NOAA Technical Memorandum NMFS-NE-271

US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2020

US DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, Massachusetts
July 2021

NOAA Technical Memorandum NMFS-NE-271

This series represents a secondary level of scientific publishing. All issues employ thorough internal scientific review; some issues employ external scientific review. Reviews are transparent collegial reviews, not anonymous peer reviews. All issues may be cited in formal scientific communications.

US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2020

Sean A Hayes[1], Elizabeth Josephson[1], Katherine Maze-Foley[6], Patricia E Rosel[3], and Jennifer Turek[1], Editors

with contributions from (listed alphabetically)

Barbie Byrd[4], Samuel Chavez-Rosales[1], Timothy VN Cole[1], Lance P Garrison[5], Joshua Hatch[1], Allison Henry[1], Stacey C Horstman[5], Jenny Litz[5], Marjorie C Lyssikatos[1], Keith D Mullin[2], Christopher Orphanides[1], Joel Ortega-Ortiz[5], Richard M Pace[1], Debra L Palka[1], Jessica Powell[5], Gina Rappucci[5], and Frederick W Wenzel[1]

[1] National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543
[2] National Marine Fisheries Service, P.O. Drawer 1207, Pascagoula, MS 39568
[3] National Marine Fisheries Service, 646 Cajundome Blvd. Suite 234, Lafayette, LA 70506
[4] National Marine Fisheries Service, 101 Pivers Island, Beaufort, NC 28516
[5] National Marine Fisheries Service, 263 13th Ave. South, St. Petersburg, FL 33701
[6] National Marine Fisheries Service, 75 Virginia Beach Drive, Miami, FL 33149

US DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, Massachusetts
July 2021

Editorial Notes

Information Quality Act Compliance: In accordance with section 515 of Public Law 106-554, the Northeast Fisheries Science Center completed both technical and policy reviews for this report. These predissemination reviews are on file at the NEFSC Editorial Office.

Species Names: The NEFSC Editorial Office's policy on the use of species names in all technical communications is generally to follow the American Fisheries Society's lists of scientific and common names for fishes, mollusks, and decapod crustaceans and to follow the Society for Marine Mammalogy's guidance on scientific and common names for marine mammals. Exceptions to this policy occur when there are subsequent compelling revisions in the classifications of species, resulting in changes in the names of species.

Statistical Terms: The NEFSC Editorial Office's policy on the use of statistical terms in all technical communications is generally to follow the International Standards Organization's handbook of statistical methods.

ACKNOWLEDGMENTS

The authors wish to acknowledge advice, comments and valuable contributions provided by the Northeast Fisheries Science Center Fisheries Sampling Branch; members of the Northeast and Southeast Marine Mammal Stranding Networks; Mendy Garron, Amanda Johnson, David Gouveia, and Allison Rosner of the Northeast Regional Office; Kevin Barry, Sabrina Bowen-Stevens, John Carlson, Ruth Ewing, Wayne Hoggard, Aleta Hohn, Alyssa Mathers, Blair Mase, Wayne McFee, Errol Ronje, Elizabeth Scott-Denton, Carrie Sinclair, and Melissa Soldevilla of the Southeast Fisheries Science Center; Jarita Davis, Michael Simpkins, and Jon Hare of the Northeast Fisheries Science Center; Laura Engleby, Kara Shervanick, and Elizabeth Stratton of the Southeast Regional Office; Brian Balmer, Todd Speakman, and Lori Schwacke of the National Marine Mammal Foundation; William McLellan of University of North Carolina Wilmington; and Robert Kenney, Jack Lawson, Richard Merrick, Michael Moore, Genny Nesslage, Kathryn Ono, James 'Buddy' Powell, Andy Read, Ana Sirovic, Erin Summers, and Randall Wells of the Atlantic Scientific Review Group. We also thank the Marine Mammal Commission, Center for Biological Diversity, Whale and Dolphin Conservation, Fisheries and Oceans Canada, Natural Resources Defense Council, and the Maine Lobstermen's Association for their constructive comments and advice.

EXECUTIVE SUMMARY

Under the 1994 amendments of the Marine Mammal Protection Act (MMPA), the National Marine Fisheries Service (NMFS) and the United States Fish and Wildlife Service (USFWS) were required to generate stock assessment reports (SARs) for all marine mammal stocks in waters within the U.S. Exclusive Economic Zone (EEZ). The first reports for the Atlantic (includes the Gulf of Mexico) were published in July 1995 (Blaylock *et al.* 1995). The MMPA requires NMFS and USFWS to review these reports annually for strategic stocks of marine mammals and at least every three years for stocks determined to be non-strategic. Included in this report as appendices are: a summary of serious injury/mortality estimates of marine mammals in observed U.S. fisheries (Appendix I), a summary of NMFS records of large whale human-caused serious injury and mortality (Appendix II), detailed fisheries information (Appendix III), summary tables of abundance estimates generated over recent years and the surveys from which they are derived (Appendix IV), a summary of observed fisheries bycatch (Appendix V), and estimates of human-caused mortality resulting from the *Deepwater Horizon* oil spill (Appendix VI).

Table 1 contains a summary, by species, of the information included in the stock assessments, and also indicates those that have been revised since the 2019 publication. Most of the changes incorporate new information into sections on population size and/or mortality estimates. A total of 32 of the Atlantic and Gulf of Mexico stock assessment reports were revised for 2020. The revised SARs include 12 strategic and 20 non-strategic stocks.

This report was prepared by staff of the Northeast Fisheries Science Center (NEFSC) and Southeast Fisheries Science Center (SEFSC). NMFS staff presented the reports at the February 2020 meeting of the Atlantic Scientific Review Group (ASRG), and subsequent revisions were based on their contributions and constructive criticism. This is a working document and individual stock assessment reports will be updated as new information becomes available and as changes to marine mammal stocks and fisheries occur. The authors solicit any new information or comments which would improve future stock assessment reports.

INTRODUCTION

Section 117 of the 1994 amendments to the Marine Mammal Protection Act (MMPA) requires that an annual stock assessment report (SAR) for each stock of marine mammals that occurs in waters under USA jurisdiction, be prepared by the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS), in consultation with regional Scientific Review Groups (SRGs). The SRGs are a broad representation of marine mammal and fishery scientists and members of the commercial fishing industry mandated to review the marine mammal stock assessments and provide advice to the NOAA Assistant Administrator for Fisheries. The reports are then made available on the *Federal Register* for public review and comment before final publication.

The MMPA requires that each SAR contain several items, including: (1) a description of the stock, including its geographic range; (2) a minimum population estimate, a maximum net productivity rate, and a description of current population trend, including a description of the information upon which these are based; (3) an estimate of the annual human-caused mortality and serious injury of the stock, and, for a strategic stock, other factors that may be causing a decline or impeding recovery of the stock, including effects on marine mammal habitat and prey; (4) a description of the commercial fisheries that interact with the stock, including the estimated number of vessels actively participating in the fishery and the level of incidental mortality and serious injury of the stock by each fishery on an annual basis; (5) a statement categorizing the stock as strategic or not, and why; and (6) an estimate of the potential biological removal (PBR) level for the stock, describing the information used in the calculation. The MMPA also requires that SARs be updated annually for stocks which are specified as strategic stocks, or for which significant new information is available, and once every three years for non-strategic stocks.

Following enactment of the 1994 amendments, the NMFS and USFWS held a series of workshops to develop guidelines for preparing the SARs. The first set of stock assessments for the Atlantic Coast (including the Gulf of Mexico) were published in July 1995 in the *NOAA Technical Memorandum* series (Blaylock *et al.* 1995). In April 1996, NMFS held a workshop to review proposed additions and revisions to the guidelines for preparing SARs (Wade and Angliss 1997). Guidelines developed at these and subsequent workshops were followed in preparing the SARs. In 1997 and 2004 SARs were not produced.

In this document, major revisions and updating of the SARs were completed for stocks for which significant new information was available. These are identified by the April 2021 date-stamp at the top right corner at the beginning of each report.

REFERENCES

Blaylock, R.A., J.W. Hain, L.J. Hansen, D.L. Palka and G.T. Waring. 1995. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments. NOAA Tech. Memo. NMFS-SEFSC-363. 211pp.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

TABLE OF CONTENTS

TABLE 1. A SUMMARY OF ATLANTIC MARINE MAMMAL STOCK ASSESSMENT REPORTS FOR STOCKS OF MARINE MAMMALS UNDER NMFS AUTHORITY THAT OCCUPY WATERS UNDER USA JURISDICTION.

Total annual mortality serious injury (M/SI) and annual fisheries M/SI are mean annual figures for the period 2014–2018. Nest = estimated abundance, Nmin = minimum abundance estimate, CV = coefficient of variation, Rmax = maximum productivity rate, Fr = recovery factor, PBR = potential biological removal, unk = unknown, and undet = undetermined (PBR for species with outdated abundance estimates is considered "undetermined").

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
1	North Atlantic right whale	Western North Atlantic	Y	412	0	408	0.04	0.1	0.8	18.6	6.85	Y	2019	2018	Total M/SI presented here is model-derived. As this has not been broken down by cause, the fishery M/SI reported here is observed interactions only.	NEC
2	Humpback whale	Gulf of Maine	N	1,396	0	1,380	0.065	0.5	22	12.15	7.75	N	2019	2016	Revised SAR was presented in draft stages but withdrawn for final due to delay in publication of supporting documents.	NEC
3	Fin whale	Western North Atlantic	Y	6,802	0.24	5,573	0.04	0.1	11	2.35	1.55	Y	2019	2016		NEC
4	Sei whale	Nova Scotia	Y	6,292	1.02	3,098	0.04	0.1	6.2	1.2	0.4	Y	2019	2016		NEC
5	Minke whale	Canadian East Coast	Y	21,968	0.31	17,002	0.04	0.5	170	10.6	9.15	N	2019	2016		NEC
6	Blue whale	Western North Atlantic	N	unk	unk	402	0.04	0.1	0.8	0	0	Y	2019	1980-2008		NEC
7	Sperm whale	North Atlantic	N	4,349	0.28	3,451	0.04	0.1	3.9	0	0	Y	2019	2016		NEC
8	Dwarf sperm whale	Western North Atlantic	N	7,750	0.38	5,689	0.04	0.4	46	0	0	N	2019	2016	Estimate for *Kogia spp.* Only.	SEC
9	Pygmy sperm whale	Western North Atlantic	N	7,750	0.38	5,689	0.04	0.4	46	0	0	N	2019	2016	Estimate for *Kogia spp.* Only.	SEC
10	Killer whale	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2014	2016		NEC
11	Pygmy killer whale	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2016		SEC
12	False killer whale	Western North Atlantic	N	1,791	0.56	1,154	0.04	0.5	12	0	0	N	2019	2016		SEC
13	Northern bottlenose whale	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2014	2016		NEC
14	Cuvier's beaked whale	Western North Atlantic	N	5,744	0.36	4,282	0.04	0.5	43	0.2	0	N	2019	2016		NEC
15	Blainville’s beaked whale	Western North Atlantic	N	10,107	0.27	8,085	0.04	0.5	81	0.2	0	N	2019	2016	Estimates for *Mesoplodon spp.*	NEC
16	Gervais beaked whale	Western North Atlantic	N	10,107	0.27	8,085	0.04	0.5	81	0	0	N	2019	2016	Estimates for *Mesoplodon spp.*	NEC
17	Sowerby’s beaked whale	Western North Atlantic	N	10,107	0.27	8,085	0.04	0.5	81	0	0	N	2019	2016	Estimates for *Mesoplodon spp.*	NEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
18	True's beaked whale	Western North Atlantic	N	10,107	0.27	8,085	0.04	0.5	81	0.2	0.2	N	2019	2016	Estimates for *Mesoplodon spp.*	NEC
19	Melon-headed whale	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2016		SEC
20	Risso's dolphin	Western North Atlantic	N	35,493	0.19	30,289	0.04	0.5	303	54.3	53.9 (0.24)	N	2019	2016		NEC
21	Pilot whale, long-finned	Western North Atlantic	N	39,215	0.30	30,627	0.04	0.5	306	21	21 (0.22)	N	2019	2016		NEC
22	Pilot whale, short-finned	Western North Atlantic	N	28,924	0.24	23,637	0.04	0.5	236	160	160 (0.12)	N	2019	2016		SEC
23	Atlantic white-sided dolphin	Western North Atlantic	N	93,233	0.71	54,443	0.04	0.5	544	26	26 (0.20)	N	2019	2016		NEC
24	White-beaked dolphin	Western North Atlantic	N	536,016	0.31	415,344	0.04	0.5	4,153	0	0	N	2019	2016		NEC
25	Common dolphin	Western North Atlantic	Y	172,974	0.21	145,216	0.04	0.5	1,452	399	399 (0.05)	N	2019	2016		NEC
26	Atlantic spotted dolphin	Western North Atlantic	N	39,921	0.27	32,032	0.04	0.5	320	0	0	N	2019	2016		SEC
27	Pantropical spotted dolphin	Western North Atlantic	N	6,593	0.52	4,367	0.04	0.5	44	0	0	N	2019	2016		SEC
28	Striped dolphin	Western North Atlantic	N	67,036	0.29	52,939	0.04	0.5	529	0	0	N	2019	2016		NEC
29	Fraser's dolphin	Western North Atlantic	N	unk	unk	unk	0.04	0.5	unk	0	0	N	2019	2016		SEC
30	Rough-toothed dolphin	Western North Atlantic	N	136	1.0	67	0.04	0.5	0.7	0	0	N	2018	2016		SEC
31	Clymene dolphin	Western North Atlantic	N	4,237	1.03	2,071	0.04	0.5	21	0	0	N	2019	2016		SEC
32	Spinner dolphin	Western North Atlantic	N	4,102	0.99	2,045	0.04	0.5	20	0	0	N	2019	2016		SEC
33	Common bottlenose dolphin	Western North Atlantic, Offshore	N	62,851	0.23	51,914	0.04	0.5	519	28	28 (0.34)	N	2019	2016	Estimates may include sightings of the coastal form.	SEC
34	Common bottlenose dolphin	Western North Atlantic, Northern Migratory Coastal	Y	6,639	0.41	4,759	0.04	0.5	48	12.2–21.5	12.2–21.5	Y	2017	2016		SEC
35	Common bottlenose dolphin	Western North Atlantic, Southern Migratory Coastal	Y	3,751	0.60	2,353	0.04	0.5	24	0–18.3	0–18.3	Y	2017	2016		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
36	Common bottlenose dolphin	Western North Atlantic, S. Carolina, Georgia Coastal	N	6,027	0.34	4,569	0.04	0.5	46	1.4-1.6	1.0-1.2	Y	2017	2017		SEC
37	Common bottlenose dolphin	Western North Atlantic, Northern Florida Coastal	N	877	0.49	595	0.04	0.5	6.0	0.6	0	Y	2017	2017		SEC
38	Common bottlenose dolphin	Western North Atlantic, Central Florida Coastal	N	1,218	0.35	913	0.04	0.5	9.1	0.4	0.4	Y	2017	2017		SEC
39	Common bottlenose dolphin	Northern North Carolina Estuarine System	Y	823	0.06	782	0.04	0.5	7.8	7.2-30	7.0-29.8	Y	2017	2017		SEC
40	Common bottlenose dolphin	Southern North Carolina Estuarine System	Y	unk	unk	unk	0.04	0.5	undet	0.4	0.4	Y	2017	2017		SEC
41	Common bottlenose dolphin	Northern South Carolina Estuarine System	N	unk	unk	unk	0.04	0.5	unk	0.2	0.2	Y	2015	n/a		SEC
42	Common bottlenose dolphin	Charleston Estuarine System	N	unk	unk	unk	0.04	0.5	undet	unk	unk	Y	2015	2005, 2006		SEC
43	Common bottlenose dolphin	Northern Georgia, Southern South Carolina Estuarine System	N	unk	unk	unk	0.04	0.5	unk	1.4	1.4	Y	2015	n/a		SEC
44	Common bottlenose dolphin	Central Georgia Estuarine System	N	192	0.04	185	0.04	0.5	1.9	unk	unk	Y	2015	2008, 2009		SEC
45	Common bottlenose dolphin	Southern Georgia Estuarine System	N	194	0.05	185	0.04	0.5	1.9	unk	unk	Y	2015	2008, 2009		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
46	Common bottlenose dolphin	Jacksonville Estuarine System	N	unk	unk	unk	0.04	0.5	unk	1.2	1.2	Y	2015	n/a		SEC
47	Common bottlenose dolphin	Indian River Lagoon Estuarine System	N	unk	unk	unk	0.04	0.5	unk	4.4	4.4	Y	2015	n/a		SEC
48	Common bottlenose dolphin	Biscayne Bay	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2013	n/a		SEC
49	Common bottlenose dolphin	Florida Bay	N	unk	unk	unk	0.04	0.5	undet	unk	unk	N	2013	2003		SEC
50	Harbor porpoise	Gulf of Maine, Bay of Fundy	Y	95,543	0.31	74,034	0.046	0.5	851	217	217 (0.15)	N	2019	2016		NEC
51	Harbor seal	Western North Atlantic	Y	75,834	0.15	66,884	0.12	0.5	2,006	350	338 (0.12)	N	2019	2012		NEC
52	Gray seal	Western North Atlantic	Y	27,131	0.19	23,158	0.12	1.0	1,389	4,729	946 (0.11)	N	2019	2016		NEC
53	Harp seal	Western North Atlantic	N	unk	unk	unk	0.12	1.0	unk	232,422	65 (0.21)	N	2019	n/a		NEC
54	Hooded seal	Western North Atlantic	N	unk	unk	unk	0.12	0.75	unk	1,680	0.6 (1.12)	N	2018	n/a		NEC
55	Sperm whale	Gulf of Mexico	Y	1,180	0.22	983	0.04	0.1	2.0	9.6	0.2 (1.0)	Y	2015	2017, 2018		SEC
56	Bryde's whale	Gulf of Mexico	Y	51	0.5	34	0.04	0.1	0.1	0.5	0	Y	2017	2017, 2018	Total M/SI is a minimum estimate and does not include Fisheries M/SI.	SEC
57	Cuvier's beaked whale	Gulf of Mexico	Y	18	0.75	10	0.04	0.5	0.1	5.2	0	N	2012	2017, 2018		SEC
58	Blainville's beaked whale	Gulf of Mexico	Y	98	0.46	68	0.04	0.5	0.7	5.2	0	N	2012	2017, 2018	Estimates for *Mesoplodon spp.*	SEC
59	Gervais' beaked whale	Gulf of Mexico	Y	20	0.98	10	0.04	0.5	0.1	5.2	0	N	2012	2017, 2018		SEC
60	Common bottlenose dolphin	Gulf of Mexico, Continental Shelf	N	51,192	0.10	46,926	0.04	0.5	469	0.8	0.6	N	2015	2011, 2012		SEC
61	Common bottlenose dolphin	Gulf of Mexico, Eastern Coastal	N	12,388	0.13	11,110	0.04	0.5	111	1.6	1.6	N	2015	2011, 2012		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
62	Common bottlenose dolphin	Gulf of Mexico, Northern Coastal	N	7,185	0.21	6,044	0.04	0.5	60	0.4	0.4	N	2015	2011, 2012	M/SI is a minimum count and does not include projected mortality estimates for 2012–2016 due to the DWH oil spill.	SEC
63	Common bottlenose dolphin	Gulf of Mexico, Western Coastal	N	20,161	0.17	17,491	0.04	0.5	175	0.6	0.6	N	2015	2011, 2012		SEC
64	Common bottlenose dolphin	Gulf of Mexico, Oceanic	Y	7,462	0.31	5,769	0.04	0.5	58	32	0	N	2014	2017, 2018		SEC
65	Common bottlenose dolphin	Laguna Madre	N	80	1.57	unk	0.04	0.5	undet	0.4	0.2	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
66	Common bottlenose dolphin	Neuces Bay, Corpus Christi Bay	N	58	0.61	unk	0.04	0.5	undet	0	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
67	Common bottlenose dolphin	Copano Bay, Aransas Bay, San Antonio Bay, Redfish Bay, Espiritu Santo Bay	N	55	0.82	unk	0.04	0.5	undet	0.2	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
68	Common bottlenose dolphin	Matagorda Bay, Tres Palacios Bay, Lavaca Bay	N	61	0.45	unk	0.04	0.5	undet	0.4	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
69	Common bottlenose dolphin	West Bay	N	48	0.03	46	0.04	0.5	0.5	0.2	0.2	N	2019	2014, 2015		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
70	Common bottlenose dolphin	Galveston Bay, East Bay, Trinity Bay	N	152	0.43	unk	0.04	0.5	undet	0.4	0.4	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
71	Common bottlenose dolphin	Sabine Lake	N	0	-	-	0.04	0.4	undet	0.2	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
72	Common bottlenose dolphin	Calcasieu Lake	N	0	-	-	0.04	0.4	undet	0.2	0.2	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
73	Common bottlenose dolphin	Vermilion Bay, West Cote Blanche Bay, Atchafalaya Bay	N	0	-	-	0.04	0.4	undet	0	0	Y	2018	1992	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
74	Common bottlenose dolphin	Terrebonne, Timbalier Bay Estuarine System	N	3,870	0.15	3,426	0.04	0.4	27	0.2	0	N	2018	2016		SEC
75	Common bottlenose dolphin	Barataria Bay Estuarine System	N	2,306	0.09	2,138	0.04	0.4	17	160	0.8	Y	2017	2010-2014		SEC
76	Common bottlenose dolphin	Mississippi River Delta	N	332	0.93	170	0.04	0.4	1.4	32.7	0	Y	2018	2011, 2012	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
77	Common bottlenose dolphin	Mississippi Sound, Lake Borgne, Bay Boudreau	N	3,046	0.06	2,896	0.04	0.4	23	310	1.0	Y	2017	2012		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
78	Common bottlenose dolphin	Mobile Bay, Bonsecour Bay	N	122	0.34	unk	0.04	0.4	undet	36.6	0.8	Y	2018	1993	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
79	Common bottlenose dolphin	Perdido Bay	N	0	-	-	0.04	0.4	undet	0.6	0.2	Y	2018	1993	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
80	Common bottlenose dolphin	Pensacola Bay, East Bay	N	33	0.80	unk	0.04	0.4	undet	0.2	0.2	Y	2018	1993	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
81	Common bottlenose dolphin	Chocta-whatchee Bay	N	179	0.04	unk	0.04	0.5	undet	0.4	0.4	Y	2015	2007		SEC
82	Common bottlenose dolphin	St. Andrew Bay	N	199	0.09	185	0.04	0.4	1.5	0.2	0.2	N	2019	2016		SEC
83	Common bottlenose dolphin	St. Joseph Bay	N	142	0.17	123	0.04	0.4	1.0	unk	unk	N	2019	2011		SEC
84	Common bottlenose dolphin	St. Vincent Sound, Apalachicola Bay, St. George Sound	N	439	0.14	unk	0.04	0.4	undet	0	0	Y	2018	2007		SEC
85	Common bottlenose dolphin	Apalachee Bay	N	491	0.39	unk	0.04	0.4	undet	0	0	Y	2018	1993	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
86	Common bottlenose dolphin	Waccasassa Bay, Withla-coochee Bay, Crystal Bay	N	unk	-	unk	0.04	0.4	undet	0	0	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
87	Common bottlenose dolphin	St. Joseph Sound, Clearwater Harbor	N	unk	-	unk	0.04	0.4	undet	0.4	0.4	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
88	Common bottlenose dolphin	Tampa Bay	N	unk	-	unk	0.04	0.4	undet	0.6	0.6	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
89	Common bottlenose dolphin	Sarasota Bay, Little Sarasota Bay	N	158	0.27	126	0.04	0.4	1.0	0.6	0.6	N	2018	2015	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
90	Common bottlenose dolphin	Pine Island Sound, Charlotte Harbor, Gasparilla Sound, Lemon Bay	N	826	0.09	unk	0.04	0.4	undet	1.6	1.0	Y	2018	2006	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
91	Common bottlenose dolphin	Caloosa-hatchee River	N	0	-	-	0.04	0.4	undet	0.4	0.4	Y	2018	1985	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
92	Common bottlenose dolphin	Estero Bay	N	unk	-	unk	0.04	0.4	undet	0.2	0	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
92	Common bottlenose dolphin	Chokoloskee Bay, Ten Thousand Islands, Gullivan Bay	N	unk	-	unk	0.04	0.4	undet	0	0	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
94	Common bottlenose dolphin	Whitewater Bay	N	unk	-	unk	0.04	0.4	undet	0	0	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
95	Common bottlenose dolphin	Florida Keys (Bahia Honda to Key West)	N	unk	-	unk	0.04	0.4	undet	0	0	Y	2018	n/a	Details for this stock are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks.	SEC
96	Atlantic spotted dolphin	Gulf of Mexico	N	unk	unk	unk	0.04	0.5	undet	42	42 (0.45)	N	2015	2003, 2004		SEC
97	Pantropical spotted dolphin	Gulf of Mexico	Y	37,195	0.24	30,377	0.04	0.5	304	241	0	N	2015	2017, 2018		SEC
98	Striped dolphin	Gulf of Mexico	Y	1,817	0.56	1,172	0.04	0.5	12	13	0	Y	2012	2017, 2018		SEC
99	Spinner dolphin	Gulf of Mexico	Y	2,991	0.54	1,954	0.04	0.5	20	113	0	Y	2012	2017, 2018		SEC
100	Rough-toothed dolphin	Gulf of Mexico	Y	unk	n/a	unk	0.04	0.4	undet	39	0.8 (1.00)	N	2016	2017, 2018		SEC
101	Clymene dolphin	Gulf of Mexico	Y	513	1.03	250	0.04	0.5	2.5	8.4	0	Y	2012	2017, 2018		SEC
102	Fraser's dolphin	Gulf of Mexico	Y	213	1.03	104	0.04	0.5	1.0	unk	0	N	2012	2017, 2018		SEC
103	Killer whale	Gulf of Mexico	Y	267	0.75	152	0.04	0.5	1.5	unk	0	N	2012	2017, 2018		SEC
104	False killer whale	Gulf of Mexico	Y	494	0.79	276	0.04	0.5	2.8	2.2	0	N	2012	2017, 2018		SEC

ID	Species	Stock Area	Updated this Year	Nest	Nest CV	Nmin	Rmax	Fr	PBR	Total Annual M/SI	Annual Fish. M/SI CV	Strategic Status	SAR of Last Update	Last Survey Year	Comments	NMFS Ctr.
105	Pygmy killer whale	Gulf of Mexico	Y	613	1.15	283	0.04	0.5	2.8	1.6	0	N	2012	2017, 2018		SEC
106	Dwarf sperm whale	Gulf of Mexico	Y	336	0.35	253	0.04	0.5	2.5	31	0	N	2012	2017, 2018	Estimate for *Kogia spp.* only.	SEC
107	Pygmy sperm whale	Gulf of Mexico	Y	336	0.35	253	0.04	0.5	2.5	31	0	N	2012	2017, 2018	Estimate for *Kogia spp.* only.	SEC
108	Melon-headed whale	Gulf of Mexico	Y	1,749	0.68	1,039	0.04	0.5	10	9.5	0	N	2012	2017, 2018		SEC
109	Risso's dolphin	Gulf of Mexico	Y	1,974	0.46	1,368	0.04	0.5	14	5.3	0	N	2015	2017, 2018		SEC
110	Pilot whale, short-finned	Gulf of Mexico	Y	1,321	0.43	934	0.04	0.4	7.5	3.9	0.4 (1.00)	N	2015	2017, 2018	Nbest includes all *Globicephala sp.*, though it is presumed that only short-finned pilot whales are present in the Gulf of Mexico.	SEC
111	Sperm Whale	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.1	unk	unk	unk	Y	2010	n/a		SEC
112	Common bottlenose dolphin	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC
113	Cuvier's beaked whale	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC
114	Pilot whale, short-finned	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC
115	Spinner dolphin	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC
116	Atlantic spotted dolphin	Puerto Rico and U.S. Virgin Islands	N	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	2011	n/a		SEC

NORTH ATLANTIC RIGHT WHALE (*Eubalaena glacialis*): Western Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The western North Atlantic right whale population ranges primarily from calving grounds in coastal waters of the southeastern U.S. to feeding grounds in New England waters and the Canadian Bay of Fundy, Scotian Shelf, and Gulf of St. Lawrence (Figure 1). Mellinger *et al.* (2011) reported acoustic detections of right whales near the nineteenth-century whaling grounds east of southern Greenland, but the number of whales and their origin is unknown. Knowlton *et al.* (1992) reported several long-distance movements as far north as Newfoundland, the Labrador Basin, and southeast of Greenland. Resightings of photographically identified individuals have been made off Iceland, in the old Cape Farewell whaling ground east of Greenland (Hamilton *et al.* 2007), in northern Norway (Jacobsen *et al.* 2004), in the Azores (Silva *et al.* 2012), and off Brittany in northwestern France (New England Aquarium unpub. Catalog record). These long-range matches indicate an extended range for at least some individuals. Records from the Gulf of Mexico (Moore and Clark 1963, Schmidly *et al.* 1972, Ward-Geiger *et al*. 2011) represent individuals beyond the primary calving and wintering ground in the waters of the southeastern U.S. East Coast. The location of much of the population is unknown during the winter.

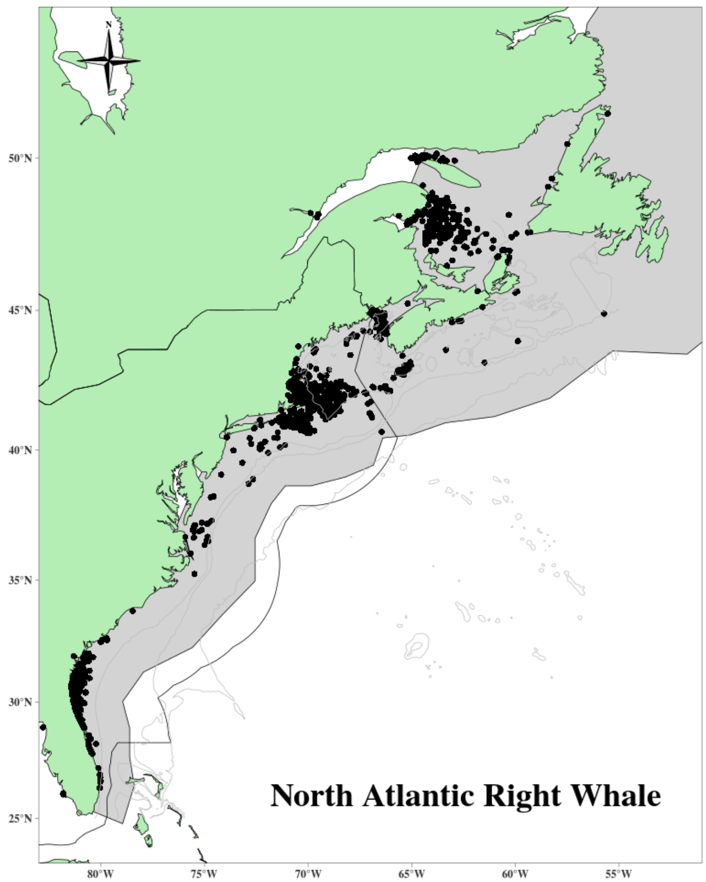


Figure 1. Approximate range (shaded area) and distribution of sightings (dots) of known North Atlantic right whales 2014–2018.

Davis *et al.* (2017) recently pooled together detections from a large number of passive acoustic devices and documented broad-scale use of U.S. eastern seaboard during much of the year.

Passive acoustic studies of right whales have demonstrated their year-round presence in the Gulf of Maine (Morano *et al.* 2012, Bort *et al.* 2015), New Jersey (Whitt *et al.* 2013), and Virginia (Salisbury *et al*. 2016). Additionally, right whales were acoustically detected off Georgia and North Carolina in 7 of 11 months monitored (Hodge *et al*. 2015).

Movements within and between habitats are extensive, and the waters off the mid-Atlantic states are an important migratory corridor. In 2000, one whale was photographed in Florida waters on 12 January, then again 11 days later (23 January) in Cape Cod Bay, less than a month later off Georgia (16 February), and back in Cape Cod Bay on 23 March, effectively making the round-trip migration to the Southeast and back at least twice during the winter season (Brown and Marx 2000). Results from satellite-tagging studies clearly indicate that sightings separated by perhaps two weeks should not necessarily be assumed to indicate a stationary or resident animal. Instead, telemetry data have shown lengthy excursions, including into deep water off the continental shelf (Mate *et al.* 1997, Baumgartner and Mate 2005).

Systematic visual surveys conducted off the coast of North Carolina during the winters of 2001 and 2002 sighted 8 calves, suggesting the calving grounds may extend as far north as Cape Fear (W.A. McLellan, Univ. of North Carolina Wilmington, pers. comm.). Four of those calves were not sighted by surveys conducted farther south. One of the females photographed was new to researchers, having effectively eluded identification over the period of its maturation. In 2016, the Southeastern U.S. Calving Area Critical Habitat was expanded north to Cape Fear, North Carolina. There is also at least one case of a calf apparently being born in the Gulf of Maine (Patrician *et al*. 2009) and another neonate was detected in Cape Cod Bay in 2012 (Center for Coastal Studies, Provincetown, MA USA, unpub. data).

New England waters are important feeding habitats for right whales, where they feed primarily on copepods (largely of the genera *Calanus* and *Pseudocalanus*). Right whales must locate and exploit extremely dense patches of zooplankton to feed efficiently (Mayo and Marx 1990). These dense zooplankton patches are likely a primary characteristic of the spring, summer, and fall right whale habitats (Kenney *et al.* 1986, 1995). While feeding in the coastal waters off Massachusetts has been better studied than in other areas, right whale feeding has also been observed on the margins of Georges Bank, in the Great South Channel, in the Gulf of Maine, in the Bay of Fundy, and over the Scotian Shelf (Baumgartner *et al.* 2007). The characteristics of acceptable prey distribution in these areas are beginning to emerge (e.g., Baumgartner *et al.* 2003, Baumgartner and Mate 2003). The National Marine Fisheries Service (NMFS) and Center for Coastal Studies aerial surveys during the springs of 1999–2011 found right whales along the Northern Edge of Georges Bank, in the Great South Channel, in Georges Basin, and in various locations in the Gulf of Maine including Cashes Ledge, Platts Bank, and Wilkinson Basin. In 2016, the Northeastern U.S. Foraging Area Critical Habitat was expanded to include nearly all U.S. waters of the Gulf of Maine based on the presence of the physical and biological features required for right whale foraging (81 FR 4837, 26 February 2016).

Analysis of sightings data has shown that the right whales' utilization of these areas within the Gulf of Maine had a strong seasonal component (Pace and Merrick 2008). Although whales were consistently found in these locations, studies also highlight the high interannual variability in right whale use of some habitats (Pendleton *et al.* 2009, Ganley *et al.* 2019). An important shift in habitat use patterns in 2010 was highlighted in an analysis of right whale acoustic presence along the U.S. Eastern seaboard from 2004 to 2014 (Davis *et al.* 2017). This shift was also reflected in visual survey data in the greater Gulf of Maine region. Between 2012 and 2016, visual surveys detected fewer individuals in the Great South Channel (NMFS unpublished data) and the Bay of Fundy (Davies *et al.* 2019), while the number of individuals using Cape Cod Bay in spring increased (Mayo *et al.* 2018). In addition, right whales apparently abandoned the Jordan Basin in the central Gulf of Maine in winter (Cole *et al.* 2013), but have since been seen in large numbers in a region south of Martha's Vineyard and Nantucket Islands (Leiter *et al.* 2017), an area outside of the 2016 Northeastern U.S. Foraging Area Critical Habitat. Since 2013, increased detections and survey effort in the Gulf of St. Lawrence indicate right whale presence in late spring through early fall (Cole *et al.* 2016; Khan *et al.* 2016, 2018). Aerial surveys of the Gulf of St. Lawrence during the summers of 2015, 2017, and 2018, documented at least 34, 105, and 131 unique individuals using the region, respectively (NMFS unpublished data).

Genetic analyses based upon direct sequencing of mitochondrial DNA (mtDNA) have identified 7 mtDNA haplotypes in the western North Atlantic right whale, including heteroplasmy that led to the declaration of the seventh haplotype (Malik *et al.* 1999, McLeod and White 2010). Schaeff *et al.* (1997) compared the genetic variability of North Atlantic and southern right whales (*E. australis*), and found the former to be significantly less diverse, a finding broadly replicated by Malik *et al.* (2000). The low diversity in North Atlantic right whales might indicate inbreeding, but no definitive conclusion can be reached using current data. Modern and historic genetic population structures were compared using DNA extracted from museum and archaeological specimens of baleen and bone. This work suggested that the eastern and western North Atlantic populations were not genetically distinct (Rosenbaum *et al.* 1997, 2000). However, the virtual extirpation of the eastern stock and its lack of recovery in the last hundred years strongly suggest population subdivision over a protracted (but not evolutionary) timescale. Genetic studies concluded that the principal loss of genetic diversity occurred prior to the 18th century (Waldick *et al.* 2002). However, revised conclusions that nearly all the remains in the North American Basque whaling archaeological sites were bowhead whales (*Balaena mysticetus*) and not right whales (Rastogi *et al.* 2004, McLeod *et al.* 2008) contradict the previously held belief that Basque whaling during the 16th and 17th centuries was principally responsible for the loss of genetic diversity.

High-resolution (i.e., using 35 microsatellite loci) genetic profiling improved our understanding of genetic variability, the number of reproductively active individuals, reproductive fitness, parentage, and relatedness of individuals (Frasier *et al.* 2007a, 2009). One finding of the genetic studies is the importance of obtaining biopsy samples from calves on the calving grounds. Between 1990 and 2010, only about 60% of all known calves were seen with their mothers in summering areas when their callosity patterns are stable enough to reliably make a photo-ID match later in life. The remaining 40% were not seen on a known summering ground. Because the calf's genetic profile is the only reliable way to establish parentage, if the calf is not sampled when associated with its mother early on, then it is not possible to link it with a calving event or to its mother, and information such as age and familial relationships is lost. From 1980 to 2001, there were 64 calves born that were not sighted later with their mothers and thus unavailable to provide age-specific mortality information (Frasier *et al.* 2007a). An additional interpretation of paternity analyses is that the population size may be larger than was previously thought. Fathers for only 45% of known calves have been genetically determined; yet, genetic profiles were available for 69% of all photo-identified males (Frasier 2005). The conclusion was that the majority of these calves must have different fathers that cannot be accounted for by the unsampled males, therefore the population of males must be larger (Frasier 2005, Frasier *et al.* 2007b). However, a recent study comparing photo-identification and pedigree genetic data for animals known or presumed to be alive during 1980–2016 found that the presumed alive estimate is similar to the actual abundance of this population, which indicates that the majority of the animals have been photo-identified (Fitzgerald 2018).

POPULATION SIZE

The western North Atlantic right whale stock size is based on a published state-space model of the sighting histories of individual whales identified using photo-identification techniques (Pace *et al*. 2017). Sightings histories were constructed from the photo-ID recapture database as it existed in October 2019, which included photographic information up through January 2018. Using a hierarchical, state-space Bayesian open population model of these histories produced a median abundance value (Nest) of 412 individuals (95%CI: 403–424; Table 1). As with any statistically-based estimation process, uncertainties exist in the estimation of abundance because it is based on a probabilistic model that makes certain assumptions about the structure of the data. Because the statistically-based uncertainty is asymmetric about N, the credible interval (CI) is used to characterize that uncertainty (as opposed to a CV that may appear in other stock assessment reports).

Table 1. Best and minimum abundance estimates for the western North Atlantic right whale (Eubalaena glacialis) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	95% Credible Interval	60% Credible Interval	N_{min}	F_r	R_{max}	PBR
412	403–424	408–416	408	0.1	0.04	0.8

Historical Abundance

The total North Atlantic right whale population size pre-whaling is estimated between 9,075 and 21,328 based on extrapolation of spatially explicit models of carrying capacity in the North Pacific (Monserrat *et al.* 2015). Basque whalers were thought to have taken right whales during the 1500s in the Strait of Belle Isle region (Aguilar 1986), however, genetic analysis has shown that nearly all of the remains found in that area are, in fact, those of bowhead whales (Rastogi *et al.* 2004, Frasier *et al.* 2007a). This stock of right whales may have already been substantially reduced by the time colonists in Massachusetts started whaling in the 1600s (Reeves *et al.* 2001, 2007). A modest but persistent whaling effort along the coast of the eastern U.S. lasted three centuries, and the records include one report of 29 whales killed in Cape Cod Bay in a single day in January 1700. Reeves *et al*. (2007) calculated that a minimum of 5,500 right whales were taken in the western North Atlantic between 1634 and 1950, with nearly 80% taken in a 50-year period between 1680 and 1730. They concluded "there were at least a few thousand whales present in the mid-1600s." The authors cautioned, however, that the record of removals is incomplete, the results were preliminary, and refinements are required. Based on back calculations using the present population size and growth rate, the population may have numbered fewer than 100 individuals by 1935 when international protection for right whales came into effect (Hain 1975, Reeves *et al.* 1992, Kenney *et al.* 1995). However, little is known about the population dynamics of right whales in the intervening years.

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% credible interval about the median of the posterior abundance estimates using the methods of Pace *et al*. (2017). This is roughly equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The median estimate of abundance for western North Atlantic right whales is 412. The minimum population estimate as of January 2018 is 408 individuals (Table 1).

Current Population Trend

The population growth rate reported for the period 1986–1992 by Knowlton *et al.* (1994) was 2.5% (CV=0.12), suggesting that the stock was recovering slowly, but that number may have been influenced by discovery phenomenon as existing whales were recruited to the catalog. Work by Caswell *et al.* (1999) suggested that crude survival probability declined from about 0.99 in the early 1980s to about 0.94 in the late 1990s. The decline was statistically significant. Additional work conducted in 1999 was reviewed by the IWC workshop on status and trends in this population (IWC 2001); the workshop concluded based on several analytical approaches that survival had indeed declined in the 1990s. Although capture heterogeneity could negatively bias survival estimates, the workshop concluded that this factor could not account for the entire observed decline, which appeared to be particularly marked in adult females. Another workshop was convened by NMFS in September 2002, and it reached similar conclusions regarding the decline in the population (Clapham 2002). At the time, the early part of the recapture series had not been examined for excessive retrospective recaptures which had the potential to positively bias the earliest estimates of survival as the catalog was being developed.

Examination of the abundance estimates for the years 1990–2011 (Figure 2) suggests that abundance increased at about 2.8% per annum from posterior median point estimates of 270 individuals in 1990 to 481 in 2011, but that there was a 100.00% chance that abundance declined from 2011 to 2018 when the final estimate was 412 individuals. The overall abundance decline between 2011 and 2018 was 14.35% (CI=11.67%–16.60%). There has been a considerable change in right whale habitat use patterns in areas where most of the population had been observed in previous years (e.g. Davies *et al*. 2017), exposing the population to additional anthropogenic threats (Hayes *et al.* 2018). This apparent change in habitat use has the effect that, despite relatively constant effort to find whales in traditional areas, the chance of photographically capturing individuals has decreased. However, the methods in Pace *et al.* (2017) account for changes in capture probability.

There were 17 right whale mortalities in 2017 (Daoust *et al.* 2018). This number exceeds the largest estimated mortality rate during the past 25 years. Further, despite high survey effort, only 5 and 0 calves were detected in 2017 and 2018, respectively.

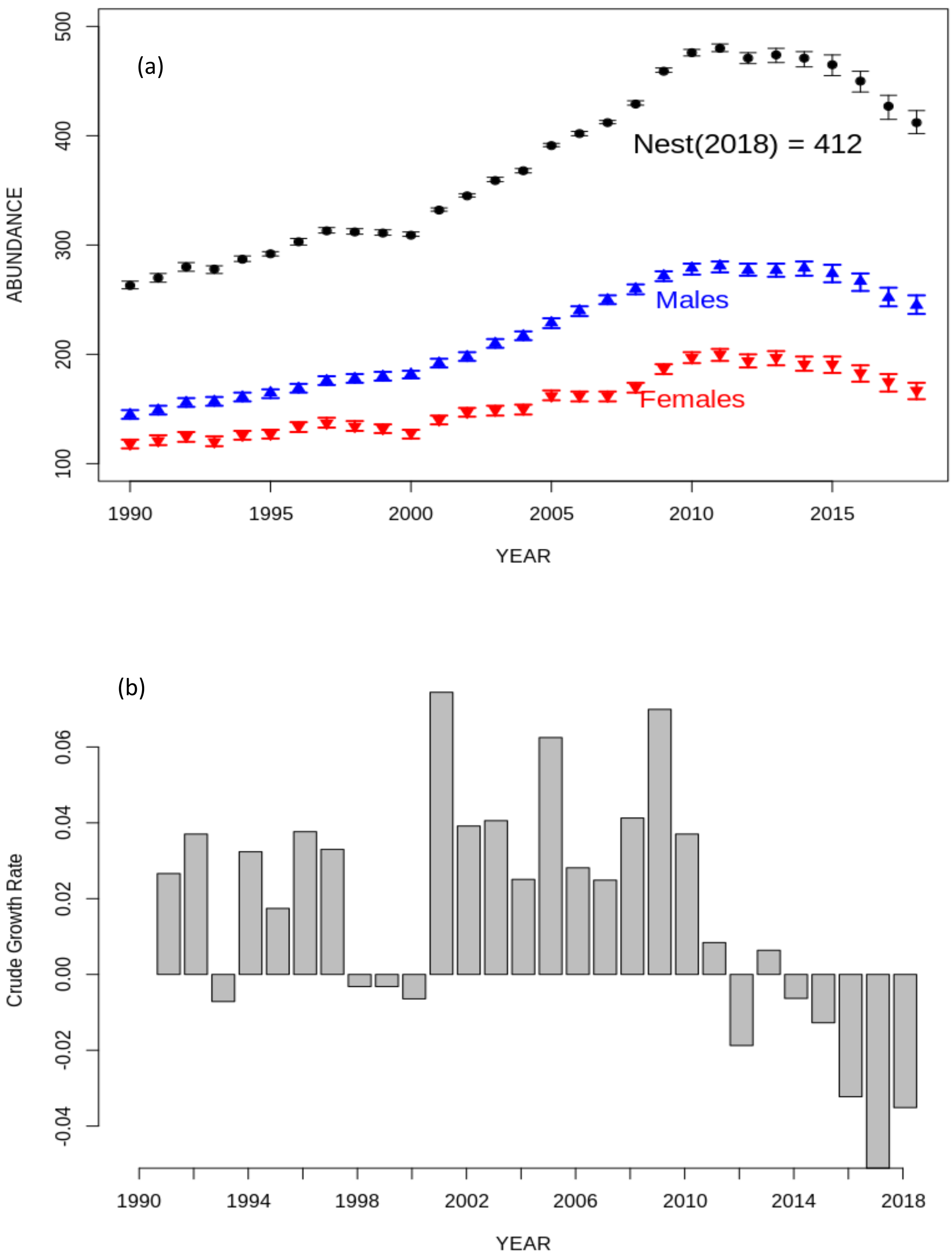


Figure 2. (a) Abundance estimates for North Atlantic right whales. Estimates are the median values of a posterior distribution from modeled capture histories. Also shown are sex-specific abundance estimates. Cataloged whales may include some but not all calves produced each year. (b) Crude annual growth rates from the abundance values.

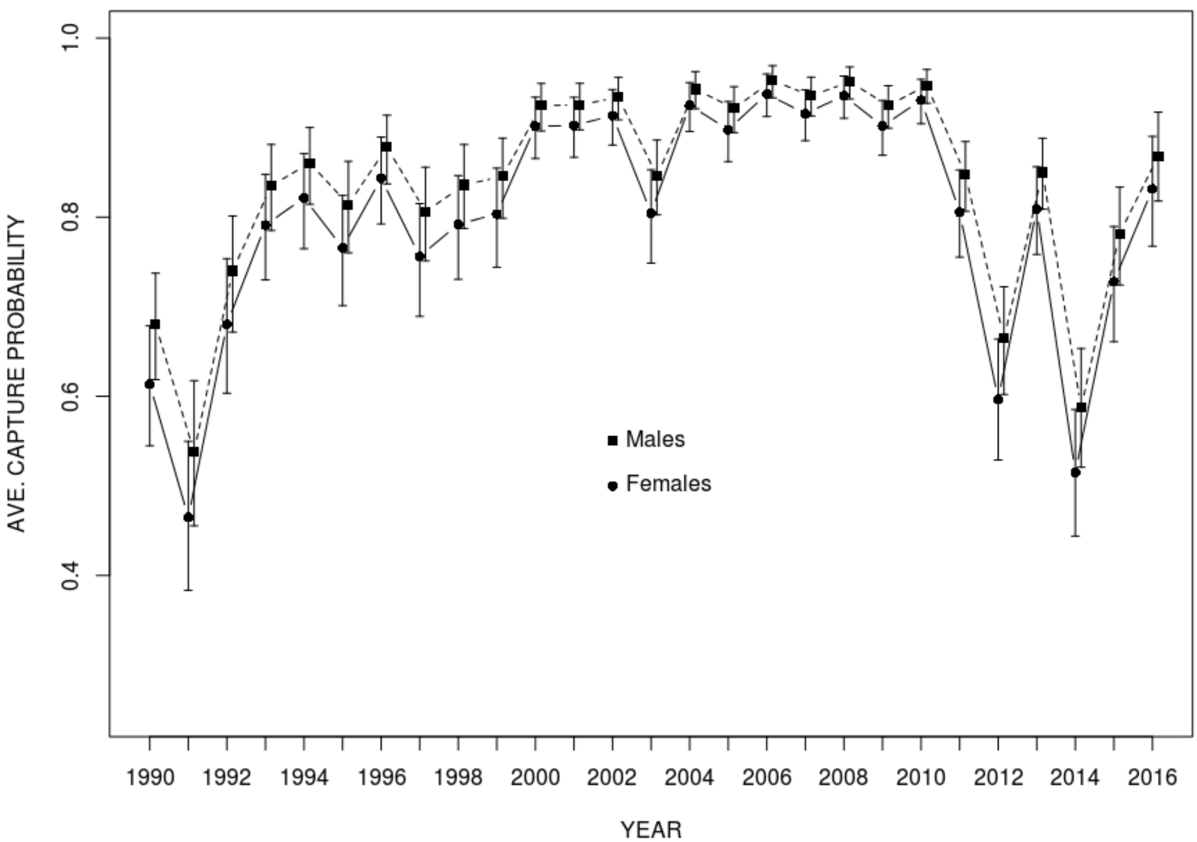


Figure 3. Estimated recapture probability and associated 95% credible intervals of North Atlantic right whales 1990–2016 based on a Bayesian MRR model allowing random fluctuation among years for survival rates, treating capture rates as fixed effects over time, and using both observed and known states as data (from Pace et al. 2017).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

During 1980–1992, at least 145 calves were born to 65 identified females. The number of calves born annually ranged from 5 to 17, with a mean of 11.2 (SE=0.90). The reproductively active female pool was static at approximately 51 individuals during 1987–1992. Mean calving interval, based on 86 records, was 3.67 years. There was an indication that calving intervals may have been increasing over time, although the trend was not statistically significant (P=0.083) (Knowlton *et al.* 1994). Since 1993, calf production has been more variable than a simple stochastic model would predict.

