjust the rate of change by adding back the fisheries take and rerunning the PVA. The results of these two analyses are then compared by using the probability of quasi-extinction at 100 years to assess the impact of the takes in the Atlantic sea scallop fisheries.

At the outset, we point out three caveats to the interpretation of these analyses. First, the current negative nesting beach trends are at odds with some in-water survey results (e.g., Epperly et al. 2007). Secondly, the current negative trend in adult female abundance has likely been

[1] Index beaches are a limited series of beaches which are regularly monitored for nesting activity. In Florida, the Index Nesting Beach Survey (INBS) has coordinated a detailed monitoring program since 1989 to measure seasonal productivity, allowing comparisons between beaches and between years. In Florida, 33 beaches (of 190 surveyed beaches) are included in the INBS program. Similar programs exist in states further north.

influenced by mortality events that have occurred over several decades. As such, a model based on current nesting beach trends may overestimate the effect of current takes on the likelihood of extinction for the population. Finally, we stress that our analyses should not be used to assess the likely fate of the population but should only be used to assess the impact of the fisheries for Atlantic sea scallops on the population trajectory of adult female loggerhead sea turtles. A thorough review of loggerhead population trends is provided by Witherington et al. (2006, in review).

METHODS

Data

Population trend data

A time series of population counts (or some index of the population) was needed through 2005 to estimate the population trend for the PVA. The time series needed to be longer than 10 years for the PVA to be more than marginally useful (Morris et al. 1999; Morris and Doak 2002).

Loggerhead nest counts (a proxy for the adult female population) are available for southeastern US index nesting beaches from 1989 to 2005 for the Northern (NC, SC, and GA) and Peninsular Florida subpopulations (NMFS in review, FWRI 2007). These are the subpopulations with the greatest nesting populations. Two other southeastern United States subpopulations have index beach nest counts available from 1996 (Dry Tortugas FL) and 1998 (Northern Gulf [AL, FL]) onwards (NMFS in review). These are the two smallest subpopulations, and since at least 1996 they have constituted a small fraction of the population (e.g., in 2005 they accounted for only 3% of the total number of index beach nests). Because nest counts were available for only a relatively brief period, these two subpopulations were excluded from the trend analysis for 1989-2005. Note that we did include the nest counts for all four subpopulations as part of a supporting analysis for the 1996-2005 period. Finally, these count data were used directly, without any adjustments for remigration[2] or nests per female, to determine the population trend.

Current abundance data

An estimate of adult female abundance in 2005 was necessary for use as the starting point for the PVA. The 2005 estimate of adult female abundance was derived by first summing nest counts from all beaches surveyed in the southeastern United States, including all beaches surveyed in 2005 in NC, SC, GA, FL, and AL (NMFS in review, FWRI 2007, SCDNR 2007). Only index beach nests counts were available for the Dry Tortugas and Northern Gulf subpopulations, so the total nest count is biased low. We then adjusted the sum to estimate adult females:

$$N_{AF} = (\text{Number of nests/Nests per female}) * \text{Remigration interval}$$

[2] Remigration is used here to mean the number of years between visits by adult females to nesting beaches and is not to be confused with the repeat visits within a single year which are included in the nests per female estimate.

Use of a constant value for nests per female and remigration interval is problematic as both parameters vary to some degree. For example, limited food resources can lead to decreased reproductive fitness because of natural and human driven fluctuations in prey availability. Moreover, if the age structure of the population changes, the number of nests per female will change. The available datasets do not characterize this variability, nor is it known whether such variability is random or associated with environmental change. Because of these uncertainties, we generally used conservative parameter values.

Estimates of nests per female vary widely, in part because of observational issues. Estimates adjusted for missed nesting suggest the mean number of nests per female per season in US waters ranges from 2.8 to 4.2 (Frazer and Richardson 1985; Schroeder et al. 2003). We used 4.2 nests per female.

Published estimates for the average remigration intervals of WNA loggerhead sea turtles on US beaches vary from 2.5 to 2.7 years (Richardson et al. 1978; Bjorndal et al. 1983; Schroeder et al. 2003). We used the 2.5 year remigration estimate.

Fishery mortality data

Estimates of loggerhead bycatch in the US Atlantic sea scallop fisheries are available for 2003-2005 for scallop dredge gear and for 2004-2005 for scallop trawl gear (Murray 2004a, 2004b, 2005, 2007). There is a wide range amongst the annual values, and two approaches for deriving an estimate for our model were considered. One approach was based on using the mean annual sea scallop dredge fishery bycatch for 2003-2005 ([749+180+0]/3=310; Murray 2004b, 2007) added to the midpoint of the range of estimated sea scallop trawl fishery bycatch from six bycatch estimates for 2004-2005 (136 turtles; Murray 2007) as the estimate of average annual total loggerhead sea turtles caught in the sea scallop fisheries (446 turtles). An additional 20 loggerheads were estimated to have been caught in groundfish bottom trawl fisheries where sea scallops were the primary catch (Murray 2006). Summing across fisheries suggests that the annual loggerhead bycatch in sea scallop related fisheries in 2004-2005 might be 466 animals.

The second approach used the take estimates in the Atlantic Sea Scallop Fishery Management Plan (FMP) Biological Opinion. This included only the 2003-2004 sea scallop dredge fishery bycatch (biennially 929 loggerhead sea turtles) added to one of the sea scallop trawl fishery bycatch estimates (268 loggerhead sea turtles biennially) and the 20 turtles estimated to be taken annually in groundfish bottom trawls for an average annual bycatch of 619 loggerhead sea turtles in the fishery.

We used the value of 619 loggerhead sea turtles as our estimate of the annual bycatch in the sea scallop fisheries of loggerhead sea turtles of various age and sex classes.