During 1990–2017, at least 447 calves were born into the population. The number of calves born annually ranged from 1 to 39, and averaged 16 but was highly variable (SD=8.9). No calves were born the winter of 2017–2018. The fluctuating abundance observed from 1990 to 2017 makes interpreting a count of calves by year less clear than measuring population productivity, which we index by the number of calves detected/estimated abundance (Apparent Productivity Index, or API). Productivity for this stock has been highly variable over time and has been characterized by periodic swings in per capita birth rates (Figure 3). Notwithstanding the high variability observed, as expected for a small population, productivity in North Atlantic right whales lacks a definitive trend. Corkeron *et al.* (2018) found that during 1990–2016, calf count rate increased at 1.98% per year with outlying years of very high and low calf production. This is approximately a third of that found for three different southern right whale (*Eubalaena australis)* populations during the same time period (5.3–7.2%).

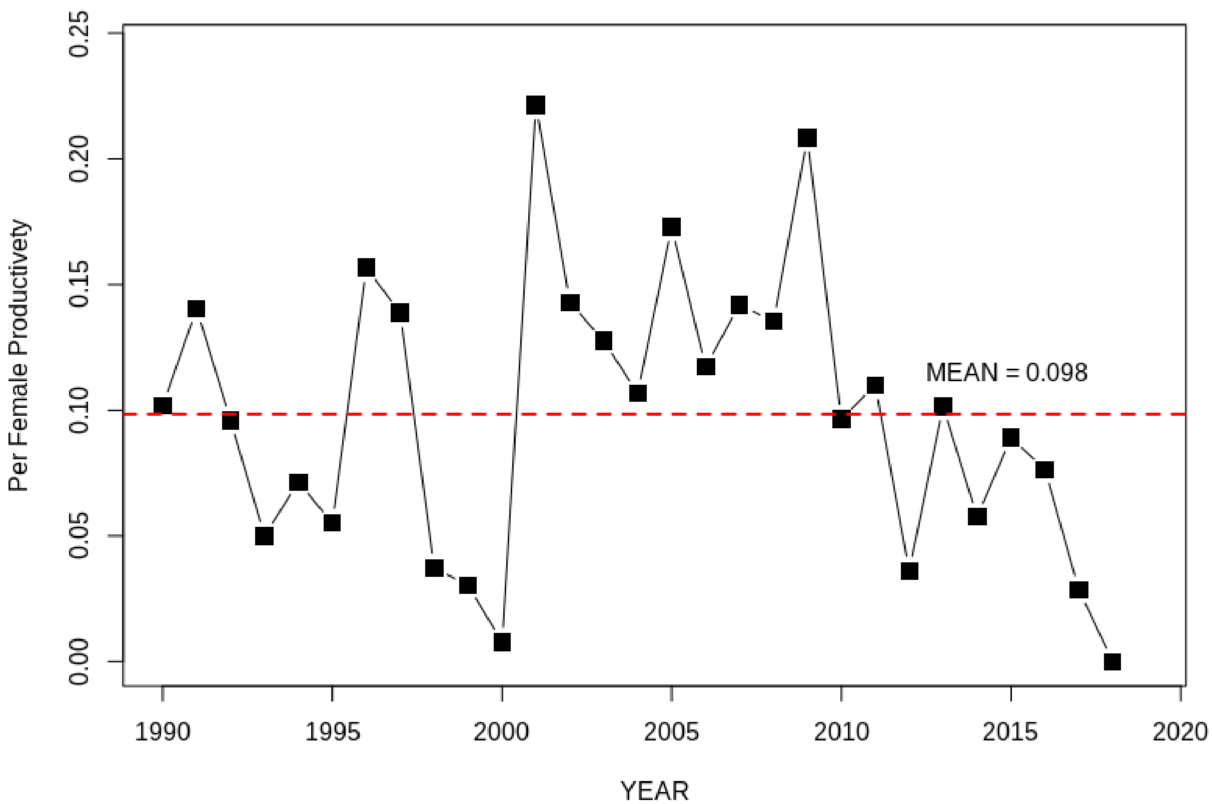


Figure 4. Productivity in the North Atlantic right whale population as characterized by calves detected/estimated number of females.

The available evidence suggests that at least some of the observed variability in the calving rates of North Atlantic right whales is related to variability in nutrition (Fortune *et al.* 2013) and possibly increased energy expenditures related to non-lethal entanglements (Rolland *et al.* 2016, Pettis *et al.* 2017, van der Hoop 2017, Christiansen *et al*. 2020).

An analysis of the age structure of this population suggests that it contains a smaller proportion of juvenile whales than expected (Hamilton *et al.* 1998, IWC 2001), which may reflect lowered recruitment and/or high juvenile mortality. Calf and perinatal mortality was estimated by Browning *et al.* (2010) to be between 17 and 45 animals during the period 1989 and 2003. In addition, it is possible that the apparently low reproductive rate is due in part to an unstable age structure or to reproductive dysfunction in some females. However, few data are available on either factor and senescence has not been documented for any baleen whale.

The maximum net productivity rate is unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be the default value of 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995). Projection models suggest that this rate could be 4% per year if female survival was the highest recorded over the time series from Pace *et al.* (2017). Reviewing the available literature, Corkeron *et al.* (2018) showed that female mortality is primarily anthropogenic, and concluded that anthropogenic mortality has limited the recovery of North Atlantic right whales. In a similar effort, Kenney (2018) back-projected a series of scenarios that varied entanglement mortality from observed to zero. Using a scenario with zero entanglement mortality, which included 15 'surviving' females, and a five year calving interval, the projected population size including 26 additional calf births would have been 588 by 2016. Single-year production has exceeded 0.04 in this population several times, but those outputs are not likely sustainable given the 3-year minimum interval required between successful calving events and the small fraction of reproductively active females. This is likely related to synchronous calving that can occur in capital breeders under variable environmental conditions. Hence, uncertainty exists as to whether the default value is representative of maximum net productivity for this stock, but it is unlikely that it is much higher than the default.

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is the product of minimum population size, one-half the maximum net productivity rate and a recovery factor for endangered, depleted, threatened stocks, or stocks of unknown status relative to OSP (MMPA Sec. 3. 16 U.S.C. 1362, Wade and Angliss 1997). The recovery factor for right whales is 0.1 because this species is listed as endangered under the Endangered Species Act (ESA). The minimum population size is 408. The maximum productivity rate is 0.04, the default value for cetaceans. PBR for the western North Atlantic stock of the North Atlantic right whale is 0.8 (Table 1).

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2014 through 2018, the average annual detected (i.e. observed) human-caused mortality and serious injury to right whales averaged 8.15 (Table 2). This is derived from two components: 1) incidental fishery entanglement records at 6.85 per year, and 2) vessel strike records averaging 1.3 per year.

Injury determinations are made based upon the best available information; these determinations may change with the availability of new information (Henry *et al.* 2021). Only records considered to be confirmed human-caused mortalities or serious injuries are reported in the observed mortality and serious injury (M/SI) rows of Table 2.

Annual rates calculated from detected mortalities should be considered a negatively-biased accounting of human-caused mortality; they represent a definitive lower bound. Detections are irregular, incomplete, and not the result of a designed sampling scheme. Research on other cetaceans has shown the actual number of deaths can be several times higher than that observed (Wells *et al.* 2015, Williams *et al.* 2011). The hierarchical Bayesian, state-space model used to estimate North Atlantic right whale abundance (Pace *et al.* 2017) can also be used to estimate total mortality. The estimated annual rate of total mortality using this modeling approach is 18.6 animals for the period 2013−2017 (Pace *et al.* 2021). This estimated total mortality accounts for detected mortality and serious injury, as well as undetected (cryptic) mortality within the population. Figure 5 compares the observed mortality and serious injury totals for the years 2000−2017 to the estimates of total mortality from the state-space model. The detection rate of mortality and serious injury for the 5-year period 2013−2017 was 51% of the model's annual mortality estimates (based on methods from Pace *et al*. 2021). The estimated mortality for 2018 is not yet available because it is derived from a comparison with the population estimate for 2019, which, in turn, is contingent on the processing of all photographs collected through 2019 for incorporation into the state-space model of the sighting histories of individual whales. At this time, we are unable to apportion estimated mortality by cause (fishery interaction vs. vessel strike) or by nationality (occurring in U.S vs. Canadian waters). However, an analysis of right whale mortalities between 2003 and 2018 found that of the examined non-calf carcasses for which cause of death could be determined, all mortality was human-caused (Sharpe *et al.* 2019). Based on these findings, 100% of the estimated mortality of 18.6 animals per year is assumed to be human-caused. This estimate of total annual human-caused mortality may be somewhat positively biased (i.e., a slight overestimate) given that some calf mortality is likely not human-caused.

Table 2. Average annual observed and estimated human-caused mortality and serious injury for the North Atlantic right whale (Eubalaena glacialis). Observed values are from confirmed interactions. Estimated total mortality is model-derived (Pace et al. 2017). Injuries prevented are a result of successful disentanglement efforts.

Years	Source	Annual Average
2014–2018	Observed incidental fishery interactions	6.85
2014–2018	Observed vessel collisions	1.30
2014–2018	Observed total human-caused M/SI	8.15
2013–2017	Estimated total mortality	18.6
2014–2018	SI prevented	1.2

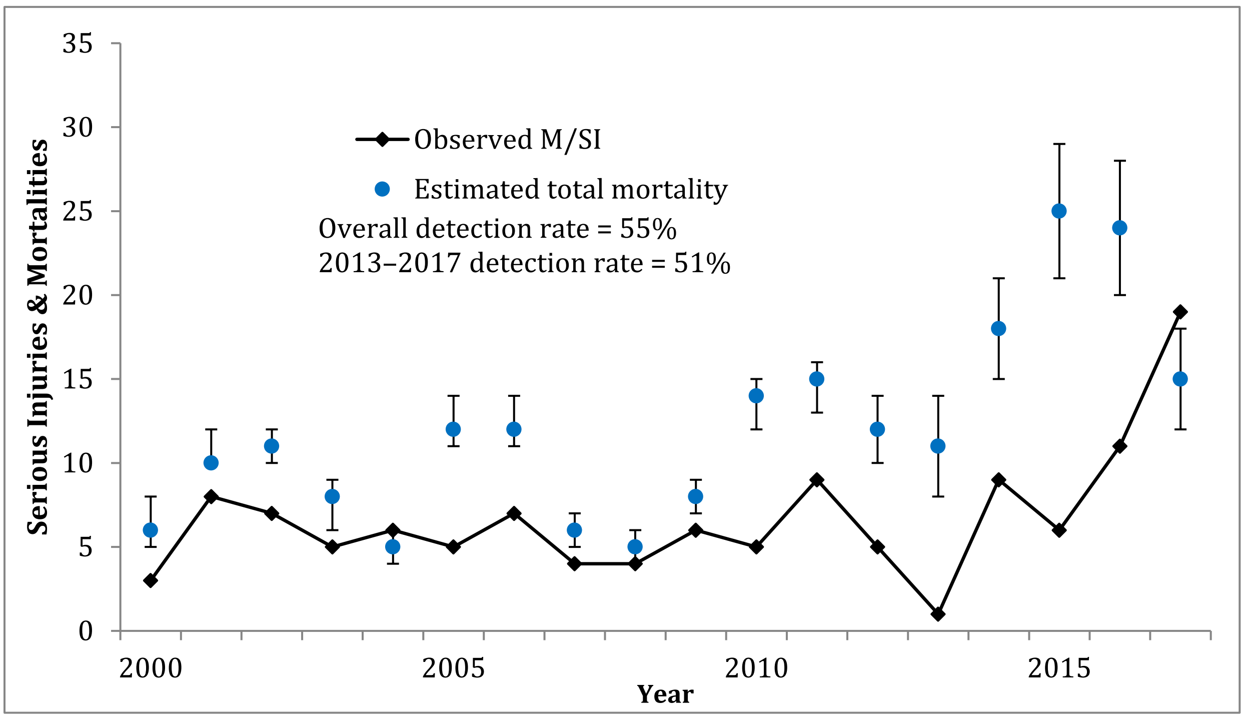


Figure 5. Time series of observed annual total mortality and serious injury (M/SI; black line) versus estimated total mortalities (blue points with associated error bars).

The small population size and low annual reproductive rate of right whales suggest that human sources of mortality have a greater effect relative to population growth rates than for other whales (Corkeron *et al.* 2018). The principal factor believed to be preventing growth and recovery of the population is entanglement with fishing gear (Kenny 2018). Between 1970 and 2018, a total of 124 right whale mortalities was recorded (Knowlton and Kraus 2001; Moore *et al.* 2005; Sharp *et al.* 2019). Of these, 18 (14.5%) were neonates that were believed to have died from perinatal complications or other natural causes. Of the remainder, 26 (21.0%) resulted from vessel strikes, 26 (21.0%) were related to entanglement in fishing gear, and 54 (43.5%) were of unknown cause. At a minimum, therefore, 42% of the observed total for the period and 43% of the 102 non-calf deaths were attributable to human impacts (calves accounted for six deaths from vessel strikes and two from entanglements). However, when considering only those cases where cause of death could be determined, 100% of non-calf mortality was human-caused. A recent analysis of human-caused serious injury and mortality during 2000–2017 shows that entanglement injuries have been increasing steadily over the past twenty years while injuries from vessel strikes have shown no specific trend despite several reported cases in 2017 (Hayes *et al.* 2018).

The details of a particular mortality or serious injury record often require a degree of interpretation (Moore *et al.* 2005, Sharp *et al.* 2019). The cause of death is based on analysis of the available data; additional information may result in revisions. When reviewing Table 3 below, several factors should be considered: 1) a vessel strike or entanglement may have occurred at some distance from the location where the animal is detected/reported; 2) the mortality or injury may involve multiple factors; for example, whales that have been both vessel struck and entangled are not uncommon; 3) the actual vessel or gear type/source is often uncertain; and 4) entanglements may involve several types of gear. Beginning with the 2001 Stock Assessment Report, Canadian records have been incorporated into the mortality and serious injury rates to reflect the effective range of this stock. However, because whales have been known to carry gear for long periods of time and over great distances before being detected, and recovered gear is often not adequately marked, it can be difficult to assign some entanglements to the country of origin.

It should be noted that entanglement and vessel collisions may not seriously injure or kill an animal directly, but may weaken or otherwise affect its reproductive success (van der Hoop *et al.* 2017, Corkeron *et al.* 2018). The NMFS serious injury determinations for large whales commonly include animals carrying gear when these entanglements are constricting or are determined to interfere with foraging (Henry *et al*. 2021). Successful disentanglement and subsequent resightings of these individuals in apparent good health are criteria for downgrading an injury to non-serious. However, these and other non-serious injury determinations should be considered to fully understand anthropogenic impacts to the population, especially in cases where females' fecundity may be affected.

Fishery-Related Mortality and Serious Injury

Not all mortalities are detected, but reports of known mortality and serious injury relative to PBR as well as total human impacts are contained in the records maintained by the New England Aquarium and the NMFS Greater Atlantic and Southeast Regional Offices. Records are reviewed and those determined to be human-caused are detailed in Table 3. Information from an entanglement event often does not include the detail necessary to assign the entanglements to a particular fishery or location.

Although disentanglement is often unsuccessful or not possible for many cases, there are several documented cases of entanglements for which the intervention by disentanglement teams averted a likely serious-injury determination. See Table 2 for annual average of serious injuries prevented by disentanglement.

Whales often free themselves of gear following an entanglement event, and as such scarring may be a better indicator of fisheries interaction than entanglement records. A review of scars detected on identified individual right whales over a period of 30 years (1980–2009) documented 1,032 definite, unique entanglement events on the 626 individual whales identified (Knowlton *et al*. 2012). Most individual whales (83%) were entangled at least once, and over half of them (59%) were entangled more than once. Hamilton *et al* (2019) estimated that 30.25% of the population was entangled annually between 2010 and 2017. Juveniles and calves were entangled at higher rates than were adults. Scarring rates suggest that entanglements occur at about an order of magnitude more often than detected from observations of whales with gear on them. Analyses of whales carrying entangling gear also suggest that entanglement wounds have become more severe since 1990, possibly due to increased use of stronger lines in fixed fishing gear (Knowlton *et al*. 2016).

Knowlton *et al.* (2012) concluded from their analysis of entanglement scarring rates from 1980–2009 that efforts of the prior decade to reduce right whale entanglement had not worked. Using a completely different data source (observed mortalities of eight large whale species, 1970–2009), van der Hoop *et al*. (2012) arrived at a similar conclusion. Similarly, Pace *et al.* (2015), analyzing entanglement rates and serious injuries due to entanglement during 1999–2009, found no support that mitigation measures implemented prior to 2009 had been effective at reducing takes due to commercial fishing. Since 2009, new entanglement mitigation measures (72 FR 193, 05 October 2007; 79 FR 124, 27 June 2014) have been implemented as part of the Atlantic Large Whale Take Reduction Plan, but their effectiveness has yet to be evaluated. One difficulty in assessing mitigation measures is the need for a statistically-significant time series to determine effectiveness.

Other Mortality

Vessel strikes are a major cause of mortality and injury to right whales (Kraus 1990, Knowlton and Kraus 2001, van der Hoop *et al.* 2012). Records from 2014 through 2018 have been summarized in Table 3. Early analyses of the effectiveness of the vessel-strike rule were reported by Silber and Bettridge (2012). Recently, van der Hoop *et al.* (2015) concluded that large whale mortalities due to vessel strikes decreased inside active seasonal management areas (SMAs) and increased outside inactive SMAs. Analysis by Laist *et al.* (2014) incorporated an adjustment for drift around areas regulated under the vessel-strike rule and produced weak evidence that the rule was effective inside the SMAs. When simple logistic regression models fit using maximum likelihood-based estimation procedures were applied to previously reported vessel strikes between 2000 and 2017 (Henry *et al.* 2021), there was no apparent trend (Hayes *et al.* 2018).

An Unusual Mortality Event was established for North Atlantic right whales in June 2017 due to elevated strandings along the Northwest Atlantic Ocean coast, especially in the Gulf of St. Lawrence region of Canada (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2020-north-atlantic-right-whale-unusual-mortality-event). There were 20 dead whales documented through December 2018, with 12 whales having evidence of vessel strike or entanglement as the preliminary cause of death. Additionally, seven free-swimming whales were documented as being seriously injured due to entanglements during the time period. Therefore, through December 2018, the number of whales included in the UME was 27, including 20 dead and 7 seriously injured free-swimming whales.

Table 3. Confirmed human-caused mortality and serious injury records of right whales: 2014–2018[a]

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
01/15/2014	Serious Injury	4394	off Ossabaw Island, GA	EN	1	XU	NP	No gear present but new ent. injuries indicating prior constricting gear on both pectorals and at fluke insertion. Injury to left ventral fluke. Evidence of health decline. No resights post Feb/2014.
04/01/2014	Serious Injury	1142	off Atlantic City, NJ	EN	1	XU	NR	Constricting rostrum wrap with line trailing to at least mid-body. Resighted in 2018. Health decline evident.
04/09/2014	Prorated Injury	-	Cape Cod Bay, MA	VS	0.52	US	-	Animal surfaced underneath a research vessel while it was underway (39 ft at 9 kts). Small amount of blood and some lacerations of unknown depth on lower left flank.
06/29/2014	Serious Injury	1131	off Cape Sable Island, NS	EN	1	XC	NR	At least 1, possibly 2, embedded rostrum wraps. Remaining configuration unclear but extensive. Animal in extremely poor condition: emaciated, heavy cyamid coverage, overall pale skin. No resights.
09/04/2014	Serious Injury	4001	off Grand Manan, NB	EN	1	XC	NR	Free-swimming with constricting rostrum wrap. Remaining configuration unknown. No resights post Oct/2014.
09/04/2014	Mortality	-	Far south of St. Pierre & Miquelon, off the south coast of NL	EN	1	XC	NR	Carcass with constricting line around rostrum and body. No necropsy conducted, but evidence of extensive, constricting entanglement supports entanglement as COD.
09/17/2014	Serious Injury	3279	off Grand Manan, NB	EN	1	XC	NR	Free-swimming with heavy, green line overhead cutting into nares. Remaining config. unk. In poor overall condition: heavy cyamids on head and blowholes. Left blowhole appears compromised. No resights.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
09/27/2014	Mortality	-	off Nantucket, MA	EN	1	US	NR	Fresh carcass with multiple lines wrapping around head, pectoral, and peduncle. Appeared to be anchored. No necropsy conducted, but extensive, constricting entanglement supports entanglement as COD.
12/18/2014	Serious Injury	3670	off Sapelo Sound, GA	EN	1	XU	NP	No gear present but new, healing entanglement injuries. Severe injuries to lip, peduncle and fluke edges. Poss. damage to right pectoral. Resights indicate health decline.
04/06/2015	Serious Injury	CT04CCB14	Cape Cod Bay, MA	EN	1	XU	NP	Encircling laceration at fluke insertion with potential to affect major artery. Source of injury likely constricting entanglement. No gear present. Evidence of health decline. No resights.
06/13/2015	Prorated Injury	-	off Westport, NS	EN	.75	XC	NR	Line through mouth, trailing 300-400m ending in 2 balloon-type buoys. Full entanglement configuration unknown. No resights.
09/28/2015	Prorated Injury	-	off Cape Elizabeth, ME	EN	.75	XU	NR	Unknown amount of line trailing from flukes. Attachment point(s) and configuration unknown. No resights.
11/29/2015	Serious Injury	3140	off Truro, MA	EN	1	XU	NR	New, significant ent. injuries indicating constricting wraps. No gear visible. In poor cond. with grey skin and heavy cyamid coverage. No resights.
01/29/2016	Serious Injury	1968	off Jupiter Inlet, FL	EN	1	XU	NP	No gear present, but evidence of recent entanglement of unknown configuration. Significant health decline: emaciated, heavy cyamid coverage, damaged baleen. Resighted in April 2017 still in poor cond.
05/19/2016	Serious Injury	3791	off Chatham, MA	EN	1	XU	NP	New entanglement injuries on peduncle. Left pectoral appears compromised. No gear seen. Significant health decline: emaciated with heavy cyamid coverage. No resights post Aug/2016.
05/03/2016	Mortality	4681	Morris Island, MA	VS	1	US	-	Fresh carcass with 9 deep ventral lacerations. Multiple shorn and/or fractured vertebral and skull bones. Destabilized thorax. Edema, blood clots, and hemorrhage associated with injuries. Proximate COD=sharp trauma. Ultimate COD= exsanguination.
07/26/2016	Serious Injury	1427	Gulf of St Lawrence, QC	EN	1	XC	NP	No gear present, but new entanglement injuries on peduncle and fluke insertions. No gear present. Resights show subsequent health decline: gray skin, rake marks, cyamids.
08/01/2016	Serious Injury	3323	Bay of Fundy, NS	EN	1	XC	NP	No gear present, but new, severe entanglement injuries on peduncle, fluke insertions, and leading edges of flukes. No gear present. Significant health decline: emaciated, cyamids patches, peeling skin. No resights.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
08/13/2016	Serious Injury	4057	Bay of Fundy, NS	EN	1	CN	PT	Free-swimming with extensive entanglement. Two heavy lines through mouth, multiple loose body wraps, multiple constricting wraps on both pectorals with lines across the chest, jumble of gear by left shoulder. Partially disentangled: left with line through mouth and loose wraps at right flipper that are expected to shed. Significant health decline: extensive cyamid coverage. Current entanglement appears to have exacerbated injuries from previous entanglement (see 16Feb2014 event). No resights.
08/16/2016	Prorated Injury	1152	off Baccaro, NS	EN	0.75	XC	NR	Free-swimming with line and buoy trailing from unknown attachment point(s). No resights.
08/28/2016	Serious Injury	2608	off Brier Island, NS	EN	1	XC	NR	Free-swimming with constricting wraps around rostrum and right pectoral. Line trails 50 ft aft of flukes. Significant health decline: heavy cyamid coverage and indication of fluke deformity. No resights.
08/31/2016	Mortality	4320	Sable Island, NS	EN	1	CN	PT	Decomposed carcass with multiple constricting wraps on pectoral with associated bone damage consistent with chronic entanglement.
09/23/2016	Mortality	3694	off Seguin Island, MA	EN	1	CN	PT	Fresh, floating carcass with extensive, constricting entanglement. Thin blubber layer and other findings consistent with prolonged stress due to chronic entanglement. Gear previously reported as unknown.
12/04/2016	Prorated Injury	3405	off Sandy Hook, NJ	EN	0.75	XU	NE	Lactating female. Free-swimming with netting crossing over blowholes and one line over back. Full configuration unknown. Calf not present, possibly already weaned. No resights. Gear type previously reported as NR.
04/13/2017	Mortality	4694	Cape Cod Bay, MA	VS	1	US	-	Carcass with deep hemorrhaging and muscle tearing consistent with blunt force trauma.
06/19/2017	Mortality	1402	Gulf of St Lawrence, QC	VS	1	CN	-	Carcass with acute internal hemorrhaging consistent with blunt force trauma.
06/21/2017	Mortality	3603	Gulf of St Lawrence, QC	EN	1	CN	PT	Fresh carcass found anchored in at least 2 sets of gear. Multiple lines through mouth and constricting wraps on left pectoral. Glucorticoid levels support acute entanglement as COD.
06/23/2017	Mortality	1207	Gulf of St Lawrence, QC	VS	1	CN	-	Carcass with acute internal hemorrhaging consistent with blunt force trauma.
07/04/2017	Serious Injury	3139	off Nantucket, MA	EN	1	XU	NP	No gear present, but evidence of recent extensive, constricting entanglement and health decline. No resights.
07/06/2017	Mortality	-	Gulf of St Lawrence, QC	VS	1	CN	-	Carcass with fractured skull and associated hemorrhaging. Glucorticoid levels support acute blunt force trauma as COD.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
07/19/2017	Serious Injury	4094	Gulf of St Lawrence, QC	EN	1	CN	PT	Line exiting right mouth, crossing over back, ending at buoys aft of flukes. Non-constricting configuration, but evidence of significant health decline. No resights.
07/19/2017	Mortality	2140	Gulf of St Lawrence, QC	VS	1	CN	-	Fresh carcass with acute internal hemorrhaging. Glucorticoid levels support acute blunt force trauma as COD.
08/06/2017	Mortality	-	Martha's Vineyard, MA	EN	1	XU	NP	No gear present, but evidence of constricting wraps around both pectorals and flukes with associated tissue reaction. Histopathology results support entanglement as COD.
09/15/2017	Mortality	4504	Gulf of St Lawrence, QC	EN	1	CN	PT	Anchored in gear with extensive constricting wraps with associated hemorrhaging.
10/23/2017	Mortality	-	Nashawena Island, MA	EN	1	XU	NP	No gear present, but evidence of extensive ent involving pectorals, mouth, and body. Hemorrhaging associated with body and right pectoral injuries. Histo results support entanglement as COD.
01/22/2018	Mortality	3893	55 nm E of Virginia Beach, VA	EN	1	CN	PT	Extensive, severe constricting entanglement including partial amputation of right pectoral accompanied by severe proliferative bone growth. COD - chronic entanglement.
02/15/2018	Serious Injury	3296	33 nm E of Jekyll Island, GA	EN	1	XU	NP	No gear present, but extensive recent injuries consistent with constricting gear on right flipper, peduncle, and leading fluke edges. Large portion of right lip missing. Extremely poor condition - emaciated with heavy cyamid load. No resights.
07/13/2018	Prorated Injury	3312	25.6 nm E of Miscou Island, NB	EN	0.75	CN	NR	Free swimming with line through mouth and trailing both sides. Full configuration unknown - unable to confirm extent of flipper involvement. No resights.
07/30/2018	Prorated Injury	3843	13 nm E of Grand Manan, NB	EN	0.75	XC	GU	Free-swimming with buoy trailing 70ft behind whale. Attachment point(s) unknown. Severe, deep, raw injuries on peduncle & head. Partial disentanglement. Resighted with line exiting left mouth and no trailing gear. Possible rostrum and left pectoral wraps, but unable to confirm. Improved health, but final configuration unclear. No additional resights.
08/25/2018	Mortality	-	Martha's Vineyard, MA	EN	1	XU	NP	No gear present. Evidence of constricting pectoral wraps with associated hemorrhaging. COD - acute entanglement
10/14/2018	Mortality	3515	134 nm E of Nantucket, MA	EN	1	XU	NP	No gear present, but evidence of constricting wraps across ventral surface and at pectorals. COD - acute, severe entanglement.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
12/20/2018	Prorated Injury	2310	Nantucket, MA	EN	0.75	XU	NR	Free-swimming with open bridle through mouth. Resight in Apr2019 shows configuration changed, but unable to determine full configuration. Health appears stable.No additional resights
12/24/2018	Serious Injury	3208	South of Nantucket, MA	EN	1	XU	NP	No gear present. Evidence of new, healed, constricting body wrap. Health decline evident - grey, lesions, thin.

Assigned Cause	Five-year Mean (US/CN/XU/XC)
Vessel strike	1.3 (0.50/0.80/0.00/0.00)
Entanglement	6.85 (0.20/1.55/3.25/1.85)

a. For more details on events, see Henry *et al.* 2021.
b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. CN=Canada, US=United States, XC=Unassigned 1st sight in CN, XU=Unassigned 1st sight in US.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

HABITAT ISSUES

Baumgartner *et al.* (2017) discuss that ongoing and future environmental and ecosystem changes may displace *C. finmarchicus*, or disrupt the mechanisms that create very dense copepod patches upon which right whales depend. One of the consequences of this may be a shift of right whales into different areas with additional anthropogenic impacts to the species. Record *et al.* (2019) described the effects of a changing oceanographic climatology in the Gulf of Maine on the distribution of right whales and their prey. The warming conditions in the Gulf have altered the availability of late stage *C. finmarchicus* to right whales, resulting in a sharp decline in sightings in the Bay of Fundy and Great South Channel over the last decade (Record *et al.* 2019, Davies *et al*. 2019), and an increase in sightings in Cape Cod Bay (Ganley *et al.* 2019).

In addition, construction noise and vessel traffic from planned development of offshore wind in southern New England and the mid-Atlantic could result in communication masking, increased risk of vessel strike or avoidance of wind energy areas. Offshore wind turbines could also influence the hydrodynamics of seasonal stratification and ocean mixing, which, in turn, could influence shelf-wide primary production and copepod distribution (Broström 2008, Carpenter *et al.* 2016, Afsharian *et al*. 2020).

STATUS OF STOCK

The size of this stock is considered to be extremely low relative to OSP in the U.S. Atlantic EEZ. This species is listed as endangered under the ESA and has been declining since 2011 (Pace *et al.* 2017). The North Atlantic right whale is considered one of the most critically endangered populations of large whales in the world (Clapham *et al.* 1999, NMFS 2017). The total level of human-caused mortality and serious injury is unknown, but the reported (and clearly biased low) human-caused mortality and serious injury was a minimum of 6.65 right whales per year from 2013 through 2017. Given that PBR has been calculated as 0.8, human-caused mortality or serious injury for this stock must be considered significant. This is a strategic stock because the average annual human-related mortality and serious injury exceeds PBR, and also because the North Atlantic right whale is an endangered species. All ESA-listed species are classified as strategic by definition; therefore, any uncertainties discussed above will not affect the status of stock.

REFERENCES CITED

Afsharian, S., P.A. Taylor and L. Momayez. 2020. Investigating the potential impact of wind farms on Lake Erie. J. Wind Eng. Ind. Aerod. 198:104049. https://doi.org/10.1016/j.jweia.2019.104049

Aguilar, A. 1986. A review of old Basque whaling and its effect on the right whales of the North Atlantic. Rep. Int. Whal. Comm. (Special Issue) 10:191–199.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Baumgartner, M.F. and B.R. Mate. 2005. Summer and fall habitat of North Atlantic right whales (*Eubalaena glacialis*) inferred from satellite telemetry. Can. J. Fish. Aquat. Sci. 62:527–543.

Bort, J., S. Van Parijs, P. Stevick, E. Summers and S. Todd. 2015. North Atlantic right whale *Eubalaena glacialis* vocalization patterns in the central Gulf of Maine from October 2009 through October 2010. Endanger. Species Res. 26:271–280.

Broström, G. 2008. On the influence of large wind farms on the upper ocean circulation. J. Mar. Syst. 74:585–591.

Brown, M.W. and M.K. Marx. 2000. Surveillance, monitoring and management of North Atlantic right whales, *Eubalaena glacialis*, in Cape Cod Bay, Massachusetts: January to mid-May, 2000. Final report. Division of Marine Fisheries. Boston, Massachusetts. 52pp. http://www.mass.gov/eea/docs/dfg/dmf/programsandprojects/rwhale00.pdf

Browning, C.L., R.M. Rolland and S.D. Kraus. 2010. Estimated calf and perinatal mortality in western North Atlantic right whales (*Eubalaena glacialis*). Mar. Mamm. Sci. 26:648–662.

Carpenter, J.R., L. Merckelbach, U. Callies, S. Clark, L. Gaslikova and B. Baschek. 2016. Potential impacts of offshore wind farms on North Sea stratification. PLoS One 11:e0160830.

Caswell, H., S. Brault and M. Fujiwara. 1999. Declining survival probability threatens the North Atlantic right whale. Proc. Natl. Acad. Sci. USA 96:3308–3313.

Christiansen, F., S.M. Dawson, J.W. Durban, H. Fearnbach, C.A. Miller, L. Bejder, M. Uhart, M. Sironi, P. Corkeron, W. Rayment, E. Leunissen, E. Haria, R. Ward, H.A. Warick, I. Kerr, M.S. Lynn, H.M. Pettis and M.J. Moore. 2020. Population comparison of right whale body condition reveals poor state of the North Atlantic right whale. Mar. Ecol. Prog. Ser. 640:1–16.

Clapham, P.J. (Ed.). 2002. Report of the working group on survival estimation for North Atlantic right whales. Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543.

Cole, T.V.N., P. Hamilton, A.G. Henry, P. Duley, R.M. Pace III, B.N. White and T. Frasier. 2013. Evidence of a North Atlantic right whale *Eubalaena glacialis* mating ground. Endanger. Species Res. 21:55–64.

Cole, T.V.N., P. Duley, M. Foster, A. Henry and D.D. Morin. 2016. 2015 Right Whale Aerial Surveys of the Scotian Shelf and Gulf of St. Lawrence. Northeast Fish. Sci. Cent. Ref. Doc. 16-02. 14pp.

Corkeron, P., P. Hamilton, J. Bannister, P. Best, C. Charlton, K.R. Groch, K. Findlay, V. Rowntree, E. Vermeulen, and R.M. Pace. 2018. The recovery of North Atlantic right whales, *Eubalaena glacialis*, has been constrained by human-caused mortality. R. Soc. Open Sci. 5:180892.

Davies, K.T.A., M.W. Brown, P.K. Hamilton, A.R Knowlton, C.T. Taggart, A.S.M. Vanderlaan. 2019. Variation in North Atlantic right whale (*Eubalaena glacialis*) occurrence in the Bay of Fundy, Canada, over three decades. Endanger. Species Res. 39:159–171.

Davis, G.E., M.F. Baumgartner, J.M. Bonnell, J. Bell, C. Berchok, J.B. Thornton, S. Brault, G. Buchanan, R.A. Charif, D. Cholewiak, C.W. Clark, P. Corkeron, J. Delarue, K. Dudzinski, L. Hatch, J. Hildebrand, L. Hodge, H. Klinck, S. Kraus, B. Martin, D.K. Mellinger, H. Moors-Murphy, S. Nieukirk, D.P. Nowacek, S. Parks, A.J. Read, A.N. Rice, D. Risch, A. Širović, M. Soldevilla, K. Stafford, J.E. Stanistreet, E. Summers, S. Todd, A. Warde and S.M. Van Parijs. 2017. Long-term passive acoustic recordings track the changing distribution of North Atlantic right whales (*Eubalaena glacialis*) from 2004 to 2014. Sci. Rep. 7:13460.

Daoust, P.-Y., E.L. Couture, T. Wimmer and L. Bourque. 2018. Incident Report: North Atlantic right whale mortality event in the Gulf of St. Lawrence, 2017. Collaborative report produced *by:* Canadian Wildlife Health Cooperative, Marine Animal Response Society, and Fisheries and Oceans Canada. 256pp.

Fitzgerald, Kayla. 2018. Combining genetic and photo-identification data to improve abundance estimates for the North Atlantic right whale. Master's Thesis. Saint Mary's University, Halifax, Nova Scotia. 32pp.

Fortune, S.M.E., A.W. Trites, C.A. Mayo, D.A.S. Rosen and P.K. Hamilton. 2013. Energetic requirements of North Atlantic right whales and the implications for species recovery. Mar. Ecol. Prog. Ser. 478:253–272.

Frasier, T.R. 2005. Integrating genetic and photo-identification data to assess reproductive success in the North Atlantic right whale (*Eubalaena glacialis*). PhD thesis. McMaster University, Hamilton, Ontario. 197 pp.

Frasier, T.R., B.A. McLeod, R.M. Gillett, M.W. Brown and B.N. White. 2007a. Right whales past and present as revealed by their genes. Pages 200–231 *in*: S.D. Kraus and R.M. Rolland (Eds.). The urban whale: North Atlantic right whales at the crossroads. Harvard University Press, Cambridge, Massachusetts.

Frasier, T.R., P.K. Hamilton, M.W. Brown, L.A. Conger, A.R. Knowlton, M.K. Marx, C.K. Slay, S.D. Kraus and B.N. White. 2007b. Patterns of male reproductive success in a highly promiscuous whale species: The endangered North Atlantic right whale. Mol. Ecol. 16:5277–5293.

Frasier, T.R., P.K. Hamilton, M.W. Brown, S.D. Kraus and B.N. White. 2009. Sources and rates of errors in methods of individual identification for North Atlantic right whales. J. Mammal. 90(5):1246–1255.

Ganley, L.C., S. Brault, and C.A. Mayo. 2019. What we see is not what there is: Estimating North Atlantic right whale *Eubalaena glacialis* local abundance. Endanger. Species Res. 38:101–113.

Hain, J.H.W. 1975. The international regulation of whaling. Marine Affairs J. 3:28–48.

Hamilton, P.K., A.R. Knowlton and M.K. Marx. 2007. Right whales tell their own stories: The photo-identification catalog. Pages 75–104 *in*: S.D. Kraus and R.M. Rolland (Eds.). The urban whale: North Atlantic right whales at the crossroads. Harvard University Press, Cambridge, Massachusetts.

Hamilton, P.K., A.R. Knowlton, M.K. Marx and S.D. Kraus. 1998. Age structure and longevity in North Atlantic right whales *Eubalaena glacialis* and their relation to reproduction. Mar. Ecol. Prog. Ser. 171:285–292.

Hamilton, P.K., A.R. Knowlton, M.N. Hagbloom, K.R. Howe, H.M. Pettis, M.K. Marx, M.A. Zani and S.D. Kraus. 2019. Maintenance of the North Atlantic right whale catalog, whale scarring and visual health databases, anthropogenic injury case studies, and near real-time matching for biopsy efforts, entangled, injured, sick or dead right whales. New England Aquarium, Boston, MA. Contract No. 1305M2-18-P-NFFM-0108.

Hayes, S.A., S. Gardner, L. Garrison, A. Henry and L. Leandro. 2018. North Atlantic right whales – Evaluating their recovery challenges in 2018. NOAA Tech Memo NMFS-NE 247. 24pp.

Henry, A.G., T.V.N. Cole, L. Hall, W. Ledwell, D. Morin and A. Reid. 2021. Mortality and serious injury determinations for baleen whale stocks along the Gulf of Mexico, United States East Coast and Atlantic Canadian Provinces, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-07.

Hodge, K., C. Muirhead, J. Morano, C. Clark and A. Rice. 2015. North Atlantic right whale occurrence near wind energy areas along the mid-Atlantic US coast: Implications for management. Endanger. Species Res. 28:225–234.

IWC [International Whaling Commission]. 2001. Report of the workshop on the comprehensive assessment of right whales: A worldwide comparison. J. Cetacean Res. Manage. (Special Issue) 2:1–60.

Jacobsen, K., M. Marx and N. Øien. 2004. Two-way trans-Atlantic migration of a North Atlantic right whale (*Eubalaena glacialis*). Mar. Mamm. Sci. 20:161–166.

Johnson, A., G. Salvador, J. Kenney, J. Robbins, S. Kraus, S. Landry and P. Clapham. 2005. Fishing gear involved in entanglements of right and humpback whales. Mar. Mamm. Sci. 21:635–645.

Kenney, R.D. 2018. What if there were no fishing? North Atlantic right whale population trajectories without entanglement mortality. Endanger. Species Res. 37:233–237.

Kenney, R.D., M.A.M. Hyman, R.E. Owen, G.P. Scott and H.E. Winn. 1986. Estimation of prey densities required by western North Atlantic right whales. Mar. Mamm. Sci. 2:1–13.

Kenney, R.D., H.E. Winn and M.C. Macaulay. 1995. Cetaceans in the Great South Channel, 1979–1989: Right whale (*Eubalaena glacialis*). Cont. Shelf Res. 15:385–414.

Knowlton, A.R. and S.D. Kraus. 2001. Mortality and serious injury of North Atlantic right whales (*Eubalaena glacialis*) in the North Atlantic Ocean. J. Cetacean Res. Manage. (Special Issue) 2:193–208.

Knowlton, A.R., S.D. Kraus and R.D. Kenney. 1994. Reproduction in North Atlantic right whales (*Eubalaena glacialis*). Can. J. Zool. 72:1297–1305.

Knowlton, A.R., J. Sigurjonsson, J.N. Ciano and S.D. Kraus. 1992. Long-distance movements of North Atlantic right whales (*Eubalaena glacialis*). Mar. Mamm. Sci. 8:397–405.

Knowlton, A.R., P.K. Hamilton, M.K. Marx, H.M. Pettis and S.D. Kraus. 2012. Monitoring North Atlantic right whale *Eubalaena glacialis* entanglement rates: A 30 year retrospective. Mar. Ecol. Prog. Ser. 466:293–302.

Knowlton, A.R., J. Robbins, S. Landry, H.A. McKenna, S.D. Kraus and T.B. Werner. 2016. Effects of fishing rope strength on the severity of large whale entanglements. Conserv. Biol. 30:318–328. https://doi.org/10.1111/cobi.12590

Kraus, S.D. 1990. Rates and potential causes of mortality in North Atlantic right whales (*Eubalaena glacialis*). Mar. Mamm. Sci. 6:278–291.

Laist, D.W., A.R. Knowlton and D. Pendleton. 2014. Effectiveness of mandatory vessel speed limits for protecting North Atlantic Right Whales. Endanger. Species Res. 23:133–147.

Leiter, S.M, K.M. Stone, J.L. Thompson, C.M. Accardo, B.C. Wikgren, M.A. Zani, T.V.N. Cole, R.D. Kenney, C.A. Mayo and S.D. Kraus. 2017. North Atlantic right whale *Eubalaena glacialis* occurrence in offshore wind energy areas near Massachusetts and Rhode Island, USA. Endang. Species Res. 34:45–59.

Malik, S., M.W. Brown, S.D. Kraus, A. Knowlton, P. Hamilton and B.N. White. 1999. Assessment of genetic structuring and habitat philopatry in the North Atlantic right whale (*Eubalaena glacialis*). Can. J. Zool. 77:1217–1222.

Malik, S., M.W. Brown, S.D. Kraus and B.N. White. 2000. Analysis of mitochondrial DNA diversity within and between North and South Atlantic right whales. Mar. Mamm. Sci. 16:545–558.

Mate, B.M., S.L. Nieukirk and S.D. Kraus. 1997. Satellite-monitored movements of the northern right whale. J. Wildl. Manage. 61:1393–1405.

Mayo, C.A. and M.K. Marx. 1990. Surface foraging behaviour of the North Atlantic right whale, *Eubalaena glacialis*, and associated zooplankton characteristics. Can. J. Zool. 68:2214–2220.

Mayo, C.A, L. Ganley, C.A. Hudak, S. Brault, M.K. Marx, E. Burke and M.W. Brown. 2018. Distribution, demography, and behavior of North Atlantic right whales (*Eubalaena glacialis*) in Cape Cod Bay, Massachusetts, 1998–2013. Mar. Mamm. Sci. 34(4):979–996.

McLeod, B., M. Brown, M. Moore, W. Stevens, S.H. Barkham, M. Barkham and B. White. 2008. Bowhead whales, and not right whales, were the primary target of 16th- to 17th-century Basque whalers in the western North Atlantic. Arctic. 61:61–75.

McLeod, B.A. and B.N. White. 2010. Tracking mtDNA heteroplasmy through multiple generations in the North Atlantic right whale (*Eubalaena glacialis*). J. Hered. 101:235–239.

Mellinger, D.K, S.L.Nieukirk, K. Klink, H. Klink, R.P. Dziak, P.J. Clapham and B Brandsdóttir. 2011. Confirmation of right whales near a nineteenth-century whaling ground east of southern Greenland. Biol. Lett. 7:411–413.

Miller, C., D. Reeb, P. Best, A. Knowlton, M. Brown and M. Moore. 2011. Blubber thickness in right whales *Eubalaena glacialis* and *Eubalaena australis* related with reproduction, life history status and prey abundance. Mar. Ecol. Prog. Ser. 438:267–283.

Monserrat, S., M.G. Pennino, T.D. Smith, R.R. Reeves, C.N. Meynard, D.M. Kaplan and A.S.L. Rodrigues. 2015. A spatially explicit estimate of the prewhaling abundance of the endangered North Atlantic right whale. Conserv. Biol. 30:783–791.

Moore, J.C. and E. Clark. 1963. Discovery of right whales in the Gulf of Mexico. Science. 141:269.

Moore, M.J., A.R. Knowlton, S.D. Kraus, W.A. McLellan and R.K. Bonde. 2005. Morphometry, gross morphology and available histopathology in North Atlantic right whale (*Eubalaena glacialis*) mortalities. J. Cetacean Res. Manag. 6:199–214.

Morano, J.L., A.N. Rice, J.T. Tielens, B.J. Estabrook, A. Murray, B.L. Roberts and C.W. Clark. 2012. Acoustically detected year-round presence of right whales in an urbanized migration corridor. Conserv. Biol. 26:698–707.

NMFS [National Marine Fisheries Service]. 2015. Critical habitat for endangered North Atlantic right whale. Federal Register. 80:9314–9345.

NMFS [National Marine Fisheries Service]. 2017. North Atlantic right whale (*Eubalaena glacialis*) 5-year review: Summary and evaluation. NMFS Greater Atlantic Regional Fisheries Office, Gloucester, Massachusetts. 34pp. Available from:
https://www.greateratlantic.fisheries.noaa.gov/protected/final_narw_5-year_review_2017.pdf

Pace, R.M, III, T.V.N. Cole and A.G. Henry. 2015. Incremental fishing gear modifications fail to significantly reduce large whale serious injury rates. Endanger. Species Res. 26:115–126.

Pace, R.M., III, P.J. Corkeron and S.D. Kraus. 2017. State-space mark-recapture estimates reveal a recent decline in abundance of North Atlantic right whales. Ecol. Evol. 7:8730–8741. https://doi.org/10.1002/ece3.3406

Pace, RM, III, R. Williams, S.D. Kraus, A.R. Knowlton and H.M. Pettis. 2021. Cryptic mortality of North Atlantic right whales. Conserv. Sci. Pract. https://doi.org/10.1111/csp2.346

Patrician, M.R., I.S. Biedron, H.C. Esch, F.W. Wenzel, L.A. Cooper, P.K. Hamilton, A.H. Glass and M.F. Baumgartner. 2009. Evidence of a North Atlantic right whale calf (*Eubalaena glacialis*) born in northeastern U.S. waters. Mar. Mamm. Sci. 25:462–477.

Pettis, H.M., R.M. Rolland, P.K. Hamilton, A.R. Knowlton, E.A. Burgess and S.D. Kraus. 2017. Body condition changes arising from natural factors and fishing gear entanglements in North Atlantic right whales *Eubalaena glacialis*. Endang. Species Res. 32:237–249.

Rastogi, T., M.W. Brown, B.A. McLeod, T.R. Frasier, R. Grenier, S.L. Cumbaa, J. Nadarajah and B.N. White. 2004. Genetic analysis of 16th-century whale bones prompts a revision of the impact of Basque whaling on right and bowhead whales in the western North Atlantic. Can. J. Zool. 82:1647–1654.

Read, A.J. 1994. Interactions between cetaceans and gillnet and trap fisheries in the northwest Atlantic. Gillnets and cetaceans. Rep. Int. Whal. Comm. (Special Issue) 15:133–147.

Record, N.R., J.A. Runge, D.E. Pendleton, W.M. Balch, K.T.A. Davies, A.J. Pershing, C.L. Johnson, K. Stamieszkin, R. Ji, Z. Feng, S.D. Kraus, R.D. Kenney, C.A. Hudak, C.A. Mayo, C. Chen, J.E. Salisbury and C.R.S. Thompson. 2019. Rapid climate-driven circulation changes threaten conservation of endangered North Atlantic right whales. Oceanography. 32(2):162–169. https://doi.org/ 10.5670/oceanog.2019.201

Reeves, R.R., J.M. Breiwick and E. Mitchell. 1992. Pre-exploitation abundance of right whales off the eastern United States. Pages 5–7 *in*: J. Hain (Ed.). The right whale in the western North Atlantic: A science and management workshop, 14–15 April 1992, Silver Spring, Maryland. Northeast Fish. Sci. Cent. Ref. Doc. 92-05.

Reeves, R.R., R. Rolland and P. Clapham (Eds). 2001. Report of the workshop on the causes of reproductive failure in North Atlantic right whales: New avenues of research. Northeast Fish. Sci. Cent. Ref. Doc. 01-16. 46pp.

Reeves, R.R., T. Smith and E. Josephson. 2007. Near-annihilation of a species: Right whaling in the North Atlantic. Pages 39–74 *in*: S.D. Kraus and R.M. Rolland (Eds.). The urban whale: North Atlantic right whales at the crossroads. Harvard University Press, Cambridge, MA.

Rolland, R.M., R.S. Schick, H.M. Pettis, A.R. Knowlton, P.K. Hamilton, J.S. Clark and S.D. Krauss. 2016. Health of North Atlantic right whales *Eubalaena glacialis* over three decades: From individual health to demographic and population health trends. Mar. Ecol. Prog. Ser. 542:265–282.