This <u>total</u> loggerhead sea turtle bycatch estimate (N_B=619 turtles) then needed to be adjusted downward to estimate the annual mortality of <u>adult female</u> loggerheads (N_{AF}) associated with the US sea scallop fisheries:

$$N_{AF} = (N_B * F_{US} * F_M * F_{M-F} * F_L) + (N_B * F_{US} * [1-F_M] * F_{IM-F} * F_{IM-R} * F_L)$$

where:

F_{US} = proportion of the bycatch from the US population

F_M = proportion of bycatch mature

F_{M-F} = proportion of the adult bycatch assumed to be female

F_{IM-F} = proportion of the immature bycatch assumed to be female

F_{IM-R} = relative reproductive value of juvenile neritic turtles

F_L = proportion of the bycatch considered as lethal takes

Again, where there was a range of parameter values, we selected the value that generated the greatest impact by the sea scallop fisheries on the loggerhead population:

1. F_{US} - Genetic samples taken from loggerhead sea turtles captured in the sea scallop fisheries indicated that 88-93% of the animals are from the US nesting population (Haas et al. in review). This is comparable to the ~92% reported by Bass et al. (2004) for the Albemarle-Pamlico Sounds area of NC. We used a value of 93%.
2. F_M - Loggerheads captured in both gear types are expected to be of the same age classes. Loggerhead sea turtles observed bycaught in sea scallop fisheries ranged in size from 62 cm to 107 cm curved carapace length (CCL)(mean = 79.2 cm CCL, SD = 11.6, NE Fishery Observer Program database). The cutoff between sexually immature and mature loggerhead sea turtles appears is in the range of 87 to 100 cm CCL (NMFS in review; SEFSC 2001). CCL data were available for 42 turtles taken in the fishery; 35 (83.3%) were less than 87 cm CCL. As such, we used 0.833 as the proportion of immatures taken in the fisheries.
3. F_{M-F} and F_{IM-F} – There are few data available on the sex classes of loggerheads bycaught in the sea scallop fisheries. We, therefore, used data available from loggerhead captures and strandings. These data suggest that the mature and immature sex ratio in Northeast waters is approximately two females per male (TEWG 2000).
4. F_{IM-R} – Estimated bycatch of immature loggerheads was adjusted to account for the natural mortality expected prior to their recruitment as breeding adults. Wallace et al. (in press) present estimates in the range of 0.28 to 0.32 for the relative reproductive value of the neritic juvenile stage of loggerhead sea turtles found stranded along the US Atlantic coast (mean CCL = 78.5, SD = 16.6). Given the similarity in size of these loggerheads to those taken in the sea scallop fishery (mean CCL = 79.2, SD = 11.6), it appears reasonable to use this estimation of reproductive value for immature juvenile turtles taken in the sea scallop fishery. We, therefore, used 0.32 as the estimate for juvenile reproductive value.
5. F_L - Observer reports from the 2003-2005 fisheries suggest that the percentage of loggerhead sea turtles released alive and uninjured was 22.7-25% for scallop dredge gear and 100% for trawl gear (Murray 2004a, 2004b, 2005, 2007). This compares to the 36% and 88.5% used in the Atlantic Sea Scallop FMP Biological Opinion. We, therefore, used 0.227 and 0.885 for dredge and trawl gear, respectively.

Because of the differences in loggerhead captures in the trawl and dredge fisheries, the number of adult female mortalities was estimated separately for each fishery and then combined.

Together this series of adjustments provides an estimate of the annual mortality (in numbers) of US adult female loggerheads caused by the bycatch in the US Atlantic sea scallop fisheries.

Model

The Dennis Model is a density-independent model of population growth, which uses a diffusion approximation to compute the probability of quasi-extinction (i.e., reaching a low threshold population size) in a randomly varying environment:

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

Application of the model requires that two key parameter values be estimated to make inferences regarding population growth rates and quasi-extinction risks:

μ – the arithmetic mean of the log population growth rate
σ^2 – variance of the log population growth rate

Holmes (2001) suggests the use of running sums as a means of reducing bias associated with sampling error and stage-specific counts. We calculated running sums as:

$$R_j = N_i + N_{i+1}$$

where j=1,2,3 … (q-1), q is the number of censuses in dataset, N represents the population size, and R_j represents the population size at time j from the running sums. Without using the running sums approach (1 yr intervals), the trend was -0.0063 and the variance was 0.038. We evaluated running sums of 2 yr, 3 yr, and 4 yr to calculate the annual estimate of R_j and found that the 3 and 4 yr running sums produced the same rate of change (-0.0216), which was slightly different from the 2 yr interval (-0.0220). With the smaller variance in the trend for the 3 and 4 yr running sums (0.006 and 0.003, respectively), the result would be that a 3 or 4 yr interval would lead to reduced probabilities of quasi-extinction in 100 yrs. Following our rule of using conservative parameter values, we decided to use a 2 yr interval for the final analysis.

Then μ was calculated as:

$$\mu = (\sum log(R_{j+1}/R_j\,)/t$$

Similarly, σ^2 is calculated as the variance over the series of log (R_{i+1}/R_i) values. The μ and σ^2 are then used to estimate r (the instantaneous rate of change) and λ (Dennis et al. 1991):

$$r = \mu + \sigma^2/2$$
$$\lambda = e^{(r)}$$

Estimation of the extinction risk requires a population size at extinction (N_{ext}). The population size at extinction can assume several values, with 0 equal to the true extinction. Rather then focusing entirely on total extinction ($N_{ext} = 0$), the concept of quasi-extinction risk has been developed (Ginzburg et al. 1982), where quasi-extinction risk is the probability that a population will fall below a given threshold ($N_{ext} > 0$). There is no generally agreed upon level for quasi-extinction, though it is commonly considered to be a threshold population size below which the population would be critically endangered or effectively extinct. For large vertebrates, a variety of numerical values have been considered for this threshold (e.g., from 20 to 500). We considered using either 50 or 250 adult females as our estimate of quasi-extinction. Our reasons for considering fifty animals were: (1) there is general consensus in the conservation genetics community that large vertebrate populations cannot fall below 50 breeding animals and still maintain genetic integrity (Shaffer 1981; Franklin 1980), (2) the International Union for Conservation of Nature (IUCN)(2008) considers this to be one of the two threshold numerical values for a "critically endangered" population category, and (3) to provide comparability with the value used in the 2004 Pacific sea turtle bycatch PVA prepared by Snover (2005). IUCN uses 250 mature animals as an alternative threshold value for "critically endangered" populations when there is evidence of a population decline. Given the apparent decline in nesting in the southeastern United States, it appears reasonable to use 250 as our threshold value for quasi-extinction. The IUCN includes all mature animals in this value and not just adult females, so using 250 adult females as the threshold provides a doubly conservative threshold.