Rosenbaum, H.C., M.S. Egan, P.J. Clapham, R.L. Brownell, Jr. and R. DeSalle. 1997. An effective method for isolating DNA from non-conventional museum specimens. Mol. Ecol. 6:677–681.

Rosenbaum, H.C., M.S. Egan, P.J. Clapham, R.L. Brownell, Jr., S. Malik, M.W. Brown, B.N. White, P. Walsh and R. DeSalle. 2000. Utility of North Atlantic right whale museum specimens for assessing changes in genetic diversity. Conserv. Biol. 14:1837–1842.

Salisbury, D., C.W. Clark and A.N. Rice. 2016. Right whale occurrence in Virginia coastal waters: Implications of endangered species presence in a rapidly developing energy market. Mar. Mamm. Sci. 32:508–519. https://doi.org/10.1111/mms.12276

Schaeff, C.M., S.D. Kraus, M.W. Brown, J. Perkins, R. Payne and B.N. White. 1997. Comparison of genetic variability of North and South Atlantic right whales (*Eubalaena*) using DNA fingerprinting. Can. J. Zool. 75:1073–1080.

Schmidly, D.J., C.O. Martin and G.F. Collins. 1972. First occurrence of a black right whale (*Balaena glacialis*) along the Texas coast. Southwest. Nat. 17:214–215.

Sharp, S.M., W.A. McLellan, D.S. Rotstein, A.M Costidis, S.G. Barco, K. Durham, T.D. Pitchford, K.A. Jackson, P.-Y. Daoust, T. Wimmer, E.L. Couture, L. Bourque, T. Frasier, D. Fauquier, T.K. Rowles, P.K. Hamilton, H. Pettis and M.J. Moore. 2019. Gross and histopathologic diagnoses from North Atlantic right whale *Eubalaena glacialis* mortalities between 2003 and 2018. Dis. Aquat. Org. 135(1):1–31.

Silber, G. K. and S. Bettridge. 2012. An assessment of the final rule to implement vessel speed restrictions to reduce the threat of vessel collisions with North Atlantic right whales. NOAA Tech. Memo. NMFS-OPR-48. 114pp.

Silva, M.A., L. Steiner, I. Cascão, M.J. Cruz, R. Prieto, T. Cole, P.K. Hamilton and M.F. Baumgartner. 2012. Winter sighting of a known western North Atlantic right whale in the Azores. J. Cetacean Res. Manage. 12:65–69.

van der Hoop, J.M., M.J. Moore, S.G. Barco, T.V. Cole, P.-Y. Daoust, A.G. Henry, D.F. McAlpine, W.A. McLellan, T. Wimmer and A.R. Solow. 2013. Assessment of management to mitigate anthropogenic effects on large whales. Conserv. Biol. 27:121–133.

van der Hoop, J.M., A.S.M. Vanderlaan, T.V.N. Cole, A.G. Henry, L. Hall, B. Mase-Guthrie, T. Wimmer and M.J. Moore. 2015. Vessel strikes to large whales before and after the 2008 Ship Strike Rule. Conserv. Lett. 8:24–32.

van der Hoop, J.M., P. Corkeron and M.J. Moore. 2017. Entanglement is a costly life-history stage in large whales. Ecol. Evol. 7:92–106. https://doi.org/10.1002/ece3.2615

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp. Available from: https://repository.library.noaa.gov/view/noaa/15963

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31:355–368.

Waldick, R.C., S.D. Kraus, M. Brown and B.N. White. 2002. Evaluating the effects of historic bottleneck events: An assessment of microsatellite variability in the endangered, North Atlantic right whale. Mol. Ecol. 11:2241–2250.

Ward-Geiger, L.I., A.R. Knowlton, A.F. Amos, T.D. Pitchford, B. Mase-Guthrie and B.J. Zoodsma. 2011. Recent sightings of the North Atlantic right whale in the Gulf of Mexico. Gulf Mex. Sci. 29:74–78.

Whitt, A.D., K. Dudzinski and J.R. Laliberté. 2013. North Atlantic right whale distribution and seasonal occurrence in nearshore waters off New Jersey, USA, and implications for management. Endanger. Species Res. 20:59–69.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

Williams, B.K., J.D. Nichols and M.J. Conroy. 2002. Analysis of animal populations, modeling, estimation and decision making. Academic Press. San Diego, California.

FIN WHALE (*Balaenoptera physalus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Fin whales have a global distribution, with populations found from temperate to polar regions in all ocean basins (Edwards *et al.* 2015). Within the Northern Hemisphere, populations in the North Pacific and North Atlantic oceans can be considered at least different subspecies, if not different species (Archer *et al.* 2019). The Scientific Committee of the International Whaling Commission (IWC) has proposed stock boundaries for North Atlantic fin whales. Fin whales off the eastern United States, Nova Scotia, and the southeastern coast of Newfoundland are believed to constitute a single stock under the present IWC scheme (Donovan 1991). Although the stock identity of North Atlantic fin whales has received much recent attention from the IWC, understanding of stock boundaries remains uncertain. The existence of a subpopulation structure was suggested by local depletions that resulted from commercial overharvesting (Mizroch *et al.* 1984).

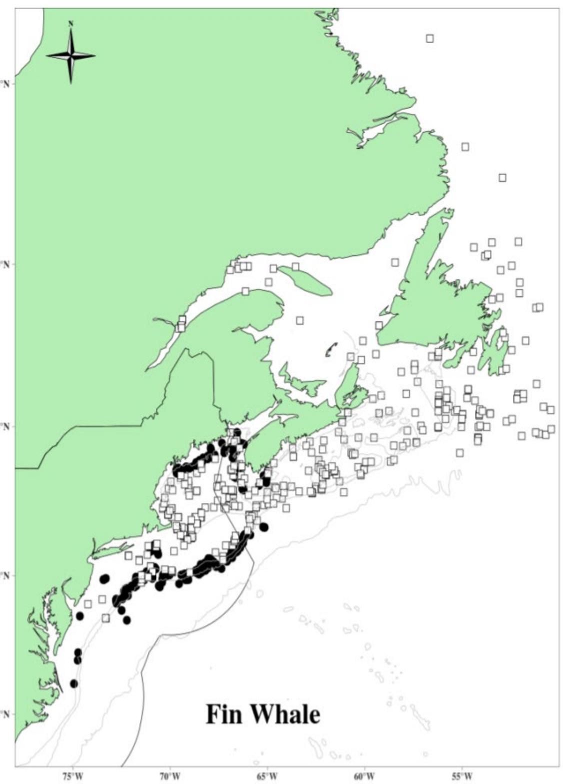


Figure 1. Distribution of fin whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 1,000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

A genetic study conducted by Bérubé *et al.* (1998) using both mitochondrial and nuclear DNA provided strong support for an earlier population model proposed by Kellogg (1929) and others. This postulates the existence of several subpopulations of fin whales in the North Atlantic and Mediterranean with limited gene flow among them. Bérubé *et al.* (1998) also proposed that the North Atlantic population showed recent divergence due to climatic changes (i.e., postglacial expansion), as well as substructuring over even relatively short distances. The genetic data are consistent with the idea that different subpopulations use the same feeding ground, a hypothesis that was also originally proposed by Kellogg (1929). More recent genetic studies have called into question conclusions drawn from early allozyme work (Olsen *et al.* 2014) and North Atlantic fin whales show a very low rate of genetic diversity throughout their range excluding the Mediterranean (Pampoulie *et al*. 2008).

Fin whales are common in waters of the U. S. Atlantic Exclusive Economic Zone (EEZ), principally from Cape Hatteras northward (Figure 1). In a globally-scaled review of sightings data, Edwards *et al.* (2015) found evidence to confirm the presence of fin whales in every season throughout much of the U.S. EEZ north of 35º N; however, densities vary seasonally. Fin whales accounted for 46% of the large whales and 24% of all cetaceans sighted over the continental shelf during aerial surveys (CETAP 1982) between Cape Hatteras and Nova Scotia during 1978–1982. While much remains unknown, the magnitude of the ecological role of the fin whale is impressive. In this region, fin whales are the dominant large cetacean species during all seasons, having the largest standing stock, the largest food requirements, and therefore the largest influence on ecosystem processes of any cetacean species (Hain *et al.* 1992, Kenney *et al.* 1997). Acoustic detections of fin whale singers augment and confirm these visual sighting conclusions for males. Recordings from Massachusetts Bay, New York Bight, and deep-ocean areas detected some level of fin whale singing from September through June (Watkins *et al*. 1987, Clark and Gagnon 2002, Morano *et al*. 2012). These acoustic observations from both coastal and deep-ocean regions support the conclusion that male fin whales are broadly distributed throughout the western North Atlantic for most of the year.

New England and Gulf of St. Lawrence waters represent major feeding grounds for fin whales. There is evidence of site fidelity by females, and perhaps some segregation by sexual, maturational, or reproductive class in the feeding area (Agler *et al.* 1993, Schleimer *et. al.* 2019). Hain *et al.* (1992) showed that fin whales measured photogrammetrically off the northeastern U.S., after omitting all individuals smaller than 14.6 m (the smallest whale taken in Iceland), were significantly smaller (mean length=16.8 m; P<0.001) than fin whales taken in Icelandic whaling (mean=18.3 m). Seipt *et al.* (1990) reported that 49% of identified fin whales sighted on the Massachusetts Bay area feeding grounds were resighted within the same year, and 45% were resighted in multiple years. The authors suggested that fin whales on these grounds exhibited patterns of seasonal occurrence and annual return that in some respects were similar to those shown for humpback whales. This was reinforced by Clapham and Seipt (1991), who showed maternally-directed site fidelity for fin whales in the Gulf of Maine. Hain *et al.* (1992), based on an analysis of neonate stranding data, suggested that calving takes place during October to January in latitudes of the U.S. mid-Atlantic region; however, it is unknown where calving, mating, and wintering occur for most of the population. Results from the Navy's SOSUS program (Clark 1995, Clark and Gagnon 2002) indicated a substantial deep-ocean distribution of fin whales. It is likely that fin whales occurring in the U.S. Atlantic EEZ undergo migrations into Canadian waters, open-ocean areas, and perhaps even subtropical or tropical regions (Edwards *et al.* 2015, Silve *et al.* 2019). However, the popular notion that entire fin whale populations make distinct annual migrations like some other mysticetes has questionable support in the data; in the North Pacific, year-round monitoring of fin whale calls found no evidence for large-scale migratory movements (Watkins *et al.* 2000).

POPULATION SIZE

The best available current abundance estimate for fin whales in the North Atlantic stock is 6,802 (CV=0.24), sum of the 2016 NOAA shipboard and aerial surveys and the 2016 NEFSC and Department of Fisheries and Oceans Canadia (DFO) surveys ("Central Virginia to Newfoundland/Labrador (COMBINED)" in Table 1). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area.

Earlier Abundance Estimates

Please see Appendix IV for earlier abundance estimates. As recommended in the guidelines for preparing Stock Assessment Reports (NMFS 2016), estimates older than eight years are deemed unreliable for the determination of a current PBR.

Recent Surveys and Abundance Estimates

An abundance estimate for western North Atlantic fin whales was generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020, Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance.

DFO generated fin whale estimates from a large-scale aerial survey of Atlantic Canadian shelf and shelf break habitats extending from the northern tip of Labrador to the U.S. border off southern Nova Scotia in August and September of 2016 (Table 1; Lawson and Gosselin 2018). A total of 29,123 km of effort was flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf stratum and 21,037 km over the Newfoundland/Labrador stratum. The Bay of Fundy/Scotian shelf portion of the fin whale population was estimated at 2,235 (CV=0.41) and the Newfoundland/Labrador portion at 2,177 (CV=0.47). The Newfoundland estimate was derived from Twin Otter data using two-team mark-recapture multi-covariate distance sampling methods. The Gulf of St. Lawrence estimate was derived from the Skymaster data using single team multi-covariate distance sampling with left truncation (to accommodate the obscured area under the plane) where size-bias was also investigated, and the Otter-based perception bias correction was applied. An availability bias correction factor, which was based on the cetaceans' surface intervals, was applied to both abundance estimates.

Table 1. Summary of recent abundance estimates for western North Atlantic fin whales with month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and coefficient of variation (CV). The estimate considered best is in bold font.

Month/Year	Area	Nest	CV
Jun−Sep 2016	Florida to lower Bay of Fundy	3,006	0.40
Aug−Sep 2016	Bay of Fundy/Scotian Shelf	2,235	0.413
Aug−Sep 2016	Newfoundland/Labrador	2,177	0.465
Jun−Sep 2016	**Central Virginia to Newfoundland/Labrador (COMBINED)**	**6,802**	**0.24**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for fin whales is 6,802 (CV=0.24). The minimum population estimate for the western North Atlantic fin whale is 5,573 (Table 2).

Current Population Trend

A trend analysis has not been conducted for this stock. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and variable survey design. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). However, a decline in the abundance of fin whales within the northern Gulf of St. Lawrence has been noted for that portion of the stock (Schleimer *et al.* 2019). There is current work to standardize the strata-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. Based on photographically identified fin whales, Agler *et al.* (1993) estimated that the gross annual reproduction rate was 8%, with a mean calving interval of 2.7 years.

For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 5,573. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.10 because the fin whale is listed as endangered under the Endangered Species Act (ESA). PBR for the western North Atlantic fin whale is 11.

Table 2. Best and minimum abundance estimates for the western North Atlantic fin whale (Balaenoptera physalus) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
6,802	0.24	5,573	0.1	0.04	11

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual estimated average human-caused mortality and serious injury for the western North Atlantic fin whale for the period 2014−2018 is presented in Table 3 (Henry *et al.* 2021). Annual rates calculated from detected mortalities should not be considered an unbiased representation of human-caused mortality, but they represent a definitive lower bound. Detections are haphazard and not the result of a designed sampling scheme. As such they represent a minimum estimate of human-caused mortality which is almost certainly biased low. The size of this bias is uncertain.

Table 3. Average annual observed and estimated human-caused and natural mortality and serious injury for the western North Atlantic fin whale (Balaenoptera physalus).

Years	Source	Annual Average
2014−2018	Incidental fishery interactions	1.55
2014−2018	Vessel collisions	0.80
Total		**2.35**

Fishery-Related Serious Injury and Mortality

United States

U.S. fishery interaction records for large whales come through two main sources—dedicated fishery observer data and opportunistic reports collected in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database. No confirmed fishery-related mortalities or serious injuries of fin whales have been reported in the NMFS Sea Sampling bycatch database (fishery observers) during this reporting period. Records of stranded, floating, or injured fin whales for the reporting period in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database with substantial evidence of fishery interactions causing injury or maorality are presented in Table 4 (Henry *et al.* 2021). These records are not statistically quantifiable in the same way as the observer fishery records, and they almost surely undercount entanglements for the stock.

Canada

The audited Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database also contains records of fin whales first reported in Canadian waters or attributed to Canada, of which the confirmed mortalities and serious injuries from the current reporting period are reported in Table 4.

Table 4. Confirmed human-caused mortality and serious injury records of fin whales (Balaenoptera physalus) where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2014–2018[a]

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
04/12/2014	Mortality	-	Port Elizabeth, NJ	VS	1	US	-	Fresh carcass on bow of vessel. Large external abrasions w/ associated hemorrhage and skeletal fractures along right side.
05/13/2014	Mortality	-	Rocky Harbour, NL	EN	1	CN	PT	Fresh carcass hog-tied in gear.
06/23/2014	Prorated Injury	-	off Chatham, MA	EN	0.75	XU	NR	Free-swimming, trailing 200ft of line. Attachment point(s) unknown. No resights.
08/20/2014	Prorated Injury	-	off Provincetown, MA	EN	0.75	XU	NR	Free-swimming, trailing buoy & 200ft of line aft of flukes. Attachment point(s) unknown. No resights.
10/05/2014	Mortality	-	off Manasquan, NJ	VS	1	US	-	Large area of hemorrhage along dorsal, ventral, and right lateral surfaces consistent with blunt force trauma.
06/06/2015	Serious Injury	-	off Bar Harbor, ME	EN	1	XU	NR	Free-swimming with 2 buoys and 80 ft of line trailing from fluke. Line cutting deeply into right fluke blade. Emaciated. No resights.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
07/06/2016	Prorated Injury	-	off Truro, MA	EN	0.75	XU	NR	Free-swimming with line trailing 60-70 ft aft of flukes. Attachment point(s) and configuration unknown. No resights.
07/08/2016	Prorated Injury	-	off Virginia Beach, VA	EN	0.75	XU	H/MF	Free-swimming with and lures in tow along left flipper area. Attachment point(s) and configuration unknown. No resights.
12/14/2016	Prorated Injury	-	off Provincetown, MA	EN	0.75	XU	NR	Free-swimming with buoy trailing 6–8ft aft of flukes. Attachment point(s) and configuration unknown. No resights.
05/30/2017	Mortality	-	Port Newark, NJ	VS	1	US	-	Fresh carcass on bow of 656 ft vessel. Speed at strike unknown.
08/25/2017	Mortality	-	off Miscou Island, QC	EN	1	CN	PT	Fisher found fresh carcass when hauling gear. Entangled at 78m depth, 51m from trap. Full configuration unknown, but unlikely to have drifted post-mortem in to gear.
06/22/2018	Mortality	-	16.5 nm E of Gaspe, QC	EN	1	CN	NP	No gear present. Fresh carcass with evidence of constricting entanglement across ventral pleats and peduncle with raw injuries to fluke. Evidence of associated bruising. No necropsy, but COD due to entanglement most parsimonious.
10/14/2018	Mortality	Ladders	Cape Cod Bay	VS	1	US	-	Floating carcass with great white shark actively scavenging. Landed on 18 Oct. Necropsied on 19 Oct. Left side not examined due to remote location & no heavy equipment. Additional exam conducted on 30 Oct. Evidence of blunt force trauma - fractured mandibles and rostrum with associated hemorrhaging. Histopathology results support findings.

Assigned Cause	Five-year Mean (US/CN/XU/XC)
Vessel strike	0.8 (0.8/0/0/0)
Entanglement	1.55 (0/0.6/0.95/0)

a. For more details on events see Henry *et al.* 2021.
b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. US=United States, XU=Unassigned 1st sight in US, CN=Canada, XC=Unassigned 1st sight in CN.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

Other Mortality

Death or injury as a result of vessel collision has significant anthropogenic impact on this stock (Schleimer *et al.* 2019). Known vessel strike cases are reported in Table 4.

HABITAT ISSUES

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Pierce *et al.* 2008; Jepson *et al.* 2016; Hall *et al.* 2018; Murphy *et al.* 2018), but research on contaminant levels for the western north Atlantic stock of fin whales is lacking.

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in or predicted for plankton species and commercially important fish stocks (Nye *et al*. 2009, Head *et al*. 2010, Pinsky *et al*. 2013, Poloczanska *et al.* 2013, Hare *et al*. 2016, Grieve *et al.* 2017, Morley *et al.* 2018) and cetacean species (e.g., MacLeod 2009; Sousa *et al.* 2019). There is uncertainty in how, if at all, the distribution and population size of this species will respond to these changes and how the ecological shifts will affect human impacts to the species.

STATUS OF STOCK

This is a strategic stock because the fin whale is listed as an endangered species under the ESA. NMFS records represent coverage of only a portion of the area surveyed for the population estimate for the stock. The total fishery-related mortality and serious injury for this stock derived from the available records is likely biased low and is not less than 10% of the calculated PBR. Therefore, entanglement rates cannot be considered insignificant and approaching a zero mortality and serious injury rate. The status of this stock relative to OSP in the U.S. Atlantic EEZ is unknown. There are insufficient data to determine the population trend for fin whales. Because the fin whale is ESA-listed, uncertainties with regard to the negatively biased estimates of human-caused mortality and the incomplete survey coverage relative to the stock's defined range would not change the status of the stock.

REFERENCES CITED

Agler, B.A., R.L. Schooley, S.E. Frohock, S.K. Katona and I.E. Seipt. 1993. Reproduction of photographically identified fin whales, *Balaenoptera physalus*, from the Gulf of Maine. J. Mamm. 73:577–587.

Archer, F.I., R.L. Brownell, Jr., B.L. Hancock-Hanser, P.A. Morin, K.M. Robertson, K.K. Sherman, J. Calambokidis, J. Urbán R., P.E. Rosel, S.A. Mizroch, S. Panigada and B.L. Taylor. 2019. Revision of fin whale *Balaenoptera physalus* (Linnaeus, 1758) subspecies using genetics. J. Mamm. 100:1653–1670.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp. https://repository.library.noaa.gov/view/noaa/6219

Bérubé, M., A. Aguilar, D. Dendanto, F. Larsen, G.N. di Sciara, R. Sears, J. Sigurjónsson, J. Urban-R. and P.J. Palsbøll. 1998. Population genetic structure of North Atlantic, Mediterranean and Sea of Cortez fin whales, *Balaenoptera physalus* (Linnaeus 1758): Analysis of mitochondrial and nuclear loci. Mol. Ecol. 15:585–599.

CETAP [Cetacean and Turtle Assessment Program]. 1982. A characterization of marine mammals and turtles in the mid- and North Atlantic areas of the U.S. outer continental shelf, final report #AA551-CT8-48, Cetacean and Turtle Assessment Program, University of Rhode Island. Bureau of Land Management, Washington, DC. 576pp.

Clapham, P.J. and I.E. Seipt. 1991. Resightings of independent fin whales, *Balaenoptera physalus*, on maternal summer ranges. J. Mamm. 72:788–790.

Clark, C.W. 1995. Application of U.S. Navy underwater hydrophone arrays for scientific research on whales. Rep. Int. Whal. Comm. 45:210–212.

Clark, C.W. and G.C. Gagnon. 2002. Low-frequency vocal behaviors of baleen whales in the North Atlantic: Insights from IUSS detections, locations and tracking from 1992 to 1996. J. Underwater Acoust. (U.S. Navy) 52:609–640.

Donovan, G.P. 1991. A review of IWC stock boundaries. Rep. Int. Whal. Comm. (Special Issue) 13:39–68.

Edwards, E.F., C. Hall, T.J. Moore, C. Sheredy and J. Redfern. 2015. Global distribution of fin whales (*Balaenoptera physalus*) in the post-whaling era (1980 to 2012). Mamm Rev. 45:197–214

Garrison, L.P. 2016. Abundance of marine mammals in waters of the U.S. East Coast during summer 2011. PRBD Contribution #PRBD-2016-08. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, Miami, FL. 21pp.

Garrison, L.P. 2020. Abundance of cetaceans along the southeast U.S. east coast from a summer 2016 vessel survey. PRD Contribution Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, Miami, FL. 17pp.

Grieve, B.D., J.A. Hare and V.S. Saba. 2017. Projecting the effects of climate change on *Calanus finmarchicus* distribution within the US Northeast continental shelf. Sci. Rep. 7:6264.

Hain, J.H.W., M.J. Ratnaswamy, R.D. Kenney and H.E. Winn. 1992. The fin whale, *Balaenoptera physalus*, in waters of the northeastern United States continental shelf. Rep. Int. Whal. Comm. 42:653–669.

Hall, A.J., B.J. McConnell, L.J. Schwacke, G.M. Ylitalo, R. Williams and T.K. Rowles. 2018. Predicting the effects of polychlorinated biphenyls on cetacean populations through impacts on immunity and calf survival. Environ. Poll. 233:407–418.

Hare, J.A., W.E. Morrison, M.W. Nelson, M.M. Stachura, E.J. Teeters, R.B. Griffis, M.A. Alexander, J.D. Scott, L. Alade, R.J. Bell, A.S. Chute, K.L. Curti, T.H. Curtis, D. Kurcheis, J.F. Kocik, S.M. Lucey, C.T. McCandless, L.M. Milke, D.E. Richardson, E. Robillard, H.J. Walsh, M.C. McManus, K.E. Maranick and C.A. Griswold. 2016. A vulnerability assessment of fish and invertebrates to climate change on the Northeast U.S. continental shelf. PLoS ONE 11:e0146756.

Head, E.J.H. and P. Pepin. 2010. Spatial and inter-decadal variability in plankton abundance and composition in the Northwest Atlantic (1958–2006). J. Plankton Res. 32:1633–1648. https://doi.org/10.1371/journal.pone.0146756.s014

Henry, A.G., T.V.N. Cole, L. Hall, W. Ledwell, D. Morin and A. Reid. 2021. Mortality determinations for baleen whale stocks along the Gulf of Mexico, United States East Coast and Atlantic Canadian Provinces, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 20-06.

Jepson, P.D., R. Deaville, J.L. Barber, A. Aguilar, A. Borrell, S. Murphy, J. Barry, A. Brownlow, J. Barnett, S. Berrow and A.A. Cunningham. 2016. PCB pollution continues to impact populations of orcas and other dolphins in European waters. Sci. Rep.-U.K. 6:18573.

Kellogg, R. 1929. What is known of the migration of some of the whalebone whales. Ann. Rep. Smithsonian Inst. 1928:467–494.

Kenney, R.D., G.P. Scott, T.J. Thompson and H.E. Winn. 1997. Estimates of prey consumption and trophic impacts of cetaceans in the USA northeast continental shelf ecosystem. J. Northw. Atl. Fish. Sci. 22:155–171.

Laake, J.L., J. Calambokidis, S.D. Osmek and D.J. Rugh. 1997. Probability of detecting harbor porpoise from aerial surveys: Estimating g (0). J. Wildl. Manage. 61:63–75.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Advanced distance sampling, Oxford University Press, New York, New York.

Lawson, J. and J-F. Gosselin. 2018. Estimates of cetacean abundance from the 2016 NAISS aerial surveys of eastern Canadian waters, with a comparison to estimates from the 2007 TNASS NAAMCO SC/25/AE/09. 40pp.

MacLeod, C.D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: A review and synthesis. Endang. Species Res. 7:125–136.

Mizroch, S.A., D.W. Rice and J.M. Breiwick. 1984. The fin whale, *Balaenoptera physalus*. Mar. Fish. Rev. 46:20–24.

Morano, J.L., D.P. Salisbury, A.N. Rice, K.L. Conklin, K.L. Falk and C.W. Clark. 2012. Seasonal changes in fin whale (*Balaenoptera physalus*) 20-Hz song in the western North Atlantic Ocean. J. Acoust. Soc. Am. 132:1207–1212.

Morley, J.W., R.L. Selden, R.J. Latour, T.L. Frolicher, R.J. Seagraves and M.L. Pinsky. 2018. Projecting shifts in thermal habitat for 686 species on the North American continental shelf. PLoS ONE 13(5):e0196127.

Murphy, S., R.J. Law, R. Deaville, J.Barnett, M.W. Perkins, A. Brownlow, R. Penrose, N.J. Davison, J.L. Barber and P.D. Jepson. 2018. Organochlorine contaminants and reproductive implication in cetaceans: A case study of the common dolphin. Pages 3–38 *in* M.C. Fossi and C. Panti (Eds.). Marine mammal ecotoxicology: Impacts of multiple stressors on population health. Academic Press, New York, New York.

Nye, J., J. Link, J. Hare and W. Overholtz. 2009. Changing spatial distribution of fish stocks in relation to climate and population size on the Northeast United States continental shelf. Mar. Ecol. Prog. Ser. 393:111–129.

Olsen, M. T., C. Pampoulie, A.K. Daníelsdóttir, E. Lidh, M. Bérubé, G.A. Víkingsson and P.J. Palsbøll. 2014. Fin whale MDH-1 and MPI allozyme variation is not reflected in the corresponding DNA sequences. Ecol. Evol. 4:1787–1803. https://doi.org/10.1002/ece3.1046

NOAA [National Oceanic and Atmospheric Administration]. 2012. National process for distinguishing serious from non-serious injuries of marine mammals. Federal Register 77:3233−3275. Available from: https://www.fisheries.noaa.gov/webdam/download/64690371

NMFS [National Marine Fisheries Service]. 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23pp. Available from: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks

NEFSC [Northeast Fisheries Science Center] and SEFSC [Southeast Fisheries Science Center]. 2018. Annual report of a comprehensive assessment of marine mammal, marine turtle, and seabird abundance and spatial distribution in US Waters of the Western North Atlantic Ocean. Northeast Fish. Sci. Cent. Ref. Doc. 18-04. 141pp. Available from: https://repository.library.noaa.gov/view/noaa/22040

Palka, D.L. and P.S. Hammond. 2001. Accounting for responsive movement in line transect estimates of abundance. Can. J. Fish. Aquat. Sci. 58:777–787.

Palka, D. 2020. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2016 line transect surveys conducted by the Northeast Fisheries Science Center. Northeast Fish. Sci. Cent. Ref. Doc. 20-05.

Pampoulie, C., A. Daníelsdóttir, M. Bérubé, P. Palsbøll, A. Arnarson, T. Gunnlaugsson, D. Olafsdottir, N. Øien, L. Witting and G. Vikingsson. 2008. Lack of genetic divergence among samples of the North Atlantic fin whale collected at feeding grounds: Congruence among microsatellite loci and mtDNA in the new Icelandic dataset. Unpublished Scientific Committee meeting document SC/60/PFI11. International Whaling Commission, Cambridge, UK. 17pp.

Pierce, G.J. M.B. Santos, S. Murphy, J.A. Learmonth, A.F. Zuur, E. Rogan, P. Bustamante, F. Caurant, V. Lahaye, V. Ridoux, B.N. Zegers, A. Mets, M. Addink, C. Smeenk, T. Jauniaux, R.J. Law, W. Dabin, A. López, J.M. Alonso Farré, A.F. González, A. Guerra, M. García-Hartmann, R.J. Reid, C.F. Moffat, C. Lockyer and J.P. Boon. 2008. Bioaccumulation of persistent organic pollutants in female common dolphins (*Delphinus delphis*) and harbour porpoises (*Phocoena phocoena*) from western European seas: Geographical trends, causal factors and effects on reproduction and mortality. Environ. Poll. 153:401–415.

Pinsky, M.L., B. Worm, M.J. Fogarty, J.L. Sarmiento and S.A. Levin. 2013. Marine taxa track local climate velocities, Science 341:1239–1242.

Poloczanska, E.S., C.J. Brown, W.J. Sydeman, W. Kiessling, D.S. Schoeman, P.J. Moore, K. Brander, J.F. Bruno, L.B. Buckley, M.T. Burrows, C.M. Duarte, B.S. Halpern, J. Holding, C.V. Kappel, M.I. O'Connor, J.M. Pandolfi, C. Parmesan, F. Schwing, S.A. Thompson and A.J. Richardson. 2013. Global imprint of climate change on marine life. Nat. Clim. Change 3:919–925.

Schleimer, A., C. Ramp, J. Delarue, A. Carpentier, M. Bérubé, P.J. Palsbøll, R. Sears and P.S. Hammond. 2019. Decline in abundance and apparent survival rates of fin whales (*Balaenoptera physalus*) in the northern Gulf of St. Lawrence. Ecol. Evol. 9:4231–4244.

Seipt, I.E., P.J. Clapham, C.A. Mayo and M.P. Hawvermale. 1990. Population characteristics of individually identified fin whales *Balaenoptera physalus* in Massachusetts Bay. Fish. Bull. 88:271–278.

Silva, M.A., A. Borrell, R. Prieto, P. Gauffier, M. Bérubé, P.J. Palsbøll and A. Colaço. 2019. Stable isotopes reveal winter feeding in different habitats in blue, fin and sei whales migrating through the Azores. R. Soc. Open Sci. 6:181800. http://doi.org/10.1098/rsos.181800

Sousa, A., F. Alves, A. Dinis, J. Bentz, M.J. Cruz and J.P. Nunes. 2019. How vulnerable are cetaceans to climate change? Developing and testing a new index. Ecol. Indic. 98:9–18.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23:157–175.

Thomas L, J.L. Laake, E. Rexstad, S. Strindberg, F.F.C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, M.L. Burt, S.L. Hedley, J.H. Pollard, J.R.B. Bishop and T.A. Marques. 2009. Distance 6.0. Release 2. [Internet]. University of St. Andrews, Research Unit for Wildlife Population Assessment, St. Andrews, UK. Available from: http://distancesampling.org/Distance/

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Watkins, W.A., P. Tyack, K.E. Moore and J.E. Bird. 1987. The 20-Hz signals of finback whales (*Balaenoptera physalus*). J. Acoust. Soc. Am. 82:1901–1912.

Watkins, W.A., M.A. Daher, G.M. Reppucci, J.E. George, D.L. Martin, N.A. DiMarzio and D.P. Gannon. 2000. Seasonality and distribution of whale calls in the North Pacific. Oceanography 13:62–67.

SEI WHALE (*Balaenoptera borealis borealis*): Nova Scotia Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Mitchell and Chapman (1977) reviewed the sparse evidence on stock identity of western North Atlantic sei whales, and suggested two stocks—a Nova Scotia stock and a Labrador Sea stock. The range of the Nova Scotia stock includes the continental shelf waters of the northeastern U.S. and extends northeastward to south of Newfoundland. The Scientific Committee of the International Whaling Commission (IWC), while adopting these general boundaries, noted that the stock identity of sei whales (and indeed all North Atlantic whales) was a major research problem (Donovan 1991). Telemetry evidence indicates a migratory corridor between animals foraging in the Labrador Sea and the Azores, based on seven individuals tagged in the Azores during spring migration (Prieto *et al*. 2014). These data support the idea of a separate foraging ground in the Gulf of Maine and Nova Scotia. However, recent genetic work did not reveal stock structure in the North Atlantic based on both mitochondrial DNA and microsatellite analyses, though the authors acknowledge that they cannot rule out the presence of multiple stocks (Huijser *et al*. 2018). Therefore, in the absence of clear evidence to the contrary, the proposed IWC stock definition is provisionally adopted, and the "Nova Scotia stock" is used here as the management unit for this stock assessment. The IWC boundaries for this stock are from the U.S. east coast to Cape Breton, Nova Scotia, thence east to longitude 42° W. A key uncertainty in the stock structure definition is due to the sparse availability of data to discern the relationship between animals from the Nova Scotia stock and other North Atlantic stocks and to determine if the Nova Scotia stock contains multiple demographically independent populations.

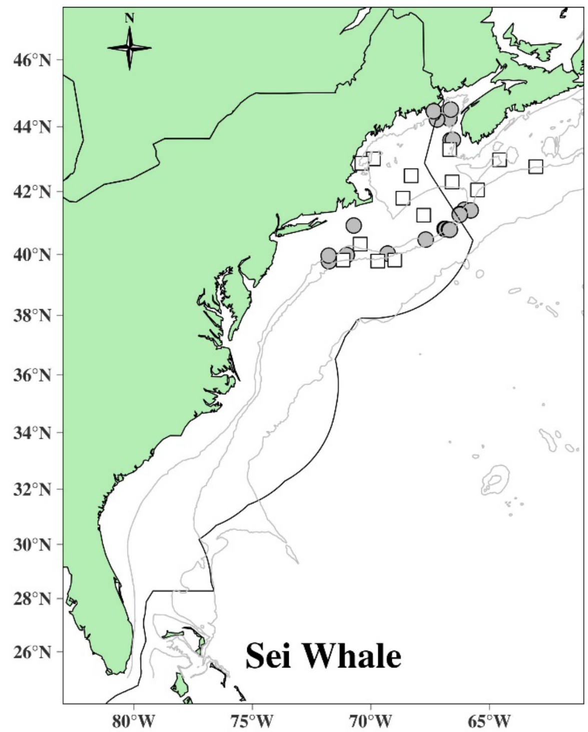


Figure 1. Distribution of sei whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-m, 1000-m and 4000-m depth contours.

Habitat suitability analyses suggest that the recent distribution patterns of sei whales in U.S. waters appear to be related to water that are cool (<10°C), with high levels of chlorophyll and inorganic carbon, and where the mixed layer depth is relatively shallow (<50m; Palka *et al*. 2017, Chavez-Rosales *et al*. 2019). Sei whales have often been found in the deeper waters characteristic of the continental shelf edge region (Mitchel 1975, Hain *et al.* 1985). During the spring/summer feeding season, existing data indicate that a major portion of the Nova Scotia sei whale stock is centered in northerly waters, perhaps on the Scotian Shelf (Mitchell and Chapman 1977). Based on analysis of records from the Blandford, Nova Scotia whaling station, where 825 sei whales were taken between 1965 and 1972, Mitchell (1975) described two "runs" of sei whales, in June–July and in September–October. He speculated that the sei whale stock migrates from south of Cape Cod and along the coast of eastern Canada in June and July, and returns on a southward migration again in September and October; however, the details of such a migration remain unverified.

The southern portion of the species' range during spring and summer includes the northern portions of the U.S. Atlantic Exclusive Economic Zone (EEZ)—the Gulf of Maine and Georges Bank. NMFS aerial surveys since 1999 have found concentrations of sei whales along the northern edge of Georges Bank in the spring. Spring is the period of greatest abundance in U.S. waters, with sightings concentrated along the eastern margin of Georges Bank, into the Northeast Channel area, south of Nantucket, and along the southwestern edge of Georges Bank, for example in the area of Hydrographer Canyon (CETAP 1982, Kraus *et al.* 2016, Roberts *et al.* 2016, Palka *et al.* 2017, Cholewiak *et al.* 2018).

The wintering habitat for sei whales remains largely unknown. In passive acoustic monitoring (PAM) conducted off Georges Bank in 2015–2016, sei whales calls were consistently detected from late fall through the winter along the southern Georges Bank region, off Heezen and Oceanographer Canyons (Cholewiak *et al.* 2018). Sei whale calls were also sporadically detected at PAM sites from Cape Hatteras southward. This included sparsely detected sei whale calls on the Blake Plateau during November–February in 2015 and 2016 (Cholewiak *et al.* 2018).

The general offshore pattern of sei whale distribution is disrupted during episodic incursions into shallower, more inshore waters. Although known to eat fish in other oceans (Flinn *et al.* 2002), North Atlantic sei whales are largely planktivorous, feeding primarily on euphausiids and copepods (Flinn *et al.* 2002). A review of prey preferences by Horwood (1987) showed that, in the North Atlantic, sei whales seem to prefer copepods over all other prey species. In Nova Scotia, sampled stomachs from captured sei whales showed a clear preference for copepods between June and October, and euphausiids were taken only in May and November (Mitchell 1975). Sei whales are reported in some years in more inshore locations, such as the Great South Channel (in 1987 and 1989) and Stellwagen Bank (in 1986) areas (Payne *et al.* 1990). An influx of sei whales into the southern Gulf of Maine occurred in the summer of 1986 (Schilling *et al.* 1993). Such episodes, often punctuated by years or even decades of absence from an area, have been reported for sei whales from various places worldwide (Jonsgård and Darling 1977).

POPULATION SIZE

The average spring 2010–2013 abundance estimate of 6,292 (CV=1.015) is considered the best available for the Nova Scotia stock of sei whales because it was derived from surveys covering the largest proportion of the range (Halifax, Nova Scotia to Florida), during the season when they are the most prevalent in U.S. waters (in spring), using only recent data (2010–2013), and correcting aerial survey data for availability bias. However, this estimate must be considered uncertain because all of the known range of this stock was not surveyed, because of uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas, and because of issues in the data collection (ambiguous identification between fin and sei whales) and analysis (in particular, how best to handle the ambiguous sightings, low encounter rates, and defining the most appropriate species-specific availability bias correction factor).

Earlier Abundance Estimates

Please see Appendix IV for earlier abundance estimates. As recommended in the GAMMS Workshop Report (Wade and Angliss 1997), estimates older than eight years are deemed unreliable for determination of the current PBR.

Recent Surveys and Abundance Estimates

An estimate of 6,292 (CV=1.015) was the springtime (March–May) average abundance estimate generated from spatially- and temporally-explicit density models derived from visual two-team abundance survey data collected between 2010 and 2013 (Table 1; Palka *et al.* 2017). This estimate is for waters between Halifax, Nova Scotia and Florida, where the highest densities of animals were predicted to be on the Scotia shelf outside of U.S. waters. Over 25,000 km of shipboard and over 99,000 km of aerial visual line-transect survey data collected in all seasons in Atlantic waters from Florida to Nova Scotia during 2010–2014 were divided into 10x10 km^2 spatial grid cells and 8-day temporal time periods. Mark-recapture covariate distance sampling was used to estimate abundance in each spatial-temporal cell which was corrected for perception bias. These density estimates and spatially- and temporally-explicit static and dynamic environmental data were used in Generalized Additive Models (GAMs) to develop spatially- and temporally-explicit animal density-habitat statistical models. These estimates were also corrected by platform- and species-specific availability bias correction factors that were based on dive time patterns.

An abundance estimate of 28 (CV=0.55) sei whales was generated from a summer shipboard and aerial survey conducted during 27 June–28 September 2016 (Table 1; Palka 2020) within a region covering 425,192 km^2. The estimate is only for waters along the continental shelf break from New Jersey to south of Nova Scotia. The aerial portion included 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout U.S. waters. The shipboard portion included 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the outer limit of the EEZ). Both sighting platforms used a two-team data collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers 2004). The estimates were also corrected for availability bias.

Comprehensive aerial surveys of Canadian east coast waters in 2007 and 2016 identified only 7 sei whales, suggesting a population of a few hundred animals or less, and a substantial reduction from pre-whaling numbers. The population is currently thought to number fewer than 1,000 in eastern Canadian waters (https://www.canada.ca/en/environment-climate-change/services/committee-status-endangered-wildlife.html).

Seasonal average habitat-based density estimates generated by Roberts *et al*. (2016) produced abundance estimates of 627 (CV=0.14) for spring in U.S. waters only and 717 (CV=0.30) for summer in waters from the mouth of Gulf of St. Lawrence to Florida. These were based on data from 1995–2013. Their models were created using GAMs, with environmental covariates projected to 10x10 km grid cells. Three model versions were fit to the data, including a climatological model with 8-day estimates of covariates, a contemporaneous model, and a combination of the two. Several differences in modeling methodology result in abundance estimates that are different than the estimates generated from the above surveys.

Table 1. Summary of recent abundance estimates for Nova Scotia sei whales with month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and coefficient of variation (CV). Estimate considered best is bolded.

Month/Year	Area	N_{est}	CV
Apr−Jun 1999−2013	Maine to Florida in U.S. waters only	627	0.14
Jul−Sep 1995−2013	Gulf of St Lawrence entrance to Florida	717	0.30
Mar−May 2010−2013	**Halifax, Nova Scotia to Florida**	**6,292**	**1.015**
Jun−Aug 2016	Continental shelf break waters from New Jersey to south of Nova Scotia	28	0.55

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for the Nova Scotia stock sei whales is 6,292 (CV=1.015). The minimum population estimate is 3,098.

Current Population Trend

A trend analysis has not been conducted for this stock. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long survey interval. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). There is current work to standardize the strata-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 3,098. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.10 because the sei whale is listed as endangered under the Endangered Species Act (ESA). PBR for the Nova Scotia stock of the sei whale is 6.2 (Table 2).

Table 2. Best and minimum abundance estimates for Nova Scotia sei whales (Balaenoptera borealis borealis) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r.) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
6,292	1.02	3,098	0.1	0.04	6.2

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The most recent 5-year average human-caused mortality and serious injury rates are summarized in Table 3. Annual rates calculated from detected mortalities should not be considered unbiased estimates of human-caused mortality, but they represent definitive lower bounds. Detections are haphazard, incomplete, and not the result of a designed sampling scheme. As such they represent a minimum estimate of human-caused mortality which is almost certainly biased low.

Table 3: ***The total annual observed average human-caused mortality and serious injury for Nova Scotia sei whales (Balaenoptera borealis borealis).***

Years	Source	Annual Average
2014−2018	Incidental fishery interactions	0.40
2014−2018	Vessel collisions	0.80
Total		**1.20**

Fishery-Related Serious Injury and Mortality

U.S. fishery interaction records for large whales come from two main sources—dedicated fishery-observer data and opportunistic reports collected in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/ stranding database. No confirmed fishery-related mortalities or serious injuries of sei whales have been reported in the NMFS Sea Sampling bycatch database (fishery observers). Records of stranded, floating, or injured sei whales for the reporting period in the audited Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database with substantial evidence of fishery interactions causing injury or mortality are presented below (Table 4).

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
05/04/2014	Mortality	-	Hudson River, NY	VS	1	US	-	Fresh carcass on bow of vessel. Extensive skeletal fractures w/ associated hemorrhage along right side.
05/07/2014	Mortality	-	Delaware River, PA	VS	1	US	-	Fresh carcass on bow of vessel.
08/14/2014	Mortality	-	James River, VA	VS	1	US	-	Live stranded and died. Emaciated. Fragment of plastic DVD case in stomach. Broken bones w/ associated hemorrhaging. Proximate COD: starvation by ingestion of plastic debris. Ultimate COD: blunt trauma from vessel strike
07/25/2016	Mortality	-	Hudson River, Newark, NJ	VS	1	US	-	Fresh carcass on bow of ship (>65 ft). Speed at strike unknown.
05/11/2017	Serious Injury	-	Cape Lookout Bight, NC	EN	1	XU	-	Free-swimming, emaciated, and carrying a large mass of heavily fouled gear consisting of line & buoys crossing over back. Full configuration unknown, but evidence of significant health decline.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
03/12/2018	Mortality	-	Fanny Keys, FL	EN	1	XU	NR	Carcass with line exiting left side of mouth, across rostrum, and entering right side. Bundle of frayed line lodged in baleen mid-rostrum. Severely emaciated, extensive scavenging. Partial necropsy conducted. Partial healing of lesions + epibiotic growth on line + emaciation = chronic entanglement. Gear not recovered

Assigned Cause	Five-year Mean (US/CN/XU/XC)
Vessel Strike	0.80 (0.80/0/0/0)
Entanglement	0.40 (0/0/0.40/0)

a. For more details on events, see Henry *et al.* 2021.
b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. US=United States, XU=Unassigned 1st sight in US, CN=Canada, XC=Unassigned 1st sight in CN.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

Other Mortality

Records with substantial evidence of vessel collision causing serious injury or mortality are presented in Table 4.

HABITAT ISSUES

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Pierce *et al.* 2008; Jepson *et al.* 2016; Hall *et al.* 2018; Murphy *et al.* 2018), but research on contaminant levels for the Nova Scotia stock of sei whales is lacking.

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in or predicted for plankton species and commercially important fish stocks (Nye *et al*. 2009, Pinsky *et al*. 2013, Poloczanska *et al.* 2013, Hare *et al*. 2016, Grieve *et al.* 2017, Morley *et al.* 2018) and cetacean species (e.g., MacLeod 2009, Sousa *et al.* 2019). There is uncertainty in how, if at all, the distribution and population size of this species will respond to these changes and how the ecological shifts will affect human impacts to the species.

STATUS OF STOCK

This is a strategic stock because the sei whale is listed as an endangered species under the ESA. The total U.S. fishery-related mortality and serious injury for this stock derived from the available records was less than 10% of the calculated PBR, and therefore could be considered insignificant and approaching a zero mortality and serious injury rate. However, evidence for fisheries interactions with large whales are subject to imperfect detection, and caution should be used in interpreting these results. The status of this stock relative to OSP in the U.S. Atlantic EEZ is unknown. There are insufficient data to determine population trends for sei whales.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.
CETAP [Cetacean and Turtle Assessment Program]. 1982. A characterization of marine mammals and turtles in the mid- and north Atlantic areas of the U.S. outer continental shelf, final report. Bureau of Land Management, Washington, DC. #AA551-CT8-48. 538pp.
Chavez-Rosales, S., D.L. Palka, L.P. Garrison and E.A. Josephson. 2019. Environmental predictors of habitat suitability and occurrence of cetaceans in the western North Atlantic Ocean. Sci. Rep. 9. https://doi.org/10.1038/s41598-019-42288-6

Cholewiak, D., D. Palka, S. Chavez-Rosales, G. Davis, E. Josephson, S. Van Parijs and S. Weiss. 2018. Updates on sei whale (*Balaenoptera borealis*) distribution, abundance estimates, and acoustic occurrence in the western North Atlantic. Unpublished Scientific Committee meeting document SC/67B/NH07. International Whaling Commission. Cambridge, UK.

Donovan, G.P. 1991. A review of IWC stock boundaries. Rep. Int. Whal. Comm. (Special Issue) 13:39–68.

Flinn, R.D., A.W. Trites and E.J. Gregr. 2002. Diets of fin, sei, and sperm whales in British Columbia: An analysis of commercial whaling records, 1963–1967. Mar. Mamm Sci. 18:663–679.

Grieve, B.D., J.A. Hare and V.S. Saba. 2017. Projecting the effects of climate change on *Calanus finmarchicus* distribution within the US Northeast continental shelf. Sci. Rep. 7:6264.

Hain, J.H.W., M.A. Hyman, R.D. Kenney and H.E. Winn. 1985. The role of cetaceans in the shelf-edge region of the northeastern United States. Mar. Fish. Rev. 47:13–17.

Hall, A.J., B.J. McConnell, L.J. Schwacke, G.M. Ylitalo, R. Williams and T.K. Rowles. 2018. Predicting the effects of polychlorinated biphenyls on cetacean populations through impacts on immunity and calf survival. Environ. Poll. 233:407–418.

Hare, J.A., W.E. Morrison, M.W. Nelson, M.M. Stachura, E.J. Teeters, R.B. Griffis, M.A. Alexander, J.D. Scott, L. Alade, R.J. Bell, A.S. Chute, K.L. Curti, T.H. Curtis, D. Kurcheis, J.F. Kocik, S.M. Lucey, C.T. McCandless, L.M. Milke, D.E. Richardson, E. Robillard, H.J. Walsh, M.C. McManus, K.E. Maranick, C.A. Griswold. 2016. A vulnerability assessment of fish and invertebrates to climate change on the Northeast U.S. continental shelf. PLoS ONE 11:e0146756. https://doi.org/10.1371/journal.pone.0146756.s014

Head, E.J.H. and P. Pepin. 2010. Spatial and inter-decadal variability in plankton abundance and composition in the Northwest Atlantic (1958–2006). J. Plankton Res. 32:1633–1648.

Henry, A.G., M. Garron, D. Morin, A. Smith, A. Reid, W. Ledwell and T. Cole. 2021. Serious injury and mortality determinations for baleen whale stocks along the Gulf of Mexico, United States east coast and Atlantic Canadian provinces, 2014–2018. NEFSC Ref. Doc. 21-07.

Huijser, L.A., M. Bérubé, A.A. Cabrera, R. Prieto, M.A Silva, J. Robbins, N. Kanda, L.A. Pastene, M. Goto, H. Yoshida and G.A. Víkingsson. 2018. Population structure of North Atlantic and North Pacific sei whales (*Balaenoptera borealis*) inferred from mitochondrial control region DNA sequences and microsatellite genotypes. Conserv. Genet. 19:1007−1024.

Jepson, P.D., R. Deaville, J.L. Barber, A. Aguilar, A. Borrell, S. Murphy, J. Barry, A. Brownlow, J. Barnett, S. Berrow and A.A. Cunningham. 2016. PCB pollution continues to impact populations of orcas and other dolphins in European waters. Sci. Rep.-U.K. 6:18573.

Jonsgård, Å. and K. Darling. 1977. On the biology of the eastern North Atlantic sei whale, *Balaenoptera borealis* Lesson. Rep. Int. Whal. Comm. (Special Issue) 1:124–129.

Kraus, S.D., S. Leiter, K. Stone, B. Wikgren, C. Mayo, P. Hughes, R.D. Kenney, C.W. Clark, A.N. Rice, B. Estabrook and J. Tielens. 2016. Northeast large pelagic survey collaborative aerial and acoustic surveys for large whales and sea turtles. OCS Study BOEM 2016-054. US Department of the Interior, Bureau of Ocean Energy Management, Sterling, Virginia. 117pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108−189 *in*: Advanced distance sampling, S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Lawson, J.W. and J.-F. Gosselin. 2009. Distribution and preliminary abundance estimates for cetaceans seen during Canada's Marine Megafauna Survey — A component of the 2007 TNASS. Can. Sci. Advisory Sec. Res. Doc. 2009/031. 33pp.

Mitchell, E. 1975. Preliminary report on Nova Scotia fishery for sei whales (*Balaenoptera borealis*). Rep. Int. Whal. Comm. 25:218–225.

Mitchell, E. and D.G. Chapman. 1977. Preliminary assessment of stocks of northwest Atlantic sei whales (*Balaenoptera borealis*). Rep. Int. Whal. Comm. (Special Issue) 1:117–120.

Morley, J.W., R.L. Selden, R.J. Latour, T.L. Frolicher, R.J. Seagraves and M.L. Pinsky. 2018. Projecting shifts in thermal habitat for 686 species on the North American continental shelf. PLoS ONE 13(5):e0196127.

Murphy, S., R.J. Law, R. Deaville, J.Barnett, M W. Perkins, A. Brownlow, R. Penrose, N.J. Davison, J.L. Barber and P.D. Jepson. 2018. Organochlorine contaminants and reproductive implication in cetaceans: A case study of the common dolphin. Pages 3–38 *in:* M.C. Fossi and C. Panti (Eds.). Marine mammal ecotoxicology: Impacts of multiple stressors on population health. Academic Press, New York, New York.

NOAA [National Oceanic and Atmospheric Administration]. 2012. National process for distinguishing serious from non-serious injuries of marine mammals. Federal Register 77:3233−3275 Available from: http://www.nmfs.noaa.gov/op/pds/documents/02/238/02-238-01.pdf

Nye, J., J. Link, J. Hare and W. Overholtz. 2009. Changing spatial distribution of fish stocks in relation to climate and population size on the Northeast United States continental shelf. Mar. Ecol. Prog. Ser. 393:111–129.

Palka, D. 2020. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2016 line transect surveys conducted by the Northeast Fisheries Science Center. Northeast Fish. Sci. Cent. Ref. Doc. 20-05.

Palka, D.L., S. Chavez-Rosales, E. Josephson, D. Cholewiak, H.L. Haas, L. Garrison, M. Jones, D. Sigourney, G. Waring, M. Jech, E. Broughton, M. Soldevilla, G. Davis, A. DeAngelis, C.R. Sasso, M.V.Winton, R.J. Smolowitz, G. Fay, E. LaBrecque, J.B. Leiness, Dettloff, M. Warden, K. Murray and C. Orphanides. 2017. Atlantic Marine Assessment Program for Protected Species: 2010–2014. OCS Study BOEM 2017-071. US Dept. of the Interior, Bureau of Ocean Energy Management, Atlantic OCS Region, Washington, DC. 211pp. https://espis.boem.gov/final%20reports/5638.pdf

Payne, P.M., D.N. Wiley, S.B. Young, S. Pittman, P.J. Clapham and J.W. Jossi. 1990. Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in selected prey. Fish. Bull. 88:687–696.

Pierce, G.J. M.B. Santos, S. Murphy, J.A. Learmonth, A.F. Zuur, E. Rogan, P. Bustamante, F. Caurant, V. Lahaye, V. Ridoux, B.N. Zegers, A. Mets, M. Addink, C. Smeenk, T. Jauniaux, R.J. Law, W. Dabin, A. López, J.M. Alonso Farré, A.F. González, A. Guerra, M. García-Hartmann, R.J. Reid, C.F. Moffat, C. Lockyer and J.P. Boon. 2008. Bioaccumulation of persistent organic pollutants in female common dolphins (*Delphinus delphis*) and harbour porpoises (*Phocoena phocoena*) from western European seas: Geographical trends, causal factors and effects on reproduction and mortality. Environ. Poll. 153:401–415.

Pinsky, M.L., B. Worm, M.J. Fogarty, J.L. Sarmiento and S.A. Levin. 2013. Marine taxa track local climate velocities. Science 341:1239–1242.

Poloczanska, E.S., C.J. Brown, W.J. Sydeman, W. Kiessling, D.S. Schoeman, P.J. Moore, K. Brander, J.F. Bruno, L.B. Buckley, M.T. Burrows, C.M. Duarte, B.S. Halpern, J. Holding, C.V. Kappel, M.I. O'Connor, J.M. Pandolfi, C. Parmesan, F. Schwing, S.A. Thompson and A.J. Richardson. 2013. Global imprint of climate change on marine life. Nat. Clim. Change 3:919–925.

Prieto, R., M.A. Silva, G.T. Waring and J.M.A. Gonçalves. 2014. Sei whale movements and behaviour in the North Atlantic inferred from satellite telemetry. Endang. Species Res. 26:103–113.

Roberts, J.J., B.D. Best, L. Mannocci, E. Fujioka, P.N. Halpin, D.L. Palka, L.P. Garrison, K.D. Mullin, T.V.N. Cole, C.B. Khan, W.A. McLellan, D.A. Pabst and G.G. Lockhart. 2016. Habitat-based cetacean density models for the US Atlantic and Gulf of Mexico. Sci. Rep. 6:22615.

Schilling, M.R., I. Seipt, M.T. Weinrich, S.E. Frohock, A.E. Kuhlberg and P.J. Clapham. 1993. Behavior of individually identified sei whales, *Balaenoptera borealis*, during an episodic influx into the southern Gulf of Maine in 1986. Fish. Bull. 90:749–755.

Sousa, A., F. Alves, A. Dinis, J. Bentz, M.J. Cruz and J.P. Nunes. 2019. How vulnerable are cetaceans to climate change? Developing and testing a new index. Ecol. Indic. 98:9–18.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23:157–175.

Thomas L., J.L. Laake, E. Rexstad, S. Strindberg, F.F.C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, M.L. Burt, S.L. Hedley, J.H. Pollard, J.R.B. Bishop and T.A. Marques. 2009. Distance 6.0. Release 2. [Internet]. University of St. Andrews (UK): Research Unit for Wildlife Population Assessment. Available from: http://distancesampling.org/Distance/

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp. Available from: https://repository.library.noaa.gov/view/noaa/15963

COMMON MINKE WHALE (*Balaenoptera acutorostrata acutorostrata*): Canadian East Coast Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Minke whales have a cosmopolitan distribution in temperate, tropical and high-latitude waters. They are common and widely distributed within the U.S. Atlantic Exclusive Economic Zone (EEZ; CETAP 1982). There appears to be a strong seasonal component to minke whale distribution on both the continental shelf and in deeper, off-shelf waters. Spring to fall are times of relatively widespread and common acoustic occurrence on the shelf (e.g., Risch *et al.* 2013), while September through April is the period of highest acoustic occurrence in deep-ocean waters throughout most of the western North Atlantic (Clark and Gagnon 2002, Risch *et al.* 2014). In New England waters the whales are most abundant during the spring-to-fall period. Records based on visual sightings and summarized by Mitchell (1991) hinted at a possible winter distribution in the West Indies, and in the mid-ocean south and east of Bermuda, a suggestion that has been validated by acoustic detections throughout broad ocean areas off the Caribbean from late September through early June (Clark and Gagnon 2002, Risch *et al.* 2014).

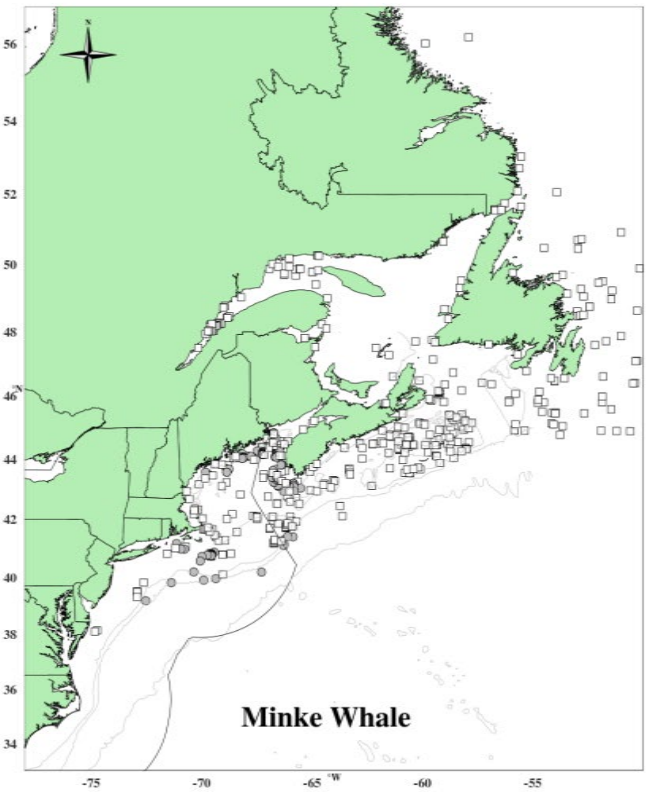


Figure 1. Distribution of minke whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-mm 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

In the North Atlantic, there are four recognized populations—Canadian East Coast, west Greenland, central North Atlantic, and northeastern North Atlantic (Donovan 1991). These divisions were defined by examining segregation by sex and length, catch distributions, sightings, marking data, and pre-existing ICES boundaries. However, there were very few data from the Canadian East Coast population. Anderwald *et al.* (2011) found no evidence for geographic structure comparing these putative populations but did, using individual genotypes and likelihood assignment methods, identify two cryptic stocks distributed across the North Atlantic. Until better information is available, common minke whales off the eastern coast of the United States are considered to be part of the Canadian East Coast stock, which inhabits the area from the western half of the Davis Strait (45ºW) to the Gulf of Mexico.

In summary, key uncertainties about stock structure are due to the limited understanding of the distribution, movements, and genetic structure of this stock. It is unknown whether the stock may contain multiple demographically independent populations that should be separate stocks. To date, no analyses of stock structure within this stock have been performed.

POPULATION SIZE

The best available current abundance estimate for common minke whales in the Canadian East Coast stock is the sum of the 2016 NEFSC and Department of Fisheries and Oceans Canada (DFO) surveys: 24,202 (CV=0.30). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area. This is assumed to be the majority of the Canadian East Coast stock. The 2016 estimate is larger than those from 2011 because the 2016 estimate is derived from a survey area extending from Newfoundland to Florida, which is about 1,300,000 km^2 larger than the 2011 survey area. In addition, some of the 2016 survey estimates in U.S. waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected.

A key uncertainty in the population size estimate is the precision and accuracy of the availability bias correction factor that was applied. More information on the spatio-temporal variability of the species' dive profile is needed.

Earlier Estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. As recommended in the 2016 guidelines for preparing stock assessment reports (NMFS 2016), estimates older than eight years are deemed unreliable for the determination of the current PBR.

Recent Surveys and Abundance Estimates

An abundance estimate of 2,802 (CV=0.81) minke whales was generated from a shipboard and aerial survey conducted during 27 June–28 September 2016 (Palka 2020) in a region covering 425,192 km^2. The aerial portion included 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout the U.S. waters. The shipboard portion consisted of 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the U.S. EEZ). Both sighting platforms used a two-team data collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers 2004). The estimates were also corrected for availability bias.

Abundance estimates of 6,158 (CV=0.40) minke whales from the Canadian Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf region and 13,008 (CV=0.46) minke whales from the Newfoundland/Labrador region were generated from an aerial survey conducted by the Department of Fisheries and Oceans, Canada (DFO). This survey covered Atlantic Canadian shelf and shelf-break waters extending from the northern tip of Labrador to the U.S. border off southern Nova Scotia in August and September of 2016 (Lawson and Gosselin 2018). A total of 29,123 km was flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf stratum using two Cessna Skymaster 337s and 21,037 km were flown over the Newfoundland/Labrador stratum using a DeHavilland Twin Otter. The Newfoundland estimate was derived from the Twin Otter data using two-team mark-recapture multi-covariate distance sampling methods. The Gulf of St. Lawrence estimate was derived from the Skymaster data using single-team multi-covariate distance sampling with left truncation (to accommodate the obscured area under the plane) where size-bias was also investigated, and the Otter-based perception bias correction was applied. An availability bias correction factor, which was based on the cetaceans' surface intervals, was applied to both abundance estimates.

Table 1. Summary of recent abundance estimates for the Canadian East Coast stock of common minke whales (Balaenoptera acutorostrata acutorostrata) by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and and coefficient of variation (CV). The best estimate is in bold font.

Month/Year	Area	N_{est}	CV
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	2,802	0.81
Aug–Sep 2016	Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf	6,158	0.40
Aug–Sep 2016	Newfoundland/Labrador	13,008	0.46
Jun–Sep 2016	**Central Virginia to Labrador – COMBINED**	**21,968**	**0.31**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for the Canadian East Coast stock of common minke whales is 21,968 animals (CV=0.30). The minimum population estimate is 17,022 animals.

Current Population Trend

A trend analysis has not been conducted for this stock. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and variable survey design (see Appendix IV for a survey history of this stock). For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). There is current work to standardize the strata-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. Life history parameters that could be used to estimate net productivity are that females mature between 6 and 8 years of age, and pregnancy rates are approximately 0.86 to 0.93. Based on these parameters, the mean calving interval is between 1 and 2 years. Calves are probably born during October to March after 10 to 11 months gestation and nursing lasts for less than 6 months. Maximum ages are not known, but for Southern Hemisphere minke whales maximum age appears to be about 50 years (IWC 1991).

For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995). Key uncertainties about the maximum net productivity rate are due to the limited understanding of the stock-specific life history parameters; thus the default value was used.

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 17,022. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5, the default value for stocks of unknown status relative to OSP and with the CV of the average mortality estimate less than 0.3 (Wade and Angliss 1997). PBR for the Canadian East Coast common minke whale is 170 (Table 2).

Table 2. Best and minimum abundance estimates for the Canadian East Coast stock of common minke whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
21,968	0.31	17,022	0.5	0.04	170

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Data to estimate the mortality and serious injury of common minke whales come from the Northeast Fisheries Science Center Observer Program, the At-Sea Monitor Program, and from records of strandings and entanglements in U.S. and Canadian waters. For the purposes of this report, mortalities and serious injuries from reports of strandings and entanglements considered to be confirmed human-caused mortalities or serious injuries are shown in Table 4 while those recorded by the Observer or At-Sea Monitor Programs are shown in Table 5. Summary statistics are shown in Table 3.

Table 3. The total annual estimated average human-caused mortality and serious injury for the Canadian East Coast stock of common minke whales.

Years	Source	Annual Avg.
2014−2018	Incidental fishery interactions non-observed	8.95
2014−2018	U.S. fisheries using observer data	0.2
2014−2018	Vessel collisions	1.20
2014−2018	Other human interaction	0.2
Total		**10.55**

Fishery-Related Serious Injury and Mortality

United States

U.S. fishery interaction records for large whales come through two main sources: dedicated fishery observer data and opportunistic reports collected in the Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database. One confirmed fishery-related mortality or serious injury of minke whales has been reported in the NMFS Sea Sampling bycatch database (fishery observers) during this reporting period (Table 4). A review of the records of stranded, floating, or injured minke whales for the reporting period 2014 through 2018 on file at NMFS found records in the audited Greater Atlantic Regional Fisheries Office/NMFS entanglement/stranding database with substantial evidence of fishery interactions causing injury or mortality (presented in Table 5; Henry *et al.* 2021). These records are not statistically quantifiable in the same way as the observer fishery records, and they almost surely undercount entanglements for the stock.

Mid-Atlantic Gillnet

In December 2016, one minke whale mortality was observed in mid-Atlantic gillnet gear. A mortality estimate was not expanded to the entire fishery because the observed mortality was such a rare event. See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Table 4. From observer program data, summary of the incidental mortality of Canadian East Coast stock of common minke whales (Balaenoptera acutorostrata acutorostrata) by commercial fishery including the years sampled, the type of data used, the annual observer coverage, the mortalities and serious injuries recorded by on-board observers, the estimated annual serious injury and mortality, the estimated CV of the annual mortality, and the mean annual combined mortality.

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury[c]	Est. Mort.	Est. Combined Mortality	Est. CVs	Mean Combined Annual Mortality	CV of Mean
	2014		0.05	0	0	0	0	0	0		
	2015	Obs.	0.06	0	0	0	0	0	0		
Mid-Atl.	2016	Data,	0.08	0	1	0	1	1	0	0.2	na
Gillnet	2017	Weighout	0.09	0	0	0	0	0	0		
	2018		0.09	0	0	0	0	0	0		
Total										**0.2**	

a. Observer data (Obs. Data) are used to measure bycatch rates and the data are collected within the Northeast Fisheries Observer Program. NEFSC collects weighout (Weighout) landings data that are used as a measure of total effort for the U.S. gillnet fisheries. Mandatory vessel trip report (VTR; Trip Logbook) data are used to determine the spatial distribution of fishing effort in the Northeast sink gillnet fishery.
b. Observer coverage for the U.S. Northeast gillnet fisheries is based on tons of fish landed.
c. Serious injuries were evaluated for the current period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2021).

Other Fisheries

Confirmed mortalities and serious injuries of common minke whales in the last five years as recorded in the audited Greater Atlantic Regional Office/NMFS entanglement/stranding database are reported in Table 5. One of the serious injury entanglement cases reported in Table 5 was a non-fishery interaction (strapping) and so 0.2 was subtracted from the total entanglement 5-year average. Most cases in which gear was recovered and identified involved gillnet or pot/trap gear.

Canada

Read (1994) reported interactions between common minke whales and gillnets in Newfoundland and Labrador, in cod traps in Newfoundland, and in herring weirs in the Bay of Fundy. Hooker *et al.* (1997) summarized bycatch data from a Canadian fisheries observer program that placed observers on all foreign fishing vessels operating in Canadian waters, on between 25% and 40% of large Canadian fishing vessels (greater than 100 feet long), and on approximately 5% of smaller Canadian fishing vessels. During 1991 through 1996, no common minke whales were observed taken. More current observer data are not available.

Other Fisheries

Mortalities and serious injuries that were likely a result of an interaction with Canadian fisheries are detailed in Table 5.

Table 5. Confirmed human-caused mortality and serious injury records of common minke whales (Balaenoptera acutorostrata acutorostrata): 2014–2018[a]

Date[b]	Fate	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
06/09/2014	Mortality	-	off Truro, MA	EN	1	US	PT	Fresh carcass anchored, hog-tied in gear. COD: peracute underwater entrapment.
07/02/2014	Mortality	-	Northumberland Strait, NB	EN	1	CN	NR	Carcass with constricting gear around lower jaw. Large open injury at attachment point on the left side.
07/10/2014	Prorated Injury	-	10 nm SE of Southport, ME	EN	0.75	XU	NR	Free-swimming, trailing 2 buoys. Attachment point(s) unknown.
07/12/2014	Serious Injury	-	10 nm S of Southampton, NY	EN	1	XU	NR	Free-swimming with yellow plastic strapping cutting into top and sides of rostrum. No trailing gear.
07/17/2014	Mortality	-	South Addison, ME	EN	1	XU	NP	Fresh carcass with line impression across ventral surface & evidence of constricting gear around peduncle and fluke insertion. Bruising evident at fluke injuries. No gear present.
07/29/2014	Mortality	-	5 nm E of Herring Cove, NS	VS	1	CN	-	Live animal w/ tongue completely ballooned out, forcing its jaws 90 degrees apart. Found dead at same location the next day. Carcass recovered with two traps & constricting line around the peduncle. Necropsy found indication of blunt trauma to right jaw. Animal anchored in gear was subsequently struck by a vessel (primary cause of death).
12/24/2014	Mortality	-	Dam Neck, VA	VS	1	US	-	Fresh carcass with broken ribs & fractured vertebrae w/ extensive hemorrhage & edema.
03/26/2015	Serious Injury	-	off Cape Canaveral, FL	EN	1	XU	NR	Evidence of constricting rostrum wrap, but unable to determine if gear still present. Emaciated.
04/16/2015	Mortality	-	Lockes Island, Shelburne, NS	EN	1	CN	NP	Fresh carcass with evidence of constricting wraps. No gear present. Robust, pregnant, fish in stomach and intestines. No other abnormalities noted.
05/09/2015	Mortality	-	Duck, NC	EN	1	XU	GU	Live stranded and euthanized. Embedded gear cutting into bone of mandible. Emaciated.
06/06/2015	Mortality	-	Coney Island, NY	VS	1	US	-	Fresh carcass with deep lacerations to throat area and head missing. Large area of bruising on dorsal surface.
06/14/2015	Prorated Injury	-	off Chatham, MA	EN	.75	XU	NR	Free-swimming with acorn buoy trailing 20-30 ft. Attachment point(s) and configuration unknown.
06/23/2015	Prorated Injury	-	off Ingonish, NS	EN	.75	CN	PT	Entangled in traps and buoys. Partially disentangled by fisherman. Original and final configuration unknown.
07/07/2015	Mortality	-	off Funk Island, NL	EN	1	CN	PT	Found at 340m depth in between two pots. Gear through mouth and wrapped around peduncle.
08/18/2015	Mortality	-	Roseville, PEI	EN	1	CN	NP	Evidence of constricting body, peduncle, and fluke wraps. No gear present. No necropsy but robust body condition supports entanglement as COD.
09/01/2015	Mortality	-	Gloucester, MA	EN	1	US	NP	Evidence of extensive, constricting gear with associated hemorrhaging. No gear present.

Date[b]	Fate	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
09/21/2015	Mortality	-	Cape Wolfe, Burton, PEI	EN	1	CN	NP	Evidence of constricting body wraps. No gear present. No necropsy but experts state peractute underwater entrapment most parsimonious.
12/06/2015	Mortality	-	off Port Joli, NS	EN	1	CN	PT	Live animal anchored in gear. Carcass recovered 4 days later.
05/03/2016	Mortality	-	Biddeford, ME	EN	1	US	PT	Line through mouth with evidence of constriction across ventral pleats and at peduncle. Hemorrhaging associated with these lesions.
07/21/2016	Serious Injury	-	Digby, NS	EN	1	XC	GU	Free-swimming with netting deeply embedded in rostrum. Disentangled, but significant health decline.
08/15/2016	Mortality	-	off Seguin Island, ME	EN	1	US	NR	Line exiting mouth leading to weighted/anchored gear.
08/30/2016	Mortality	-	3.1 nm SW of Matinicus Island, ME	EN	1	US	PT	Fresh carcass anchored in gear with evidence of constricting wraps at peduncle and fluke insertions
11/02/2016	Prorated Injury	-	Bonne Bay, Gros Morne National Park, NL	EN	0.75	XC	NR	Free-swimming and towing gear. Attachment point(s) and configuration unknown. No resights post 06Nov2016.
04/27/2017	Mortality	-	Staten Island, NY	VS	1	US	-	Evidence of bruising on dorsal and right scapular region. Histopathology results support blunt trauma from vessel strike most parsimonious as COD.
07/06/2017	Mortality	-	Manomet Point, MA	EN	1	US	PT	Live animal anchored in gear. Witnessed becoming entangled in second set. Gear hauled and animal found deceased with line through mouth and constricting wraps on peduncle.
07/22/2017	Mortality	-	Piscataqua River, NH	EN	1	US	NP	Evidence of multiple constricting wraps on lower jaw and ventral pleats with associated hemorrhaging. No gear present.
08/09/2017	Mortality	-	off Plymouth, MA	EN	1	US	NP	Evidence of constricting entanglement at fluke insertion, across fluke blades and ventral pleats. No necropsy but fresh carcass with extensive injuries supports COD of entanglement as most parsimonious.
08/11/2017	Prorated Injury	-	off York, ME	EN	0.75	US	NR	Partially disentangled from anchoring gear. Final configuration unknown.
08/12/2017	Mortality	-	off Tremont, ME	EN	1	US	GU	Fresh carcass of a pregnant female in gear. Constricting wrap injuries with associated hemorrhaging on dorsal and ventral surfaces and flukes.
08/14/2017	Mortality	-	Pt. Judith, RI	EN	1	US	NP	Evidence of constricting entanglement along left side with associated hemorrhaging. Found floating in stationary offshore fishing trap, but not entangled in trap gear. No gear present on animal.
08/17/2017	Mortality	-	Rye, NH	EN	1	US	NR	Evidence of constricting wraps on fluke blades and peduncle. Documented with line in baleen, but not present at time of necropsy. Limited necropsy, but extent of injuries and robust animal with evidence of recent feeding supports COD of entanglement as most parsimonious.
08/28/2017	Mortality	-	off Portland, ME	EN	1	US	PT	Fresh carcass anchored in gear. Endline wrapped around mouth and laceration from constricting gear on peduncle. Mud on flippers and mouth.

Date[b]	Fate	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
08/30/2017	Mortality	-	off North Cape, PEI	EN	1	CN	NR	Fresh carcass in gear. Full configuration unclear, but complex enough to not have drifted into post-mortem.
09/04/2017	Mortality	-	St. Carroll's, NL	EN	1	CN	NE	Alive in herring net. Found dead the next day. Fisher pulled carcass ashore and removed the net.
09/06/2017	Mortality	-	Newport, RI	VS	1	US	-	Hemorrhaging at left pectoral, left body, and aft of blowholes. Histopathology results support blunt trauma from vessel strike as COD.
09/17/2017	Mortality	-	Henry Island, NS	EN	1	CN	NR	Fresh carcass with gear in mouth and around flukes. Evidence of constricting wrap on dorsum. No necropsy, but configuration complex enough that unlikely to have drifted into gear post-mortem.
09/26/2017	Prorated Injury	-	off Richbuctou, NB	EN	0.75	CN	NR	Animal initially anchored in gear then not resighted. Unable to confirm if gear free, partially entangled, or drowned.
09/27/2017	Mortality	-	5.7nm NE of Richbuctou, NB	EN	1	CN	NP	No gear present. Fresh carcass with evidence of constricting wraps.
10/10/2017	Mortality	-	off Rockland, ME	EN	1	US	PT	Entangled in 2 different sets of gear. Constricting wrap around lower jaw. Found at depth when fisher hauled gear.
02/09/2018	Mortality	-	Tiverton, Long Island, NS	EN	1	XC	NP	No gear present. Evidence of constricting body, flipper, and peduncle wraps. No necropsy conducted, but COD from entanglement most parsimonious.
05/25/2018	Mortality	-	Digby, NS	VS	1	CN	-	Fresh carcass in harbor with large area of hemorrhage aft of blowholes. Necropsy did not state COD, but blunt trauma from vessel strike most parsimonious.
06/11/2018	Mortality	-	Cape Dauphin, NS	EN	1	CN	PT	Fresh, pregnant carcass anchored in gear.
06/19/2018	Mortality	-	East Point, PEI	EN	1	CN	NP	No gear present. Fresh, pregnant carcass with evidence of extensive constricting body and peduncle wraps with associated hemorrhaging.
06/22/2018	Prorated Injury	-	4.5 nm N of Grand Manan, NB	EN	0.75	XC	NR	Full configuration unclear — line across back, one buoy under left pectoral and another trailing 30–40ft aft. Reported as anchored but unable to confirm. Response team was not able to relocate.
06/24/2018	Mortality	-	Wellfleet, MA	EN	1	XU	GN	Evidence of extensive constricting body and mouth wraps with associated hemorrhaging. Deep lacerations at fluke insertion from constricting gear. COD - peracute underwater entrapment.
07/07/2018	Mortality	-	1.6 nm E of Newcastle, NH	EN	1	US	PT	Anchored in gear with line through mouth and wrapping around body. Associated bruising at right corner of mouth. COD - peracute underwater entrapment.
07/22/2018	Mortality	-	Cape Neddick, ME	EN	1	XU	NP	No necropsy, but evidence of constricting wrap at fluke insertion with associated hemorrhaging. Histopathology confirms pre-mortem human-induced trauma.
07/28/2018	Mortality	-	Biddeford, ME	EN	1	XU	NP	No gear present, but evidence of constricting gear with associated bruising at mouth, around body and peduncle.
08/06/2018	Prorated Injury	-	Fish Cove Point, NL	EN	0.75	CN	NE	Free-swimming towing net with float attached. Member of public cut off float. Original and final configuration unknown.

Date[b]	Fate	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
08/29/2018	Prorated Injury	-	7.5 nm SE of Chatham, MA	EN	0.75	XU	NR	Free-swimming with buoy near flukes, full configuration unknown.
09/03/2018	Mortality	-	Nancy Head, Campobello, NB	EN	1	CN	WE, SE	Live animal entrapped. Failed attempt by fisher to remove animal with seine. Animal became entangled in seine and drowned.
09/16/2018	Mortality	-	0.7 nm SSE of Rye, NH	EN	1	US	PT	Fresh carcass anchored in gear. Constricting body, jaw, peduncle, and fluke wraps with associated hemorrhaging.
11/07/2018	Mortality	-	Tangier Island, VA	EN	1	XU	NE	Constricting gear with associated hemorrhaging partly amputating tip of rostrum. Poor body condition. COD - chronic entanglement.
12/25/2018	Mortality	-	Yarmouth Bar, NS	EN	1	XC	NP	No gear present. Evidence of constricting entanglement on head, ventral pleats, peduncle and flukes. No necropsy, but COD from entanglement most parsimonious.

Assigned Cause	Five-Year Mean (US/CN/XU/XC)
Vessel strike	1.2 (0.8/0.4/0/0)
Entanglement	8.95 (3.15/2.85/2.05/0.9)

a. For more details on events, see Henry *et al.* 2021.
b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.
c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012).
d. US=United States, XU=Unassigned 1st sight in U.S., CN=Canada, XC=Unassigned 1st sight in CN.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.
f. Assigned cause: EN=entanglement, VS=vessel strike, ET=entrapment (summed with entanglement).

Other Mortality

North Atlantic common minke whales have been and continue to be hunted. From the Canadian East Coast population, documented whaling occurred from 1948 to 1972 with a total kill of 1,103 animals (IWC 1992). Animals from other North Atlantic common minke populations (e.g., Iceland) are presently being hunted.

United States

Common minke whales inhabit coastal waters during much of the year and are thus susceptible to collision with vessels. Vessel strike interactions in U.S. and Canadian waters are reported in Table 5. In January 2017, a minke whale Unusual Mortality Event (UME) was declared for the U.S. Atlantic coast due to elevated numbers of mortalities. From January 2017 to December 2018, 57 minke whales stranded between Maine and South Carolina. Preliminary findings in several of the whales have shown evidence of human interactions or infectious disease. This most recent UME is ongoing (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2020-minke-whale-unusual-mortality-event-along-atlantic-coast; Accessed 31July2020). Anthropogenic mortalities and serious injuries that occurred in 2017 and 2018 as part of this UME are included in Table 5.

Canada

The Nova Scotia Stranding Network documented whales and dolphins stranded on the coast of Nova Scotia between 1991 and 1996 (Hooker *et al.* 1997). Researchers with the Department of Fisheries and Oceans Canada documented strandings on the beaches of Sable Island (Lucas and Hooker 2000). Common minke whales stranded on the coast of Nova Scotia were recorded by the Marine Animal Response Society (MARS) and the Nova Scotia Stranding Network (Tonya Wimmer/Andrew Reid, pers. comm.).

The Whale Release and Strandings program report common minke whale stranding mortalities in Newfoundland and Labrador (Ledwell and Huntington 2013, 2014, 2015, 2017, 2018). Those that have been determined to be human-caused serious injury or mortality are included in Table 5.

HABITAT ISSUES

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in and predicted for a range of plankton species and commercially important fish stocks (Nye *et al.* 2009, Head *et al.* 2010, Pinsky *et al.* 2013, Poloczanska *et al.* 2013, Hare *et al.* 2016, Grieve *et al.* 2017, Morley *et al*. 2018) and cetacean species (e.g., MacLeod 2009; Sousa *et al*. 2019). There is uncertainty in how, if at all, the distribution and population size of this species will respond to these changes and how the ecological shifts will affect human impacts to the species.

Human-made noises have been shown to impact common minke whales. A study in the Northwest Atlantic, investigated the potential of vessel noise to mask baleen whale vocalizations and found an 80% loss of communication space for minke whale pulse trains relative to historical "quiet" conditions (Cholewiak *et al*. 2018). Minke whales have been observed to respond to mid-frequency active sonar and other training activities by reducing or ceasing calling and by exhibiting avoidance behaviors (Harris *et al.* 2019, Martin *et al.* 2015). In addition, they have strongly avoided acoustic deterrent devices that were used as noise mitigation of construction activities (McGarry *et al*. 2017).

Although levels of persistent organic pollutants are decreasing in many cetacean species, elevated concentrations of persistent organic pollutants and emerging halogenated flame retardants have been reported in tissues of minke whales in the St. Lawrence Estuary in Canada that may affect the regulation of the thyroid and/or steroid axes (Simond *et al.* 2019).

STATUS OF STOCK

Common minke whales are not listed as threatened or endangered under the Endangered Species Act, and the Canadian East Coast stock is not considered strategic under the Marine Mammal Protection Act. The total U.S. fishery-related mortality and serious injury for this stock is less than 10% of the calculated PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of common minke whales relative to OSP in the U.S. Atlantic EEZ is unknown.

It is expected that the uncertainties described above will have little effect on the designation of the status of the entire stock. Even though the estimate of human-caused mortality and serious injury in this assessment (8 animals) is negatively biased due to using strandings and entanglement data as the primary source, it is well below the PBR calculated from the abundance estimate for the U.S. and Canadian portion of the Canadian East Coast common minke whale stock's habitat.

REFERENCES CITED

Anderwald, P., A.K. Daníelsdóttir, T. Haug, F. Larsen, V. Lesage, R.J. Reid, G.A. Víkingsson and A.R. Hoelzel. 2011. Possible cryptic stock structure for minke whales in the North Atlantic: Implications for conservation and management. Biol. Conserv. 144:2479–2489.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

CETAP [Cetacean and Turtle Assessment Program]. 1982. A characterization of marine mammals and turtles in the mid- and North Atlantic areas of the U.S. outer continental shelf. Final report #AA551-CT8-48. Cetacean and Turtle Assessment Program, University of Rhode Island. Bureau of Land Management, Washington, DC. 576pp.

Cholewiak, D., C.W. Clark, D. Ponirakis, A. Frankel, L.T. Hatch and D. Risch. 2018. Communicating amidst the noise: Modeling the aggregate influence of ambient and vessel noise on baleen whale communication space in a national marine sanctuary. Endanger. Species Res. 36:59–75. https://doi.org/10.3354/esr00875

Clark, C.W. and G.C. Gagnon. 2002. Low-frequency vocal behaviors of baleen whales in the North Atlantic: Insights from IUSS detections, locations and tracking from 1992 to 1996. J. Underwater Acoust. (USN) 52(3):609–640.

Donovan, G.P. 1991. A review of IWC stock boundaries. Rep. Int. Whal. Comm. (Special Issue) 13:39–68.

Grieve, B.D., J.A. Hare and V.S. Saba. 2017. Projecting the effects of climate change on *Calanus finmarchicus* distribution within the US Northeast continental shelf. Sci. Rep. 7:6264.

Hare, J.A., W.E. Morrison, M.W. Nelson, M.M. Stachura, E.J. Teeters, R.B. Griffis, M.A. Alexander, J.D. Scott, L. Alade, R.J. Bell, A.S. Chute, K.L. Curti, T.H. Curtis, D. Kurcheis, J.F. Kocik, S.M. Lucey, C.T. McCandless, L.M. Milke, D.E. Richardson, E. Robillard, H.J. Walsh, M.C. McManus, K.E. Maranick and C.A. Griswold. 2016. A vulnerability assessment of fish and invertebrates to climate change on the Northeast U.S. continental shelf. PLoS ONE 11:e0146756.

Harris, C.M., S.W. Martin, C. Martin, T.A. Helbel, E.E.Henderson, C.G.M. Paxton and L. Thomas. Changes in the spatial distribution of acoustically derived minke whale (*Balaenoptera acutorostrata*) tracks in response to Navy training. Aquatic Mammals 45:661–674.

Head, E.J.H. and P. Pepin. 2010. Spatial and inter-decadal variability in plankton abundance and composition in the Northwest Atlantic (1958–2006). J. Plankton Res. 32:1633–1648.

Henry, A.G., M. Garron, D. Morin, A. Smith, A. Reid, W. Ledwell and T. Cole. 2021. Serious injury and mortality determinations for baleen whale stocks along the Gulf of Mexico, United States east coast and Atlantic Canadian provinces, 2014–2018. NEFSC Ref. Doc. 21-07.

Hooker, S.K., R.W. Baird and M.A. Showell. 1997. Cetacean strandings and bycatches in Nova Scotia, eastern Canada, 1991–1996. Unpublished Scientific Committee meeting document SC/49/O5. International Whaling Commission, Cambridge, UK.

IWC [International Whaling Commission]. 1991. Appendix 11. Biological parameters of North Atlantic minke whales. *in:* Annex F: Report of the sub-committee on North Atlantic minke whales. Rep. Int. Whal. Comm. 41:160.

IWC [International Whaling Commission]. 1992. Annex K: Report of the working group on North Atlantic minke whales. Rep. Int. Whal. Comm. 42:246–251.

Josephson, E., F. Wenzel and M.C. Lyssikatos. 2021. Serious injury determinations for small cetaceans and pinnipeds caught in commercial fisheries off the northeast U.S. coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-04. 33pp.

Lawson, J. and J-F. Gosselin. 2018 Estimates of cetacean abundance from the 2016 NAISS aerial surveys of eastern Canadian waters, with a comparison to estimates from the 2007 TNASS. NAAMCO SC/25/AE/09.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Advanced distance sampling. Oxford University Press, New York, New York.

Ledwell, W. and J. Huntington. 2014. Incidental entrapments and entanglements of cetaceans and leatherback sea turtles, strandings, ice entrapments reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2014. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 23pp.

Ledwell, W. and J. Huntington. 2015. Incidental entrapments and entanglements of cetaceans and leatherback sea turtles, strandings, ice entrapments reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2015. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 22pp.

Ledwell, W. and J. Huntington. 2017. Incidental entrapments and entanglements of cetaceans and leatherback sea turtles, strandings, ice entrapments reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2016. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 22pp.

Ledwell, W. and J. Huntington. 2018. Incidental entrapments and entanglements of cetaceans and leatherback sea turtles, strandings, ice entrapments reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2017. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 24pp.

Lucas, Z.N. and S.K. Hooker. 2000. Cetacean strandings on Sable Island, Nova Scotia, 1970–1998. Can. Field-Nat. 114:46–61.

MacLeod, C.D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: A review and synthesis. Endang. Species Res. 7:125–136.

Martin, S.W., C.R. Martin, B.M. Matsuyama and E.E. Henderson. 2015. Minke whales (*Balaenoptera acutorostrata*) respond to navy training. J. Acoust. Soc. Am. 137:2533–2541. https://doi.org/10.1121/1.4919319

McGarry, T., O. Boisseau, S. Stephenson and R. Compton. 2017. Understanding the effectiveness of acoustic deterrent devices (ADDs) on minke whale (*Balaenoptera acutorostrata*), a low-frequency cetacean. ORJIP Project 4 Report EOR0692. London: The Carbon Trust.

Mitchell, E.D. 1991. Winter records of the minke whale (*Balaenoptera acutorostrata* Lacepede 1804) in the southern North Atlantic. Rep. Int. Whal. Comm. 41:455–457.

Morley, J.W., R.L. Selden, R.J. Latour, T.L. Frolicher, R.J. Seagraves and M.L. Pinsky. 2018. Projecting shifts in thermal habitat for 686 species on the North American continental shelf. PLoS ONE 13(5):e0196127.

NMFS [National Marine Fisheries Service]. 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23pp.
Available from: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks

Nye, J., J. Link, J. Hare and W. Overholtz. 2009. Changing spatial distribution of fish stocks in relation to climate and population size on the Northeast United States continental shelf. Mar. Ecol. Prog. Ser. 393:111–129.

Orphanides, C.D. 2020. Estimates of cetacean and pinniped bycatch during 2017 in the New England and mid-Atlantic sink gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 20-03. 16pp.

Palka, D. 2020. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2016 line transect surveys conducted by the Northeast Fisheries Science Center. Northeast Fish. Sci. Cent. Ref. Doc. 20-05.

Pinsky, M.L., B. Worm, M.J. Fogarty, J.L. Sarmiento and S.A. Levin. 2013. Marine taxa track local climate velocities. Science 341:1239–1242.

Poloczanska, E.S., C.J. Brown, W.J. Sydeman, W. Kiessling, D.S. Schoeman, P.J. Moore, K. Brander, J.F. Bruno, L.B. Buckley, M.T. Burrows, C.M. Duarte, B.S. Halpern, J. Holding, C.V. Kappel, M.I. O'Connor, J.M. Pandolfi, C. Parmesan, F. Schwing, S.A. Thompson and A.J. Richardson. 2013. Global imprint of climate change on marine life. Nat. Clim. Change 3:919–925.

Read, A.J. 1994. Interactions between cetaceans and gillnet and trap fisheries in the northwest Atlantic. Rep. Int. Whal. Comm. (Special Issue) 15:133–147.

Risch, D., C.W. Clark, P.J. Dugan, M. Popescu, U. Siebert and S.M. VanParijs. 2013. Minke whale acoustic behavior and multi-year seasonal and diel vocalization patterns in Massachusetts Bay, USA. Mar. Ecol. Prog. Ser. 489:279–295.

Risch, D., M. Castellote, C.W. Clark, G.E. Davis, P.J. Dugan, L.E.W. Hodge, A. Kumar, K. Lucke, D.K. Mellinger, S.L. Nieukirk, C.M. Popescu, C. Ramp, A.J. Read, A.N. Rice, M.A. Silva, U. Siebert, K.M. Stafford, H. Verdaat and S.M. VanParijs. 2014. Seasonal migrations of North Atlantic minke whales: Novel insights from large-scale passive acoustic monitoring networks. Mov. Ecol. 2:24.

Rodriguez, G., R. Reyes, N.M. Hammerman and J.E. Garcia-Hernandez. 2018. Cetacean sightings in Puerto Rican waters: Including the first underwater photographic documentation of a minke whale (*Balaenoptera acutorostrata*). Latin American J. of Aquatic Mammals 13:26–36.

Simond, A.E., M. Houde, V. Lesage, R. Michaud, D. Zbinden and J. Verreault. 2019. Associations between organohalogen exposure and thyroid- and steroid-related gene response in St. Lawrence Estuary belugas and minke whales. Mar. Poll. Bull. 145:174–184.

Sousa, A., F. Alves, A. Dinis, J. Bentz, M.J. Cruz and J.P. Nunes. 2019. How vulnerable are cetaceans to climate change? Developing and testing a new index. Ecol. Indic. 98:9–18.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23:157–175.

Thomas L., J.L. Laake, E. Rexstad, S. Strindberg, F.F.C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, M.L. Burt, S.L. Hedley, J.H. Pollard, J.R.B. Bishop and T.A. Marques. 2009. Distance 6.0. Release 2. [Internet]. University of St. Andrews (UK): Research Unit for Wildlife Population Assessment. Available from: http://distancesampling.org/Distance/

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

COMMON DOLPHIN (*Delphinus delphis delphis*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

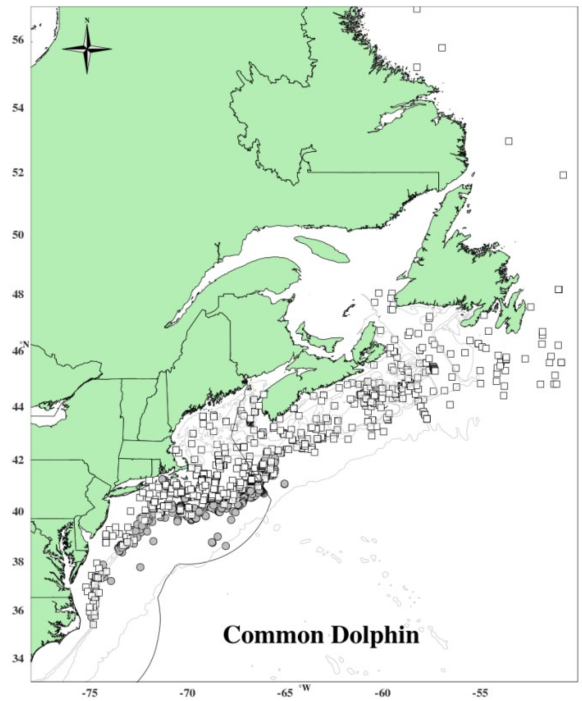


Figure 1. Distribution of common dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2010, 2011, 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 1000-m and 4000-m depth contours.

The common dolphin (*Delphinus delphis delphis*) may be one of the most widely distributed species of cetaceans, as it is found world-wide in temperate and subtropical seas. In the North Atlantic, common dolphins are commonly found along the shoreline of Massachusetts in mass-stranding events (Bogomolni *et al.* 2010, Sharp *et al.* 2014). At-sea sightings have been concentrated over the continental shelf between the 100-m and 2000-m isobaths and over prominent underwater topography and east to the mid-Atlantic Ridge (29˚W) (Doksaeter *et al.* 2008, Waring *et al*. 2008). Common dolphins have been noted to be associated with Gulf Stream features (CETAP 1982, Selzer and Payne 1988, Waring *et al.* 1992, Hamazaki 2002). The species is less common south of Cape Hatteras, although schools have been reported as far south as the Georgia/South Carolina border (32ºN; Jefferson *et al*. 2009). They exhibit seasonal movements, where they are found from Cape Hatteras northeast to Georges Bank (35˚ to 42˚N) during mid-January to May (Hain *et al.* 1981, CETAP 1982, Payne *et al.* 1984), although some animals tagged and released after stranding in winters of 2010–2012 used habitat in the Gulf of Maine north to almost 44˚N (Sharp *et al*. 2016). Common dolphins move onto Georges Bank, Gulf of Maine, and the Scotian Shelf from mid-summer to autumn. Selzer and Payne (1988) reported very large aggregations (greater than 3,000 animals) on Georges Bank in autumn. Migration onto the Scotian Shelf and continental shelf off Newfoundland occurs during summer and autumn when water temperatures exceed 11ºC (Sergeant *et al.* 1970, Gowans and Whitehead 1995).

Westgate (2005) tested the proposed one-population-stock model using a molecular analysis of mitochondrial DNA (mtDNA), as well as a morphometric analysis of cranial specimens. Both genetic analysis and skull morphometrics failed to provide evidence ($p>0.05$) of more than a single population in the western North Atlantic, supporting the proposed one-stock model. However, when western and eastern North Atlantic common dolphin mtDNA and skull morphology were compared, both the cranial and mtDNA results showed evidence of restricted gene flow ($p<0.05$) indicating that these two areas are not panmictic. Cranial specimens from the two sides of the North Atlantic differed primarily in elements associated with the rostrum. These results suggest that common dolphins in the western North Atlantic are composed of a single panmictic group whereas gene flow between the western and eastern North Atlantic is limited (Westgate 2005, 2007). This was further supported by Mirimin *et al.* (2009) who investigated genetic variability using both nuclear and mitochondrial genetic markers and observed no significant genetic differentiation between samples from within the western North Atlantic region, which may be explained by seasonal shifts in distribution between northern latitudes (summer months) and southern latitudes (winter months). However, the authors point out that some uncertainty remains if the same population was sampled in the two different seasons.

POPULATION SIZE

The current best abundance estimate for Western North Atlantic stock of common dolphins is 172,947 (CV=0.21) which is the total of Canadian and U.S. surveys conducted in 2016 (Table 1). This estimate, derived from shipboard and aerial surveys, covers most of this stock's known range. Because the survey areas did not overlap, the estimates from the three surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area. The 2016 estimate is larger than those from 2011 because the 2016 estimate is derived from a survey area extending from Newfoundland to Florida, which is about 1,300,000 km^2 larger than the 2011 survey area. In addition, some of the 2016 survey estimates in US waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected.

Earlier Estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. As recommended in the guidelines for preparing Stock Assessment Reports (NMFS 2016), estimates older than eight years are deemed unreliable to determine a current PBR.

Recent Surveys and Abundance Estimates

Abundance estimates of 48,723 (CV=0.48) for the Newfoundland/Labrador portion and 43,124 (CV=0.28) for the Bay of Fundy/Scotian Shelf/Gulf of St. Lawrence portion of the stock area were generated from the Canadian Northwest Atlantic International Sightings Survey (NAISS) survey conducted in August–September 2016 (Table 1). This large-scale aerial survey covered Atlantic Canadian shelf and shelf break habitats from the northern tip of Labrador to the U.S border off southern Nova Scotia (Lawson and Gosselin 2018). Line-transect density and abundance analyses were completed using Distance 7.1 release 1 (Thomas *et al*. 2010).

Abundance estimates of 80,227 (CV=0.31) and 900 (CV=0.57) common dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020, Palka 2020). One survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and consisted of 5,354 km of on-effort trackline along the shelf break and offshore to the outer limit of the U.S. EEZ (NEFSC and SEFSC 2018). The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the outer limit of the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort (NEFSC and SEFSC 2018). Both surveys utilized two visual teams and an independent observer approach to estimate detection probability on the trackline (Laake and Borchers 2004). Mark-recapture distance sampling was used to estimate abundance. Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1. Summary of recent abundance estimates for western North Atlantic common dolphin (Delphinus delphis delphis) by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and coefficient of variation (CV). The estimate considered best in in bold font.

Month/Year	Area	N_{est}	CV
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	80,227	0.31
Jun–Aug 2016	Florida to Central Virginia	900	0.57
Jun–Sep 2016	Newfoundland/Labrador	48,723	0.48
Jun–Sep 2016	Bay of Fundy/Scotian Shelf/Gulf of St. Lawrence	43,124	0.28
Jun–Sep 2016	**Florida to Newfoundland/Labrador (COMBINED)**	**172,974**	**0.21**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for common dolphins is 172,974 animals (CV=0.21), derived from the 2016 aerial and shipboard surveys. The minimum population estimate for the western North Atlantic common dolphin is 145,216.

Current Population Trend

A trend analysis has not been conducted for this stock. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long survey interval (see Appendix IV for a survey history of this stock). For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). There is current work to standardize the strata-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There is limited published life-history information that could be used to estimate net productivity. Westgate (2005) and Westgate and Read (2007) have provided reviews with a number of known parameters. There is a peak in parturition during July and August with an average birth date of 28 July. Gestation lasts about 11.7 months and lactation lasts at least a year. Given these results, western North Atlantic female common dolphins likely average 2–3 year calving intervals. Females become sexually mature earlier (8.3 years and 200 cm) than males (9.5 years and 215 cm) as males continue to increase in size and mass. There is significant sexual dimorphism present with males being on average about 9% larger in body length.