Morris and Doak (2002) describe the probability of reaching a quasi-extinction threshold (N_{ext}) by using the following function:

$$g(t|\,\mu, \sigma^2, d) = \frac{d}{\sqrt{2\pi\sigma^2 t^3}} \exp\left[\frac{-(d+\mu t)^2}{2\sigma^2 t}\right]$$

with d = $\log(N_0/N_{ext})$, and N_0 is the population size at the beginning of the analysis period. To calculate the total probability of reaching N_{ext} at some future time T, the cumulative distribution function (which is the preceding function integrated from t = 0 to T) is applied:

$$G(T|\,\mu, \sigma^2, d) = \exp\left[\frac{-2\hat{\mu} d}{\hat{\sigma}^2}\right]\Phi\left[\frac{-d+\hat{\mu}T}{\sqrt{\hat{\sigma}^2 T}}\right] + \Phi\left[\frac{-d-\hat{\mu}T}{\sqrt{\hat{\sigma}^2 T}}\right]$$

where $\Phi(z)$ is the standard normal cumulative distribution function (Morris and Doak 2002).

Morris and Doak (2002) outlined an approach for deriving the quasi-extinction time cumulative distribution function confidence intervals by using bootstrap estimation procedures. We used a similar approach, sampling from a random distribution drawn from within the 95% confidence interval for μ and σ^2 and replicated 1000 times to estimate the confidence intervals around the cumulative probability of reaching N_{ext} at some future time T.

Modeling Steps

The base model (with fisheries bycatch) was run over a 1,000 yr period with the estimates of μ, σ^2, N_0 beginning in 2005 and quasi-extinction threshold of 250 adult female loggerheads (Dennis et al. 1991; Holmes 2001; Morris and Doak 2002; Snover 2005). The 1,000 year time horizon was necessary so that we could determine the median time to extinction. Quasi-extinction likelihoods were then bootstrapped under baseline conditions to derive confidence intervals. The μ for each bootstrap iteration was drawn from a normally distributed random sampling of μ values lying within the 95% confidence interval around the original μ.

The model was modified to add back in the annual loggerhead bycatch in the Atlantic sea scallop fisheries. First, we adjusted the annual estimated bycatch in the fisheries (dredge and trawl) of loggerhead sea turtles for all age and sex classes to derive an estimate of total adult females removed from the population. We then calculated the rate of adult female removals for 2005 by dividing the bycatch by the total adult female population in 2005. This rate was then added into the population instantaneous growth rate (r) for each year from 1989 to 2005, and a revised μ and σ^2 was calculated. The model (without fishery bycatch) was then run with the revised estimates of μ, σ^2, and N_0. We bootstrapped quasi-extinction likelihoods under the new model's conditions to derive confidence intervals.

Evaluation of Results

The primary metric we used to compare the results of the two PVAs (with and without the fishery mortalities) was the cumulative probability of quasi-extinction at 100 years (based on recommendations on acceptable risk of extinction in DeMaster et al. 2004). Secondary metrics included the number of bootstrap replicates with a probability of extinction > 0.05 in 100 years and the median times to extinction[3]. We analyzed the sensitivity of the 1989-2005 model to changes in the population trend by comparison with the trend from 1996-2005. We also compared extinction probabilities at take levels that were two and ten times the documented levels of takes in the sea scallop fisheries.

RESULTS

Population Trends to Present

Loggerhead nest counts from the Northern and Peninsular subpopulations were summed (Fig. 1) and analyzed to develop the annual rates (λ) of population change for 1989-2005 (Table 1). The trend ($\mu = -0.022$, $\sigma^2 = 0.012$, Table 2) for 1989-2005 for the US Atlantic Ocean loggerhead adult female population suggests the adult female population is declining.

We used an estimate of 58,602[4] nests in 2005 in the southeastern United States (North Carolina to Alabama). This produced an estimate of 34,881 adult females when adjusted for nests per female (4.2 nests per female) and remigration interval (2.5 years).

The annual sea scallop fisheries bycatch mortality of adult female loggerheads was estimated to be 102 turtles (97 in the dredge fishery and 5 in the trawl fisheries). This estimate was derived from the total annual take of 619 loggerheads prorated for area of origin (0.930 from United States), maturity (0.833 immature), female proportion (0.67), reproductive value of juveniles (0.32), and fishery specific mortality (dredge = 0.773 and trawl = 0.115).

Given the 2005 population estimate of 34,881 adult females and a fishery-induced mortality of 102 adult females per year, the rate of adult female removals in the sea scallop

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[3] The time when the quasi-extinction probability is 0.50

[4] This includes 2005 counts for <u>all</u> beaches in the Northern (NC = 560, SC = 4,233, GA = 1,145 nests) and Peninsular Florida (51,636 nests) subpopulations and <u>index</u> beaches in the Dry Tortugas (159 nests) and Northern Gulf (869 nests) subpopulations (NMFS in review; FWRI 2007; SCDNR 2007).

fishery was 0.0029 in 2005. These mortalities were added back into the population to produce a revised 1989-2005 μ of -0.019 (σ^2 = 0.012, Table 2).

Viability Analyses

Using the 1989-2005 model, the risk of quasi-extinction (N_{ext} = 250 adult females) at 100 years was 0.01 (Table 2, Fig. 2) with a median time to extinction of 207 years (Table 2). Over 1000 iterations of the model, 258 produced a probability of extinction at 100 years greater than 0.05.