Due to uncertainties about the stock-specific life-history parameters, the maximum net productivity rate was assumed to be the default value for cetaceans of 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 145,216 animals. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5, the default value for stocks of unknown status and with the CV of the average mortality estimate less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic stock of common dolphin is 1,452.

Table 2. Best and minimum abundance estimates for the western North Atlantic common dolphin (Delphinus delphis delphis) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
172,974	0.21	145,216	0.5	0.04	1,452

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Average annual estimated fishery-related mortality or serious injury to this stock during this reporting period are presented in Table 3.

Table 3. The total annual estimated average human-caused mortality and serious injury for the western North Atlantic common dolphin (Delphinus delphis delphis).

Years	Source	Annual Average	CV
2014–2018	U.S. fisheries using observer data	399	0.1
2014–2018	Research mortalities	0.2	
Total		**399**	

Uncertainties not accounted for include the potential that the observer coverage was not representative of the fishery during all times and places. There are no major known sources of unquantifiable human-caused mortality or serious injury for this stock.

Northeast Sink Gillnet

Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Hatch and Orphanides 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

Common dolphins were taken in observed trips during most years. Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Hatch and Orphanides 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Bottom Trawl

This fishery is active in New England waters in all seasons. Annual common dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos *et al.* 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Bottom Trawl

Annual common dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos *et al.* 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Pelagic Longline

Pelagic longline bycatch estimates of common dolphins for 2014–2018 were documented in Garrison and Stokes (2016, 2017, 2020a, 2020b). There is a high likelihood that dolphins released alive with ingested gear or gear wrapped around appendages will not survive (Wells *et al.* 2008). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Research Takes

In October 2016, the University of Rhode Island, Graduate School of Oceanography reported the incidental capture/drowning of a 206 cm female, common dolphin during a routine, weekly research trawl fishing trip in Narragansett Bay, Rhode Island. The incident was reported four ways: 1) Mystic Aquarium, Mystic, Connecticut, 2) NOAA GARFO Office, Gloucester, Massachusetts, 3) NOAA law enforcement, and 4) the NOAA Protected Species Branch, Woods Hole, Massachusetts. A complete necropsy was conducted at the Wood Hole Oceanographic Institution, Woods Hole, Massachusetts.

Table 4. Summary of the incidental serious injury and mortality of North Atlantic common dolphins (Delphinus delphis delphis) by commercial fishery including the years sampled, the type of data used, the annual observer coverage, the serious injuries and mortalities recorded by on-board observers, the estimated annual serious injury and mortality, the combined serious injury and mortality estimate, the estimated CV of the annual combined serious injury and mortality and the mean annual serious injury and mortality estimate (CV in parentheses).

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury[d]	Observed Mortality	Estimated Serious Injury[d]	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
Northeast Sink Gillnet	2014	Obs.	0.18	0	11	0	111	0	0.47	94 (0.19)
	2015	Trip	0.14	0	3	0	55	0	0.54	
	2016	Logbook,	0.10	0	8	0	80	0	0.38	
	2017	Allocated	0.12	0	20	0	133	0	0.28	
	2018	Dealer Data	0.11	0	10	0	93	0	0.45	
Mid-Atlantic Gillnet	2014	Obs. Data, Weighout	0.05	0	1	0	17	17	0.86	17 (0.34)
	2015		0.06	0	3	0	30	30	0.55	
	2016		0.08	0	1	0	7	7	0.97	
	2017		0.09	1	1	11	11	22	0.71	
	2018		0.09	0	1	0	7.7	7.7	0.54	
Northeast Bottom Trawl [c]	2014	Obs. Data, Logbook Data	0.19	0	3	0	17	17	0.53	17 (0.26)
	2015		0.19	0	4	0	22	22	0.45	
	2016		0.12	0	2	0	16	16	0.46	
	2017		0.12	0	0	0	0	0	0	
	2018		0.12	0	4	0	28	28	0.54	
Mid-Atlantic Bottom Trawl [c]	2014	Obs. Data, Dealer Data	0.08	3	38	24	305	329	0.29	268 (0.13)
	2015		0.09	0	26	0	250	250	0.32	
	2016		0.10	0	22	0	177	177	0.33	
	2017		0.10	0	66	0	380	380	0.23	
	2018		0.12	1	34	5	200	205	0.54	
Pelagic Longline	2014	Obs. Data, Logbook Data	0.10	0	0	0	0	0	0	3.1 (0.67)
	2015		0.12	1	0	9.05	0	9.05	1	
	2016		0.15	0	0	0	0	0	0	
	2017		0.12	1	0	4.92	0	4.92	1	
	2018		0.10	1	0	1.44	0	1.44	1	
Total										**399 (0.1)**

a. Observer data (Obs. Data), used to measure bycatch rates, are collected within the Northeast Fisheries Observer Program and At-sea Monitoring Program. NEFSC collects landings data (unallocated Dealer Data or Allocated Dealer Data) which are used as a measure of total landings and mandatory Vessel Trip Reports (VTR; Trip Logbook) are used to determine the spatial distribution of landings and fishing effort.

b. Observer coverage is defined as the ratio of observed to total metric tons of fish landed for the gillnet fisheries and the ratio of observed to total trips for bottom trawl and mid-Atlantic mid-water trawl (including pair trawl) fisheries.

c. Fishery related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos *et al.* 2021).

d. Serious injuries were evaluated for the period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2021).

Other Mortality

Common dolphins reported stranded between Maine and Florida are reported in Table 5 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 20 November 2019). The total includes mass-stranded common dolphins in Massachusetts during 2014 (a total of 14 in 4 events), 2015 (a total of 37 in 13 events), 2016 (a total of 35 animals in nine events), 2017 (over 90 animals in 20 events), and 2018 (a total of 28 animals in nine events). Animals released or last sighted alive include 12 animals in 2014, 9 in 2015, 23 in 2016, 70 in 2017 and 18 in 2018. In 2014, five cases were classified as human interaction, one of which was a fishery interaction. In 2015, two cases were classified as human interactions, both in Rhode Island. Seven cases in 2016 were coded as human interaction, one of which was a fishery interaction. Six cases in 2017 were coded as human interaction, two of which were classified as fishery interactions and one of which was classified as a boat collision. In 2018, five cases were coded as human interactions, three of which involved fishing gear. In an analysis of mortality causes of stranded marine mammals on Cape Cod and southeastern Massachusetts between 2000 and 2006, Bogomolni (2010) reported that 61% of stranded common dolphins were involved in mass-stranding events, and 37% of all the common dolphin stranding mortalities were disease-related.

The Marine Animal Response Society of Nova Scotia reported 3 common dolphins stranded in 2014, 2 in 2015, 5 in 2016, 5 in 2017 and 5 in 2018 (Tonya Wimmer/Andrew Reid, pers. comm.).

Table 5. Common dolphin (Delphinus delphis delphis) reported strandings along the U.S. Atlantic coast, 2014–2018.

State	2014	2015	2016	2017	2018	Totals
New Hampshire	0	1	1	2	0	4
Massachusetts[a]	38	40	67	166	61	372
Rhode Island[b]	6	7	4	5	4	26
Connecticut	0	2	1	1	0	4
New York	11	3	3	15	11	43
New Jersey	8	3	5	0	2	18
Delaware	0	2	0	0	0	2
Maryland	0	1	0	0	0	1
Virginia	9	2	0	1	3	15
North Carolina	6	4	1	0	3	14
Totals	**78**	**65**	**82**	**190**	**84**	**499**

Stranding data probably underestimate the extent of fishery-related mortality and serious injury because all of the marine mammals that die or are seriously injured may not wash ashore, nor will all of those that do wash ashore necessarily show signs of entanglement or other fishery interaction. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of fishery interaction. However, a recently published human interaction manual (Barco and Moore 2013) and case criteria for human interaction determinations (Moore *et al.* 2013) should help with this.

HABITAT ISSUES

The chronic impacts of contaminants (polychlorinated biphenyls [PCBs] and chlorinated pesticides [DDT, DDE, dieldrin, etc.]) on marine mammal reproduction and health are of concern (e.g., Pierce *et al.* 2008; Jepson *et al.* 2016; Hall *et al.* 2018; Murphy *et al.* 2018), but research on contaminant levels for the western north Atlantic stock of common dolphins is lacking.

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in or predicted for plankton species and commercially important fish stocks (Nye *et al*. 2009, Head *et al*. 2010, Pinsky *et al*. 2013, Poloczanska *et al.* 2013, Hare *et al*. 2016, Grieve *et al.* 2017, Morley *et al.* 2018) and cetacean species (e.g., MacLeod 2009; Sousa *et al.* 2019). There is uncertainty in how, if at all, the distribution and population size of this species will respond to these changes and how the ecological shifts will affect human impacts to the species.

STATUS OF STOCK

Common dolphins are not listed as threatened or endangered under the Endangered Species Act, and the Western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The 2014–2018 average annual human-related mortality does not exceed PBR. The total U.S. fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The status of common dolphins, relative to OSP, in the U.S. Atlantic EEZ is unknown.

REFERENCES CITED

Barco, S. and K.T. Moore. 2013. Handbook for recognizing, evaluating and documenting human interactions in stranded cetaceans and pinnipeds. NOAA Tech. Memo. NMFS-SWFSC-510. 102pp. http://swfsc.noaa.gov/publications/TM/SWFSC/NOAA-TM-NMFS-SWFSC-510.pdf.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Bogomolni, A.L., K.R. Pugliares, S.M. Sharp, K. Patchett, C.T. Harry, J.M. LaRocque, K.M. Touhey and M. Moore. 2010. Mortality trends of stranded marine mammals on Cape Cod and southeastern Massachusetts, USA, 2000 to 2006. Dis. Aquat. Org. 88:143–155.

Buckland, S.T., D.R. Anderson, K.P. Burnham, J.L. Laake, D.L. Borchers and L. Thomas. 2001. Introduction to distance sampling: Estimating abundance of biological populations. Oxford University Press, London, UK. 432pp.

CETAP [Cetacean and Turtle Assessment Program]. 1982. A characterization of marine mammals and turtles in the mid- and North Atlantic areas of the U.S. outer continental shelf. Final report # AA551-CT8-48. Cetacean and Turtle Assessment Program, University of Rhode Island. Bureau of Land Management, Washington, DC. 576pp.

Doksaeter, L., E. Olsen, L. Nottestad and A. Ferno. 2008 Distribution and feeding ecology of dolphins along the mid-Atlantic Ridge between Iceland and the Azores. Deep Sea Res. II 55:243–253.

Garrison, L.P. 2020. Abundance of cetaceans along the southeast U.S. east coast from a summer 2016 vessel survey. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRD Contribution # PRD-2020-04. 17pp.

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp.

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 67pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. PRBD Contribution # PRBD-2020-05. 61pp. https://repository.library.noaa.gov/view/noaa/26506.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. PRBD Contribution # PRBD-2020-08. 56pp.

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Gowans, S. and H. Whitehead. 1995. Distribution and habitat partitioning by small odontocetes in the Gully, a submarine canyon on the Scotian Shelf. Can. J. Zool. 73:1599–1608.

Grieve, B.D., J.A. Hare and V.S. Saba. 2017. Projecting the effects of climate change on *Calanus finmarchicus* distribution within the US Northeast continental shelf. Sci. Rep. 7:6264.

Hain, J.H.W., R.K. Edel, H.E. Hays, S.K. Katona and J.D. Roanowicz. 1981. General distribution of cetaceans in the continental shelf waters of the northeastern United States. Pages II.1–II.345 *in:* Cetacean and Turtle Assessment Program, University of Rhode Island. A characterization of marine mammals and turtles in the mid- and North Atlantic areas of the US outer continental shelf, annual report for 1979, AA551-CT8-48. Bureau of Land Management, Washington, DC.

Hall, A.J., B.J. McConnell, L.J. Schwacke, G.M. Ylitalo, R. Williams and T.K. Rowles. 2018. Predicting the effects of polychlorinated biphenyls on cetacean populations through impacts on immunity and calf survival. Environ. Poll. 233:407–418.

Hamazaki, T. 2002. Spatiotemporal prediction models of cetacean habitats in the mid-western North Atlantic Ocean (from Cape Hatteras, North Carolina, U.S.A to Nova Scotia, Canada). Mar. Mamm. Sci. 18:920–939.

Hare, J.A., W.E. Morrison, M.W. Nelson, M.M. Stachura, E.J. Teeters, R.B. Griffis, M.A. Alexander, J.D. Scott, L. Alade, R.J. Bell, A.S. Chute, K.L. Curti, T.H. Curtis, D. Kurcheis, J.F. Kocik, S.M. Lucey, C.T. McCandless, L.M. Milke, D.E. Richardson, E. Robillard, H.J. Walsh, M.C. McManus, K.E. Maranick and C.A. Griswold. 2016. A vulnerability assessment of fish and invertebrates to climate change on the Northeast U.S. continental shelf. PLoS ONE 11:e0146756.

Head, E.J.H. and P. Pepin. 2010. Spatial and inter-decadal variability in plankton abundance and composition in the Northwest Atlantic (1958–2006). J. Plankton Res. 32:1633–1648. https://doi.org/10.1093/plankt/fbq090

Hatch, J.M. and C.D. Orphanides. 2016. Estimates of cetacean and pinniped bycatch in the 2014 New England sink and Mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 16-05. 22pp.

Jefferson, T.A., D. Fertl, J. Bolanos-Jimenez and A.N. Zerbini. 2009. Distribution of common dolphins (*Delphinus* spp.) in the western North Atlantic: A critical re-examination. Mar. Biol. 156:1109–1124.

Jepson, P.D., R. Deaville, J.L. Barber, A. Aguilar, A. Borrell, S. Murphy, J. Barry, A. Brownlow, J. Barnett, S. Berrow and A.A. Cunningham. 2016. PCB pollution continues to impact populations of orcas and other dolphins in European waters. Sci. Rep.-U.K. 6:18573.

Josephson, E., F. Wenzel and M.C. Lyssikatos. 2021. Serious injury determinations for small cetaceans and pinnipeds caught in commercial fisheries off the northeast U.S. coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-04. 33pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Advanced distance sampling. Oxford University Press, New York, New York.

Laake, J.L., J. Calambokidis, S.D. Osmek and D.J. Rugh. 1997. Probability of detecting harbor porpoise from aerial surveys: Estimating (0). J. Wildl. Manage. 61:63–75.

Lawson J. and J-F. Gosselin. 2018. Estimates of cetacean abundance from the 2016 NAISS aerial surveys of eastern Canadian waters, with a comparison to estimates from the 2007 TNASS. NAMMCO SC/25/AE/09. 40pp.

Lens, S. 1997. Interactions between marine mammals and deep water trawlers in the NAFO regulatory area. Unpublished meeting document C.M. 1997/Q:08. International Council for the Exploration of the Sea, Copenhagen, Denmark. 10pp.

Lyssikatos, M.C., S. Chavez-Rosales and J. Hatch. 2020. Estimates of cetacean and pinniped bycatch in Northeast and mid-Atlantic bottom trawl fisheries, 2013–2017. Northeast Fish. Sci. Cent. Ref. Doc. 20-04. 11pp.

MacLeod, C.D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: A review and synthesis. Endang. Species Res. 7:125–136.

Mirimin L., A. Westgate, E. Rogan, P. Rosel, A. Read, J. Coughlan and T. Cross. 2009. Population structure of short-beaked common dolphins (*Delphinus delphis*) in the North Atlantic Ocean as revealed by mitochondrial and nuclear genetic markers. Mar. Biol. 156:821–834. https://doi.org/10.1007/s00227-008-1120-y

Moore, M., J. van der Hoop, S. Barco, A. Costidis, F. Gulland, P. Jepson, K. Moore, S. Raverty and W. McLellan. 2013. Criteria and case definitions for serious injury and death of pinnipeds and cetaceans caused by anthropogenic trauma. Dis. Aquat. Org. 103:229–264.

Morley, J.W., R.L. Selden, R.J. Latour, T.L. Frolicher, R.J. Seagraves and M.L. Pinsky. 2018. Projecting shifts in thermal habitat for 686 species on the North American continental shelf. PLoS ONE 13(5):e0196127.

Murphy, S., R.J. Law, R. Deaville, J. Barnett, M.W. Perkins, A. Brownlow, R. Penrose, N.J. Davison, J.L. Barber and P.D. Jepson. 2018. Organochlorine contaminants and reproductive implication in cetaceans: A case study of the common dolphin. Pages 3–38 *in* M.C. Fossi and C. Panti (Eds.). Marine mammal ecotoxicology: Impacts of multiple stressors on population health. Academic Press, New York, New York.

NMFS [National Marine Fisheries Service]. 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23pp. Available from: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

NEFSC [Northeast Fisheries Science Center] and SEFSC [Southeast Fisheries Science Center]. 2018. Annual report of a comprehensive assessment of marine mammal, marine turtle, and seabird abundance and spatial distribution in US waters of the western North Atlantic Ocean. Northeast Fish. Sci. Cent. Ref. Doc. 18-04. 141pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/atlantic-marine-assessment-program-protected-species

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: Time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Nye, J., J. Link, J. Hare and W. Overholtz. 2009. Changing spatial distribution of fish stocks in relation to climate and population size on the northeast United States continental shelf. Mar. Ecol. Prog. Ser. 393:111–129.

Orphanides, C.D. 2019. Estimates of cetacean and pinniped bycatch in the 2016 New England sink gillnet fishery and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 19-04. 12pp.

Orphanides, C.D. 2020. Estimates of cetacean and pinniped bycatch in the 2017 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 20-03. 16pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Orphanides, C.D. 2021. Estimates of cetacean and pinniped bycatch in the 2018 New England sink gillnet fishery and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 21-01. 12pp.

Orphanides, C.D. and J. Hatch. 2017. Estimates of cetacean and pinniped bycatch in the 2015 New England sink gillnet fishery and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 17-18. 21pp.

Palka, D.L. 2012. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2011 line transect survey. Northeast Fish. Sci. Cent. Ref. Doc. 12-29. 37pp. Available from: https://repository.library.noaa.gov/view/noaa/4312

Palka, D. 2020. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2016 line transect surveys conducted by the Northeast Fisheries Science Center. Northeast Fish. Sci. Cent. Ref. Doc. 20-05.

Payne, P.M., L.A. Selzer and A.R. Knowlton. 1984. Distribution and density of cetaceans, marine turtles and seabirds in the shelf waters of the northeast U.S., June 1980–Dec. 1983, based on shipboard observations. Final report, contract NA81FAC00023. National Marine Fisheries Service, Woods Hole, Massachusetts. 245pp.

Pierce, G.J., M.B. Santos, S. Murphy, J.A. Learmonth, A.F. Zuur, E. Rogan, P. Bustamante, F. Caurant, V. Lahaye, V. Ridoux, B.N. Zegers, A. Mets, M. Addink, C. Smeenk, T. Jauniaux, R.J. Law, W. Dabin, A. López, J.M. Alonso Farré, A.F. González, A. Guerra, M. García-Hartmann, R.J. Reid, C.F. Moffat, C. Lockyer and J.P. Boon. 2008. Bioaccumulation of persistent organic pollutants in female common dolphins (*Delphinus delphis*) and harbour porpoises (*Phocoena phocoena*) from western European seas: Geographical trends, causal factors and effects on reproduction and mortality. Env. Poll. 153:401–415.

Pinsky, M.L., B. Worm, M.J. Fogarty, J.L. Sarmiento and S.A. Levin. 2013. Marine taxa track local climate velocities. Science 341:1239–1242.

Poloczanska, E.S., C.J. Brown, W.J. Sydeman, W. Kiessling, D.S. Schoeman, P.J. Moore, K. Brander, J.F. Bruno, L.B. Buckley, M.T. Burrows, C.M. Duarte, B.S. Halpern, J. Holding, C.V. Kappel, M.I. O'Connor, J.M. Pandolfi, C. Parmesan, F. Schwing, S.A. Thompson and A.J. Richardson. 2013. Global imprint of climate change on marine life. Nat. Clim. Change 3:919–925.

Selzer, L.A. and P.M. Payne. 1988. The distribution of white-sided (*Lagenorhynchus acutus*) and common dolphins (*Delphinus delphis*) vs. environmental features of the continental shelf of the northeastern United States. Mar. Mamm. Sci. 4:141–153.

Sergeant, D.E., A.W. Mansfield and B. Beck. 1970. Inshore records of Cetacea for eastern Canada, 1949–68. J. Fish. Res. Board Can. 27:1903–1915.

Sharp, S.M., J.S. Knoll, M.J. Moore, K.M. Moore, C.T. Harry, J. M Hoppe, M.E. Niemeyer, I. Robinson, K.S. Rose, W.B. Sharp and D. Rotstein. 2014, Hematological, biochemical, and morphological parameters as prognostic indicators for stranded common dolphins (*Delphinus delphis)* from Cape Cod, Massachusetts, U.S.A. Mar. Mamm. Sci. 30:864–887.

Sharp, S.M., C.T. Harry, J.M. Hoppe, K.M. Moore, M.E. Niemeyer, I. Robinson, K.S. Rose, W.B. Sharp, S. Landry, J. Richardson and M.J. Moore 2016. A comparison of post release survival parameters between single and mass stranded delphinids from Cape Cod, Massachusetts, U.S.A. Mar. Mamm. Sci. 32:161–180.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23:157–175.

Thomas L, J.L. Laake, E. Rexstad, S. Strindberg, F.F.C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, M.L. Burt, S.L. Hedley, J.H. Pollard, J.R.B. Bishop and T.A. Marques. 2009. Distance 6.0. Release 2. [Internet]. University of St. Andrews, Research Unit for Wildlife Population Assessment. St. Andrews, UK. Available from: http://distancesampling.org/Distance/

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Waring, G.T., C.P. Fairfield, C.M. Ruhsam and M. Sano. 1992. Cetaceans associated with Gulf Stream features off the northeastern USA shelf. Unpublished meeting document C.M. 1992/N:12. International Council for the Exploration of the Sea, Copenhagen, Denmark.

Waring, G.T., L. Nottestad, E. Olsen, H. Skov and G. Vikingsson. 2008. Distribution and density estimates of cetaceans along the mid-Atlantic Ridge during summer 2004. J. Cetacean Res. Manage. 10:137–146.

Wells, R.S., J.B. Allen, S. Hofmann, K. Bassos-Hull, D.A. Fauquier, N.B. Barros, R.E. DeLynn, G. Sutton, V. Socha and M.D. Scott. 2008. Consequences of injuries on survival and reproduction of common bottlenose dolphins (*Tursiops truncatus*) along the west coast of Florida. Mar. Mamm. Sci. 24:774–794.

Westgate, A.J. 2005. Population structure and life history of short-beaked common dolphins (*Delphinus delphis*) in the North Atlantic. Ph.D thesis. Nicholas School of the Environment and Earth Sciences, Duke University, Beaufort, North Carolina.

Westgate, A.J. 2007. Geographic variation in cranial morphology of short-beaked common dolphins (*Delphinus delphis*) from the North Atlantic. J. Mamm. 88:678–688.

Westgate, A.J. and A.J. Read. 2007. Reproduction in short-beaked common dolphins (*Delphinus delphis*) from the western North Atlantic. Mar. Biol. 150:1011–1024.

COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Western North Atlantic Northern Migratory Coastal Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are found in estuarine, coastal, continental shelf, and oceanic waters of the western North Atlantic (wNA). Distinct morphological forms have been identified in offshore and coastal waters of the wNA off the U.S. East Coast: a smaller morphotype present in estuarine, coastal, and shelf waters from Florida to approximately Long Island, New York, and a larger, more robust morphotype present further offshore in deeper waters of the continental shelf and slope from Florida to Canada (Mead and Potter 1995). The two morphotypes also differ in parasite load and prey preferences (Mead and Potter 1995), and show significant genetic divergence at both mitochondrial and nuclear DNA markers (Hoelzel *et al*. 1998, Kingston and Rosel 2004, Kingston *et al.* 2009, Rosel *et al*. 2009). The level of genetic divergence is greater than that seen between some other dolphin species (Kingston and Rosel 2004, Kingston *et al.* 2009) suggesting the two morphotypes in the wNA may represent different subspecies or species. The larger morphotype comprises the wNA Offshore Stock of common bottlenose dolphins. Spatial distribution data (Kenney 1990, Garrison *et al.* 2017a), tag-telemetry studies (Garrison *et al.* 2017b), photo-identification (photo-ID) studies (e.g., Zolman 2002; Speakman *et al.* 2006; Stolen *et al.* 2007; Mazzoil *et al.* 2008), and genetic studies (Caldwell 2001, Rosel *et al.* 2009, Litz *et al.* 2012) indicate that the coastal morphotype comprises multiple stocks distributed in coastal and estuarine waters of the U.S. East Coast. The Northern Migratory Coastal Stock is one such stock and one of only two (the other being the Southern Migratory Coastal Stock) thought to make broad-scale, seasonal migrations in coastal waters of the wNA.

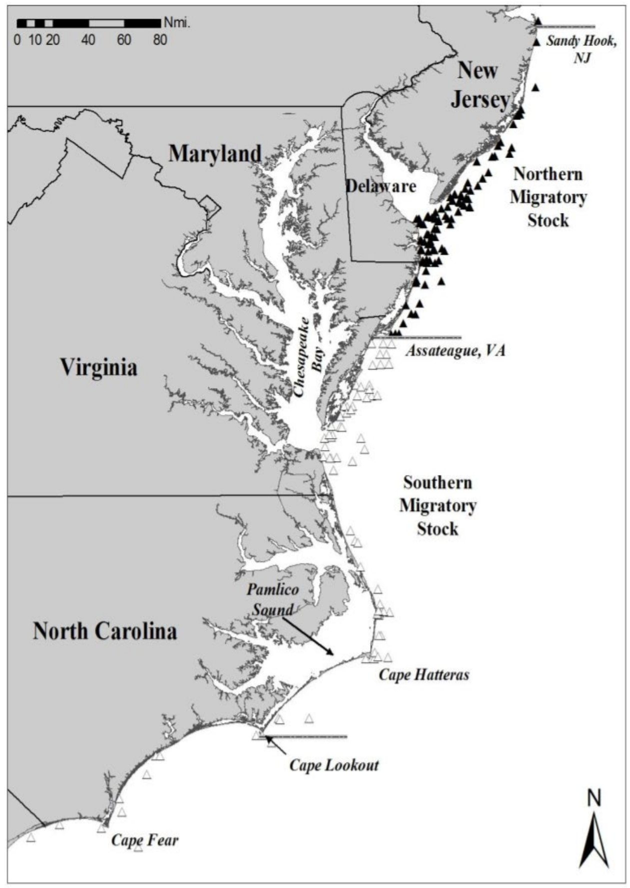


Figure 1. The distribution of common bottlenose dolphin sightings in coastal waters from North Carolina to New Jersey during July–August 2002, 2004, 2010, 2011, and 2016. Sighting locations from aerial surveys are plotted as triangle symbols. Sightings ascribed to the Northern Migratory Coastal Stock are shown as filled symbols; those from the Southern Migratory Coastal Stock as open symbols. Horizontal lines intersecting the coast denote the southern boundary for each stock in warm water months.

This stock exhibits spatiotemporal overlap with multiple common bottlenose stocks in the wNA. The stock is best defined by its distribution during warm water months (best described by July and August) when it overlaps with the fewest stocks. During warm water months, this stock occupies coastal waters from the shoreline to approximately the 20-m isobath between Assateague, Virginia, and Long Island, New York (Figure 1; Garrison *et al.* 2017b). The stock migrates in late summer and fall and, during cold water months (best described by January and February), occupies coastal waters from approximately Cape Lookout, North Carolina, to the North Carolina/Virginia border (Garrison *et al.* 2017b). Four common bottlenose dolphins tagged during 2003 and 2004 off the coast of New Jersey in late summer moved south to North Carolina and inhabited waters near and just south of Cape Hatteras during cold water months. These animals then returned to coastal waters of New Jersey in the following warm water months (Garrison *et al.* 2017b). Similarly, a dolphin tagged in late September 1998 off Virginia Beach, Virginia, moved south to waters between Cape Hatteras and Cape Lookout during cold water months (Garrison *et al.* 2017b). Photo-ID data also support that central North Carolina is the southern limit for this stock in winter (Urian *et al.* 1999). There are no matches from long-term photo-ID studies between sites in New Jersey and those south of Cape Lookout (Urian *et al.* 1999). Historically, common bottlenose dolphins have been rarely observed during cold water months in coastal waters north of the North Carolina/Virginia border, and their northern distribution in winter appears to be limited by water temperatures <9.5ºC (Garrison *et al.* 2016). During aerial and ship surveys off the New Jersey coast in 2008 and 2009, no sightings of common bottlenose dolphins were made during November–February; bottlenose dolphins were sighted from early March to mid-October and were most abundant during May–August (Whitt *et al.* 2015). Seasonal variation in the densities of animals observed off Virginia Beach, Virginia, supports the seasonal migration of dolphins northward during warm water months and then south during cold water months (Barco and Swingle 1996). Genetic analyses using mitochondrial and nuclear microsatellite data also indicated significant differentiation between common bottlenose dolphins occupying coastal waters north of the North Carolina/Virginia border to New Jersey during warm water months and those in southern North Carolina and further south (Rosel *et al*. 2009). Toth *et al.* (2012) suggested the Northern Migratory Coastal stock may be further partitioned in waters off of New Jersey. Two clusters of visual sightings that differed in the presence of a commensal soft-stalked barnacle, *Xenobalanus globicipitis*, in avoidance behavior, and in "base coloration" were identified. One cluster inhabited waters 0–1.9 km from shore while the other cluster inhabited waters 1.9–6 km from shore (Toth *et al.* 2012). Additional studies are needed to determine whether the two clusters should be considered demographically independent.

The distribution of the Northern Migratory Coastal Stock overlaps in certain seasons with several other common bottlenose dolphin stocks. Overlap with the Southern Migratory Coastal Stock in coastal waters of northern North Carolina and Virginia is possible during spring and fall migratory periods, but the degree of overlap is unknown and it may vary depending on annual water temperature (Garrison *et al.* 2016). When the stock has migrated in cold water months to coastal waters from just north of Cape Hatteras, North Carolina, to just south of Cape Lookout, North Carolina, it overlaps spatially with the Northern North Carolina Estuarine System (NNCES) Stock (Garrison *et al.* 2017b). Depending on the timing of the northward migration in the spring, it may overlap with the NNCES stock in coastal waters (<1 km from shore) as far north as Virginia Beach, Virginia, and the mouth of the Chesapeake Bay. It may also overlap with the Southern North Carolina Estuarine System Stock (Garrison *et al.* 2017b) in nearshore coastal waters south of Cape Hatteras in winter, although the degree of overlap with the latter stock is not well defined. This stock may also overlap to some degree with the wNA Offshore Stock of common bottlenose dolphins. A combined genetic and logistic regression analysis that incorporated depth, latitude, and distance from shore was used to model the probability that a particular common bottlenose dolphin group seen in coastal waters was of the coastal morphotype (Garrison *et al.* 2017a). North of Cape Hatteras during summer months, there is strong separation between the coastal and offshore morphotype (Kenney 1990, Garrison *et al.* 2017a), and the coastal morphotype is nearly completely absent in waters >20 m depth. South of Cape Hatteras, the regression analysis indicated that the coastal morphotype occurs at lower densities over the continental shelf, in waters >20 m deep where it overlaps to some degree with the offshore morphotype. For the purposes of defining stock boundaries and identifying bycaught dolphins, the offshore boundary of the Northern Migratory Coastal Stock is defined as the 20-m isobath in summer north of Cape Hatteras and the 200-m isobath in winter between Cape Hatteras and Cape Lookout.

POPULATION SIZE

The best available abundance estimate for the Northern Migratory Coastal Stock of common bottlenose dolphins in the western North Atlantic is 6,639 (CV=0.41; Table 1; Garrison *et al.* 2017a). This estimate was derived from aerial surveys conducted during the summer of 2016 covering coastal and shelf waters from Assateague, Virginia to Sandy Hook, New Jersey.

Background

Estimating the abundance of the Northern Migratory Coastal Stock is complicated by the spatiotemporal overlap the stock has with other coastal and estuarine common bottlenose dolphins as described above. Summer surveys are best for estimating the abundance for the Northern Migratory Coastal Stock because it overlaps least with other coastal, estuarine, and offshore stocks of common bottlenose dolphins during warm water months. Abundance for the Northern Migratory Coastal Stock is estimated using summer sightings made in the 0–20 m depth stratum during summer aerial surveys north of Assateague, Virginia (37.9°N) to Sandy Hook, New Jersey (40.3°N). The definition of the southern summer boundary and inter-annual variation in stock distribution are significant unquantified sources of uncertainty.

Earlier Abundance Estimates (>8 years old)

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions.

Recent Surveys and Abundance Estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters along the U.S. East Coast from southeastern Florida (26.9°N) to Sandy Hook, New Jersey (40.3°N), during the summer of 2016 (see Garrison *et al.* 2017a for survey design). The survey was conducted along tracklines oriented perpendicular to the shoreline and spaced latitudinally at 20-km intervals, and covered waters from the shoreline to the continental shelf break (Garrison *et al.* 2017a). The survey also included more closely spaced "fine-scale" tracklines in waters offshore of New Jersey and Virginia within areas being evaluated for the placement of offshore energy installations (Garrison *et al.* 2017a).

As with previous surveys, the 2016 survey was conducted using a two-team approach to develop estimates of detection probabilities using the independent observer approach with Distance analysis (Laake and Borchers 2004). The detection functions from the 2016 and two previous surveys indicated a decreased probability of detection near the trackline. The sighting data were therefore "left-truncated" by analyzing only sightings occurring greater than 100 m from the trackline during the 2016 survey (see Buckland *et al.* 2001 for left-truncation methodology). The independent observer method assuming point independence was used to estimate detection probability on the trackline. This estimate accounts for the probability of detecting a marine mammal group conditional on it being available to both survey teams. Covariates that may influence detection probabilities (e.g., sea state, glare, cloud cover, visibility) were incorporated into both the mark-recapture and distance function components of the detection models (Laake and Borchers 2004, Garrison *et al.* 2017a). The resulting abundance estimate is negatively biased due to the effects of animals spending some time underwater where they are not available to the survey teams. However, due to the relatively short dive times of bottlenose dolphins (Klatsky *et al.* 2007) and the large group sizes, it is likely that this bias is small (Garrison *et al.* 2017a).

The abundance estimate for the 2016 summer aerial survey was 6,639 (CV=0.41; Table 1; Garrison *et al.* 2017a). Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution.

Table 1. Abundance estimate for the western North Atlantic Northern Migratory Coastal Stock of common bottlenose dolphins. Month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and coefficient of variation (CV).

Month/Year	Area	N_{est}	CV
July–August 2016	Assateague, Virginia (37.9°N) to Sandy Hook, New Jersey (40.3°N)	6,639	0.41

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate for the Northern Migratory Coastal Stock of common bottlenose dolphins is 6,639 (CV=0.41). The resulting minimum population size estimate is 4,759 (Table 2).

Current Population Trend

Available surveys allow an analysis of trend in population size for coastal stocks of common bottlenose dolphins. A standardized analytical approach accounting for variation in survey execution and environmental conditions was used to derive unbiased abundance estimates for each survey (Garrison *et al.* 2017a). A weighted generalized linear model was used to evaluate trends in population size by stock using abundance estimates from surveys conducted in the summers of 2002, 2004, 2010, 2011 and 2016. Abundance estimates were weighted by the inverse of their standard error, which reduces the influence of less certain estimates (Neter *et al.* 1983). Stock was treated as a fixed factor, and surveys were grouped into three periods to test for long term trends in population size: 2002–2004, 2010–2011 and 2016. Period was also included as a fixed factor in the model along with the interaction between stock and period. Contrasts were specified to test for differences in abundance between periods for each stock (Garrison *et al.* 2017a). For the Northern Migratory Coastal Stock, the resulting mean abundance estimate for 2002–2004 was 8,597 (CV=0.53), and that for 2010–2011 was 15,232 (CV=0.35). There was no significant difference between these estimates and the estimate of 6,639 (CV=0.41) for 2016. There is limited power to detect a significant change given the high CV of the estimates, interannual variability in spatial distribution and stock abundance between 2002 and 2004, and the availability of only one recent survey (Garrison *et al.* 2017a).

An analysis of coast-wide (New Jersey to Florida) trends in abundance for common bottlenose dolphins was conducted. A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model included a linear term for survey year and an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant ($p=0.0308$) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse-variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison *et al.* 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME (see Strandings section).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are not known for the Northern Migratory Coastal Stock. The maximum net productivity rate is assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations likely do not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a "recovery" factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997, Wade 1998). The minimum population size of the Northern Migratory Coastal Stock of common bottlenose dolphins is 4,759. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because this stock is depleted. PBR for this stock of common bottlenose dolphins is 48 (Table 2).

Table 2. Best and minimum abundance estimates for the western North Atlantic Northern Migratory Coastal Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
6,639	0.41	4,759	0.5	0.04	48

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Northern Migratory Coastal Stock during 2014–2018 is unknown. The mean annual fishery-related mortality and serious injury for observed fisheries and strandings identified as fishery-related ranged between 12.2 and 21.5. No additional mortality and serious injury was documented from other human-caused sources (e.g., fishery research) and therefore, the minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 ranged between 12.2 and 21.5 (Tables 3a, 3b and 3c). This range reflects several sources of uncertainty and is a minimum due to the following five factors: 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected and recovered by the stranding network (Peltier *et al.* 2012, Wells *et al.* 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between this stock and other common bottlenose dolphin stocks in North Carolina and Virginia introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities and serious injuries were ascribed to a stock or stocks by comparing the season and geographic location of the take/stranding to the stock boundaries and geographic range delimited for each stock (Lyssikatos and Garrison 2018).

Fishery Information

There are eight commercial fisheries that interact, or that potentially could interact, with this stock. These include both the Category I mid-Atlantic gillnet and northeast sink gillnet fisheries, five Category II fisheries (Chesapeake Bay inshore gillnet, Virginia pound net, mid-Atlantic menhaden purse seine, Atlantic blue crab trap/pot, and mid-Atlantic haul/beach seine), and the Category III Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery. Detailed fishery information is presented in Appendix III.

Note: Animals reported in the sections to follow were ascribed to a stock or stocks of origin following methods described in Maze-Foley et al. (2019). These include strandings, observed takes (through an observer program), *fisherman self-reported takes (through the Marine Mammal Authorization Program), research takes, and opportunistic at-sea observations.*

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the Northern Migratory Coastal Stock throughout its range. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001, Lyssikatos and Garrison 2018). This fishery is currently observed by the Northeast Fisheries Observer Program, and previously was observed by both the Northeast and Southeast Fisheries Observer Programs (through 2016). The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, 26 April 2006, Available from: https://www.federalregister.gov/documents/2006/04/26/06-3909/taking-of-marine-mammals-incidental-to-commercial-fishing-operations-bottlenose-dolphin-take). In addition, two amendments to the BDTRP were implemented in 2008 and 2012 regarding gear restrictions for medium-mesh gillnets in North Carolina waters (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan). Mortality estimates for the period 2002–2006 (immediately prior to implementation of the BDTRP) and 2007–2011 are available in the 2015 stock assessment report for the Northern Migratory Coastal Stock (Waring *et al.* 2016). The current report covers the most recent available five-year estimate (NMFS 2016) for 2014–2018.

Mortality estimation for this stock is difficult for three main reasons: 1) observed takes are statistically rare events, 2) the Northern Migratory Coastal, Southern Migratory Coastal, NNCES, and Southern North Carolina Estuarine System stocks of common bottlenose dolphin overlap in coastal waters of North Carolina and Virginia at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are insufficient to consistently detect rare bycatch events (Lyssikatos and Garrison 2018). To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2014–2018: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data, and 2) a pooled CPUE approach (where all observer data from the most recent five years were combined into one sample to estimate CPUE; Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018).

During the most recent five-year time period, 2014–2018, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 5.16% in state waters (0–3 miles from shore) and 9.95% in federal waters (3–200 miles from shore; Lyssikatos 2021). During these trips, observers documented five entangled dolphins that may have been from the Northern Migratory Coastal Stock, two off of North Carolina, two off of New Jersey, and one off of Virginia. In April 2018, the NEFOP observed one mortality in medium-mesh gillnet gear off the coast of Virginia that was ascribed solely to the Northern Migratory Coastal Stock. Also in October 2017, the NEFOP observed one mortality in medium-mesh gillnet gear off the coast of New Jersey ascribed solely to the Northern Migratory Coastal Stock. In February 2017, the NEFOP documented a dolphin entangled in small-mesh gillnet gear off the coast of North Carolina that was released alive, and it could not be determined if the animal was seriously injured. The animal was ascribed to the Northern Migratory Coastal and NNCES stocks. In August 2015, the NEFOP observed one mortality in medium-mesh gillnet gear off the coast of New Jersey that was ascribed solely to the Northern Migratory Coastal Stock. In January 2015, one mortality was observed by the NEFOP off Hatteras, North Carolina, entangled in a medium-mesh gillnet gear. This dolphin was ascribed to the Northern Migratory Coastal and NNCES stocks (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). The resultant five-year mean minimum and maximum mortality estimates (2014–2018) were 11.8 (CV=0.18) and 19.5 (CV=0.14) animals per year, respectively (Table 3a; Lyssikatos 2021).

Historical stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear attached (Byrd *et al.* 2014, Waring *et al.* 2016, Lyssikatos and Garrison 2018). Six mortalities and one live animal were documented entangled in gillnet gear during the current five-year period, 2014–2018, that may have been from the Northern Migratory Coastal Stock (two of these mortalities were also documented by the Marine Mammal Authorization Program). The live animal was disentangled and released alive but it could not be determined whether the animal was seriously injured (Maze-Foley and Garrison 2020). From 2014 to 2018, 12 dead, stranded dolphins were recovered with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Seven of the 12 cases were ascribed to the Northern Migratory Coastal Stock alone, two were ascribed to the Northern and Southern Migratory Coastal stocks, and three were ascribed to the Northern Migratory Coastal and NNCES stocks.

Northeast Sink Gillnet

During 2014–2018, there were four documented mortalities self-reported through the Marine Mammal Authorization Program for the New England sink gillnet fishery that may have been from the Northern Migratory Coastal Stock. All four mortalities were ascribed to the Northern Migratory Coastal and Offshore Stocks, and included one case from August 2017 of two dolphins entangled in the same gillnet, and a separate case from November 2017 of two dolphins entangled in the same gillnet. This fishery is observed by the NEFOP and the Northeast Fisheries At-Sea Monitoring Program (ASM; see Orphanides and Hatch 2017), however, no observed takes have been assigned to the Northern Migratory Coastal Stock and there is no bycatch estimate for this stock. The four self-reported mortalities are included in the annual human-caused mortality and serious injury total for this stock (Table 3b).

Chesapeake Bay Inshore Gillnet

During 2014–2018, there was one documented stranding of a common bottlenose dolphin entangled in inshore gillnet gear in Chesapeake Bay. In 2015, in Virginia, a stranded animal was found entangled in gillnet gear (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). This animal was ascribed to the Northern and Southern Migratory Coastal stocks, and is included in the annual human-caused mortality and serious injury total for this stock (Table 3b) as well as in the stranding database and stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). There is no observer coverage of this fishery within Maryland waters of Chesapeake Bay; within Virginia waters of Chesapeake Bay, there is a low level of observer coverage (<1%). No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years. Six other dead, stranded common bottlenose dolphins were recovered within Chesapeake Bay with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Five of these animals were ascribed solely to the Northern Migratory Coastal Stock, and one was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks.

Virginia Pound Net

During 2014–2018, there were no documented mortalities or serious injuries involving pound net gear in Virginia. However, one dead, stranded dolphin was recovered with twisted twine markings indicative of interactions with pound net gear, but it is unknown whether the interactions with the gear contributed to the death of this animal and this case is not included in the annual human-caused mortality and serious injury total for this stock. This stranding was ascribed solely to the Northern Migratory Coastal Stock, and it occurred inside estuarine waters near the mouth of the Chesapeake Bay in March 2016 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available. An amendment to the BDTRP was implemented in 2015 requiring gear restrictions for VA pound nets in estuarine and coastal state waters of Virginia to reduce bycatch (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan).

Mid-Atlantic Menhaden Purse Seine

During 2014–2018, there were no documented mortalities or serious injuries in mid-Atlantic menhaden purse seine gear of common bottlenose dolphins that could be ascribed to the Northern Migratory Coastal Stock. The mid-Atlantic menhaden purse seine fishery historically reported an annual incidental take of one to five common bottlenose dolphins (NMFS 1991, pp. 5–73). There has been very limited federal observer coverage since 2008. No observer coverage was allocated to this fishery during 2014–2018. Because there is no systematic observer program for this fishery, no estimate of bycatch mortality is available.

Atlantic Blue Crab Trap/Pot

During 2014–2018, stranding data documented four cases of common bottlenose dolphins entangled in trap/pot gear that could be ascribed to the Northern Migratory Coastal Stock. Two cases were serious injuries and for the remaining two cases, it could not be determined whether the animals were seriously injured. One serious injury occurred in 2017 in unidentified trap/pot gear and was ascribed solely to the Northern Migratory Coastal Stock. The second serious injury occurred in 2017 in commercial blue crab trap/pot gear and was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. The serious injuries are included in the annual human-caused mortality and serious injury total for this stock (Table 3b). Also in 2017, there was one entanglement in unidentified trap/pot gear ascribed to the Northern and Southern Migratory Coastal stocks. In 2018, there was an entanglement in unidentified trap/pot gear that was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. For both of these cases, it could not be determined whether the animals were seriously injured. All of the cases were included in the stranding database and in the stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). Details regarding the serious injury determinations can be found in Maze-Foley and Garrison (2020). Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots. However, stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002, Burdett and McFee 2004).

Mid-Atlantic Haul/Beach Seine

During 2014–2018, one serious injury of a common bottlenose dolphin occurred associated with the mid-Atlantic haul/beach seine fishery that could be ascribed to the Northern Migratory Coastal Stock. During 2014, a common bottlenose dolphin was found within a haul seine net in Virginia and released alive seriously injured (Maze-Foley and Garrison 2020). The animal was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. This case was included in the stranding database and in the stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019) as well as in the annual human-caused mortality and serious injury total for this stock (Table 3b). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2014–2018. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Hook and Line (Rod and Reel)

During 2014–2018, stranding data identified four mortalities and one serious injury of common bottlenose dolphins that could be ascribed to the Northern Migratory Coastal Stock for which hook and line gear entanglement or ingestion were documented. For one mortality, available evidence suggested the hook and line gear interaction contributed to the cause of death (2018, Maryland). This animal was ascribed solely to the Northern Migratory Coastal Stock. For a second mortality that was also ascribed solely to the Northern Migratory Coastal Stock, the carcass was in a state of advanced decomposition and it could not be determined whether the hook and line gear interaction contributed to cause of death (2018, Delaware). For a third mortality, available evidence suggested the hook and line gear interaction did not contribute to the cause of death (2016, Virginia). This animal was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. The fourth mortality was in a state of advanced decomposition and it could not be determined whether the hook and line gear interaction contributed to cause of death (2014, Virginia; Maze-Foley *et al.* 2019). This mortality was ascribed solely to the Northern Migratory Coastal Stock. In addition, there was one live animal documented with an entanglement (2017, Virginia), and this animal was considered seriously injured (Maze-Foley and Garrison 2020). It was ascribed to the Northern and Southern Migratory Coastal stocks. All of these cases were included in the stranding database and in the stranding totals presented in Table 4

(Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). The 2018 mortality for which evidence suggested the gear contributed to the cause of death and the 2017 serious injury are included in the annual human-caused mortality and serious injury total for this stock (Table 3b).

It should be noted that, in general, it cannot be determined if rod and reel (hook and line) gear originated from a commercial (i.e., commercial fisherman, charter boat, or headboat) or recreational angler because the gear type used by both sources is typically the same. Also, it is not possible to estimate the total number of interactions with hook and line gear because there is no systematic observer program. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Other Mortality

Historically, there have been occasional mortalities of common bottlenose dolphins during activities including during directed live capture-release studies, turtle relocation trawls, and fisheries surveys (Waring *et al.* 2016); however, none were documented during 2014–2018 that could be ascribed to the Northern Migratory Coastal Stock. All mortalities and serious injuries from known human-caused sources for the Northern Migratory Coastal Stock are summarized in Tables 3a, 3b and 3c.

Table 3a. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Migratory Coastal Stock for the commercial mid-Atlantic gillnet fishery, which has an ongoing, systematic federal observer program. The years sampled, the type of data used, the annual percentage observer coverage, the observed serious injuries and mortalities recorded by on-board observers, and the mean annual estimate of mortality and serious injury and its CV are provided. Minimum and maximum values are reported due to uncertainty in the assignment of mortalities to this particular stock because there is spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	Years	Data Type	Observer Coverage	Observed Serious Injury	Observed Mortality	Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data
Mid-Atlantic Gillnet	2014 2015 2016 2017 2018	Obs. Data Logbook	3.6 5.6 9.8 7.0 6.4	0 0 0 0 0	0 2 0 1 1	Min=11.8 (0.18) Max=19.5 (0.14)
Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2014–2018)						**Min=11.8 (0.18) Max=19.5 (0.14)**

Table 3b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Migratory Coastal Stock during 2014–2018 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	**Years**	**Data Type**	**5-year Count Based on Stranding Data and the Marine Mammal Authorization Program**
Northeast Sink Gillnet	2014–2018	Federal Observer, and Marine Mammal Authorization Program	Min=0 Max=4
Chesapeake Bay Inshore Gillnet[a]	2014–2018	Limited Observer and Stranding Data	Min=0 Max=1
Virginia Pound Net[b]	2014–2018	Stranding Data	0
Mid-Atlantic Menhaden Purse Seine	2014–2018	Limited Observer and Stranding Data	0
Atlantic Blue Crab Trap/Pot	2014–2018	Stranding Data	Min=1 Max=2
Mid-Atlantic Haul/Beach Seine	2014–2018	Limited Observer and Stranding Data	Min=0 Max=1
Hook and Line[c]	2014–2018	Stranding Data	Min=1 Max=2
Mean Annual Mortality due to unobserved commercial fisheries (2014–2018)			**Min=0.4** **Max=2.0**

a Chesapeake Bay inshore gillnet interactions are included if the animal was found entangled in gillnet gear. Strandings with markings indicative of interactions with gillnet gear are not included within the table. See "Chesapeake Bay Inshore Gillnet" text for more details.
b Pound net interactions are included if the animal was found entangled in pound net gear. Strandings with twisted twine markings indicative of interactions with pound net gear are not included within the table. See "Virginia Pound Net" text for more details.
c Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death per Maze-Foley *et al.* (2019). See "Hook and Line" text for more details.

Table 3c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Migratory Coastal Stock during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2014–2018) (Table 3a)	**Min=11.8 (0.18) Max=19.5 (0.14)**
Mean Annual Mortality due to unobserved commercial fisheries (2014–2018) (Table 3b)	**Min=0.4 Max=2.0**
Research Takes (5-year Min/Max Count)	0
Other takes (5-year Min/Max Count)	0
Mean Annual Mortality due to research and other takes (2014–2018)	**0**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2014–2018)	**Min=12.2 Max=21.5**

Strandings

Between 2014 and 2018, 692 common bottlenose dolphins that were ascribed to the Northern Migratory Coastal Stock stranded along the Atlantic coast between North Carolina and New York (Table 4; Northeast Regional (NER) Marine Mammal Stranding Network; Southeast Regional (SER) Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 (SER) and 13 August 2019 (NER); Maze-Foley *et al.* 2019). There was evidence of human interaction for 80 of these strandings, of which 51 (64%) were fisheries interactions and 4 (5%) showed evidence of a boat strike. No evidence of human interaction was detected for 134 strandings, and for the remaining 478 strandings, it could not be determined if there was evidence of human interaction. It should be recognized that evidence of human interaction does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions due to spatial and temporal overlap among several common bottlenose dolphin stocks (Maze-Foley *et al.* 2019). Of the 692 strandings ascribed to the Northern Migratory Coastal Stock, 297 were ascribed solely to this stock. Therefore, the counts in Table 4 likely include some animals from the Southern Migratory Coastal, NNCES, and Offshore stocks and, therefore, overestimate the number of strandings for the Northern Migratory Coastal Stock; those strandings that could not be ascribed to the Northern Migratory Coastal Stock alone are also included in the counts for these other stocks as appropriate. Stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin, therefore, it is possible that some of the reported strandings were of the offshore form though that number is likely to be low, especially for states south of New York (Byrd *et al.* 2014).

This stock has also been impacted by two large unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994, Morris *et al.* 2015). Both UMEs included deaths of dolphins north of Assateague, Virginia, in summer, corresponding solely to the Northern Migratory Coastal Stock area. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al*. 1988), so impacts to the Northern Migratory Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988, Eguchi 2002). For the 2013–2015 UME, a total of 1614 stranded common bottlenose dolphins were recovered in the UME area which stretched from New York to Brevard County, Florida. Of these, 348 stranded dolphins were recovered from the states of New York, New Jersey, Delaware, and Maryland (https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic, accessed 13 November 2019). While some of these deaths may be attributable to the Offshore Stock, the majority likely came from the Northern Migratory Coastal Stock given their geographic location. This number is likely an underestimate of the total number of deaths for this stock, however, because it does not include animals that stranded in Virginia and North Carolina in cold water months that might have come from this stock, and not all dolphins that died during the UME would have been recovered.

Table 4. Strandings of common bottlenose dolphins during 2014–2018 from North Carolina to New York that were ascribed to the Northern Migratory Coastal Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Assignments to stock were based upon the understanding of the seasonal movements of this stock. However, in waters of North Carolina and Virginia there is likely overlap with other stocks during particular times of year. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 21 May 2019 for SER and 13 August 2019 for NER). Please note HI does not necessarily mean the interaction caused the animal's death.

State	2014			2015			2016			2017			2018			Total
	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	(2014–2018)
North Carolina	3	15	23	3	10	23	3	6	21	5	7	13	2	20	12	166
Virginia	5	5	44	8	6	55	11	6	33	9	1	46	9	1	50	289
Maryland	0	1	6	0	2	8	2	1	9	2	2	6	1	0	7	47
Delaware	0	0	7	1	2	5	0	1	8	0	1	6	2	2	17	52
New Jersey	0	2	16	0	0	24	0	7	4	3	10	7	1	10	4	88
New York	1	1	8	0	2	2	0	3	3	5	5	5	4	5	6	50
Total	**137**			**151**			**118**			**133**			**153**			**692**

[a] Strandings for North Carolina include data for November–April north of Cape Lookout when Northern Migratory Coastal animals may be in coastal waters. The stock identity of these strandings is highly uncertain and likely also includes animals from the NNCES Stock.

[b] Strandings from Virginia were ascribed to stock based upon both location and time of year. Some of the strandings ascribed to the Northern Migratory Coastal Stock could possibly be ascribed to the Southern Migratory Coastal Stock or NNCES Stock.

[c] Strandings from New York are assigned to both the Northern Migratory Coastal Stock and the Offshore Stock regardless of the month or location (coastal or sound waters) of their recovery.

HABITAT ISSUES

The coastal habitat occupied by this stock is adjacent to areas of high human densities, some industrialized areas, and waters that are heavily utilized for commercial and recreational fishing, and boating activities. The blubber of stranded dolphins examined during the 1987–1988 mortality event contained very high concentrations of organic pollutants (Kuehl *et al*. 1991). Total DDT levels measured in common bottlenose dolphins sampled in Cape May, New Jersey, were higher than 12 other sites sampled in the wNA and northern Gulf of Mexico (of 14 sites examined in total; Kucklick *et al*. 2011). Values for total PCBs exceeded toxic thresholds proposed by Kannan *et al*. (2000) and Schwacke *et al*. (2002) and may result in adverse effects on health or reproductive rates (Schwacke *et al*. 2002, Hansen *et al*. 2004, Yordy *et al*. 2010). Studies of contaminant concentrations relative to life history parameters showed higher levels of mortality in first-born offspring and higher contaminant concentrations in these calves and in primiparous females (Wells *et al*. 2005). Exposure to high PCB levels has been linked to anemia, hyperthyroidism, and immune suppression in common bottlenose dolphins in Georgia (Schwacke *et al*. 2012). The exposure to environmental pollutants and subsequent effects on population health is an area of concern.

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the Northern Migratory Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of common bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The Northern Migratory Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the Northern Migratory Coastal Stock is 48 and so the zero mortality rate goal, 10% of PBR, is 4.8. The documented mean annual human-caused mortality for this stock for 2014–2018 ranged between a minimum of 12.2 and a maximum of 21.5. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for the Northern Migratory Coastal Stock cannot be directly estimated because of the spatial overlap among the stocks of common bottlenose dolphins that occupy waters of North Carolina and Virginia resulting in uncertainty in the stock assignment of some takes, 2) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). The total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and therefore, cannot be considered to be insignificant and approaching a zero mortality and serious injury rate. The impacts of two large UMEs on the status of this stock are unknown. Analysis of trends in abundance suggests a probable decline in stock size between 2010–2011 and 2016, concurrent with a large UME in the area; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

REFERENCES CITED

Barco, S.G. and W.M. Swingle. 1996. Sighting patterns of coastal migratory bottlenose dolphins (*Tursiops truncatus*) in the nearshore waters of Virginia and North Carolina. Final Report to the Virginia Dept. of Environmental Quality, Coastal Resources Management Program through Grant #NA47OZ0287-01 from NOAA, Office of Ocean and Coastal Resource Management. 32pp.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine Mammal Stock Assessments: Guidelines for Preparation, Background and a Summary of the 1995 Assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Buckland, S.T., D.R. Andersen, K.P Burnham, J.L. Laake, D.L. Borchers and L. Thomas. 2001. Introduction to distance sampling: Estimating abundance of biological populations. Oxford University Press, New York. 432pp.

Burdett, L.G. and W.E. McFee. 2004. Bycatch of bottlenose dolphins in South Carolina, USA, and an evaluation of the Atlantic blue crab fishery categorization. J. Cetacean Res. Mange. 6(3):231–240.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Caldwell, M. 2001. Social and genetic structure of bottlenose dolphin (*Tursiops truncatus*) in Jacksonville, Florida. Ph.D. dissertation from University of Miami. 143pp.

Eguchi, T. 2002. A method for calculating the effect of a die-off from stranding data. Mar. Mamm. Sci. 18(3):698–709.

Garrison, L.P., P.E. Rosel, A.A. Hohn, R. Baird and W. Hoggard. 2016. Abundance of the coastal morphotype of bottlenose dolphin *Tursiops truncatus*, in U.S. continental shelf waters between New Jersey and Florida during winter and summer 2002. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-03. 135pp.

Garrison, L.P., K. Barry and W. Hoggard. 2017a. The abundance of coastal morphotype bottlenose dolphins on the U.S. east coast: 2002-2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-01. 37pp.

Garrison, L.P., A.A. Hohn and L.J. Hansen. 2017b. Seasonal movements of Atlantic common bottlenose dolphin stocks based on tag telemetry data. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-02.

Hansen, L.J., L.H. Schwacke, G.B. Mitchum, A.A. Hohn, R.S. Wells, E.S. Zolman and P.A. Fair. 2004. Geographic variation in polychlorinated biphenyl and organohaline pesticide concentrations in the blubber of bottlenose dolphins from the US Atlantic coast. Sci. Total Environ. 319:147–172.

Hoelzel, A.R., C.W. Potter and P.B. Best. 1998. Genetic differentiation between parapatric nearshore and offshore populations of the bottlenose dolphin. Proc. Royal Soc. London 265:1177–1183.

Kannan, K., A.L. Blankenship, P.D. Jones and J.P. Giesy. 2000. Toxicity reference values for the toxic effects of polychlorinated biphenyls to aquatic mammals. Hum. Ecol. Risk Assess. 6:181–201.

Kenney, R.D. 1990. Bottlenose dolphins off the northeastern United States. Pages 369–386. *in:* S. Leatherwood and R. R. Reeves (Eds.). The bottlenose dolphin. Academic Press, San Diego, California. 653pp.

Kingston, S.E. and P.E. Rosel. 2004. Genetic differentiation among recently diverged Delphinid taxa determined using AFLP markers. J. Hered. 95(1):1–10.
Kingston, S.E., L.D. Adams and P.E. Rosel. 2009. Testing mitochondrial sequences and anonymous nuclear markers for phylogeny reconstruction in a rapidly radiating group: Molecular systematics of the Delphininae (Cetacea: Odontoceti: Delphinidae). BMC Evol. Biol. 9:245.19pp.
Klatsky, L.J., R.S. Wells and J.C. Sweeney. 2007. Offshore bottlenose dolphins (*Tursiops truncatus*): Movement and dive behavior near the Bermuda pedestal. J. Mamm. 88:59–66.
Kucklick, J., L. Schwacke, R. Wells, A. Hohn, A. Guichard, J. Yordy, L. Hansen, E. Zolman, R. Wilson, J. Litz, D. Nowacek, T. Rowles, R. Pugh, B. Balmer, C. Sinclair and P. Rosel. 2011. Bottlenose dolphins as indicators of persistent organic pollutants in the western North Atlantic Ocean and northern Gulf of Mexico. Environ. Sci. Technol. 45:4270–4277.
Kuehl, D.W., R. Haebler and C. Potter. 1991. Chemical residues in dolphins from the US Atlantic coast including Atlantic bottlenose obtained during the 1987/1988 mass mortality. Chemosphere 22:1071–1084.
Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in:* S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Advanced distance sampling. Oxford University Press, New York.
Lipscomb, T.P., F.Y. Schulman, D. Moffett and S. Kennedy. 1994. Morbilliviral disease in Atlantic bottlenose dolphins (*Tursiops truncatus*) from the 1987–1988 epizootic. J. Wildl. Dis. 30:567–571.
Litz, J.A., C.R. Hughes, L.P. Garrison, L.A. Fieber and P.E. Rosel. 2012. Genetic structure of common bottlenose dolphins (*Tursiops truncatus*) inhabiting adjacent South Florida estuaries - Biscayne Bay and Florida Bay. J. Cetacean Res. Manage. 12(1):107–117.
Lyssikatos, M. 2021. Common bottlenose dolphin (*Tursiops truncatus*) gillnet bycatch estimates along the US mid-Atlantic Coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-02. 12pp.
Lyssikatos, M. and L.P. Garrison. 2018. Common bottlenose dolphin (*Tursiops truncatus*) gillnet bycatch estimates along the US mid-Atlantic Coast, 2007-2015. Northeast Fish. Sci. Cent. Ref. Doc. 18-07. 37pp. Available from: http://www.nefsc.noaa.gov/publications/
Maze-Foley, K. and L.P. Garrison. 2020. Serious injury determinations for small cetaceans off the southeast U.S. coast, 2014–2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2020-06. 43pp.
Maze-Foley, K., B.L. Byrd, S.C. Horstman and J.R. Powell. 2019. Analysis of stranding data to support estimates of mortality and serious injury in common bottlenose dolphin (*Tursiops truncatus truncatus*) stock assessments for the Atlantic Ocean and Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-742. 42pp. Available from: https://repository.library.noaa.gov/view/noaa/23151
Mazzoil, M., J.S. Reif, M. Youngbluth, M.E. Murdoch, S.E. Bechdel, E. Howells, S.D. McCulloch, L.J. Hansen and G.D. Bossart. 2008. Home ranges of bottlenose dolphins (*Tursiops truncatus*) in the Indian River Lagoon Florida: Environmental correlates and implications for management strategies. EcoHealth 5:278–288.
Mead, J.G. and C.W. Potter. 1995. Recognizing two populations of the bottlenose dolphin (*Tursiops truncatus*) off the Atlantic coast of North America: Morphological and ecological considerations. IBI Reports 5:31–44.
Morris, S.E., J.L. Zelner, D.A. Fauquier, T.K. Rowles, P.E. Rosel, F. Gulland and B.T. Grenfell. 2015. Partially observed epidemics in wildlife hosts: Modelling an outbreak of dolphin morbillivirus in the northwestern Atlantic, June 2013–2014. J. R. Soc. Interface 12:20150676.
Neter, J., W. Wasserman and M. Kutner. 1983. Applied linear regression models. Richard D. Irwin, Inc., Homewood, Illinois. 547pp.
NMFS [National Marine Fisheries Service]. 1991. Proposed regime to govern the interactions between marine mammals and commercial fishing operations after October 1, 1993. Draft Environmental Impact Statement, June 1991.
NMFS [National Marine Fisheries Service]. 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the MMPA. NMFS Instruction 02-204-01, February 22, 2016. Available from: http://www.nmfs.noaa.gov/op/pds/index.html
Noke, W.D. and D.K. Odell. 2002. Interactions between the Indian River Lagoon blue crab fishery and the bottlenose dolphin, *Tursiops truncatus*. Mar. Mamm. Sci 18(4):819–832.
Orphanides, C.D. and J.M. Hatch. 2017. Estimates of cetacean and pinniped bycatch in the 2015 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 17-18. 21pp. Available from: http://www.nefsc.noaa.gov/publications/
Palka, D.L. and M.C. Rossman. 2001. Bycatch estimates of coastal bottlenose dolphin (*Tursiops truncatus*) in the U.S. mid-Atlantic gillnet fisheries for 1996 to 2000. Northeast Fish. Sci. Center Ref. Doc. 01-15. 77pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Read, A.J. and K.T. Murray. 2000. Gross evidence of human-induced mortality in small cetaceans. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-15. 21pp.
Available from: https://repository.library.noaa.gov/view/noaa/3679

Rosel, P.E., L. Hansen and A.A. Hohn. 2009. Restricted dispersal in a continuously distributed marine species: Common bottlenose dolphins *Tursiops truncatus* in coastal waters of the western North Atlantic. Mol. Ecol. 18:5030–5045.

Schwacke, L.H., E.O. Voit, L.J. Hansen, R.S. Wells, G.B. Mitchum, A.A. Hohn and P.A. Fair. 2002. Probabilistic risk assessment of reproductive effects of polychlorinated biphenyls on bottlenose dolphins (*Tursiops truncatus*) from the southeast United States coast. Env. Toxic. Chem. 21:2752–2764.

Schwacke, L.H., E.S. Zolman, B.C. Balmer, S. De Guise, R.C. George, J. Hoguet, A.A. Hohn, J.R. Kucklick, S. Lamb, M. Levin, J.A. Litz, W.E. McFee, N.J. Place, F.I. Townsend, R.S. Wells and T.K. Rowles. 2012. Anaemia, hypothyroidism and immune suppression associated with polychlorinated biphenyl exposure in bottlenose dolphins (*Tursiops truncatus*). Proc. R. Soc. B. https://doi.org/10.1098/rspb.2011.0665

Scott, G.P., D.M. Burn and L.J. Hansen. 1988. The dolphin die off: Long term effects and recovery of the population. Proceedings: Oceans '88, IEEE Cat. No. 88-CH2585-8, Vol. 3:819–823.

Speakman, T., E.S. Zolman, J. Adams, R.H. Defran, D. Laska, L. Schwacke, J. Craigie and P. Fair. 2006. Temporal and spatial aspects of bottlenose dolphin occurrence in coastal and estuarine waters near Charleston, South Carolina. NOAA Tech. Memo. NOS-NCCOS-37. 243pp.

Stolen, M.K., W.N. Durden and D.K. Odell. 2007. Historical synthesis of bottlenose dolphin (*Tursiops truncatus*) stranding data in the Indian River Lagoon system, Florida, from 1977-2005. Fla. Sci. 70:45–54.

Toth, J.L., A.A. Hohn, K.W. Able and A.M. Gorgone. 2012. Defining bottlenose dolphin (*Tursiops truncatus*) stocks based on environmental, physical, and behavioral characteristics. Mar. Mamm. Sci. 28(3):461–478.

Urian, K.W., A.A. Hohn and L.J. Hansen. 1999. Status of the photo-identification catalog of coastal bottlenose dolphins of the western north Atlantic: Report of a workshop of catalog contributors. NOAA Tech. Memo. NMFS-SEFSC-425. 22pp.

Wade, P.R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Mar. Mamm. Sci. 14(1):1–37.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Waring, G.T., E. Josephson, K. Maze-Foley and P.E. Rosel. 2016. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2015. NOAA Tech. Memo. NMFS-NE-238. 512pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Wells, R.S., V. Tornero, A. Borrell, A. Aguilar, T.K. Rowles, H.L. Rhinehart, S. Hofmann, W.M. Jarman, A.A. Hohn and J.C. Sweeney. 2005. Integrating life history and reproductive success data to examine potential relationships with organochlorine compounds for bottlenose dolphins (*Tursiops truncatus*) in Sarasota Bay, Florida. Sci. Total Environ. 349:106–119.

Whitt, A.D., J.A. Powell, A.G. Richardson and J.R. Bosyk. 2015. Abundance and distribution of marine mammals in nearshore waters off New Jersey, USA. J. Cetacean Res. Manage. 15:45–59.

Yordy, J.E., M.A. Mollenhauer, R.M. Wilson, R.S. Wells, A. Hohn, J. Sweeney, LH. Schwacke, T.K. Rowles, J.R. Kucklick and M.M. Peden-Adams. 2010. Complex contaminant exposure in cetaceans: A comparative E-screen analysis of bottlenose dolphin blubber and mixtures of four persistent organic pollutants. Environ. Toxicol. Chem. 29:2143–2153.

Zolman, E.S. 2002. Residence patterns of bottlenose dolphins (*Tursiops truncatus*) in the Stono River Estuary, Charleston County, South Carolina. Mar. Mamm. Sci. 18:879–892.

COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Western North Atlantic Southern Migratory Coastal Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are found in estuarine, coastal, continental shelf, and oceanic waters of the western North Atlantic (wNA). Two distinct morphological forms have been identified in offshore and coastal waters of the wNA off the U.S. East Coast: a smaller morphotype present in estuarine, coastal, and shelf waters from Florida to approximately Long Island, New York, and a larger, more robust morphotype present further offshore in deeper waters of the continental shelf and slope (Mead and Potter 1995) from Florida to Canada. The two morphotypes also differ in parasite load and prey preferences (Mead and Potter 1995), and show significant genetic divergence at both mitochondrial and nuclear DNA markers (Hoelzel *et al*. 1998, Kingston and Rosel 2004, Kingston *et al.* 2009, Rosel *et al*. 2009). The level of genetic divergence is greater than that seen between some other dolphin species (Kingston and Rosel 2004, Kingston *et al.* 2009) suggesting the two morphotypes in the wNA may represent different subspecies or species. The larger morphotype makes up the wNA Offshore Stock of common bottlenose dolphins. Spatial distribution data (Kenney 1990, Garrison *et al.* 2017a), tag-telemetry studies (Garrison *et al.* 2017b), photo-identification (photo-ID) studies (e.g., Zolman 2002; Speakman *et al.* 2006; Stolen *et al.* 2007; Mazzoil *et al.* 2008), and genetic studies (Caldwell 2001, Rosel *et al.* 2009, Litz *et al.* 2012) indicate that the coastal morphotype comprises multiple stocks distributed in coastal and estuarine waters of the U.S. East Coast. The Southern Migratory Coastal Stock is one such stock and one of only two (the other being the Northern Migratory Coastal Stock) thought to make broad-scale, seasonal migrations in coastal waters of the wNA.

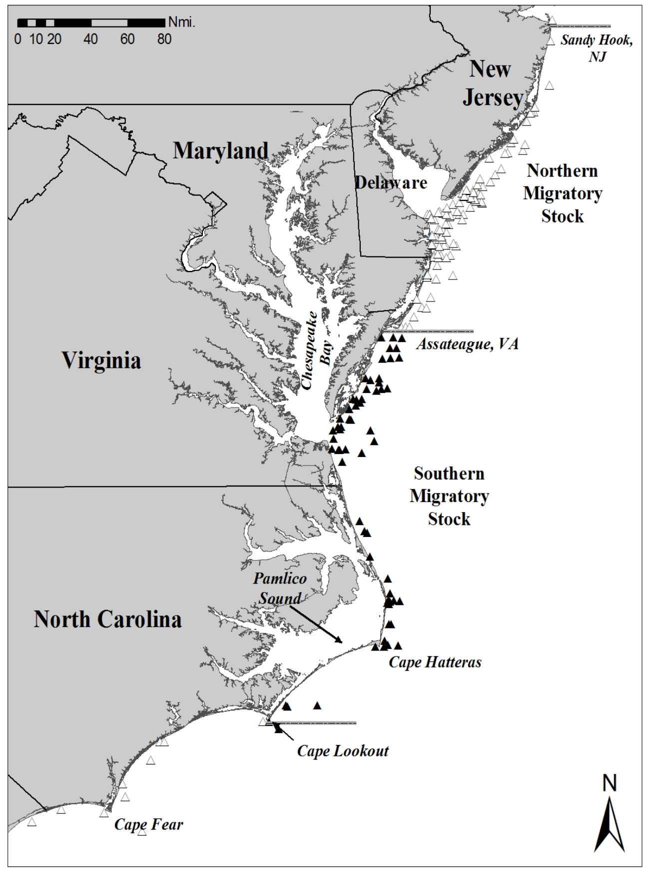


Figure 1. The distribution of common bottlenose dolphin sightings in coastal waters from North Carolina to New Jersey during July-August 2002, 2004, 2010, 2011, and 2016. Sighting locations from aerial surveys are plotted as triangle symbols. Sightings ascribed to the Southern Migratory Coastal Stock are shown as filled symbols; those from the Northern Migratory Coastal Stock as open symbols. Horizontal gray lines intersecting the coast denote the southern boundary for each stock in warm water months.

The spatial distribution and migratory movements of the Southern Migratory Coastal Stock are poorly understood and have been defined based on movement data from satellite-linked telemetry and photo-ID studies, and stable isotope studies. The distribution of this stock is best described by satellite-linked telemetry data which provided evidence for a stock of dolphins migrating seasonally along the coast between North Carolina and northern Florida (Garrison *et al.* 2017b). Telemetry data collected from two dolphins tagged in November 2004 just south of Cape Fear, North Carolina, suggested that, during October–December, this stock occupies waters of southern North Carolina (south of Cape Lookout) where it may overlap spatially with the Southern North Carolina Estuarine System (SNCES) Stock in coastal waters ≤3 km from shore (Garrison *et al.* 2017b). Based on the satellite-linked telemetery data, during January–March, the Southern Migratory Coastal Stock appears to move as far south as northern Florida where it would overlap spatially with the South Carolina/Georgia and Northern Florida Coastal stocks. During April–June, the stock moves back north to North Carolina past the tagging site to Cape Hatteras, North Carolina (Garrison *et al.* 2017b), where it overlaps, in coastal waters, with the SNCES Stock (in waters ≤3 km from shore) and the Northern North Carolina Estuarine System (NNCES) Stock (in waters ≤1 km from shore). During the warm water months of July–August, the stock is presumed to occupy coastal waters north of Cape Lookout, North Carolina, to Assateague, Virginia, including Chesapeake Bay (Figure 1) where it likely overlaps in nearshore-coastal waters of North Carolina (in waters ≤1 km from shore) and southern Chesapeake Bay waters with the NNCES Stock but the exact northern limit is unknown because the satellite-linked tags did not last beyond June (Garrison *et al.* 2017b). The northern boundary in warm water months was therefore inferred from an analysis of spatial distribution of the adjacent Northern Migratory Coastal Stock using aerial survey data and tag-telemetry data, delineating the northern boundary of the Southern Migratory Coastal Stock at the point of the southern boundary identified for the Northern Migratory Coastal Stock (Garrison *et al.* 2017b). An observed shift in spatial distribution during a summer 2004 survey indicates that the northern boundary for the Southern Migratory Coastal Stock may vary from year to year. The location of the boundary between the Northern and Southern Migratory Coastal stocks and the effects of interannual variation in spatial distribution are significant sources of uncertainty in assessing this stock (Garrison *et al.* 2017b). Stable isotope analysis conducted using biopsy samples from free-ranging animals sampled in estuarine, nearshore coastal, and offshore habitats further support migratory movement of dolphins in coastal waters between Georgia in cold water months and southern North Carolina during warm water months (Knoff 2004). Silva (2016) identified a fall increase in sightings during photo-ID surveys in coastal waters of northern South Carolina, lending further support for a migratory stock that moves seasonally through this area.

This stock may also overlap to some degree with the wNA Offshore Stock of common bottlenose dolphins. A combined genetic and logistic regression analysis that incorporated depth, latitude, and distance from shore was used to model the probability that a particular common bottlenose dolphin group seen in coastal waters was of the coastal versus offshore morphotype (Garrison *et al.* 2017a). North of Cape Hatteras during summer months, there is strong separation between the coastal and offshore morphotypes (Kenney 1990, Garrison *et al.* 2017a), and the coastal morphotype is nearly completely absent in waters >20 m depth. South of Cape Hatteras, the regression analysis indicated that the coastal morphotype is most common in waters <20 m deep, but occurs at lower densities over the continental shelf, in waters >20 m deep, where it overlaps to some degree with the offshore morphotype. For the purposes of defining stock boundaries, estimating abundance, and identifying bycaught samples, the offshore boundary of the Southern Migratory Coastal Stock is defined as the 20-m isobath north of Cape Hatteras and the 200-m isobath south of Cape Hatteras.

In summary, this stock is best designated in warm water months, when it overlaps least with other stocks, as common bottlenose dolphins of the coastal morphotype that occupy coastal waters from the shoreline to 200 m depth from Cape Lookout to Cape Hatteras, North Carolina, and coastal waters 0–20 m in depth from Cape Hatteras to Assateague, Virginia, including Chesapeake Bay. Due to the limited understanding of the distribution and movements of this stock, it is unknown whether the stock may contain multiple demographically independent populations that should be separate stocks.

It should be noted that dolphins of the coastal morphotype present in waters between 3 km from shore and the 200-m isobath from the Little River Inlet, South Carolina, to Cape Lookout, North Carolina, in summer are currently not contained within any designated stock. These dolphins could be members of the South Carolina/Georgia Coastal Stock, or the southern limit of the Southern Migratory Coastal Stock may extend further south than currently delimited. In winter, the dolphins in this region are considered members of the Southern Migratory Coastal Stock. Further research is necessary to determine the affinities of the dolphins in this region in summer.

POPULATION SIZE

The best available abundance estimate for the Southern Migratory Coastal Stock of common bottlenose dolphins in the western North Atlantic is 3,751 (CV=0.60; Table 1; Garrison *et al.* 2017a). This estimate was derived from aerial surveys conducted during the summer of 2016 covering coastal and shelf waters from Florida to New Jersey.

Background

Estimating the abundance of the Southern Migratory Coastal Stock is complicated by the spatiotemporal overlap the stock has with other coastal, estuarine, and offshore stocks of common bottlenose dolphins as described above. Summer surveys are best for estimating the abundance for this stock because it overlaps least with other coastal and estuarine common bottlenose dolphin stocks during warm water months. Based on the logistic regression described above, abundance for the Southern Migratory Coastal Stock is estimated using summer sightings made in the 0–200 m depth range between Cape Lookout (34.6°N) and Cape Hatteras, North Carolina (35.2°N), and in the 0–20 m depth range from Cape Hatteras to Assateague, Virginia (37.9°N). As noted above, the definition of the northern boundary and inter-annual variation in stock distribution are significant unquantified sources of uncertainty.

Earlier Abundance Estimates (>8 years old)

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions.

Recent Surveys and Abundance Estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters along the U.S. East Coast from southeastern Florida to Cape May, New Jersey, during the summer of 2016 (Garrison *et al.* 2017a). The survey was conducted along tracklines oriented perpendicular to the shoreline and spaced latitudinally at 20-km intervals, and covered waters from the shoreline to the continental shelf break (Garrison *et al.* 2017a).

As with previous surveys, the 2016 survey was conducted using a two-team approach to develop estimates of detection probabilities using the independent observer approach with Distance analysis (Laake and Borchers 2004). The detection functions from the 2016 and two previous surveys indicated a decreased probability of detection near the trackline. The sighting data were therefore "left-truncated" by analyzing only sightings occurring greater than 100 m from the trackline during the 2016 survey (see Buckland *et al.* 2001 for left-truncation methodology). The independent observer method assuming point independence was used to estimate detection probability on the trackline. This estimate accounts for the probability of detecting a marine mammal group conditional on it being available to both survey teams. Covariates that may influence detection probabilities (e.g., sea state, glare, cloud cover, visibility) were incorporated into both the mark-recapture and distance function components of the detection models (Laake and Borchers 2004, Garrison *et al.* 2017a). The resulting abundance estimate is negatively biased due to the effects of animals spending some time underwater where they are not available to the survey teams. However, due to the relatively short dive times of bottlenose dolphins (Klatsky *et al.* 2007) and the large group sizes, it is likely that this bias is small (Garrison *et al.* 2017a).

The abundance estimate for the 2016 summer aerial survey was 3,751 (CV=0.60; Garrison *et al.* 2017a). Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution. Another unquantified source of uncertainty in the abundance estimate is the potential overlap of this stock (during summer) with the NNCES Stock in near-shore ocean waters within 1 km from shore.

Table 1. Abundance estimate for the western North Atlantic Southern Migratory Coastal Stock of common bottlenose dolphins. Month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and coefficient of variation (CV).

Month/Year	Area	N_{est}	CV
July–August 2016	Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)	3,751	0.60

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. The best estimate for the Southern Migratory Coastal Stock of common bottlenose dolphins is 3,751 (CV=0.60). The resulting minimum population estimate is 2,353 (Table 2).

Current Population Trend

Available surveys allow an analysis of trend in population size for coastal stocks of common bottlenose dolphins. A standardized analytical approach accounting for variation in survey execution and environmental conditions was used to derive unbiased abundance estimates for each survey (Garrison *et al.* 2017a). A weighted generalized linear model was used to evaluate trends in population size by stock using abundance estimates from surveys conducted in the summers of 2002, 2004, 2010, 2011, and 2016. Abundance estimates were weighted by the inverse of their standard error, which reduces the influence of less certain estimates (Neter *et al.* 1983). Stock was treated as a fixed factor, and surveys were grouped into three periods to test for long-term trends in population size: 2002–2004, 2010–2011, and 2016. Period was also included as a fixed factor in the model along with the interaction between stock and period. Contrasts were specified to test for differences in abundance between periods for each stock (Garrison *et al.* 2017a). For the Southern Migratory Coastal Stock, the resulting mean abundance estimate for 2002–2004 was 23,206

(CV=0.25), and that for 2010–2011 was 6,694 (CV=0.62). There was no significant difference between these estimates and the estimate of 3,751 (CV=0.60) for 2016. There is limited power to detect a significant change given the high CV of the estimates, interannual variability in spatial distribution and stock abundance between 2002 and 2004, and the availability of only one recent survey (Garrison *et al.* 2017a).

An analysis of coast-wide (New Jersey to Florida) trends in abundance for common bottlenose dolphins was conducted. A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model included a linear term for survey year and an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant (p=0.0308) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse-variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison et al. 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME (see Strandings section).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are not known for the Southern Migratory Coastal Stock. The maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations likely do not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a "recovery" factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997, Wade 1998). The minimum population size of the Southern Migratory Coastal Stock of common bottlenose dolphins is 2,353. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because this stock is depleted. PBR for this stock of common bottlenose dolphins is 24 (Table 2).

Table 2. Best and minimum abundance estimates for the western North Atlantic Southern Migratory Coastal Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
3,751	0.60	2,353	0.5	0.04	24

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Southern Migratory Coastal Stock during 2014–2018 is unknown. The minimum mean annual fishery-related mortality and serious injury for observed fisheries and strandings identified as fishery-related ranged between 0 and 18.3. No additional mortality or serious injury was documented from other human-caused sources (e.g., fishery research) and therefore, the minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 ranged between 0 and 18.3 (Tables 3a, 3b and 3c). This range reflects several sources of uncertainty and is a minimum because: 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected and recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between this stock and other common bottlenose dolphin stocks throughout its range introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities and serious injuries were ascribed to a stock or stocks by comparing the season and geographic location of the take/stranding to the stock boundaries and geographic range delimited for each stock (Lyssikatos and Garrison 2018).

Fishery Information

There are 11 commercial fisheries that interact, or that potentially could interact, with this stock. These include the Category I mid-Atlantic gillnet fishery, nine Category II fisheries (Southeastern U.S. Atlantic shark gillnet, Southeast Atlantic gillnet, Chesapeake Bay inshore gillnet, Virginia pound net, Atlantic blue crab trap/pot, North

Carolina roe mullet stop net, mid-Atlantic menhaden purse seine, mid-Atlantic haul/beach seine, and Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl fisheries), and the Category III Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery. Detailed fishery information is presented in Appendix III.

Note: Animals reported in the sections to follow were ascribed to a stock or stocks of origin following methods described in Maze-Foley et al. (2019). These include strandings, observed takes (through an observer program), research takes, fisherman self-reported takes (through the Marine Mammal Authorization Program), and opportunistic at-sea observations.

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the Southern Migratory Coastal Stock in the northern part of its range. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001, Lyssikatos and Garrison 2018). This fishery is currently observed by the Northeast Fisheries Observer Program, and previously was observed by both the Northeast and Southeast Fisheries Observer Programs (through 2016). The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006, available from: https://www.federalregister.gov/documents/2006/04/26/06-3909/taking-of-marine-mammals-incidental-to-commercial-fishing-operations-bottlenose-dolphin-take). In addition, two subsequent amendments to the BDTRP were implemented in 2008 and 2012 regarding gear restrictions for medium-mesh gillnets in North Carolina waters (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan). Mortality estimates for the period (2002–2006) immediately prior to implementation of the BDTRP and 2007–2011 are available in the 2015 stock assessment report for the Northern Migratory Coastal Stock (Waring *et al.* 2016). The current report covers the most recent available five-year estimate (NMFS 2016) for 2014–2018.

Mortality estimation for this stock is difficult because: 1) observed takes are statistically rare events, 2) the Southern Migratory Coastal, Northern Migratory Coastal, NNCES, and SNCES stocks of common bottlenose dolphin overlap in coastal waters of North Carolina and Virginia at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are insufficient to consistently detect bycatch events (Lyssikatos and Garrison 2018). To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2014–2018: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data and 2) a pooled CPUE approach (where all observer data from the most recent five years were combined into one sample to estimate CPUE; Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018).

During the most recent 5-year time period, 2014–2018, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 5.16% in state waters (0–3 miles from shore) and 9.95% in federal waters (3–200 miles from shore; Lyssikatos 2021). During these trips, observers documented two dolphins (mortalities) entangled in small-mesh gillnet gear off the coast of North Carolina that may have been from the Southern Migratory Coastal Stock. One observed take (NEFOP) occurred in July 2017, and the second observed take (SEFOP) occurred in September 2014. Both takes were ascribed to the NNCES and Southern Migratory Coastal stocks (Lyssikatos 2021). The resultant 5-year mean minimum and maximum mortality estimates (2014–2018) for the Southern Migratory Coastal Stock were therefore 0 and 16.3 (CV=0.23) animals per year, respectively (Table 3a; Lyssikatos 2021).

Historical and recent stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear attached (Byrd *et al.* 2014, Waring *et al.* 2016, Lyssikatos and Garrison 2018). In July 2018, the stranding network recovered a dead dolphin entangled in gillnet gear in Virginia. This animal was ascribed to the Northern and Southern Migratory Coastal stocks. Because there is already an observer program-based bycatch estimate for the

Southern Migratory Coastal Stock for the mid-Atlantic gillnet fishery, and the bycatch estimate was not zero, the additional recovered animal was not added to the bycatch estimate. However, the overall minimum annual mortality for this stock is likely not zero. During the current 5-year period there were also seven common bottlenose dolphin strandings, five in North Carolina and two in Virginia, with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. All seven cases were ascribed to multiple stocks including the Southern Migratory Coastal Stock.

Southeastern U.S. Atlantic Shark Gillnet and Southeast Atlantic Gillnet

There have been no documented mortalities or serious injuries of common bottlenose dolphins associated with the Southeastern U.S. Atlantic Shark Gillnet or Southeast Atlantic Gillnet fisheries during 2014–2018 that could be ascribed to the Southern Migratory Coastal Stock (Mathers *et al.* 2015, 2016, 2017, 2018, 2020). These fisheries target sharks and finfish in waters between North Carolina and southern Florida. The majority of fishing effort occurs in federal waters because Florida, Georgia, and South Carolina, with limited exception, prohibit the use of gillnets in state waters. The Southeast Gillnet Observer Program observes these fisheries year-round (e.g., Mathers *et al.* 2016).

Chesapeake Bay Inshore Gillnet

During 2014–2018, stranding data documented one interaction (mortality) between a common bottlenose dolphin and inshore gillnet gear in Chesapeake Bay. In 2015, in Virginia, a dead dolphin was recovered entangled in gillnet gear (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). This animal was ascribed to the Northern and Southern Migratory Coastal stocks, and it is included in the annual human-caused mortality and serious injury total for this stock (Table 3b) as well as in the stranding database and stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). There is no observer coverage of this fishery within Maryland waters of Chesapeake Bay; however, within Virginia waters of Chesapeake Bay, there is a low level of observer coverage (<1%). No estimate of bycatch mortality is available for this fishery, and the documented interactions in this commercial gear represent a minimum known count of interactions in the last five years. Three other dead, stranded common bottlenose dolphins were recovered within Chesapeake Bay with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Two of these animals were ascribed to the Southern Migratory Coastal and NNCES stocks, and one was ascribed to the Northern and Southern Migratory Coastal and NNCES Stocks.

Virginia Pound Net

During 2014–2018, there were no documented mortalities or serious injuries involving pound net gear in Virginia. However, during 2017, one dolphin stranded with twisted twine markings indicative of interactions with pound net gear, but it is unknown whether the interactions with the gear contributed to the death of this animal, and this case is not included in the annual human-caused mortality and serious injury total for this stock. This stranding was ascribed to the Southern Migratory Coastal and NNCES stocks. It occurred inside estuarine waters near the mouth of the Chesapeake Bay in August (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). The overall impact of the Virginia pound net fishery on the Southern Migratory Coastal Stock is unknown due to the limited information on the stock's movements. Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available. An amendment to the BDTRP was implemented in 2015 requiring gear restrictions for VA pound nets in estuarine and coastal state waters of Virginia to reduce bycatch (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan).

Atlantic Blue Crab Trap/Pot

During 2014–2018, stranding data documented nine cases of common bottlenose dolphins entangled in trap/pot gear that could be ascribed to the Southern Migratory Coastal Stock. Two cases were mortalities, four were serious injuries, and for the remaining three cases, it could not be determined whether the animals were seriously injured. The mortalities occurred during 2016 in unidentified trap/pot gear and in 2015 in commercial blue crab trap/pot gear. Both mortalities were ascribed to the Southern Migratory Coastal and NNCES stocks. One serious injury occurred in 2014 in commercial blue crab trap/pot gear, and one occurred in 2015 in unidentified trap/pot gear. These two cases were ascribed to the Southern Migratory Coastal and NNCES stocks. The remaining two serious injuries occurred in 2015 and 2017 in commercial blue crab trap/pot gear; one was ascribed to the Southern Migratory Coastal and South

Carolina/Georgia Coastal stocks; the other was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. The six mortalities and serious injuries are included in the annual human-caused mortality and serious injury total for this stock (Table 3b). In addition, there were three cases where it could not be determined whether the animals were seriously injured. Two occurred in 2017. One case was ascribed to the Northern and Southern Migratory Coastal stocks, and the other was ascribed to the Southern Migratory Coastal and South Carolina/Georgia Coastal stocks. The third case occurred during 2018 and was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. All nine cases were included in the stranding database and in the stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 for SER and 13 August 2019 for NER). Details regarding the serious injury determinations can be found in Maze-Foley and Garrison (2020). Because there is no observer program, it is not possible to estimate the total number of mortalities associated with crab traps/pots and these documented interactions in this commercial gear represent a minimum known count of interactions with this fishery. Stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002, Burdett and McFee 2004).

North Carolina Roe Mullet Stop Net

During 2014–2018, there were no documented mortalities or serious injuries of common bottlenose dolphins in stop net gear. However, a dead stranded dolphin with line markings indicative of interaction with stop net gear was recovered in October 2015 ~300 yards from a stop net, but it is unknown whether the interaction with gear contributed to the death of this animal, and this case is not included in the annual human-caused mortality and serious injury total for this stock. This animal was ascribed to multiple stocks: the Southern Migratory Coastal, NNCES, and SNCES stocks. This mortality is included in the stranding database and in the stranding totals presented in Table 4 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). No estimate of bycatch mortality is available for the stop net fishery. This fishery has not had regular, ongoing federal or state observer coverage. However, the NMFS Beaufort laboratory observed this fishery in 2001–2002 (Byrd and Hohn 2010), and Duke University observed the fishery in 2005–2006 (Thayer *et al.* 2007). Entangled dolphins were not documented during these formal observations, but historical takes of dolphins entangled in stop nets occurred in 1993 and 1999 (Byrd and Hohn 2010).

Mid-Atlantic Menhaden Purse Seine

During 2014–2018, there were no documented mortalities or serious injuries in mid-Atlantic menhaden purse seine gear of common bottlenose dolphins that could be ascribed to the Southern Migratory Coastal Stock. The mid-Atlantic menhaden purse seine fishery historically reported an annual incidental take of one to five common bottlenose dolphins (NMFS 1991, pp. 5–73). There has been very limited federal observer coverage since 2008. No observer coverage was allocated to this fishery during 2014–2018. Because there is no systematic observer program for this fishery, no estimate of bycatch mortality is available.

Mid-Atlantic Haul/Beach Seine

During 2014–2018, one serious injury of a common bottlenose dolphin occurred associated with the mid-Atlantic haul/beach seine fishery that could be ascribed to the Southern Migratory Coastal Stock. During 2014, a common bottlenose dolphin was found within a haul seine net in Virginia and released alive seriously injured (Maze-Foley and Garrison 2020). The animal was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks, and is included in the annual human-caused mortality and serious injury total for this stock (Table 3b) as well as in the stranding database and stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2014–2018. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Shrimp Trawl

During 2014–2018, there were no documented mortalities or serious injuries of common bottlenose dolphins associated with the shrimp trawl fishery that could be ascribed to the Southern Migratory Coastal Stock. There has been very little systematic observer coverage of this fishery in the Atlantic during the last decade.

Hook and Line (Rod and Reel)

During 2014–2018, stranding data documented four mortalities and one serious injury that could be ascribed to the Southern Migratory Coastal Stock for which hook and line gear entanglement or ingestion were recorded. The serious injury (2017, Virginia) was ascribed to the Northern and Southern Coastal Migratory stocks (Maze-Foley and Garrison 2020). For one mortality, ascribed to the Southern Migratory Coastal and South Carolina/Georgia Coastal stocks, available evidence suggested the hook and line gear interaction contributed to the cause of death (2017, South Carolina; Maze-Foley *et al.* 2019). This serious injury and mortality are included in the annual human-caused mortality and serious injury total for this stock (Table 3b). For two of the remaining mortalities, evidence suggested the hook and line gear interactions were not a contributing factor to cause of death. Both of these mortalities occurred in 2016 (one in Virginia, one in North Carolina) and were ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. For the final mortality, ascribed to the Southern Migratory Coastal and South Carolina/Georgia Coastal stocks, it could not be determined whether the hook and line gear interaction contributed to cause of death (2017, South Carolina; Maze-Foley *et al.* 2019). All five cases were included in the stranding database and are included in the stranding totals presented in Table 4 (Northeast Regional Marine Mammal Stranding Network; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 for SER and 13 August 2019 for NER).

It should be noted that, in general, it cannot be determined if rod and reel hook and line gear originated from a commercial (i.e., commercial fisherman, charter boat, or headboat) or recreational angler because the gear type used by both sources is typically the same. Also, it is not possible to estimate the total number of interactions with hook and line gear because there is no systematic observer program. The documented interactions in this commercial gear represent a minimum known count of interactions with this fishery.

Other Mortality

Historically, there have been occasional mortalities of common bottlenose dolphins during research activities (Waring *et al.* 2016); however, none were documented during 2014–2018 that could be ascribed to the Southern Migratory Coastal Stock. All mortalities and serious injuries from known human-caused sources for the Southern Migratory Coastal Stock are summarized in Tables 3a, 3b and 3c.

Table 3a. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock for the commercial fisheries with ongoing, systematic federal observer programs. The years sampled, the type of data used, the annual percentage observer coverage, the observed serious injuries and mortalities recorded by on-board observers, and the mean annual estimate of mortality and serious injury and its CV are provided. Minimum and maximum values are reported due to uncertainty in the assignment of mortalities to this particular stock because there is spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	Years	Data Type	Observer Coverage	Observed Serious Injury	Observed Mortality	Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data
Mid-Atlantic Gillnet	2014–2018	Obs. Data Logbook	3.6, 5.6, 9.8, 7.0, 6.4	0, 0, 0, 0, 0	1, 0, 0, 1, 0	Min=0 Max=16.3 (0.23)
Southeastern U.S. Atlantic Shark Gillnet	2014–2018	Obs. Data Logbook	NA due to uncertainty in reported effort	0, 0, 0, 0, 0	0, 0, 0, 0, 0	No estimate
Southeast Atlantic Gillnet	2014–2018	Obs. Data Logbook	NA due to uncertainty in reported effort	0, 0, 0, 0, 0	0, 0, 0, 0, 0	No estimate
Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2014–2018)						**Min=0 Max=16.3 (0.23)**

Table 3b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock during 2014–2018 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	**Years**	**Data Type**	**5-year Count Based on Stranding Data**
Chesapeake Bay Inshore Gillnet[a]	2014–2018	Limited Observer and Stranding Data	Min=0 Max=1
Virginia Pound Net[b]	2014–2018	Stranding Data	Max=10
Atlantic Blue Crab Trap/Pot	2014–2018	Stranding Data	Min=0 Max=6
North Carolina Roe Mullet Stop Net[c]	2014–2018	Stranding Data	Max=10
Mid-Atlantic Menhaden Purse Seine	2014–2018	Limited Observer and Stranding Data	0
Mid-Atlantic Haul/Beach Seine	2014–2018	Limited Observer and Stranding Data	Min=0 Max=1
Shrimp Trawl	2014–2018	Limited Observer and Stranding Data	0
Hook and Line[d]	2014–2018	Stranding Data	Min=0 Max=2
Mean Annual Mortality due to unobserved commercial fisheries (2014–2018)			**Min=0** **Max=2.0**

[a] Chesapeake Bay inshore gillnet interactions are included if the animal was found entangled in gillnet gear. Strandings with markings indicative of interactions with gillnet gear are not included within the table. See "Chesapeake Bay Inshore Gillnet" text for more details.

[b] Pound net interactions are included if the animal was found entangled in pound net gear. Strandings with twisted twine markings indicative of interactions with pound net gear are not included within the table. See "Virginia Pound Net" text for more details.

[c] Stop Net interactions are included if the animal was found entangled in stop net gear. Stranding with line markings indicative of interaction with stop net gear are not included within the table. See "North Carolina Roe Mullet Stop Net" text for more details.

[d] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.

Table 3c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2014–2018) (Table 3a)	**Min=0 Max=16.3 (0.23)**
Mean Annual Mortality due to unobserved commercial fisheries (2014–2018) (Table 3b)	**Min=0 Max=2.0**
Research Takes (5-year Min/Max Count)	0
Other takes (5-year Min/Max Count)	0
Mean Annual Mortality due to research and other takes (2014–2018)	**0**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2014–2018)	**Min=0 Max=18.3**

Strandings

During 2014–2018, 565 common bottlenose dolphins stranded along the Atlantic coast between Florida and Virginia that could be ascribed to the Southern Migratory Coastal Stock (Table 4; Northeast Regional Marine Mammal Stranding Network; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 for SER and 13 August 2019 for NER; Maze-Foley *et al.* 2019). There was evidence of human interaction for 59 of these strandings, of which 43 (73%) were fisheries interactions and 1 (2%) showed evidence of a boat strike (Table 4). No evidence of human interaction was detected for 121 strandings, and for the remaining 385 strandings, it could not be determined if there was evidence of human interaction. It should be recognized that evidence of human interaction does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012, Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions due to spatial and temporal overlap among several common bottlenose dolphin stocks (Maze-Foley *et al.* 2019). Due to its migratory behavior, the Southern Migratory Coastal Stock can overlap with other common bottlenose dolphin stocks in every season. Only two of the 565 strandings ascribed to the Southern Migratory Coastal Stock were ascribed solely to this stock. Therefore, the counts in Table 4 likely include animals from other stocks and therefore overestimate the number of strandings attributable to the Southern Migratory Coastal Stock. Those strandings that could not be definitively ascribed to the Southern Migratory Coastal Stock alone are also included in the counts for these other stocks as appropriate. In addition, stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin, therefore it is possible that some of the reported strandings were of the offshore form though that number is likely to be low (Byrd *et al.* 2014).

This stock has also been impacted by three unusual mortality events (UMEs). Two events, one in 1987–1988 and one in 2013–2015, have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994, Morris *et al.* 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the Southern Migratory Coastal Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988), so impacts to the Southern Migratory Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988; Eguchi 2002). The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was 1,614 (https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic, accessed 13 November 2019). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). The number of dolphins from the Southern Migratory Coastal Stock that died in this event is unknown. Finally, a UME was declared in South Carolina during February–May 2011. Six strandings assigned to the Southern Migratory Coastal Stock were considered to be part of the UME. The cause of this UME was undetermined.