Adding the Atlantic sea scallop fisheries-related loggerhead mortalities back into the population had only a small effect on population trajectory and extinction probabilities. The μ was -0.022 and -0.019 for the analyses with and without the fishery takes. The risk of quasi-extinction at 100 years remained 0.01 (Table 2, Fig. 3). The median time to extinction grew to 240 years (Table 2). Over 1000 iterations of the model, 178 produced a probability of extinction at 100 years greater than 0.05.

Results of the two analyses were similar (Table 2, Fig. 4). Both had quasi-extinction probabilities of zero (0) at 25, 50, and 75 and a probability of 0.01 at 100 years. Median times to quasi-extinction were similar (207 years versus 240 years). The number of simulations with extinction probabilities at 100 years greater than 0.05 was 258 and 178, respectively.

Model Sensitivity

An incorrect estimate of the population trend would significantly affect the model results. Therefore, we repeated this analysis with just the 1996-2005 time series. While this would generally be considered to be too short a time series for analysis, it does provide some insight into the capability of the model to detect risk of extinctions.

Loggerhead nest counts from all four subpopulations were summed (Table 3) and analyzed to develop the annual rates (λ) of population change for 1996-2005 (Table 4). The trend (μ = -0.049, σ^2 = 0.011, Table 2) for 1996-2005 for the US Atlantic Ocean loggerhead adult female population suggests even more strongly than the 1989-2005 analysis that the adult female population is declining. Again with the 2005 population estimate of 34,881 adult females and a fishery-induced mortality of 102 adult females per year, the rate of adult female removals in the sea scallop fishery was 0.0029 in 2005. These mortalities were added back into the population to produce a revised 1996-2005 μ of -0.046 (σ^2 = 0.011, Table 4).

There was little difference between the 1996-2005 analyses with and without the sea scallop fisheries mortalities (Tables 4, Fig. 5). The population trend remains similar; μ equals 0.049 and 0.046 for the two analyses. Cumulative probabilities of extinction are identical up until approximately the 75th year, and the median times to extinction were very similar for both 1996-2005 models (i.e., 98 versus 102 years). The number of simulations with extinction probabilities at 100 years greater than 0.05 was 940 and 922, respectively.

We also evaluated the model's sensitivity to changes in fishery mortality rates. Given that the 1989-2005 model showed probabilities of extinction at 100 years equal to zero for both the original model and the model with takes added back in, it was necessary to use the 1996-2005 model for this evaluation. We compared the results of adding the loggerhead mortalities caused by the Atlantic sea scallop fisheries (102 adult females) with adding back in mortalities that were two and ten times greater than that observed in the sea scallop fisheries (Fig. 6).

Ultimately, it appears that the probability of extinction at 100 years would be reduced to zero if ten times the number of adult females estimated to be taken by the Atlantic sea scallop fisheries were added back to the population.

DISCUSSION

These results suggest that mortalities of loggerhead sea turtles in the US Atlantic sea scallop dredge and trawl fisheries are detectable but have a relatively small effect on the trajectory of the adult female components of the WNA loggerhead sea turtles over the next 100 years. The 1989-2005 population trends, with and without the mortalities, were not significantly different, and the probability of reaching the quasi-extinction threshold (250 adult females) under both scenarios was 0.01. Median times to extinction for both were greater than 200 years. The only obvious difference was in the number of bootstrap simulations with a probability of extinction > 0.05 in 100 years.

The relatively large population size of adult females (34,881), the relatively small negative trend in the adult female population over 1989-2005 (r = -0.022 per year), and the number of adult female mortalities in the fisheries (102 per year) all contribute to the lack of effect. This lack of impact occurred despite the use, wherever possible, of values which generated the greatest consequence of the sea scallop fisheries takes of loggerheads. If less stringent values had been used, the effect would have been less. Patterson and Murray (2008) provide commentary on the effect that application of the precautionary principle to a PVA may have on "robust inference" and defensible policy.

Even a model as simple as the Dennis model is sensitive to parameter values and data inputs. Values calculated or selected for μ, $N_{ext,}$ and σ^2 were all influential. With respect to μ, we found that relatively small changes in the population trend produced profound changes in the probability of quasi-extinction at 100 years. For example, doubling the rate of decline in the base model (from -0.022 to -0.049) greatly increased the probability of extinction at 100 years from 0.01 to 0.54. In contrast, the level of bycatch mortality value removed from the population would need to be much greater than that observed in the sea scallop fisheries to have a major effect on the population trajectory. The comparison of the effect of different background mortalities (Fig. 6) suggests that up to ten times the level of loggerhead mortality in the sea scallop fisheries needs to be removed to stabilize the population. This small effect is important in that it suggests the relatively steep declining trend for 1996-2005 is being driven by some other, larger source of mortality.

Recognizing the influence of the population trend to the analysis, it is important to point out our assumption that the nesting beach data used in this analysis were representative of trends of the US loggerhead population. This was a practical decision; only the index beaches are counted annually in a systematic fashion. However, there is a risk in this assumption. We noted earlier the problem of juvenile in-water counts being at odds with the nesting trends. There is also some concern about the representativeness of the nest counts. If loggerhead nesting shifts systematically between years (either inside or outside of the index beach areas), then trends in the index nesting beach data may not represent the overall trend. For example, if loggerhead nesting is becoming more aggregated at the index sites (because of issues such as habitat protection), then the estimates may be biased high. Alternatively, if turtles nest outside of the time period (for example, earlier nesting caused by warmer climate conditions), then the index site estimates would be biased low. Work underway by the loggerhead TEWG and Florida's

Fish and Wildlife Research Institute will provide a substantive review of these trends. Our focus here was with evaluating the impact of the bycatch mortality in the Atlantic sea scallop fisheries on the future of the loggerhead population, and the impact of such biases on our analysis are likely immaterial. These biases could, however, significantly influence an analysis of population status and perhaps result in inappropriate management decisions.