Table 4. Strandings of common bottlenose dolphins during 2014–2018 from Maryland to Florida that were ascribed to the Southern Migratory Coastal Stock, as well as number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Assignments to stock were based upon the understanding of the seasonal movements of this stock; however, there is likely overlap with other stocks throughout the year. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 21 May 2019 for SER and 13 August 2019 for NER). Please note HI does not necessarily mean the interaction caused the animal's death.

State	2014			2015			2016			2017			2018			Total
	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	(2014–2018)
Maryland	0	0	6	0	0	2	1	0	5	0	0	7	0	0	19	40
Virginia	5	5	49	4	5	55	11	4	35	7	1	29	6	1	52	269
North Carolina	2	25	24	7	20	12	6	13	16	1	8	10	2	9	7	162
South Carolina (Dec–Mar)	0	6	7	0	1	2	0	4	3	3	8	2	1	5	5	47
Georgia (Jan–Feb)	0	1	7	2	2	5	0	0	2	0	0	0	1	1	7	28
Florida (Jan–Feb)	0	1	6	0	0	2	0	0	5	0	0	2	0	1	2	19
Total	**144**			**119**			**105**			**78**			**119**			**565**

[a] Strandings from Virginia and Maryland were ascribed to stock based upon location and time of year with most occurring between May and September that could be ascribed to the Southern Migratory Coastal Stock. Some of these strandings could also be ascribed to the Northern Migratory Coastal Stock or NNCES Stock.

[b] Strandings from North Carolina were ascribed based on location and time of year. During summer and fall, some of these strandings could also be ascribed to the NNCES or SNCES stocks.

[c] Strandings in coastal waters from South Carolina during December–March are potentially ascribed to the Southern Migratory Coastal Stock or the South Carolina/Georgia Coastal Stock.

[d] Strandings in Georgia and northern Florida during January and February could be ascribed to the South Carolina/Georgia or the Northern Florida Coastal Stocks, respectively.

HABITAT ISSUES

The coastal habitat occupied by this stock is adjacent to areas of high human densities, some industrialized areas, and waters that are heavily utilized for commercial and recreational fishing, and boating activities. The blubber of stranded dolphins examined during the 1987–1988 mortality event contained very high concentrations of organic pollutants (Kuehl *et al*. 1991). Persistent organic pollutant levels have not been measured for this stock. Kucklick *et al.* (2011) measured total DDT and total PCB levels in common bottlenose dolphins from 13 sites in the wNA and northern Gulf of Mexico. Total DDT levels measured in common bottlenose dolphins sampled in Holden Beach, North Carolina, the site that may best represent the Southern Migratory Coastal Stock, were lower than 10 other sites sampled and total PCB levels were also lower than most other sampled sites (Kucklick *et al.* 2011), however the sample size for this site was very small (n=3).

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the Southern Migratory Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of common bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The Southern Migratory Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the Southern Migratory Coastal Stock is 24 and so the zero mortality rate goal, 10% of PBR, is 2.4. The documented mean annual human-caused mortality for this stock for 2014–2018 ranged between a minimum of 0 and a maximum of 18.3. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for the Southern Migratory Coastal Stock cannot be directly estimated because of the spatial overlap of this stock with several other stocks of common bottlenose dolphins resulting in uncertainty in the stock assignment of takes, 2) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these biases and uncertainties, there is insufficient information to determine whether or not the total fishery-related mortality and serious injury is approaching a zero mortality and serious injury rate. The impacts of two large UMEs on the status of this stock are unknown. Although there was no statistically significant difference in abundance for this stock between the 2010–2011 and 2016 surveys, a statistically significant decline in population size of all common bottlenose dolphins in coastal waters from New Jersey to Florida between 2010–2011 and 2016 was detected (Garrison *et al*. 2017a), concurrent with a large UME in the area; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine Mammal Stock Assessments: Guidelines for Preparation, Background and a Summary of the 1995 Assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Buckland, S. T., D. R. Andersen, K. P Burnham, J. L. Laake, D. L. Borchers and L. Thomas. 2001. Introduction to distance sampling: Estimating abundance of biological populations. Oxford University Press, New York. 432pp.

Burdett, L.G. and W.E. McFee. 2004. Bycatch of bottlenose dolphins in South Carolina, USA, and an evaluation of the Atlantic blue crab fishery categorization. J. Cetacean Res. Mange. 6(3):231–240.

Byrd, B.L. and A.A. Hohn. 2010. Challenges of documenting *Tursiops truncatus* Montagu (bottlenose dolphin) bycatch in the stop net fishery along Bogue Banks, North Carolina. Southeast. Nat. 9(1):47–62.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Caldwell, M. 2001. Social and genetic structure of bottlenose dolphin (*Tursiops truncatus*) in Jacksonville, Florida. Ph.D. dissertation from University of Miami. 143pp.

Eguchi, T. 2002. A method for calculating the effect of a die-off from stranding data. Mar. Mamm. Sci. 18(3):698–709.

Garrison, L.P., K. Barry and W. Hoggard. 2017a. The abundance of coastal morphotype bottlenose dolphins on the U.S. east coast: 2002–2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-01. 37pp.

Garrison, L.P., A.A. Hohn and L.J. Hansen. 2017b. Seasonal movements of Atlantic common bottlenose dolphin stocks based on tag telemetry data. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-02.

Hoelzel, A.R., C.W. Potter and P.B. Best. 1998. Genetic differentiation between parapatric nearshore and offshore populations of the bottlenose dolphin. Proc. Royal Soc. London 265:1177–1183.

Kenney, R.D. 1990. Bottlenose dolphins off the northeastern United States. Pages 369-386 *in:* S. Leatherwood and R. R. Reeves (Eds.). The bottlenose dolphin. Academic Press, San Diego, CA. 653pp.

Kingston, S.E. and P.E. Rosel. 2004. Genetic differentiation among recently diverged delphinid taxa determined using AFLP markers. J. Hered. 95(1):1–10.

Kingston, S.E., L.D. Adams and P.E. Rosel. 2009. Testing mitochondrial sequences and anonymous nuclear markers for phylogeny reconstruction in a rapidly radiating group: Molecular systematics of the Delphininae (Cetacea: Odontoceti: Delphinidae). BMC Evol. Biol. 9:245. 19pp.

Klatsky, L.J., R.S. Wells and J.C. Sweeney. 2007. Offshore bottlenose dolphins (*Tursiops truncatus*): Movement and dive behavior near the Bermuda pedestal. J. Mamm. 88:59–66.

Knoff, A.J. 2004. Bottlenose dolphin (*Tursiops truncatus*) population structure along the Atlantic coast of the United States: A stable isotope approach. Ph.D. Dissertation, University of Virginia, Charlottesville, Virginia. 156pp.

Kucklick, J., L. Schwacke, R. Wells, A. Hohn, A. Guichard, J. Yordy, L. Hansen, E. Zolman, R. Wilson, J. Litz, D. Nowacek, T. Rowles, R. Pugh, B. Balmer, C. Sinclair and P. Rosel. 2011. Bottlenose dolphins as indicators of persistent organic pollutants in the western North Atlantic Ocean and northern Gulf of Mexico. Environ. Sci. Technol. 45:4270–4277.

Kuehl, D.W., R. Haebler and C. Potter. 1991. Chemical residues in dolphins from the US Atlantic coast including Atlantic bottlenose obtained during the 1987/1988 mass mortality. Chemosphere 22:1071–1084.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in:* S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Advanced distance sampling. Oxford University Press, New York.

Lipscomb, T.P., F.Y. Schulman, D. Moffett and S. Kennedy. 1994. Morbilliviral disease in Atlantic bottlenose dolphins (*Tursiops truncatus*) from the 1987–1988 epizootic. J. Wildl. Dis. 30:567–571.

Litz, J.A., C.R. Hughes, L.P. Garrison, L.A. Fieber and P.E. Rosel. 2012. Genetic structure of common bottlenose dolphins (*Tursiops truncatus*) inhabiting adjacent South Florida estuaries - Biscayne Bay and Florida Bay. J. Cetacean Res. Manage. 12(1):107–117.

Lyssikatos, M. 2021. Common bottlenose dolphin (*Tursiops truncatus*) gillnet bycatch estimates along the US mid-Atlantic Coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-02.

Lyssikatos, M. and L.P. Garrison. 2018. Common bottlenose dolphin (*Tursiops truncatus*) gillnet bycatch estimates along the US mid-Atlantic Coast, 2007–2015. Northeast Fisheries Science Center Reference Document 18-07. Available from: http://www.nefsc.noaa.gov/publications/

Mathers, A.N., B.M. Deacy and J.K. Carlson. 2015. Catch and bycatch in U.S. Southeast gillnet fisheries, 2014. NOAA Tech. Memo. NMFS-SEFSC-675. 24pp.

Mathers, A.N., B.M. Deacy and J.K. Carlson. 2016. Catch and bycatch in U.S. Southeast gillnet fisheries, 2015. NOAA Tech. Memo. NMFS-SEFSC-690. 30pp.

Mathers, A.N., B.M. Deacy and J.K. Carlson. 2017. Catch and bycatch in U.S. Southeast gillnet fisheries, 2016. NOAA Tech. Memo. NMFS-SEFSC-713. 29pp.

Mathers, A.N., B.M. Deacy, H.E. Moncrief-Cox and J.K. Carlson. 2018. Catch and bycatch in U.S. Southeast gillnet fisheries, 2017. NOAA Tech. Memo. NMFS-SEFSC-728. 13pp.

Mathers, A.N., B.M. Deacy, H.E. Moncrief-Cox and J.K. Carlson. 2020. Catch and bycatch in U.S. Southeast gillnet fisheries, 2018. NOAA Tech. Memo. NMFS-SEFSC-743. 15pp.

Maze-Foley, K. and L.P. Garrison. 2020. Serious injury determinations for small cetaceans off the southeast U.S. coast, 2014–2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2020-06. 43pp.

Maze-Foley, K., B.L. Byrd, S.C. Horstman and J.R. Powell. 2019. Analysis of stranding data to support estimates of mortality and serious injury in common bottlenose dolphin (*Tursiops truncatus truncatus*) stock assessments for the Atlantic Ocean and Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-742. 42pp. Available from: https://repository.library.noaa.gov/view/noaa/23151

Mazzoil, M., J.S. Reif, M. Youngbluth, M.E. Murdoch, S.E. Bechdel, E. Howells, S.D. McCulloch, L.J. Hansen and G.D. Bossart. 2008. Home ranges of bottlenose dolphins (*Tursiops truncatus*) in the Indian River Lagoon, Florida: Environmental correlates and implications for management strategies. EcoHealth 5:278–288.

Mead, J.G. and C.W. Potter. 1995. Recognizing two populations of the bottlenose dolphin (*Tursiops truncatus*) off the Atlantic coast of North America: Morphological and ecological considerations. IBI Reports 5:31–44.

Morris, S.E., J.L. Zelner, D.A. Fauquier, T.K. Rowles, P.E. Rosel, F. Gulland and B.T. Grenfell. 2015. Partially observed epidemics in wildlife hosts: Modelling an outbreak of dolphin morbillivirus in the northwestern Atlantic, June 2013–2014. J. R. Soc. Interface 12:20150676.

Neter, J., W. Wasserman and M. Kutner. 1983. Applied linear regression models. Richard D. Irwin, Inc., Homewood, Illinois. 547pp.

NMFS [National Marine Fosheries Service]. 1991. Proposed regime to govern the interactions between marine mammals and commercial fishing operations after October 1, 1993. Draft Environmental Impact Statement, June 1991.

NMFS [National Marine Fosheries Service]. 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the MMPA. NMFS Instruction 02-204-01, February 22, 2016. Available from: http://www.nmfs.noaa.gov/op/pds/index.html

Noke, W.D. and D.K. Odell. 2002. Interactions between the Indian River Lagoon blue crab fishery and the bottlenose dolphin, *Tursiops truncatus*. Mar. Mamm. Sci 18(4):819–832.

Palka, D.L. and M.C. Rossman. 2001. Bycatch estimates of coastal bottlenose dolphin (*Tursiops truncatus*) in the U.S. mid-Atlantic gillnet fisheries for 1996 to 2000. Northeast Fisheries Science Center Reference Document 01-15. 77pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Read, A.J. and K.T. Murray. 2000. Gross evidence of human-induced mortality in small cetaceans. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-15. 21pp.
Available from: https://repository.library.noaa.gov/view/noaa/3679

Rosel, P.E., L. Hansen and A.A. Hohn. 2009. Restricted dispersal in a continuously distributed marine species: Common bottlenose dolphins *Tursiops truncatus* in coastal waters of the western North Atlantic. Mol. Ecol. 18:5030–5045.

Scott, G.P., D.M. Burn and L.J. Hansen. 1988. The dolphin die off: Long term effects and recovery of the population. Proceedings: Oceans '88, IEEE Cat. No. 88-CH2585-8, Vol. 3:819–823.

Silva, D.C. 2016. Use of photo-identification and mark-recapture techniques to identify characteristics of the stock structure of coastal bottlenose dolphins (*Tursiops truncatus*) off northern South Carolina. M.Sc. thesis, Coastal Carolina University, Conway, South Carolina. 115pp.

Speakman, T., E.S. Zolman, J. Adams, R.H. Defran, D. Laska, L. Schwacke, J. Craigie and P. Fair. 2006. Temporal and spatial aspects of bottlenose dolphin occurrence in coastal and estuarine waters near Charleston, South Carolina. NOAA Tech. Memo. NOS-NCCOS-37. 243pp.

Stolen, M.K., W.N. Durden and D.K. Odell. 2007. Historical synthesis of bottlenose dolphin (*Tursiops truncatus*) stranding data in the Indian River Lagoon system, Florida, from 1977–2005. Fla. Sci. 70:45–54.

Thayer, V.G., D.M. Waples and A.J. Read. 2007. Monitoring bycatch in the North Carolina stop net fishery. Final report for NMFS Fisheries Research Grant, project WC133F05SE5050. Available from: NMFS, Southeast Fisheries Science Center, 3209 Frederic St., Pascagoula, Mississippi 39568.

Wade, P.R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Mar. Mamm. Sci. 14(1):1–37.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Waring, G.T., E. Josephson, K. Maze-Foley and P.E. Rosel. 2016. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2015. NOAA Tech. Memo. NMFS-NE-238. 512pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Zolman, E.S. 2002. Residence patterns of bottlenose dolphins (*Tursiops truncatus*) in the Stono River Estuary, Charleston County, South Carolina. Mar. Mamm. Sci. 18:879–892.

COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*)
Northern North Carolina Estuarine System Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are found in estuarine, coastal, continental shelf, and oceanic waters of the western North Atlantic (wNA). Distinct morphological forms have been identified in offshore and coastal waters of the wNA off the U.S. East Coast: a smaller morphotype present in estuarine, coastal, and shelf waters from Florida to approximately Long Island, New York, and a larger, more robust morphotype present further offshore in deeper waters of the continental shelf and slope (Mead and Potter 1995) from Florida to Canada. The two morphotypes also differ in parasite load and prey preferences (Mead and Potter 1995), and show significant genetic divergence at both mitochondrial and nuclear DNA markers (Hoelzel *et al*. 1998, Kingston and Rosel 2004, Kingston *et al.* 2009, Rosel *et al*. 2009). The level of genetic divergence is greater than that seen between some other dolphin species (Kingston and Rosel 2004, Kingston *et al.* 2009) suggesting the two morphotypes in the wNA may represent different subspecies or species. The larger morphotype comprises the wNA Offshore Stock of common bottlenose dolphins. Spatial distribution data (Kenney 1990, Garrison *et al.* 2017a), tag-telemetry studies (Garrison *et al.* 2017b), photo-identification (photo-ID) studies (e.g., Zolman 2002; Speakman *et al.* 2006; Stolen *et al.* 2007; Mazzoil *et al.* 2008), and genetic studies (Caldwell 2001, Rosel *et al.* 2009, Litz *et al.* 2012) indicate that the coastal morphotype comprises multiple, demographically independent stocks distributed in coastal and estuarine waters of the wNA. The Northern North Carolina Estuarine System Stock is one such stock.

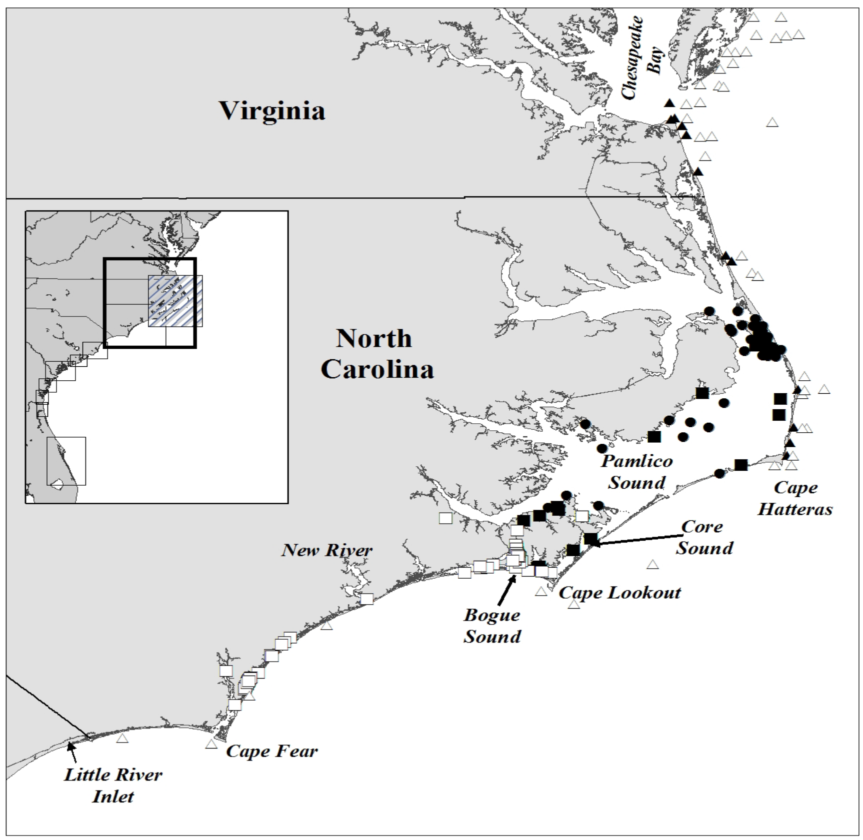


Figure 1. The distribution of bottlenose dolphins occupying coastal and estuarine waters in North Carolina and Virginia during July–August. Locations are shown from aerial surveys (triangles), satellite-linked telemetry (circles), and photo-identification studies (squares). Sightings assigned to the Northern North Carolina Estuarine System Stock are shown with filled symbols (all fall within hatched box in inset map). Photo-identification data are courtesy of Duke University and the University of North Carolina at Wilmington.

The Northern North Carolina Estuarine System (NNCES) Stock is best defined as animals that occupy primarily waters of the Pamlico Sound estuarine system (which also includes Core, Roanoke, and Albemarle sounds, and the Neuse River) during warm water months (July–August; Figure 1). Members of this stock also use coastal waters (≤1 km from shore) of North Carolina from Beaufort north to Virginia Beach, Virginia, including the Chesapeake Bay during this time period (Garrison *et al.* 2017a). Many of these animals move out of the estuaries during colder water months and occupy coastal waters (≤3 km from shore) between the New River and Oregon Inlet, North Carolina (Garrison *et al.* 2017a). However, others continue to be present in the Pamlico Sound estuarine system during cold water months (Goodman Hall *et al*. 2013). These movements and the range of this stock have been inferred from a combination of photo-ID, satellite telemetry (Garrison *et al*. 2017a, 2017b) and stable isotope (Cortese 2000) data. Eighteen animals captured and released near Beaufort, North Carolina, between 1995 and 2006 were fitted with satellite-linked transmitters and or freeze-branded and were subsequently documented, through photo-ID surveys, in waters of Pamlico Sound in warm water months (Garrison *et al.* 2017b). Satellite telemetry data from one animal tagged near Virginia Beach in September 1998 indicated that this animal moved south into waters of Pamlico Sound during October (Garrison *et al.* 2017b). This dolphin was also observed in Pamlico Sound in July 2006, providing evidence that at least some members of this stock may move into nearshore coastal waters along the northern coast of North Carolina and into coastal waters of Virginia and perhaps into Chesapeake Bay during warm water months (Garrison *et al.* 2017b). Analysis of photo-ID and satellite telemetry data indicate that a portion of the stock moves out of Pamlico Sound into coastal waters south of Cape Hatteras during cold water months (Garrison *et al.* 2017b). Telemetry and photo-ID records show that NNCES animals move as far south as the New River during January and February (Garrison *et al*. 2017b). In addition, stable isotope analysis of animals sampled along the beaches of North Carolina between Cape Hatteras and Bogue Inlet during February and March showed very low stable isotope ratios of ^{18}O relative to ^{16}O (referred to as "depleted oxygen"; Cortese 2000). One explanation for the depleted oxygen signature is a resident group of dolphins in Pamlico Sound that move into nearby coastal waters in the winter (NMFS 2001).

The distribution of the NNCES Stock overlaps in certain seasons with up to three other common bottlenose dolphin stocks. During warm water months (best defined as July and August), this stock overlaps with the Southern North Carolina Estuarine System (SNCES) Stock in estuarine waters near Beaufort, North Carolina, and in southern Pamlico Sound (Garrison *et al.* 2017b). However, SNCES Stock animals were not observed to move north of Cape Lookout in coastal waters nor into the main portion of Pamlico Sound during warm water months (Garrison *et al.* 2017b) thereby limiting the amount of overlap between the two stocks. Because the NNCES Stock also utilizes nearshore coastal waters of North Carolina north to Virginia Beach and the mouth of Chesapeake Bay, it likely overlaps with the Southern Migratory Coastal Stock in warm water months. During cold water months, the NNCES Stock overlaps in coastal waters with the Northern Migratory Coastal Stock, particularly between Cape Lookout and Cape Hatteras and may overlap with the Southern Migratory Coastal Stock between the New River and Beaufort Inlet. The timing of the seasonal movements into and out of Pamlico Sound and north along the coast likely occurs with some inter-annual variability related to seasonal changes in water temperatures and/or prey availability. Given the relatively small range of this stock and its seasonal movement in and out of the Pamlico Sound habitat, it is unlikely the stock contains multiple demographically independent populations. However, stocks of common bottlenose dolphins in other large estuaries show evidence of habitat partitioning that could suggest stock structure (Urian *et al*. 2009, Wells *et al*. 2017). To date, stock structure within this stock has not been investigated.

POPULATION SIZE

The best available abundance estimate for the NNCES Stock is 823 animals (CV=0.06; Table 1) based upon photo-ID mark-recapture surveys in summer 2013 (Gorgone *et al*. 2014). This estimate may be negatively biased as the survey did not cover all of the stock's range (i.e., coastal waters).

Earlier Abundance Estimates (>8 years old)

Read *et al.* (2003) provided the first abundance estimate of common bottlenose dolphins that occur within the estuarine portion of the NNCES Stock range. This estimate, 919 (CV=0.13, 95%CI: 730–1,190), was based on a July 2000 photo-ID mark-recapture survey of a portion of North Carolina waters inshore of the barrier islands. However, the portion of the stock that may have occurred in coastal waters (≤1 km from shore) was not accounted for in this survey. Aerial survey data from 2002 (Garrison *et al.* 2016) were therefore used to account for this portion of the stock in coastal waters. The abundance estimate for the NNCES Stock during 2000–2002 was the combined abundance from estuarine and coastal waters. This combined estimate was 1,387 (CV=0.17). Because the survey did not sample all of the estuarine waters where dolphins are known to occur, the estimate of abundance may be negatively biased. Positive bias may have been introduced through the aerial survey data because Southern Migratory Coastal Stock dolphins may have been present in the coastal strip.

A photo-ID mark-recapture study was conducted in July 2006 by Urian *et al.* (2013) using similar methods to those in Read *et al.* (2003) and included estuarine waters of North Carolina from, and including, the Little River Inlet estuary (near the North Carolina/South Carolina border) to, and including, Pamlico Sound. This survey also included coastal waters up to Cape Hatteras extending up to 1 km from shore. In order to estimate the abundance for the NNCES Stock, only sightings north of 34°46'N in central Core Sound were used (Urian *et al.* 2013). The resulting abundance estimate was 950 animals (CV=0.23, 95%CI: 516–1,384) and included a correction for the proportion of dolphins in the population with non-distinct fins (Urian *et al.* 2013). Because the survey did not include estuarine waters of Albemarle or Currituck Sounds or more northern estuarine and coastal waters, it is likely that some portion of the NNCES Stock was outside of the boundaries of the survey. Thus, the 2006 abundance estimate was most likely negatively biased.

Recent Surveys and Abundance Estimates

Photo-ID mark-recapture surveys were conducted in Pamlico, Albemarle, and Core Sounds and their tributaries during June–July 2013 to provide an abundance estimate for the NNCES Stock (Gorgone *et al.* 2014). The surveys excluded nearshore coastal waters and inshore waters at the southern extent of the NNCES range (i.e., Bogue Sound, North River, and the southernmost portion of Core Sound) to avoid potential overlap with the SNCES and Southern Migratory Coastal stocks. Estimates were obtained using closed capture-mark-recapture models and a method described by Eguchi (2014) to correct for dolphins with indistinctive fins. The resulting abundance estimate was 823 (CV=0.06; Table 1; Gorgone *et al.* 2014) and is likely to be negatively biased as not all of the stock's range (i.e., coastal waters) was covered in the survey.

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for the NNCES Stock is 823 (CV=0.06). The minimum population estimate for the NNCES Stock is 782 (Table 1).

Current Population Trend

A trend analysis has not been conducted for this stock. Gorgone *et al.* (2014) noted that the estimate from 2013 (823; CV=0.06) was similar to the previous two estimates from 2006 (950, CV=0.23) and 2000 (919, CV=0.13), but methodological differences among the estimates need to be evaluated to quantify trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. The maximum net productivity rate is assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations likely do not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size of the NNCES Stock of common bottlenose dolphins is 782. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because the stock's status relative to optimum sustainable population (OSP) is unknown (Wade and Angliss 1997). The resulting PBR for this stock is 7.8 animals (Table 1).

Table 1. Best (N_{est}) and minimum (N_{min}) abundance estimates for the Northern North Carolina Estuarine System Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR (CV=coefficient of variation).

Nest	CV Nest	Nmin	Fr	Rmax	PBR
823	0.06	782	0.50	0.04	7.8

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the NNCES Stock during 2014–2018 is unknown. The mean annual fishery-related mortality and serious injury for observed fisheries, for strandings, and for at-sea observations identified as fishery-related ranged between 7.0 and 29.8. Additional mean annual mortality and serious injury due to other human-caused sources (at-sea entanglements in debris) was 0.2. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 therefore ranged between 7.2 and 30.0 (Tables 2a, 2b and 2c). This range reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected or recovered by the stranding network (Peltier *et al*. 2012, Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between the NNCES Stock and other common bottlenose dolphin stocks introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities were assigned to a stock or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock (Lyssikatos and Garrison 2018).

Fishery Information

There are ten commercial fisheries that interact, or that potentially could interact, with this stock. These include the Category I mid-Atlantic gillnet fishery, seven Category II fisheries (Chesapeake Bay inshore gillnet, North Carolina inshore gillnet, North Carolina long haul seine, mid-Atlantic haul/beach seine, Virginia pound net, North Carolina roe mullet stop net, and Atlantic blue crab trap/pot fisheries), and two Category III fisheries (the U.S. mid-Atlantic mixed species stop seine/weir/pound net fishery, which includes the North Carolina pound net fishery, and the Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery). Detailed fishery information is presented in Appendix III.

Note: Animals reported in the sections to follow were ascribed to a stock or stocks of origin following methods described in Maze-Foley et al. (2019). These include strandings, observed takes (through an observer program), fisherman self-reported takes (through the Marine Mammal Authorization Program), research takes, and opportunistic at-sea observations.

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2019 List of Fisheries) and overlaps with the NNCES Stock. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001, Lyssikatos and Garrison 2018). This fishery is currently observed by the Northeast Fisheries Observer Program, and previously was observed by both the Northeast and Southeast Fisheries Observer Programs (through 2016). The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006; Available from: https://www.federalregister.gov/documents/2006/04/26/06-3909/taking-of-marine-mammals-incidental-to-commercial-fishing-operations-bottlenose-dolphin-take). In addition, two subsequent amendments to the BDTRP were implemented in 2008 and 2012 regarding gear restrictions for medium-mesh gillnets in North Carolina waters (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan). Mortality estimates for the period (2002–2006) immediately prior to implementation of the BDTRP and 2007–2011 are available in the 2015 stock assessment report for the NNCES Stock (Waring *et al.* 2016). The current report covers the most recent available five-year estimate (NMFS 2016) for 2014–2018.

Mortality estimation for this stock is difficult because 1) observed takes are statistically rare events, 2) the NNCES, Northern Migratory Coastal, Southern Migratory Coastal, and SNCES common bottlenose dolphin stocks overlap in coastal waters of North Carolina at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect rare bycatch events (Lyssikatos and Garrison 2018). To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2014–2018: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data; and 2) a pooled CPUE approach (where all observer data from the most recent 5 years were combined into one sample to estimate CPUE; Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018). It should be noted that effort for internal North Carolina waters (i.e., Pamlico Sound Estuary) was not included in these analyses. Federal observer sampling rates in internal waters are low and insufficient to pool with bycatch rates coming from samples collected primarily in coastal/offshore waters. Internal waters are important habitat to the NNCES so this could lead to a downward bias in bycatch mortality estimates (see North Carolina Inshore Gillnet section below).

During the most recent five-year time period, 2014–2018, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 5.16% in state waters (0–3 miles from shore) and 9.95% in federal waters (3–200 miles from shore; Lyssikatos 2021). During this timeframe, three mortalities and two cases where it could not be determined whether the animal was seriously injured were observed (Lyssikatos 2021, Lyssikatos and Garrison 2018). In February 2017, and again in May 2018, the NEFOP observed an animal entangled in a small-mesh gillnet off the coast of North Carolina that was released alive but it could not be determined whether the animal was seriously injured and therefore, it was not included in the bycatch estimate. The entangled animal from 2018 was ascribed to the NNCES Stock, and the animal from 2017 was ascribed to the NNCES and Northern Migratory Coastal stocks. In July 2017, one mortality was observed by the NEFOP off North Carolina entangled in a small-mesh gillnet and was ascribed to the NNCES and Southern Migratory Coastal stocks. In January 2015, one mortality was observed by the NEFOP off Hatteras, North Carolina, entangled in a medium-mesh gillnet and was ascribed to the NNCES and Northern Migratory Coastal stocks (Lyssikatos and Garrison 2018; this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). The third mortality was observed by the SEFOP off the coast of northern North Carolina in September 2014, and this animal was ascribed to the NNCES and Southern Migratory Coastal stocks (Lyssikatos and Garrison 2018). The animal was observed entangled in a small-mesh gillnet. The most recent five-year mean minimum and maximum mortality estimates (2014–2018) were 6.6 (CV=0.32) and 28.2 (CV=0.15) animals per year, respectively (Table 2a; Lyssikatos 2021).

Based on documented serious injury and mortality in this fishery from both federal observer coverage and other data sources, the mean annual minimum mortality is likely not zero. Historical stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear attached (Byrd *et al.* 2014, Waring *et al.* 2016). During 2014–2018, stranding data documented two mortalities entangled in a single medium-mesh gillnet off of North Carolina, and these animals were ascribed to the NNCES and Northern Migratory Coastal stocks (these animals were also self-reported by the fisherman per the Marine Mammal Authorization Program). Eight other dead, stranded common bottlenose dolphins were recovered with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. One of the eight cases was ascribed to the NNCES Stock alone, four were ascribed to both the NNCES and Southern Migratory Coastal stocks, and three cases were ascribed to both the NNCES and Northern Migratory Coastal stocks. Overall, the low level of observer coverage, rarity of observed takes, and the inability to definitively assign each observed take to stock are sources of uncertainty in the bycatch estimates for this fishery (Lyssikatos and Garrison 2018).

Chesapeake Bay Inshore Gillnet

During 2014–2018, three dead, stranded common bottlenose dolphins were recovered within Chesapeake Bay with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Two of these animals were ascribed to the Southern Migratory Coastal and NNCES stocks, and one was ascribed to the Northern and Southern Migratory Coastal and NNCES Stocks. There is no observer coverage of this fishery within Maryland waters of Chesapeake Bay; however, within Virginia waters of Chesapeake Bay, there is a low level of observer coverage (<1%). No estimate of bycatch mortality is available for this fishery.

North Carolina Inshore Gillnet

During 2014–2018, one mortality ascribed to the NNCES Stock was observed inshore entangled in small-mesh gillnet gear. Observers from the North Carolina Division of Marine Fisheries (NCDMF) recorded this incident in November 2017 (McConnaughey *et al.* 2019). The mortality is included within the annual human-caused mortality and serious injury total for the North Carolina inshore gillnet fishery (Table 2b). No estimate of bycatch mortality is available for this fishery, and the documented interaction in commercial gear represents a minimum known count of interactions with this fishery in the last five years. Five other dead, stranded common bottlenose dolphins were recovered in inshore waters with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Four of the five cases were ascribed to the NNCES Stock alone, and one case was ascribed to both the NNCES and SNCES stocks.

Previously, information on interactions between common bottlenose dolphins and the North Carolina inshore gillnet fishery was based solely on stranding data as no bycatch had been observed by state and federal observer programs. There was limited federal observer coverage (0.28%) of this fishery from May 2010 through March 2012, when NMFS observed this fishery. No common bottlenose dolphin bycatch was recorded. However, the low level of federal observer coverage in internal waters where the NNCES Stock largely resides is likely insufficient to detect bycatch events of common bottlenose dolphins if they were to occur in the inshore commercial gillnet fishery. The NCDMF has operated their own observer program since 2000 due to sea turtle bycatch in inshore gillnets. The NCDMF applied for and obtained an Incidental Take Permit (ITP) in September 2013 that covers gillnet fisheries in all internal state waters. This ITP requires monitoring of gillnets statewide in internal waters with at least 7% observer coverage of large-mesh nets during spring, summer, and fall, and at least 1% observer coverage of small mesh nets during the same seasons (U.S. Dept. of Commerce 2013, Notice of permit issuance, Fed. Register 78: 57132–57133). In November 2017 NCDMF observers recorded their first bycatch event of a common bottlenose dolphin since they began monitoring in 2000 (McConnaughey *et al.* 2019). No common bottlenose dolphin bycatch was recorded by NCDMF during 2018 (McConnaughey *et al.* 2019, Byrd *et al.* 2020).

North Carolina Long Haul Seine

There have been no documented interactions between common bottlenose dolphins of the NNCES Stock and the North Carolina long haul seine fishery during 2014–2018. The fishery includes fishing with long haul seine gear to target any species in waters off North Carolina, including estuarine waters in Pamlico and Core Sounds and their tributaries. There has not been federal observer coverage of this fishery.

Mid-Atlantic Haul/Beach Seine

During 2014–2018, stranding data documented one serious injury involving a common bottlenose dolphin and the mid-Atlantic haul/beach seine fishery in Virginia (Maze-Foley and Garrison 2020). The animal was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. The serious injury occurred during October 2014, and is included in the annual human-caused mortality and serious injury total for this stock (Table 2b) as well as in the stranding database and in the stranding totals presented in Table 3 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2014–2018. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Virginia Pound Net

During 2014–2018, there were no documented mortalities or serious injuries in pound net gear of common bottlenose dolphins that could be ascribed to the NNCES Stock (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 August 2019). However, during 2014–2018, one dolphin carcass stranded with twisted twine markings indicative of interactions with Virginia pound net gear, but no gear was attached to the carcass and it is unknown whether the interaction with the gear contributed to the death of this animal. This case was not included in the annual human-caused mortality and serious injury total for this stock (Table 2b). This stranding was ascribed to both the NNCES and Southern Migratory Coastal stock, and it was included in the stranding database and in the stranding totals presented in Table 3. Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available. The overall impact of the Virginia Pound Net fishery on the NNCES Stock is unknown due to limited information on the extent to which the stock occurs within waters inside the mouth of the Chesapeake Bay. An amendment to the BDTRP was implemented in 2015 requiring gear restrictions for VA pound nets in estuarine and coastal state waters of Virginia to reduce bycatch (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan).

North Carolina Roe Mullet Stop Net

During 2014–2018, there were no documented mortalities or serious injuries of common bottlenose dolphins in stop net gear. However, in 2015 a dead dolphin with line markings indicative of interaction with stop net gear was recovered ~300 yards from a stop net, but it is unknown whether the interaction with gear contributed to the death of this animal, and this case is therefore not included in the annual human-caused mortality and serious injury total for this stock. This animal was ascribed to the NNCES, SNCES, and Southern Migratory Costal stocks. This mortality was included in the stranding database and in the stranding totals presented in Table 3 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). This fishery has not had regular, ongoing federal or state observer coverage. However, the NMFS Beaufort laboratory observed this fishery in 2001–2002 (Byrd and Hohn 2010), and Duke University observed the fishery in 2005–2006 (Thayer *et al.* 2007). Entangled dolphins were not documented during these formal observations, but two mortalities of dolphins due to entanglement in stop nets occurred in 1993 and 1999, and were documented by the stranding network in North Carolina (Byrd and Hohn 2010).

Atlantic Blue Crab Trap/Pot

During 2014–2018, stranding data documented seven cases of common bottlenose dolphins entangled in trap/pot gear that could be ascribed to the NNCES Stock. Two cases were mortalities, four were serious injuries, and for the remaining case, it could not be determined whether the animal was seriously injured. One mortality occurred during 2016 in unidentified trap/pot gear, and the other mortality occurred during 2015 in commercial blue crab trap/pot gear. Both of the mortalities were ascribed to the NNCES and Southern Migratory Coastal stocks. One serious injury occurred in 2018 in commercial blue crab trap/pot gear, and was ascribed solely to the NNCES Stock. Two additional serious injuries occurred in 2014 in commercial blue crab trap/pot gear and in 2015 in unidentified trap/pot gear. Both of these cases were ascribed to the Southern Migratory Coastal and NNCES stocks. The remaining serious injury occurred in 2017 in commercial blue crab trap/pot gear, and was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. The two mortalities and four serious injuries are included in the annual human-caused mortality and serious injury total for this stock (Table 2b). In addition, during 2017 an animal was disentangled and released alive from commercial blue crab trap/pot gear, but it could not be determined whether the animal was seriously injured. During 2018, an animal was disentangled from unidentified trap/pot gear, released alive, and considered not seriously injured following the disentanglement. Both of these animals were ascribed to the NNCES, Northern Migratory Coastal and Southern Migratory Coastal stocks. All of the cases were included in the stranding database and in the stranding totals presented in Table 3 (Northeast Regional (NER) Marine Mammal Stranding Network; Southeast Regional (SER) Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 for SER and 13 August 2019 for NER). Details regarding the serious injury determinations can be found in Maze-Foley and Garrison (2020). Because there is no observer program, it is not possible to estimate the total number of mortalities associated with crab traps/pots. However, stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002, Burdett and McFee 2004).

North Carolina Pound Net

During 2014–2018, there were no documented mortalities or serious injuries in North Carolina pound net gear of common bottlenose dolphins (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). The North Carolina pound net fishery is included within the Category III U.S. mid-Atlantic mixed species stop seine/weir/pound net fishery. The pound net is a common fishing gear used in portions of North Carolina's estuarine waters. However, the level of interaction with common bottlenose dolphins is unknown. Between 1997 and 2018, there has only been one documented mortality (2008) in North Carolina pound net gear, and this came from stranding data (Byrd *et al*. 2014). Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with this commercial gear.

Hook and Line (Rod and Reel)

During 2014–2018, stranding data included two mortalities that could be ascribed to the NNCES Stock for which hook and line gear ingestion were documented. Both mortalities occurred in 2016, and for both, the stranding data suggested the hook and line gear interaction was not a contributing factor to cause of death (Maze-Foley *et al.* 2019). One mortality was ascribed to the NNCES and Southern Migratory Coastal stocks, and the other was ascribed to the NNCES, Northern Migratory Coastal and Southern Migratory Coastal stocks. Neither of these mortalities is included in the annual human-caused mortality and serious injury total for this stock (Table 2b).

It should be noted that, in general, it cannot be determined if rod and reel hook and line gear originated from a commercial (i.e., commercial fisherman, charter boat, or headboat) or recreational angler because the gear type used by both sources is typically the same. Also, it is not possible to estimate the total number of interactions with hook and line gear because there is no systematic observer program, so documented interactions in this gear represent a minimum known count of interactions in the last five years.

Other Mortality

Historically, there have been occasional mortalities of common bottlenose dolphins during research activities (Waring *et al.* 2016); however, none were documented during 2014–2018 that were ascribed to the NNCES Stock.

During 2015, a live animal was documented entangled in a sport toy flying ring (e.g., Aerobie or similar flying ring), and this animal was considered seriously injured (Maze-Foley and Garrison in 2020). This animal was ascribed to the NNCES Stock alone, and it is included in the annual human-caused mortality and serious injury total for this stock (Table 2c). This animal was also included within the stranding database and in the stranding totals presented in Table 3 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 SER).

In addition to animals included in the stranding database, during 2014–2018, there was one at-sea observation in the NNCES Stock area of a live common bottlenose dolphin entangled in unidentified line/fishing gear. This observation occurred in 2014, and it could not be determined if the animal was seriously injured (Maze-Foley and Garrison 2020). This animal was ascribed to the NNCES and SNCES stocks.

All mortalities and serious injuries from known sources for the NNCES Stock are summarized in Tables 2a, 2b and 2c.

Table 2a. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern North Carolina Estuarine System Stock for the commercial mid-Atlantic gillnet fishery, which has an ongoing, systematic federal observer program. The years sampled, the type of data used, the annual percentage observer coverage, the observed serious injuries and mortalities recorded by on-board observers, and the mean annual estimate of mortality and serious injury and its CV are provided. Minimum and maximum values are reported due to uncertainty in the assignment of mortalities to this particular stock because there is spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	Years	Data Type	Observer Coverage	Observed Serious Injury	Observed Mortality	Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data
Mid-Atlantic Gillnet	2014–2018	Obs. Data Logbook	3.6, 5.6, 9.8, 7.0, 6.4	0, 0, 0, 0, 0	1, 1, 0, 1, 0	Min=6.6 (0.32) Max=28.2 (0.15)
Mean Annual Mortality due to the observed mid-Atlantic gillnet commercial fishery (2014–2018)						**Min=6.6 (0.32) Max=28.2 (0.15)**

Table 2b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern North Carolina Estuarine System Stock during 2014–2018 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons. In addition, mortality due to research and other non-commercial fishery takes are included, as well as a total mean annual human-caused mortality and serious injury summed from all sources.

Fishery	Years	Data Type	5-year Count Based on Stranding Data
Chesapeake Bay Inshore Gillnet[a]	2014–2018	Limited Observer and Stranding Data	0
North Carolina Inshore Gillnet	2014–2018	Limited Federal Observer and Stranding Data	1
North Carolina Long Haul Seine	2014–2018	Stranding Data	0
Mid-Atlantic Haul/Beach Seine	2014–2018	Limited Observer and Stranding Data	Min=0 Max=1
Virginia Pound Net[b]	2014–2018	Stranding Data	0
North Carolina Roe Mullet Stop Net[c]	2014–2018	Stranding Data	Max=10
Atlantic Blue Crab Trap/Pot	2014–2018	Stranding Data	Min=1 Max=6
North Carolina Pound Net	2014–2018	Stranding Data	0
Hook and Line[d]	2014–2018	Stranding Data	0
Mean Annual Mortality due to unobserved commercial fisheries (2014–2018)			**Min=0.4 Max=1.6**

[a] Chesapeake Bay inshore gillnet interactions are included if the animal was found entangled in gillnet gear. Strandings with markings indicative of interactions with gillnet gear are not included within the table. See "Chesapeake Bay Inshore Gillnet" text for more details.

[b] Pound net interactions are included if the animal was found entangled in pound net gear. Strandings with twisted twine markings indicative of interactions with pound net gear are not included within the table. See "Virginia Pound Net" text for more details.

[c] Stop Net interactions are included if the animal was found entangled in stop net gear. Stranding with line markings indicative of interaction with stop net gear are not included within the table. See "North Carolina Roe Mullet Stop Net" text for more details.

[d] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.

Table 2c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern North Carolina Estuarine System Stock during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2014–2018) (Table 2a)	**Min=6.6 (0.32) Max=28.2 (0.15)**
Mean Annual Mortality due to unobserved commercial fisheries (2014–2018) (Table 2b)	**Min=0.4 Max=1.6**
Research Takes (5-year Min/Max Count)	0
Other takes (5-year Min/Max Count)	1
Mean Annual Mortality due to research and other takes (2014–2018)	**0.2**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2014–2018)	**Min=7.2 Max=30.0**

Strandings

Between 2014 and 2018, 480 common bottlenose dolphins stranded along coastal and estuarine waters of North Carolina, Virginia, and Maryland that could be assigned to the NNCES Stock (Table 3; Northeast Regional Marine Mammal Stranding Network, Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019 (SER) and 13 August 2019 (NER); Maze-Foley *et al.* 2019). There was evidence of human interaction for 66 of these strandings (Table 3). No evidence of human interaction was detected for 83 strandings, and for the remaining 331 strandings, it could not be determined if there was evidence of human interaction. Wells *et al.* (2015) estimated only one-third of common bottlenose dolphin carcasses in estuarine environments are recovered. In most cases, it was not possible to determine if a human interaction had occurred due to the decomposed state of the stranded animal. Of the 17 (of 144) estuarine strandings positive for human interaction, 11 (65%) of them exhibited evidence of fisheries entanglement (e.g., entanglement lesions, attached gear). Evidence of human interaction does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012, Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions where stocks overlap, particularly in coastal waters of North Carolina and Virginia, and estuarine waters near Beaufort Inlet (Maze-Foley *et al.* 2019). Of the 476 strandings ascribed to the NNCES Stock, 140 were ascribed solely to this stock. It is likely, therefore, that the counts in Table 3 include some animals from the Southern Migratory Coastal, Northern Migratory Coastal, and SNCES stocks, and thereby overestimate the number of strandings for the NNCES Stock; those strandings that could not be definitively ascribed to the NNCES Stock were also included in the counts for these other stocks as appropriate. Stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin, therefore it is possible that some of the reported strandings were of the offshore form, though that number is likely to be low (Byrd *et al*. 2014).

This stock has also been impacted by two unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994, Morris *et al.* 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the NNCES Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988) and it was estimated that 10 to 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988, Eguchi 2002). Impacts to the NNCES

Stock alone are not known. However, Scott *et al.* (1988) indicated that the observed mortalities from this event affected primarily coastal dolphins. The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was 1614 (https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic, accessed 13 November 2019). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, again suggesting that coastal stocks may have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). However, the habitat of the NNCES stock includes more nearshore coastal waters (in winter) than many estuarine stocks and so it may have been more heavily impacted by this UME than other estuarine stocks.

Table 3. Strandings of common bottlenose dolphins during 2014–2018 from North Carolina, Virginia, and Maryland that were ascribed to the Northern North Carolina Estuarine System (NNCES) Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Strandings observed in North Carolina are separated into those occurring within the Pamlico Sound estuarine system (Estuary) vs. coastal waters. Assignments to stock were based upon the understanding of the seasonal movements of this stock. However, particularly in coastal waters, there is likely overlap between the NNCES Stock and other common bottlenose dolphin stocks. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 21 May 2019 for SER and 13 August 2019 for NER). Please note HI does not necessarily mean the interaction caused the animal's death.

State	**2014**			**2015**			**2016**			**2017**			**2018**			**Total**
Type	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	**(2014–2018)**
North Carolina - Estuary	2	3	35	4	3	18	6	0	23	1	2	20	4	1	22	144
North Carolina - Coastal	2	22	27	7	15	25	8	8	18	6	7	19	2	15	13	194
Virginia[a]	5	3	16	3	3	22	7	1	18	4	0	11	5	0	20	118
Maryland[a]	0	0	4	0	0	0	0	0	0	0	0	4	0	0	16	24
Total	**119**			**100**			**89**			**74**			**98**			**480**

[a] Strandings from Virginia and Maryland include primarily waters inside Chesapeake Bay during late summer through fall. It is likely that the NNCES Stock overlaps with the Southern Migratory Coastal Stock in this area.

HABITAT ISSUES

This stock inhabits areas with significant drainage from agricultural, industrial and urban sources (Lindsey *et al.* 2014), and as such is exposed to contaminants in runoff from those sources. The blubber of 47 common bottlenose dolphins captured and released near Beaufort, North Carolina, contained levels of organochlorine contaminants, including DDT and PCBs, sufficiently high to warrant concern for the health of dolphins, and seven had unusually high levels of the pesticide methoxychlor (Hansen *et al.* 2004). Schwacke *et al.* (2002) found that the levels of polychlorinated biphenyls (PCBs) observed in female common bottlenose dolphins near Beaufort, North Carolina, would likely impair reproductive success, especially of primiparous females. In addition, exposure to high PCB levels has been linked to anemia, hyperthyroidism, and immune suppression in common bottlenose dolphins in Georgia (Schwacke *et al.* 2012). The exposure to environmental pollutants and subsequent effects on population health is an area of concern.

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act. However, this stock is considered strategic under the MMPA. PBR for the NNCES Stock is 7.8 and so the zero mortality rate goal, 10% of PBR, is 0.8. The documented mean annual human-caused mortality for this stock for 2014–2018 ranged between a minimum of 7.2 and a maximum of 30.0. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for this stock cannot be directly estimated because of the spatial overlap of several stocks of common bottlenose dolphins in North Carolina and Virginia resulting in uncertainty in the stock assignment of some takes, 2) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these uncertainties, and the fact that the maximum mean annual human-caused mortality and serious injury exceeds PBR, NMFS considers this stock strategic under the MMPA. The total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and therefore, cannot be considered to be insignificant and approaching a zero mortality and serious injury rate. The status of this stock relative to OSP is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Burdett, L.G. and W.E. McFee. 2004. Bycatch of bottlenose dolphins in South Carolina, USA, and an evaluation of the Atlantic blue crab fishery categorization. J. Cetacean Res. Manage. 6(3):231–240.

Byrd, B.L. and A.A. Hohn. 2010. Challenges of documenting Montagu (bottlenose dolphin) bycatch in the stop net fishery along Bogue Banks, North Carolina. Southeast. Nat. 9(1):47–62.

Byrd, B.L., J. K. McConnaughey and S.A. Smith. 2020. Annual sea turtle interaction monitoring of the anchored gill-net fisheries in North Carolina for incidental take permit year 2019 (1 September 2018 – 31 August 2019). North Carolina Division of Marine Fisheries. Available from: North Carolina Division of Marine Fisheries, Protected Resources Section, 3441 Arendell Street, Morehead City, North Carolina 28557.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Caldwell, M. 2001. Social and genetic structure of bottlenose dolphin (*Tursiops truncatus*) in Jacksonville, Florida. Ph.D. thesis. University of Miami. 143pp.