The quasi-extinction value selected was also influential, but not as dramatically as the population trend. We evaluated N_{ext} values of 50 and 250 adult females. With the 1989-2005 base model, the probabilities of extinction at 100 years were 0.00 and 0.01 for 50 and 250 animals, respectively. Larger differences were observed in the 1996-2005 base model, where the values were 0.07 and 0.42 respectively. The latter, larger effect is likely due to the increased negative population trend. We also considered using the percent of decline approach suggested by Snover and Heppell (in press). We estimated the probability of reaching 50% of the current population size. Although risks of reaching the threshold were much higher (0.97 and 0.95 in 100 years) than with the 50 or 250 animal threshold, there were no significant differences between the base model and the model with takes added back in. Ultimately, we decided to use an absolute value of N_{ext} = 250 adult females largely because this analysis was designed to evaluate the risk of extinction resulting from mortalities in the scallop fisheries, and 250 animals better represents a threshold extinction value than does 50% of the current population size (N_{ext} = 17,441 adult females).

The model is also sensitive to changes in the variance; as the variance increases, the probability of extinction at any point in time increases, and as the variance decreases, probabilities of extinction decrease. Here it was assumed that the variance in the population trend is largely the same with and without the sea scallop fishery takes. Violations of this assumption would not change the interpretation of the sea scallop fisheries impacts, unless the take estimates were much higher relative to the population size and the variance in the takes was large.

However, the largest issue with variance was not the influence on the outcome but the difficulty of providing meaningful tests of significance with large confidence intervals. Using bootstrap techniques produced much tighter confidence intervals, but trajectories would need to vary considerably to find statistical differences.

Finally, this analysis was undertaken to provide a simple evaluation of the effect that loggerhead bycatch in the Atlantic sea scallop fisheries could have on the future viability of the WNA loggerhead population. It was not designed to and should not be used to evaluate population status. For example, here we implicitly assume that adult female recruitment will not change in the future. This is a particularly troublesome assumption because there are data suggesting that the number of juvenile loggerhead sea turtles is increasing (e.g., Epperly et al. 2007). If the increase in juvenile abundance translates into increased adult female recruitment, then our estimates of extinction probabilities would be overestimated; however, the relationship between the models with and without fishery takes would not be fundamentally changed. A staged matrix model, incorporating age-class survival and fecundity, would provide a much better evaluation tool to assess population status (and fishery impacts).

An example of such an evaluation is provided by the US Fish and Wildlife Service's (USFWS) recent quantitative threats analysis for the Florida manatee (*Trichechus manatus latirostris*; Runge et al. 2007). The basis of this threats assessment is a comparative population viability analysis, which involves forecasting the Florida manatee population under different scenarios regarding the presence of threats, while accounting for process variation (environmental, demographic, and catastrophic stochasticity) and parametric and structural uncertainty. Several steps were required: modifying an existing population model to accommodate the threats analysis framework, updating survival rates, estimating the fractions of mortality from various causes, modeling the threats themselves, and developing metrics to measure the impact of the threats. While the conceptual process followed in our analysis of loggerhead sea turtles and that used by the USFWS are similar, the additional information available from the USFWS exercise results from a stage-based projection model for Florida manatees, incorporating environmental and demographic stochasticity, catastrophes, density-dependence, and long-term change in carrying capacity.

However, recent data to support such an analysis of loggerhead sea turtles are incomplete. A comprehensive program to collect these data should be developed and implemented so that scientific analyses, such as those presented here, can be improved and the best possible scientific advice can be provided to NOAA managers tasked with conserving both turtle populations and fisheries.

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Table 1. Counts of loggerhead sea turtle (*Caretta caretta*) nests at index beaches for 1989-2005 by subpopulation, biannual totals, and rates of change (λ and r) by year (NMFS in review, FWRI 2007).

Year	Northern (NC, SC, GA)	Peninsular Florida	Total ($N_{i)}$	Two-year Running Sum (Rj)	Rate of Change (λ)	Inst. rate of change (r)
1989	1,421	39,091	40,512			
1990	2,466	50,266	52,732	93,244		
1991	2,127	52,802	54,929	107,661	1.1546	0.14377
1992	1,844	47,567	49,411	104,340	0.9692	-0.0313
1993	931	41,808	42,739	92,150	0.8832	-0.1242
1994	2,207	51,168	53,375	96,114	1.0430	0.04212
1995	1,484	57843	59,327	112,702	1.1726	0.15921
1996	1,969	52811	54,780	114,107	1.0125	0.01239
1997	1,100	43156	44,256	99,036	0.8679	-0.1417
1998	1,812	59918	61,730	105,986	1.0702	0.06782
1999	2,173	56471	58,644	120,374	1.1358	0.1273
2000	1,475	56277	57,752	116,396	0.9670	-0.0336
2001	1,242	45941	47,183	104,935	0.9015	-0.1037
2002	1,543	38125	39,668	86,851	0.8277	-0.1891
2003	1,998	40726	42,724	82,392	0.9487	-0.0527
2004	549	29547	30,096	72,820	0.8838	-0.1235
2005	1,766	34872	36,638	66,734	0.9164	-0.0873

Table 2. Model results based on 1989-2005 2-year running sum trend with a starting population size of 34,881 adult female loggerhead sea turtles (*Caretta caretta*) and quasi-extinction threshold equal to 250 adult females for base model and model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes added back into population.

	Base Model	With Fishery Takes Added Back In
Population Trend	-0.022	-0.019
Variance of trend	0.012	0.012
Upper confidence limit	0.039	0.042
Lower confidence limit	-0.084	-0.080
Quasi-extinction risk with 95% confidence interval in parentheses		
@ 25 years	0.00 (0, 0)	0.00 (0, 0)
@ 50 years	0.00 (0, 0)	0.00 (0, 0)
@ 75 years	0.00 (0, 0.09)	0.00 (0, 0.02)
@ 100 years	0.01 (0, 0.46)	0.01 (0, 0.31)
Median time to extinction	207 years	240 years

Table 3. Counts of loggerhead sea turtle (*Caretta caretta*) nests at index beaches for 1996-2005 by subpopulation, biannual totals, and rates of change (λ and r) by year (NMFS in review, FWRI 2007). Number in italics were interpolated from adjacent counts.

Year	Northern (NC, SC, GA)	Peninsular Florida	Dry Tortugas (Florida)	Northern Gulf (FL, AL)	Total (N_i)	Running sum (Rj)	Rate of change (λ)	Inst. rate of change (r)
1996	1,969	52,811	249	*166*	55,195			
1997	1,100	43,156	258	166	44,680	99,875		
1998	1,812	59,918	249	149	62,128	106,808	1.0694	0.0671
1999	2,173	56,471	292	235	59,171	121,299	1.1357	0.1272
2000	1,475	56,277	242	181	58,175	117,346	0.9674	-0.0331
2001	1,242	45,941	213	143	47,539	105,714	0.9009	-0.1044
2002	1,543	38,125	*210*	149	40,027	87,566	0.8283	-0.1883
2003	1,998	40,726	208	95	43,027	83,054	0.9485	-0.053
2004	549	29,547	159	114	30,369	73,396	0.88371	-0.1236
2005	1,766	34,872	*159*	120	36,917	67,286	0.91675	-0.0869

Table 4. Model results based on 1996-2005 2-year running sum trend with a starting population size of 34,881 adult female loggerhead sea turtles (*Caretta caretta*), and quasi-extinction threshold equal to 250 adult females for base model and model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes added back into population.

	Base Model	With Fishery Takes Added Back In
Population trend	-0.049	-0.046
Variance of trend	0.011	0.011
Upper confidence limit	0.037	0.040
Lower confidence limit	-0.135	-0.1322
Quasi-extinction risk with 95% confidence interval in parentheses		
@ 25 years	0.00 (0, 0)	0.00 (0, 0)
@ 50 years	0.00 (0, 0.03)	0.00 (0, 0.02)
@ 75 years	0.10 (0, 0.67)	0.06 (0, 0.57)
@ 100 years	0.54 (0.02, 0.98)	0.42 (0.01, 0.996)
Median time to extinction	98 years	102 years

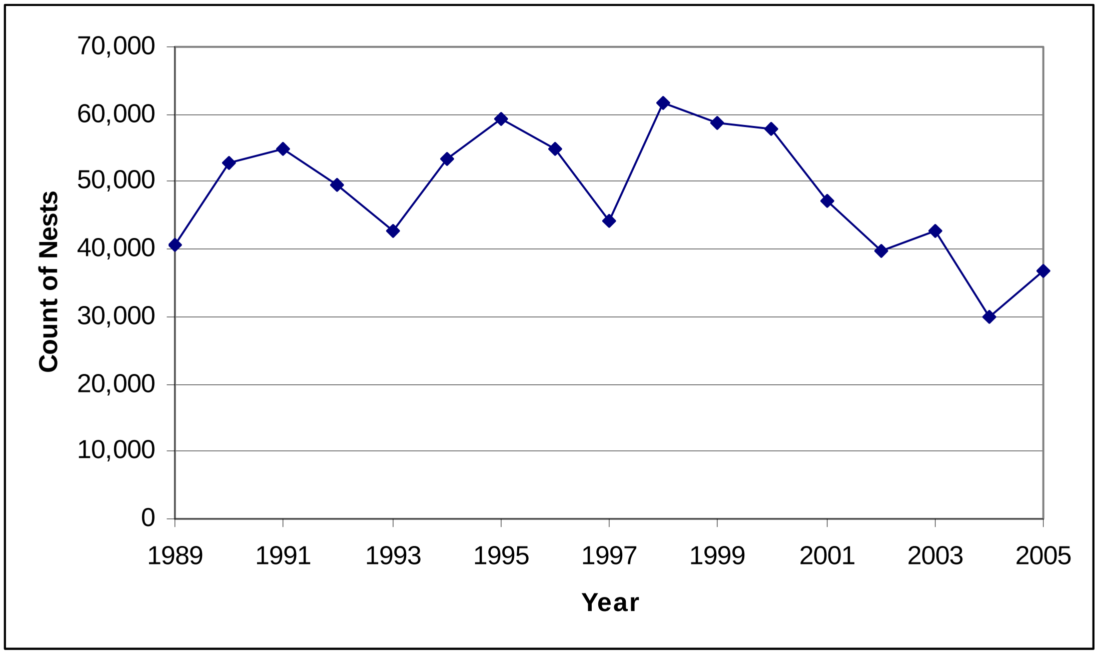

Figure 1. Number of Atlantic Ocean loggerhead sea turtle (*Caretta caretta*) nests recorded at US Northern (NC, SC, GA) and Peninsular Florida index beaches from 1989 to 2005 (NMFS in review, FWRI 2007).

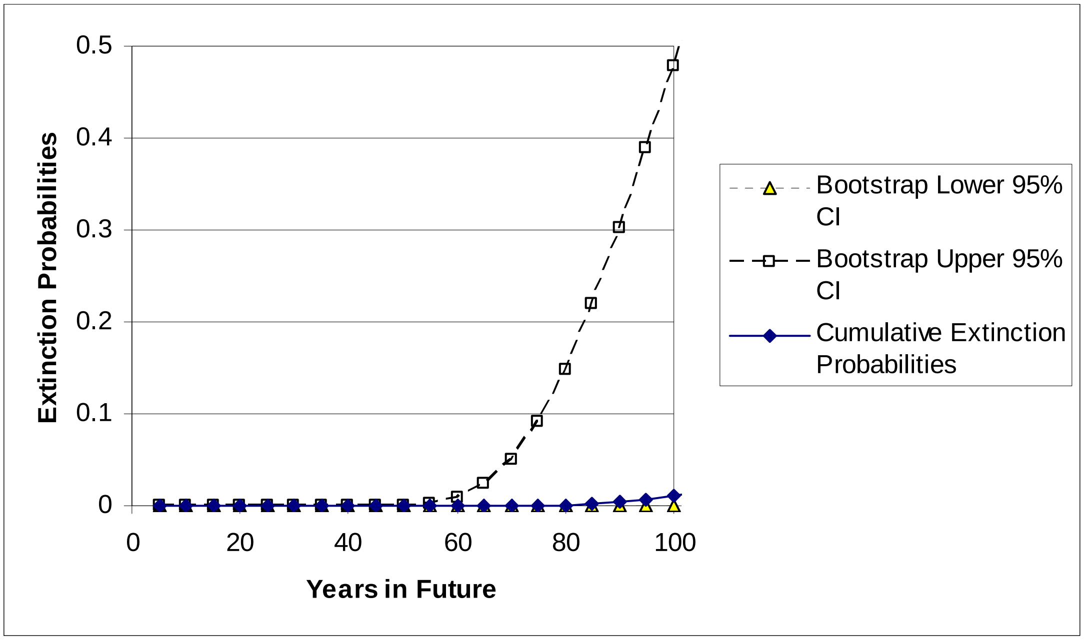

Figure 2. Cumulative quasi-extinction probabilities and confidence intervals (CI) for 1989-2005 base model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes for adult female western North Atlantic loggerhead sea turtles *(Caretta caretta)*. Quasi-extinction is equal to 250 adult female loggerhead sea turtles.

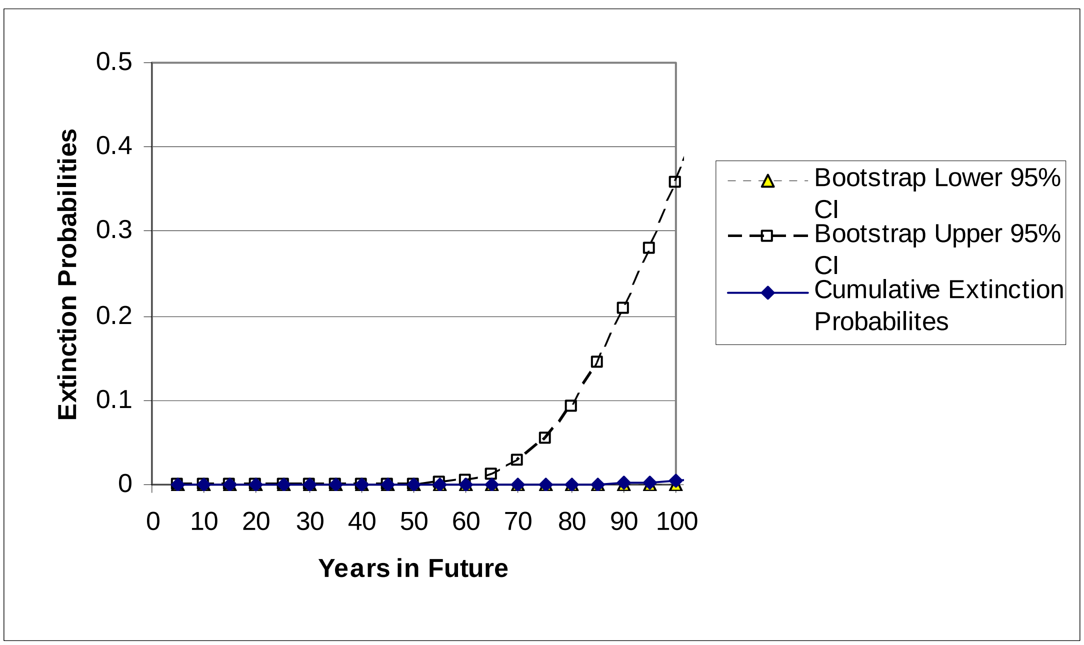

Figure 3. Cumulative quasi-extinction probabilities and confidence intervals (CI) for 1989-2005 model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes for adult female western North Atlantic loggerhead sea turtles *(Caretta caretta)* added back into population. Quasi-extinction is equal to 250 adult female loggerhead sea turtles.

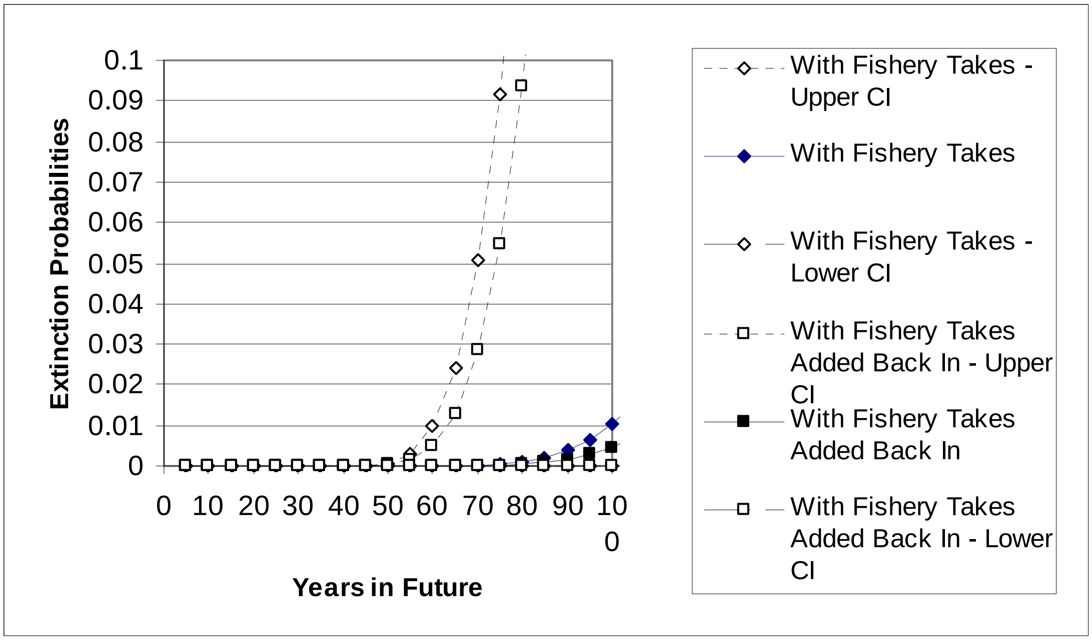

Figure 4. Comparison of cumulative quasi-extinction probabilities and confidence intervals (CI) of 1989-2005 models with and without Atlantic sea scallop (*Placopecten magellanicus*) fishery takes. Quasi-extinction is equal to 250 adult female loggerhead sea turtles *(Caretta caretta)*. Note vertical scale runs only through $P_{EX} = 0.10$.

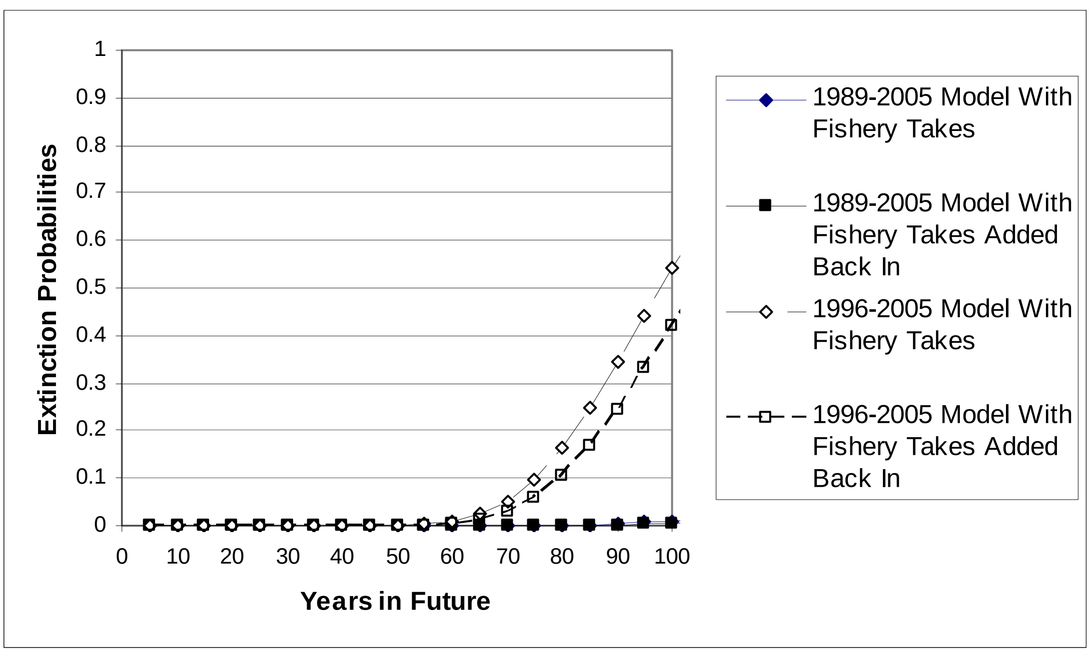

Figure 5. Extinction trajectories for models with and without Atlantic sea scallop (*Placopecten magellanicus*) fishery takes with original 1989-2005 population trajectory compared to 1996-2005 trajectory. Quasi-extinction is equal to 250 adult female loggerhead sea turtles *(Caretta caretta)*.

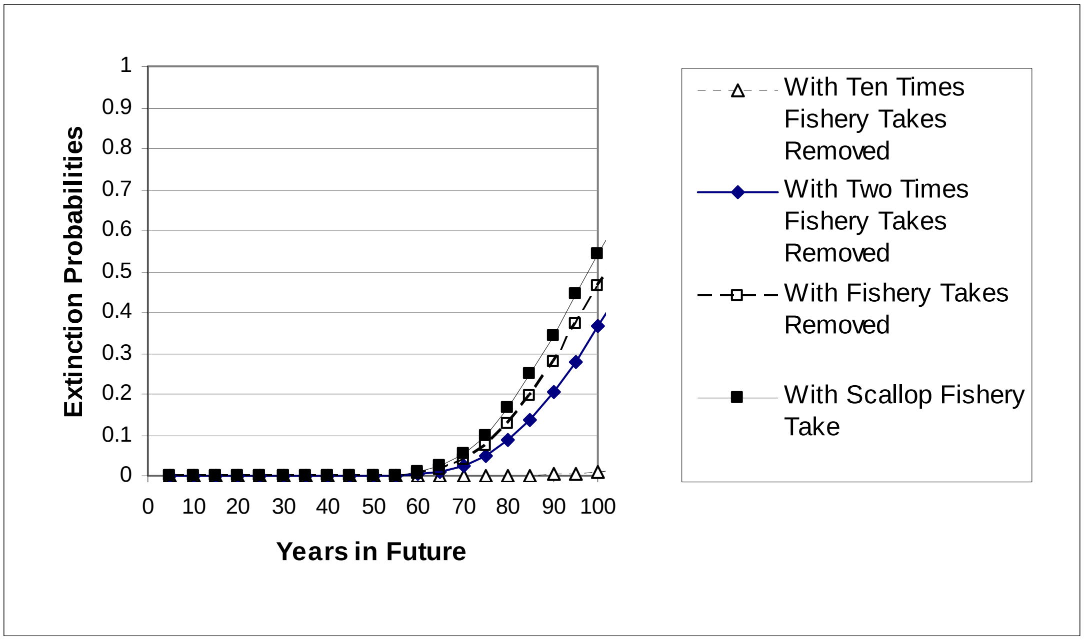

Figure 6. Cumulative quasi-extinction probabilities for 1996-2005 models with various levels of mortality removed from the trend. Fishery takes estimated as one time (the Atlantic sea scallop [*Placopecten magellanicus*] fisheries) versus two and ten times the original sea scallop fishery take level. Quasi-extinction equal to 250 adult females loggerhead sea turtles (*Caretta caretta*).

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