Cortese, N.A. 2000. Delineation of bottlenose dolphin populations in the western Atlantic Ocean using stable isotopes. Master's thesis from University of Virginia, Charlottesville. 118pp.

Eguchi, T. 2002. A method for calculating the effect of a die-off from stranding data. Mar. Mamm. Sci. 18(3):698–709.

Eguchi, T. 2014. Estimating the proportion of identifiable individuals and group sizes in photographic identification studies. Mar. Mamm. Sci. 30(3):1122–1139.

Garrison, L.P., P.E. Rosel, A.A. Hohn, R. Baird and W. Hoggard. 2016. Abundance of the coastal morphotype of bottlenose dolphin *Tursiops truncatus*, in U.S. continental shelf waters between New Jersey and Florida during winter and summer 2002. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-03. 135pp.

Garrison, L.P., K. Barry and W. Hoggard. 2017a. The abundance of coastal morphotype bottlenose dolphins on the U.S. east coast: 2002–2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-01. 37pp.

Garrison, L.P., A.A. Hohn and L.J. Hansen. 2017b. Seasonal movements of Atlantic common bottlenose dolphin stocks based on tag telemetry data. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-02.

Goodman Hall, A., J.B. McNeill, P.B. Conn, E. Davenport and A.A. Hohn. 2013. Seasonal co-occurrence of sea turtles, bottlenose dolphins, and commercial gill nets in southern Pamlico and northern Core Sounds, and adjacent coastal waters of North Carolina, USA. Endang. Species Res. 22:235–249.

Gorgone, A.M., T. Eguchi, B.L. Byrd, K.M. Altman and A.A. Hohn. 2014. Estimating the abundance of the Northern North Carolina Estuarine System Stock of common bottlenose dolphins (*Tursiops truncatus*). NOAA Tech. Memo. NMFS-SEFSC-664. 22pp.

Hansen, L.J., L.H. Schwacke, G.B. Mitchum, A.A. Hohn, R.S. Wells, E.S. Zolman and P.A. Fair 2004. Geographic variation in polychlorinated biphenyl and organochlorine pesticide concentrations in the blubber of bottlenose dolphins from the U.S. Atlantic coast. Sci. Total Environ. 319:147–172.

Hoelzel, A.R., C.W. Potter and P.B. Best. 1998. Genetic differentiation between parapatric nearshore and offshore populations of the bottlenose dolphin. Proc. Royal Soc. London 265:1177–1183.

Kenney, R.D. 1990. Bottlenose dolphins off the northeastern United States. Pages 369–386 *in:* S. Leatherwood and R. R. Reeves (Eds.). The bottlenose dolphin. Academic Press, San Diego, CA. 653pp.

Kingston, S.E. and P.E. Rosel. 2004. Genetic differentiation among recently diverged Delphinid taxa determined using AFLP markers. J. Hered. 95(1):1–10.

Kingston, S.E., L.D. Adams and P.E. Rosel. 2009. Testing mitochondrial sequences and anonymous nuclear markers for phylogeny reconstruction in a rapidly radiating group: Molecular systematics of the Delphininae (Cetacea: Odontoceti: Delphinidae). BMC Evol. Biol. 9:245. 19 pp.

Lindsey, B.D., T.M. Zimmerman, M.J. Chapman, C.A. Cravotta III and Z. Szabo. 2014. The quality of our nation's waters—Water quality in the principal aquifers of the Piedmont, Blue Ridge, and Valley and Ridge regions, eastern United States, 1993–2009. U.S. Geological Survey Circular 1354, 107pp. Available from: http://dx.doi.org/10.3133/cir1354

Lipscomb, T.P., F.Y. Schulman, D. Moffett and S. Kennedy. 1994. Morbilliviral disease in Atlantic bottlenose dolphins (*Tursiops truncatus*) from the 1987–1988 epizootic. J. Wildl. Dis. 30:567–571.

Litz, J.A., C.R. Hughes, L.P. Garrison, L.A. Fieber and P.E. Rosel. 2012. Genetic structure of common bottlenose dolphins (*Tursiops truncatus*) inhabiting adjacent South Florida estuaries - Biscayne Bay and Florida Bay. J. Cetacean Res. Manage. 12(1):107–117.

Lyssikatos, M. 2021. Common bottlenose dolphin (*Tursiops truncatus*) gillnet bycatch estimates along the US mid-Atlantic Coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-02.

Lyssikatos, M. and L.P. Garrison. 2018. Common bottlenose dolphin (*Tursiops truncatus*) gillnet bycatch estimates along the US mid-Atlantic Coast, 2007–2015. Northeast Fisheries Science Center Reference Document 18-07. Available from: http://www.nefsc.noaa.gov/publications/

Maze-Foley, K. and L.P. Garrison. 2020. Serious injury determinations for small cetaceans off the southeast U.S. coast, 2014–2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2020-06. 43pp.

Maze-Foley, K., B.L. Byrd, S.C. Horstman and J.R. Powell. 2019. Analysis of stranding data to support estimates of mortality and serious injury in common bottlenose dolphin (*Tursiops truncatus* truncatus) stock assessments for the Atlantic Ocean and Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-742. 42pp. Available from: https://repository.library.noaa.gov/view/noaa/23151

Mazzoil, M., J.S. Reif, M. Youngbluth, M.E. Murdoch, S.E. Bechdel, E. Howells, S.D. McCulloch, L.J. Hansen and G.D. Bossart. 2008. Home ranges of bottlenose dolphins (*Tursiops truncatus*) in the Indian River Lagoon Florida: Environmental correlates and implications for management strategies. EcoHealth 5:278–288.

McConnaughey, J., J. Boyd and L. Klibansky. 2019. Annual sea turtle interaction monitoring of the anchored gill-net fisheries in North Carolina for incidental take permit year 2018. North Carolina Division of Marine Fisheries. Available from: North Carolina Division of Marine Fisheries, Protected Resources Section, 3441 Arendell Street, Morehead City, North Carolina 28557.

Mead, J.G. and C.W. Potter. 1995. Recognizing two populations of the bottlenose dolphin (*Tursiops truncatus*) off the Atlantic coast of North America: Morphological and ecological considerations. IBI Reports 5:31–44.

Morris, S.E., J.L. Zelner, D.A. Fauquier, T.K. Rowles, P.E. Rosel, F. Gulland and B.T. Grenfell. 2015. Partially observed epidemics in wildlife hosts: Modelling an outbreak of dolphin morbillivirus in the northwestern Atlantic, June 2013–2014. J. R. Soc. Interface 12: 20150676.

NMFS [National Marine Fisheries Service]. 2001. Preliminary stock structure of coastal bottlenose dolphins along the Atlantic coast of the US. NMFS/SEFSC Report prepared for the Bottlenose Dolphin Take Reduction Team. Available from: NMFS, Southeast Fisheries Science Center, 75 Virginia Beach Dr., Miami, Florida 33149.

NMFS [National Marine Fisheries Service]. 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the MMPA. NMFS Instruction 02-204-01. 24pp.

Noke, W.D. and D.K. Odell. 2002. Interactions between the Indian River Lagoon blue crab fishery and the bottlenose dolphin, *Tursiops truncatus*. Mar. Mamm. Sci 18(4):819–832.

Palka, D.L. and M.C. Rossman. 2001. Bycatch estimates of coastal bottlenose dolphin (*Tursiops truncatus*) in the U.S. mid-Atlantic gillnet fisheries for 1996 to 2000. Northeast Fisheries Science Center Reference Document 01-15. 77pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Read, A.J. and K.T. Murray. 2000. Gross evidence of human-induced mortality in small cetaceans. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-15. 21pp. Available from: https://repository.library.noaa.gov/view/noaa/3679

Read, A.J., K.W. Urian, B. Wilson and D.M. Waples. 2003. Abundance of bottlenose dolphins in the bays, sounds, and estuaries of North Carolina. Mar. Mamm. Sci. 19(1):59–73.

Rosel, P.E., L. Hansen and A.A. Hohn. 2009. Restricted dispersal in a continuously distributed marine species: Common bottlenose dolphins *Tursiops truncatus* in coastal waters of the western North Atlantic. Mol. Ecol. 18:5030–5045.

Schwacke, L.H., E.O. Voit, L.J. Hansen, R.S. Wells, G.B. Mitchum, A.A. Hohn and P.A. Fair. 2002. Probabilistic risk assessment of reproductive effects of polychlorinated biphenyls on bottlenose dolphins (*Tursiops truncatus*) from the southeast United States coast. Environ. Toxicol. Chem. 21(12):2752–2764.

Schwacke, L.H., E.S. Zolman, B.C. Balmer, S. De Guise, R.C. George, J. Hoguet, A.A. Hohn, J.R. Kucklick, S. Lamb, M. Levin, J.A. Litz, W.E. McFee, N.J. Place, F.I. Townsend, R.S. Wells and T.K. Rowles. 2012. Anaemia, hypothyroidism and immune suppression associated with polychlorinated biphenyl exposure in bottlenose dolphins (*Tursiops truncatus*). Proc. R. Soc. B. https://doi.org/10.1098/rspb.2011.0665

Scott, G.P., D.M. Burn and L.J. Hansen. 1988. The dolphin die off: Long term effects and recovery of the population. Proceedings: Oceans '88, IEEE Cat. No. 88-CH2585-8, Vol. 3:819–823.

Speakman, T., E.S. Zolman, J. Adams, R.H. Defran, D. Laska, L. Schwacke, J. Craigie and P. Fair. 2006. Temporal and spatial aspects of bottlenose dolphin occurrence in coastal and estuarine waters near Charleston, South Carolina. NOAA Tech. Memo. NOS-NCCOS-37. 243pp.

Stolen, M.K., W.N. Durden and D.K. Odell. 2007. Historical synthesis of bottlenose dolphin (*Tursiops truncatus*) stranding data in the Indian River Lagoon system, Florida, from 1977–2005. Fla. Sci. 70:45–54.

Thayer, V.G., D.M. Waples and A.J. Read. 2007. Monitoring bycatch in the North Carolina stop net fishery. Final report for NMFS Fisheries Research Grant, project WC133F05SE5050. Available from: NMFS, Southeast Fisheries Science Center, 3209 Frederic St., Pascagoula, Mississippi 39568.

U.S. Department of Commerce. 2013. Notice of permit issuance. Fed. Register. 78:57132–57133.

Urian, K.W., S. Hofmann, R.S. Wells and A.J. Read. 2009. Fine-scale population structure of bottlenose dolphins (*Tursiops truncatus*) in Tampa Bay, Florida. Mar. Mamm. Sci. 25:619–638.

Urian, K.W., D.M. Waples, R.B. Tyson, L.E. Willams Hodge and A.J. Read. 2013. Abundance of bottlenose dolphins (*Tursiops truncatus*) in estuarine and near-shore waters of North Carolina, USA. J. N. C. Acad. Sci. 129:165–171.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Waring, G.T., E. Josephson, K. Maze-Foley and P.E. Rosel. 2016. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2015. NOAA Tech. Memo. NMFS-NE-238. 512pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Wells, R.S., L.H. Schwacke, T.K. Rowles, B.C. Balmer, E. Zolman, T. Speakman, F.I. Townsend, M.C. Tumlin, A. Barleycorn and K.A. Wilkinson. 2017. Ranging patterns of common bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana, following the *Deepwater Horizon* oil spill. Endanger. Spec. Res. 33:159–180.

Zolman, E.S. 2002. Residence patterns of bottlenose dolphins (*Tursiops truncatus*) in the Stono River estuary, Charleston County, South Carolina, U.S.A. Mar. Mamm. Sci. 18:879–892.

COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*) Southern North Carolina Estuarine System Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are found in estuarine, coastal, continental shelf, and oceanic waters of the western North Atlantic (wNA). Distinct morphological forms have been identified in offshore and coastal waters of the wNA off the U.S. East Coast: a smaller morphotype present in estuarine, coastal, and shelf waters from Florida to approximately Long Island, New York, and a larger, more robust morphotype present further offshore in deeper waters of the continental shelf and slope (Mead and Potter 1995) from Florida to Canada. The two morphotypes also differ in parasite load and prey preferences (Mead and Potter 1995), and show significant genetic divergence at both mitochondrial and nuclear DNA markers (Hoelzel *et al*. 1998, Kingston and Rosel 2004, Kingston *et al.* 2009, Rosel *et al*. 2009). The level of genetic divergence is greater than that seen between some other dolphin species (Kingston and Rosel 2004, Kingston *et al.* 2009) suggesting the two morphotypes in the wNA may represent different subspecies or species. The larger morphotype makes up the wNA Offshore Stock of common bottlenose dolphins. Spatial distribution data (Kenney 1990, Garrison *et al.* 2017a), tag-telemetry studies (Garrison *et al.* 2017b), photo-identification (photo-ID) studies (e.g., Zolman 2002; Speakman *et al.* 2006; Stolen *et al.* 2007; Mazzoil *et al.* 2008), and genetic studies (Caldwell 2001, Rosel *et al.* 2009, Litz *et al.* 2012) indicate that the coastal morphotype comprises multiple, demographically independent stocks distributed in coastal and estuarine waters of the wNA. The Southern North Carolina Estuarine System Stock is one such stock.

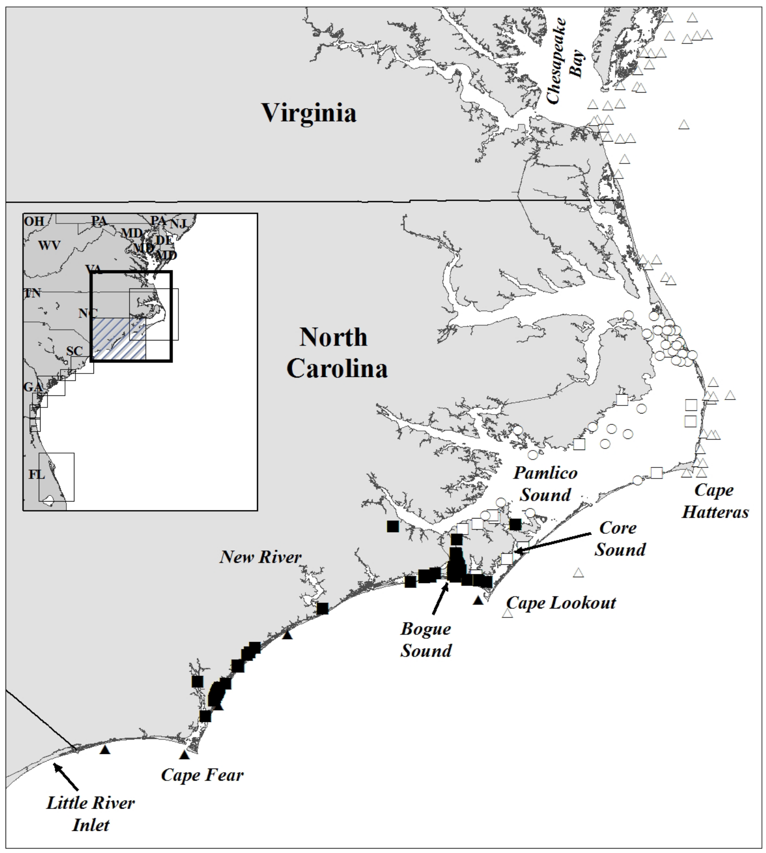


Figure 1. The distribution of bottlenose dolphins occupying coastal and estuarine waters in North Carolina and Virginia during the period July–September. Locations are shown from aerial surveys (triangles), satellite telemetry (circles) and photo-identification studies (squares). Sightings assigned to the Southern North Carolina Estuarine System stock are shown with filled symbols (all fall within hatched box in inset map). Photo-identification data are courtesy of Duke University and the University of North Carolina at Wilmington.

The Southern North Carolina Estuarine System (SNCES) Stock is best defined as animals occupying estuarine and nearshore coastal waters (≤3 km from shore) between the Little River Inlet estuary (33.9°N), inclusive of the estuary (near the North Carolina/South Carolina border), and the New River (34.5°N) during cold water months (best defined as January and February). Members of this stock do not undertake large-scale migratory movements. Instead, they expand their range only slightly northward during warmer months into estuarine waters and nearshore waters (≤3 km from shore) of southern North Carolina as far as central Core Sound and southern Pamlico Sound (Garrison *et al*. 2017b; Figure 1). These movements and the range of this stock have been inferred from a combination of telemetry, photo-ID, and genetic data (Read *et al.* 2003, Rosel *et al.* 2009, Garrison *et al.* 2017b). Two animals tagged at Holden Beach, North Carolina, just south of Cape Fear during November 2004, remained within waters of southern and central North Carolina throughout the nine-month period their tags were operational (Garrison *et al.* 2017b). Eight animals tagged and/or freeze-branded near Beaufort, North Carolina, between 1995 and 2006 were documented, using long-term photo-ID studies, to have moved south and occupied estuarine and coastal waters near Cape Fear, south of the New River during cold water months (Garrison *et al.* 2017b). A photo-ID mark-recapture survey (Read *et al.* 2003) found little movement of marked animals between the northern portion of the survey area (northern Pamlico Sound, Roanoke Sound, Albemarle Sound, and Currituck Sound) and the southern portion (Southport, Cape Fear River, New River, and Bogue Sound). The authors suggested that movement patterns, differences in group sizes, and habitats are consistent with two stocks of animals occupying estuarine waters of North Carolina (Read *et al.* 2003). SNCES animals have not been observed to move north of Cape Lookout in coastal waters nor into the northern and central portion of Pamlico Sound during warm water months (Garrison *et al.* 2017b). Finally, genetic analysis of samples from animals in waters of southern North Carolina (including known SNCES animals based on live captures and strandings of unknown stock origin between Cape Lookout and the North Carolina/South Carolina border) demonstrated significant genetic differentiation from animals occupying waters from Virginia and further north and estuarine waters of South Carolina (Rosel *et al.* 2009).

The distribution of the SNCES Stock overlaps in certain seasons with several other common bottlenose dolphin stocks. During warm water months (best defined as July and August), this stock overlaps with the Northern North Carolina Estuarine System (NNCES) Stock in estuarine waters near Beaufort, North Carolina, and in southern Pamlico Sound (Garrison *et al.* 2017b). Because this stock also utilizes nearshore coastal waters along the coast of southern North Carolina, it also overlaps with the Southern Migratory Coastal Stock as this stock makes its seasonal migratory movements (Garrison *et al*. 2017b). The timing of the seasonal contraction (and expansion) of the range of the SNCES Stock, and therefore the degree of overlap with various stocks, likely occurs with some inter-annual variability related to seasonal changes in water temperatures and/or prey availability. Given the relatively small range of this stock and its seasonal movement, it is unlikely the stock contains multiple demographically independent populations; however, structure within this stock has not been investigated.

POPULATION SIZE

The current population size of the SNCES Stock is unknown because the survey data are more than eight years old (Wade and Angliss 1997; Table 1).

Earlier Abundance Estimates (>8 years old)

Read *et al.* (2003) provided the first abundance estimate for common bottlenose dolphins occurring within the boundaries of the SNCES Stock. This estimate was based on a photo-ID mark-recapture survey of North Carolina waters inshore of the barrier islands, conducted during July 2000. Read *et al.* (2003) estimated the number of animals in the inshore waters of North Carolina occupied by the SNCES Stock at 141 (CV=0.15, 95%CI: 112–200). This estimate did not account for the portion of the stock that may have occurred in coastal waters. Summer aerial survey data from 2002 (Garrison *et al.* 2016) were therefore used to account for the portion of the stock in coastal waters. The abundance estimate for a 3-km strip from Cape Lookout to the North Carolina-South Carolina border was 2,454 (CV=0.53), yielding a total of 2,595 (CV=0.50). This estimate is likely positively biased as some animals in coastal waters may have belonged to a coastal stock.

A photo-ID mark-recapture study was conducted by Urian *et al.* (2013) in July 2006 using similar methods to those in Read *et al.* (2003) and included estuarine waters of North Carolina from, and including, the Little River Inlet estuary (near the North Carolina/South Carolina border) to, and including, Pamlico Sound. The 2006 survey also included coastal waters up to Cape Hatteras extending up to 1 km from shore. In order to estimate abundance for the SNCES Stock alone, only sightings south of 34°46'N in central Core Sound were used. The resulting abundance estimate included a correction for the proportion of dolphins with non-distinct fins in the population. The abundance estimate for the SNCES Stock based upon photo-ID mark-recapture surveys in 2006 was 188 animals (CV=0.19, 95%CI: 118–257; Urian *et al*. 2013). This estimate is probably negatively biased as the survey covered waters only to 1 km from shore and did not include habitat in southern Pamlico Sound.

Recent Surveys and Abundance Estimates

Silva *et al.* (2020) performed photo-identification (photo-ID) capture-mark-recapture (CMR) surveys in summer and winter 2014 within the estuarine waters of the SNCES stock and nearshore coastal waters. The estimated abundance in the winter survey, when the least amount of spatial overlap with other stocks is expected, was 206 (CV=0.38, 95%CI: 100–423). Each survey consisted of a single mark and recapture session and had low resight rates during the recapture session (five resights in summer, three in winter). Both surveys required extended periods of time to complete the original mark (15–20 days) and single recapture (10–30 days). In addition, the length of time between the end of the initial summer season mark and the start of the single recapture session was 19 days. These prolonged periods of time likely lead to violation of the assumption of population closure in CMR analysis as noted by the authors in particular for the summer estimate. For the winter survey, the authors note that the spatial coverage of the survey was reduced and that the distribution of the dolphins expanded outside of the survey area potentially resulting in a negative bias. Finally, the survey did not include multiple recapture sessions as suggested for CMR studies to be used for stock assessment reports (Rosel *et al.* 2011). Due to the potential bias and uncertainty associated with these estimates, the study results were not used to provide an estimate of abundance for the SNCES stock.

Minimum Population Estimate

The current minimum population estimate is unknown (Table 1). The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997).

Current Population Trend

A trend analysis has not been conducted for this stock. There are two abundance estimates from 2000/2002 and 2006. Methodological differences between the estimates need to be evaluated to quantify trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. The maximum net productivity rate is assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations likely do not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is currently undetermined. PBR is the product of the minimum population size, one-half the maximum productivity rate, and a “recovery” factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size of the SNCES Stock of common bottlenose dolphins is unknown. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because the stock's status relative to optimum sustainable population (OSP) is unknown (Table 1).

Table 1. Best (N_{est}) and minimum (N_{min}) abundance estimates for the Southern North Carolina Estuarine System Stock of common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR (CV=coefficient of variation).

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
Unknown	-	Unknown	0.5	0.04	Undetermined

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the SNCES Stock during 2014–2018 is unknown. The mean annual fishery-related mortality and serious injury estimated from observed fisheries and strandings identified as fishery-related was 0.4. No additional mortality and serious injury was documented from other human-caused sources (e.g., fishery research) and therefore, the minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was 0.4 (Tables 2a, 2b and 2c). This estimate reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are recovered by the stranding network (Peltier *et al.* 2012, Wells *et al.* 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between the SNCES Stock and other common bottlenose dolphin stocks introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities were assigned to a stock or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock (Lyssikatos and Garrison 2018).

Fishery Information

There are six commercial fisheries that interact, or that potentially could interact, with this stock. These include the Category I mid-Atlantic gillnet fishery, four Category II fisheries (North Carolina inshore gillnet, Atlantic blue crab trap/pot, North Carolina long-haul seine, and North Carolina roe mullet stop net fisheries), and the Category III Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery. Detailed fishery information is presented in Appendix III.

Note: Animals reported in the sections to follow were ascribed to a stock or stocks of origin following methods described in Maze-Foley et al. (2019). These include strandings, observed takes (through an observer program), fisherman self-reported takes (through the Marine Mammal Authorization Program), research takes, and opportunistic at-sea observations.

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the SNCES Stock. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001, Lyssikatos and Garrison 2018). This fishery is currently observed by the Northeast Fisheries Observer Program, and previously was observed by both the Northeast and Southeast Fisheries Observer Programs (through 2016). The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006, Available from: https://www.federalregister.gov/documents/2006/04/26/06-3909/taking-of-marine-mammals-incidental-to-commercial-fishing-operations-bottlenose-dolphin-take). In addition, two subsequent amendments to the BDTRP were implemented in 2008 and 2012 regarding gear restrictions for medium-mesh gillnets in North Carolina waters (https://www.fisheries.noaa.gov/national/marine-mammal-protection/bottlenose-dolphin-take-reduction-plan). Mortality estimates for the period (2002–2006) immediately prior to implementation of the BDTRP and 2007–2011 are available in the 2015 stock assessment report for the SNCES Stock (Waring *et al.* 2016). The current report covers the most recent available five-year estimate (NMFS 2016) for 2014–2018.

Mortality estimation for this stock is difficult because 1) observed takes are statistically rare events, 2) the Northern Migratory, Southern Migratory, NNCES, and SNCES common bottlenose dolphin stocks overlap in coastal waters off North Carolina and Virginia at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect rare bycatch events (Lyssikatos and Garrison 2018). To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2014–2018: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data; and 2) a pooled CPUE approach (where all observer data from the most recent five years were combined into one sample to estimate CPUE; Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018).

During the most recent five-year reporting period, 2014–2018, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 5.35% in state waters (0–3 miles from shore) and 9.95% in federal waters (3–200 miles from shore), respectively (Lyssikatos 2021). This low level of observer coverage may result in small-sample bias in the bycatch estimate because the stock is small and PBR may be less than four (NMFS 2016, Lyssikatos and Garrison 2018). During this timeframe, no common bottlenose dolphin mortalities or serious injuries that could be attributed to the SNCES Stock were observed by the NEFOP or SEFOP. The most recent five-year mean minimum and maximum mortality estimates (2014–2018) were, therefore, both unknown (Table 2a; Lyssikatos 2021).

However, based on documented serious injury and mortality in this fishery during 2014–2018 from other data sources (see Table 2a), the mean annual minimum mortality is likely not zero. In 2015, a stranded carcass was recovered with markings indicative of interaction with gillnet gear (Read and Murray 2000), but no gear was attached to the carcass and it is unknown whether the interaction with the gear contributed to the death of this animal. This case was ascribed to the SNCES and Southern Migratory Coastal stocks. Also in July 2015, through the Marine Mammal Authorization Program (MMAP), a fisherman self-reported an animal released alive following entanglement in his small-mesh gillnet in southern North Carolina. This animal was considered seriously injured (Maze-Foley and Garrison 2020) and was ascribed to the SNCES Stock. The 2015 MMAP serious injury is included in the annual human-caused mortality and serious injury total for this stock since bycatch estimates for this stock based on observer program data were zero (Table 2a). Overall, the low level of observer coverage, rarity of observed takes, and the inability to definitively assign each observed take to stock are sources of uncertainty in the bycatch estimates for this fishery.

North Carolina Inshore Gillnet

During 2014–2018, there were no documented mortalities or serious injuries within the stranding data involving inshore gillnet gear and common bottlenose dolphins that could be ascribed to the SNCES Stock (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). However, there was one case documented in which a carcass stranded with markings indicative of interaction with gillnet gear (Read and Murray 2000) but no gear was attached to the carcass and it is unknown whether the interaction with the gear contributed to the death of this animal. The case occurred in 2015 and was ascribed to the SNCES and NNCES stocks. This mortality is not included in the annual human-caused mortality and serious injury total for this stock (Table 2b), but it is included in the stranding database and in the stranding totals presented in Table 3. In addition, during 2014–2018, there was one at-sea observation of a live common bottlenose dolphin entangled in gillnet gear (in 2018) which was ascribed to the SNCES Stock, and this animal was considered seriously injured (Maze-Foley and Garrison 2020). This serious injury is included in the annual human-caused mortality and serious injury total for this stock (Table 2b).

Previously, information about interactions between common bottlenose dolphins and the North Carolina inshore gillnet fishery was based solely on stranding data as no bycatch had been observed by state and federal observer programs. There was limited federal observer coverage (0.28%) of this fishery from May 2010 through March 2012, when the NMFS observed this fishery for the first time. No common bottlenose dolphin bycatch was recorded by federal observers. The low level of federal observer coverage in internal waters where the SNCES Stock resides is likely insufficient to detect bycatch events of common bottlenose dolphins if they were to occur in the inshore commercial gillnet fishery. The North Carolina Division of Marine Fisheries (NCDMF) has operated their own observer program since 2000 due to sea turtle bycatch in inshore gillnets. The NCDMF applied for and obtained an Incidental Take Permit (ITP) in September 2013 that covers gillnet fisheries in all internal state waters. This ITP requires monitoring of gillnets statewide in internal waters with at least 7% observer coverage of large-mesh nets during spring, summer, and fall, and at least 1% observer coverage of small mesh nets during the same seasons (U.S. Dept. of Commerce 2013, Notice of permit issuance, Fed. Register 78: 57132–57133). In November 2017 NCDMF observers recorded their first bycatch event of a common bottlenose dolphin since they began monitoring in 2000 (McConnaughey *et al.* 2019), and this animal was ascribed to the NNCES Stock. No common bottlenose dolphin bycatch was recorded by NCDMF during 2018 (McConnaughey *et al.* 2019, Byrd *et al.* 2020).

Atlantic Blue Crab Trap/Pot

During 2014–2018, there were no documented mortalities or serious injuries in commercial blue crab trap/pot gear of common bottlenose dolphins that could be ascribed to the SNCES Stock (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots. However, stranding data indicate that interactions occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002, Burdett and McFee 2004).

North Carolina Long Haul Seine Fishery

There have been no documented interactions between common bottlenose dolphins of the SNCES Stock and the North Carolina long haul seine fishery during 2014–2018. The fishery includes fishing with long haul seine gear to target any species in waters off North Carolina, including estuarine waters in Pamlico and Core Sounds and their tributaries. There has not been federal observer coverage of this fishery.

North Carolina Roe Mullet Stop Net

During 2014–2018, there were no documented mortalities or serious injuries of common bottlenose dolphins in stop net gear. However, a dead stranded dolphin with line markings indicative of interaction with stop net gear was recovered in October 2015 ~300 yards from a stop net, but it is unknown whether the interaction with gear contributed to the death of this animal, and this case was not included in the annual human-caused mortality and serious injury total for this stock (Table 2b). This animal was ascribed to multiple stocks: the SNCES, NNCES, and Southern Migratory Costal stocks. This mortality is included in the stranding database and in the stranding totals presented in Table 3 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). No estimate of bycatch mortality is available for the stop net fishery. This fishery has not had regular, ongoing federal or state observer coverage. However, the NMFS Beaufort laboratory observed this fishery in 2001–2002 (Byrd and Hohn 2010), and Duke University observed the fishery in 2005–2006 (Thayer *et al.* 2007). Entangled dolphins were not documented during these formal observations, but two mortalities of dolphins due to entanglement in stop nets occurred in 1993 and 1999 and were documented by the stranding network in North Carolina (Byrd and Hohn 2010).

Hook and Line (Rod and Reel)

During 2014–2018, there were no documented mortalities or serious injuries of common bottlenose dolphins that could be ascribed to the SNCES Stock involving hook and line gear.

It should be noted that, in general, it cannot be determined if rod and reel hook and line gear originated from a commercial (i.e., commercial fisherman, charter boat, or headboat) or recreational angler because the gear type used by both sources is typically the same. Also, it is not possible to estimate the total number of interactions with hook and line gear because there is no systematic observer program, so the documented interaction in this gear represents a minimum known count of interactions in the last five years.

Other Mortality

Historically, there have been occasional mortalities of common bottlenose dolphins during research activities (Waring *et al.* 2016); however, none were documented during 2014–2018 that were ascribed to the SNCES Stock.

In addition to animals included in the stranding database and the at-sea observation mentioned above (under North Carolina Inshore Gillnet), during 2014–2018, there was one at-sea observation of a live common bottlenose dolphin entangled in unidentified line (in 2014). It could not be determined if this animal was seriously injured or not (Maze-Foley and Garrison 2020), and therefore, this animal was not included in the annual human-caused mortality and serious injury total for this stock. The animal was ascribed to the SNCES and NNCES stocks. All mortalities and serious injuries from known sources for the SNCES Stock are summarized in Tables 2a, 2b and 2c.

Table 2a. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern North Carolina Estuarine System Stock for the commercial mid-Atlantic gillnet fishery, which has an ongoing, systematic federal observer program. The years sampled, the type of data used, the annual percentage observer coverage, the observed serious injuries and mortalities recorded by on-board observers, and the mean annual estimate of mortality and serious injury and its CV are provided. Minimum and maximum values are reported due to uncertainty in the assignment of mortalities to this particular stock because there is spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	Years	Data Type	Observer Coverage	Observed Serious Injury	Observed Mortality	Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data
Mid-Atlantic Gillnet	2014–2018	Obs. Data Logbook	3.6, 5.6, 9.8, 7.7, 6.7	0, 0, 0, 0, 0	0, 0, 0, 0, 0	Unknown
5-year Count Based on Stranding Data and Fisherman Self-Reported Takes via the Marine Mammal Authorization Program						1
Mean Annual Mortality due to the observed mid-Atlantic gillnet commercial fishery (2014–2018)						**0.2**

Table 2b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern North Carolina Estuarine System Stock during 2014–2018 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	Years	Data Type	5-Year Count Based on Stranding Data and At-Sea Observations
North Carolina Inshore Gillnet[a]	2014–2018	Limited Federal and State Observers, Stranding Data, and At-Sea Observation	1
Atlantic Blue Crab Trap/Pot	2014–2018	Stranding Data	0
North Carolina Long Haul Seine	2014–2018	Stranding Data	0
North Carolina Roe Mullet Stop Net[b]	2014–2018	Stranding Data	0
Hook and Line[c]	2014–2018	Stranding Data	0
Mean Annual Mortality Due to Unobserved Commercial Fisheries (2014–2018)			**0.2**

[a] North Carolina inshore gillnet interactions are included if the animal was found entangled in gillnet gear. Strandings with line markings indicative of interaction with gillnet gear are not included within the table. See "North Carolina Inshore Gillnet" text for more details.
[b] Stop net interactions are included if the animal was found entangled in stop net gear. Stranding with line markings indicative of interaction with stop net gear are not included within the table. See "North Carolina Roe Mullet Stop Net" text for more details.
[c] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.

Table 2c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern North Carolina Estuarine System Stock during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2014–2018) (Table 2a)	**0.2**
Mean Annual Mortality due to unobserved commercial fisheries (2014–2018) (Table 2b)	**0.2**
Research Takes (5-year Min/Max Count)	0
Other takes (5-year Min/Max Count)	0
Mean Annual Mortality due to research and other takes (2014–2018)	**0**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2014–2018)	**0.4**

Strandings

Between 2014 and 2018, 53 common bottlenose dolphins stranded along coastal and estuarine waters of North Carolina that could be ascribed to the SNCES Stock (Table 3; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; Maze-Foley *et al.* 2019). There was evidence of human interaction for five of these strandings, all of which were fisheries interactions (Table 3). No evidence of human interaction was detected for 30 strandings, and for the remaining 18 strandings, it could not be determined if there was evidence of human interaction. It should be recognized that evidence of human interaction does not always indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012, Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

As described in the Stock Definition and Geographic Range section, there is spatiotemporal overlap between the SNCES Stock and the Southern Migratory Coastal Stock in coastal waters of southern North Carolina when the Southern Migratory Coastal Stock makes its seasonal migrations north and south. There is also overlap in waters from southern Pamlico Sound to Bogue Sound with the NNCES Stock during late summer and early fall. Therefore, assignment of animals to a single stock is impossible in some seasons and regions (Maze-Foley *et al.* 2019). Of the 53 strandings ascribed to the SNCES Stock, 11 were ascribed solely to this stock and one of those was identified as having evidence of both a fishery interaction and boat collision. It is likely that the counts in Table 3 include some animals from the Southern Migratory Coastal and/or NNCES Stock and therefore overestimate the number of strandings for the SNCES Stock; those strandings that could not be solely ascribed to the SNCES Stock were also included in the counts for these other stocks as appropriate. In addition, stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin. Therefore, it is possible that some of the reported strandings recorded along the coast were of the offshore form, although that number is likely to be low (Byrd *et al*. 2014).

This stock has been impacted by two unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994, Morris *et al.* 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the SNCES Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988) and it was estimated that 10 to 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988, Eguchi 2002). Impacts to the SNCES Stock alone are not known. However, Scott *et al.* (1988) indicated that the observed mortalities from this event affected primarily coastal rather than estuarine dolphins. The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was 1614 (https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic, accessed 13 November 2019). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks may have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). However, the habitat of the SNCES Stock includes more nearshore coastal waters than many estuarine stocks and so it may have been more heavily impacted by this UME than other estuarine stocks.

Table 3. Strandings of common bottlenose dolphins during 2014–2018 from North Carolina that were ascribed to the Southern North Carolina Estuarine System (SNCES) Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Strandings observed in North Carolina are separated into those occurring within estuaries vs. coastal waters. Assignments to stock were based upon the understanding of the seasonal movements of this stock. However, particularly in coastal waters, there is likely overlap between the SNCES Stock and other common bottlenose dolphin stocks. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 21 May 2019). Please note HI does not necessarily mean the interaction caused the animal's death.

State	2014			2015			2016			2017			2018			Total
	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	(2014–2018)
North Carolina - Estuary	0	1	3	1	0	0	0	1	2	0	0	3	1	1	2	15
North Carolina - Coastal	0	9	1	3	4	1	0	7	4	0	5	2	0	2	0	38
Total	**14**			**9**			**14**			**10**			**6**			**53**

HABITAT ISSUES

This stock inhabits areas with significant drainage from agricultural, industrial, and urban sources (Lindsey *et al.* 2014), and as such is exposed to contaminants in runoff from those sources. The blubber of 47 common bottlenose dolphins captured and released near Beaufort, North Carolina, contained levels of organochlorine contaminants, including DDT and PCBs, sufficiently high to warrant concern for the health of dolphins, and seven had unusually high levels of the pesticide methoxychlor (Hansen *et al.* 2004). Schwacke *et al.* (2002) found that the levels of polychlorinated biphenyls (PCBs) observed in female common bottlenose dolphins near Beaufort, North Carolina, would likely impair reproductive success, especially of primiparous females.

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act. NMFS considers the SNCES Stock to be a strategic stock under the MMPA because while the abundance of the SNCES Stock is currently unknown, based on the restricted range of the stock and previous abundance estimates it is likely small and therefore relatively few mortalities and serious injuries per year would exceed PBR. An annual average of 0.4 carcasses showing evidence of fishery interaction (primarily gillnet interactions, Table 2) were recovered within this stock's range during 2014–2018. However, this estimate is biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for this stock cannot be directly estimated because of the spatial overlap of several stocks of bottlenose dolphins in this area resulting in uncertainty in the stock assignment of takes, and 2) there are several commercial fisheries operating within this stock's boundaries and these fisheries have little to no observer coverage. In addition, the number of stranded dolphins showing evidence of fishery interactions is nearly 10% of the total number of strandings, suggesting more fishery interactions occur than are observed. Finally, Wells *et al*. (2015) estimated that only one-third of bottlenose dolphin carcasses in estuarine environments are recovered, indicating significantly more mortalities may occur than are recorded. Therefore, the documented mortalities must be considered minimum estimates of total fishery-related mortality and are of concern given the stock's restricted range and likely small abundance. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching a zero mortality and serious injury rate. The status of this stock relative to OSP is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Burdett, L.G. and W.E. McFee. 2004. Bycatch of bottlenose dolphins in South Carolina, USA, and an evaluation of the Atlantic blue crab fishery categorization. J. Cetacean Res. Manage. 6(3):231–240.

Byrd, B.L. and A.A. Hohn. 2010. Challenges of documenting *Tursiops truncatus* Montagu (bottlenose dolphin) bycatch in the stop net fishery along Bogue Banks, North Carolina. Southeast. Nat. 9(1):47–62.

Byrd, B.L., J. K. McConnaughey and S.A. Smith. 2020. Annual sea turtle interaction monitoring of the anchored gill-net fisheries in North Carolina for incidental take permit year 2019 (1 September 2018 – 31 August 2019). North Carolina Division of Marine Fisheries. Available from: North Carolina Division of Marine Fisheries, Protected Resources Section, 3441 Arendell Street, Morehead City, North Carolina 28557.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Caldwell, M. 2001. Social and genetic structure of bottlenose dolphin (*Tursiops truncatus*) in Jacksonville, Florida. Ph.D. thesis, University of Miami. 143pp.

Eguchi, T. 2002. A method for calculating the effect of a die-off from stranding data. Mar. Mamm. Sci. 18(3):698–709.

Garrison, L.P., P.E. Rosel, A.A. Hohn, R. Baird and W. Hoggard. 2016. Abundance of the coastal morphotype of bottlenose dolphin *Tursiops truncatus*, in U.S. continental shelf waters between New Jersey and Florida during winter and summer 2002. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-03. 135 pp.

Garrison, L.P., K. Barry and W. Hoggard. 2017a. The abundance of coastal morphotype bottlenose dolphins on the U.S. east coast: 2002–2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-01.

Garrison, L.P., A.A. Hohn and L.J. Hansen. 2017b. Seasonal movements of Atlantic common bottlenose dolphin stocks based on tag telemetry data. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2017-02.

Hansen, L.J., L.H. Schwacke, G.B. Mitchum, A.A. Hohn, R.S. Wells, E.S. Zolman and P.A. Fair. 2004. Geographic variation in polychlorinated biphenyl and organochlorine pesticide concentrations in the blubber of bottlenose dolphins from the U.S. Atlantic coast. Sci. Total Environ. 319:147–172.

Hoelzel, A.R., C.W. Potter and P.B. Best. 1998. Genetic differentiation between parapatric nearshore and offshore populations of the bottlenose dolphin. Proc. Royal Soc. London 265:1177–1183.

Kenney, R.D. 1990. Bottlenose dolphins off the northeastern United States. Pages 369–386 *in:* S. Leatherwood and R. R. Reeves (Eds.). The bottlenose dolphin. Academic Press, San Diego, California. 653pp.

Kingston, S.E. and P.E. Rosel. 2004. Genetic differentiation among recently diverged delphinid taxa determined using AFLP markers. J. Hered. 95(1):1–10.

Kingston, S.E., L.D. Adams and P.E. Rosel. 2009. Testing mitochondrial sequences and anonymous nuclear markers for phylogeny reconstruction in a rapidly radiating group: Molecular systematics of the Delphininae (Cetacea: Odontoceti: Delphinidae). BMC Evol. Biol. 9: 245. 19 pp.

Lipscomb, T.P., F.Y. Schulman, D. Moffett and S. Kennedy. 1994. Morbilliviral disease in Atlantic bottlenose dolphins (*Tursiops truncatus*) from the 1987–1988 epizootic. J. Wildl. Dis. 30:567–571.

Litz, J.A., C.R. Hughes, L.P. Garrison, L.A. Fieber and P.E. Rosel. 2012. Genetic structure of common bottlenose dolphins (*Tursiops truncatus*) inhabiting adjacent South Florida estuaries - Biscayne Bay and Florida Bay. J. Cetacean Res. Manage. 12(1):107–117.

Lindsey, B.D., T.M. Zimmerman, M.J. Chapman, C.A. Cravotta III and Z. Szabo. 2014. The quality of our nation's waters—Water quality in the principal aquifers of the Piedmont, Blue Ridge, and Valley and Ridge regions, eastern United States, 1993–2009. U.S. Geological Survey Circular 1354, 107pp. Available from: https://doi.org/10.3133/cir1354

Lyssikatos, M. 2021. Common bottlenose dolphin (*Tursiops truncatus*) gillnet bycatch estimates along the US mid-Atlantic Coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-02.

Lyssikatos, M. and L.P. Garrison. 2018. Common bottlenose dolphin (*Tursiops truncatus*) gillnet bycatch estimates along the US mid-Atlantic Coast, 2007–2015. Northeast Fisheries Science Center Reference Document 18-07. Available from: https://repository.library.noaa.gov/view/noaa/22756

Maze-Foley, K. and L.P. Garrison. 2020. Serious injury determinations for small cetaceans off the southeast U.S. coast, 2014–2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2020-06. 43pp.

Maze-Foley, K., B.L. Byrd, S.C. Horstman and J.R. Powell. 2019. Analysis of stranding data to support estimates of mortality and serious injury in common bottlenose dolphin (*Tursiops truncatus truncatus*) stock assessments for the Atlantic Ocean and Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-742. 42pp. Available from: https://repository.library.noaa.gov/view/noaa/23151

Mazzoil, M., J.S. Reif, M. Youngbluth, M.E. Murdoch, S.E. Bechdel, E. Howells, S.D. McCulloch, L.J. Hansen and G.D. Bossart. 2008. Home ranges of bottlenose dolphins (*Tursiops truncatus*) in the Indian River Lagoon Florida: Environmental correlates and implications for management strategies. EcoHealth 5:278–288.

McConnaughey, J., J. Boyd and L. Klibansky. 2019. Annual sea turtle interaction monitoring of the anchored gill-net fisheries in North Carolina for incidental take permit year 2018. North Carolina Division of Marine Fisheries. Available from: North Carolina Division of Marine Fisheries, Protected Resources Section, 3441 Arendell Street, Morehead City, North Carolina 28557.

Mead, J.G. and C.W. Potter. 1995. Recognizing two populations of the bottlenose dolphin (*Tursiops truncatus*) off the Atlantic coast of North America: Morphological and ecological considerations. IBI Reports 5:31–44.

Morris, S.E., J.L. Zelner, D.A. Fauquier, T.K. Rowles, P.E. Rosel, F. Gulland and B.T. Grenfell. 2015. Partially observed epidemics in wildlife hosts: Modelling an outbreak of dolphin morbillivirus in the northwestern Atlantic, June 2013–2014. J. R. Soc. Interface 12:20150676.

NMFS [National Marine Fisheries Service]. 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the MMPA. NMFS Instruction 02-204-01, February 22, 2016. Available from: http://www.nmfs.noaa.gov/op/pds/index.html

Noke, W.D. and D.K. Odell. 2002. Interactions between the Indian River Lagoon blue crab fishery and the bottlenose dolphin, *Tursiops truncatus*. Mar. Mamm. Sci 18(4):819–832.

Palka, D.L. and M.C. Rossman. 2001. Bycatch estimates of coastal bottlenose dolphin (*Tursiops truncatus*) in the U.S. mid-Atlantic gillnet fisheries for 1996 to 2000. Northeast Fisheries Science Center Reference Document 01-15. 77pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Read, A.J. and K.T. Murray. 2000. Gross evidence of human-induced mortality in small cetaceans. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-15. 21pp. Available from: https://repository.library.noaa.gov/view/noaa/3679

Read, A.J., K.W. Urian, B. Wilson and D.M. Waples 2003. Abundance of bottlenose dolphins in the bays, sounds, and estuaries of North Carolina. Mar. Mamm. Sci. 19(1):59–73.

Rosel, P.E., K.D. Mullin, L. Garrison, L. Schwacke, J. Adams, B. Balmer, P. Conn, M.J. Conroy, T. Eguchi, A. Gorgone, A. Hohn, M. Mazzoil, C. Schwartz, C. Sinclair, T. Speakman, K. Urian, N. Vollmer, P. Wade, R. Wells and E. Zolman. 2011. Photo-identification capture-mark-recapture techniques for estimating abundance of bay, sound and estuary populations of bottlenose dolphins along the U.S. East Coast and Gulf of Mexico: A workshop report. NOAA Tech. Memo. NMFS-SEFSC-621. 30pp. Available from: https://repository.library.noaa.gov/view/noaa/4014

Rosel, P.E., L. Hansen and A.A. Hohn. 2009. Restricted dispersal in a continuously distributed marine species: Common bottlenose dolphins *Tursiops truncatus* in coastal waters of the western North Atlantic. Mol. Ecol. 18:5030–5045.

Schwacke, L.H., E.O. Voit, L.J. Hansen, R.S. Wells, G.B. Mitchum, A.A. Hohn and P.A. Fair. 2002. Probabilistic risk assessment of reproductive effects of polychlorinated biphenyls on bottlenose dolphins (*Tursiops truncatus*) from the southeast United States coast. Environ. Toxicol. Chem. 21(12):2752–2764.

Scott, G.P., D.M. Burn and L.J. Hansen. 1988. The dolphin die off: Long term effects and recovery of the population. Proceedings: Oceans '88, IEEE Cat. No. 88-CH2585-8, Vol. 3:819–823.

Silva, D., R.F. Young, A. Lavin, C. O'Shea and E. Murray. 2020. Abundance and seasonal distribution of the Southern North Carolina Estuarine System Stock (USA) of common bottlenose dolphins (*Tursiops truncatus*). J. Cetacean Res. Manage. 21:33–43.

Speakman, T., E.S. Zolman, J. Adams, R.H. Defran, D. Laska, L. Schwacke, J. Craigie and P. Fair. 2006. Temporal and spatial aspects of bottlenose dolphin occurrence in coastal and estuarine waters near Charleston, South Carolina. NOAA Tech. Memo. NOS-NCCOS-37. 243pp.

Stolen, M.K., W.N. Durden and D.K. Odell. 2007. Historical synthesis of bottlenose dolphin (*Tursiops truncatus*) stranding data in the Indian River Lagoon system, Florida, from 1977–2005. Fla. Sci. 70:45–54.

Thayer, V.G., D.M. Waples and A.J. Read. 2007. Monitoring bycatch in the North Carolina stop net fishery. Final report for NMFS Fisheries Research Grant, project WC133F05SE5050. Available from: NMFS, Southeast Fisheries Science Center, 3209 Frederic St., Pascagoula, Mississippi 39568.

U.S. Department of Commerce. 2013. Notice of permit issuance. Fed. Register. 78:57132–57133.

Urian, K.W., D.M. Waples, R.B. Tyson, L.E. Willams Hodge and A.J. Read. 2013. Abundance of bottlenose dolphins (*Tursiops truncatus*) in estuarine and near-shore waters of North Carolina, USA. J. N. C. Acad. Sci. 129(4):165–171.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Waring, G.T., E. Josephson, K. Maze-Foley and P.E. Rosel. 2016. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2015. NOAA Tech. Memo. NMFS-NE-238. 512pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Zolman, E.S. 2002. Residence patterns of bottlenose dolphins (*Tursiops truncatus*) in the Stono River estuary, Charleston County, South Carolina, U.S.A. Mar. Mamm. Sci. 18:879–892.

HARBOR PORPOISE (*Phocoena phocoena phocoena*): Gulf of Maine/Bay of Fundy Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

This stock is found in U.S. and Canadian Atlantic waters. The distribution of harbor porpoises has been documented by sighting surveys, satellite telemetry data, passive acoustic monitoring, strandings and takes reported by NMFS observers in the Sea Sampling Programs. During summer (July to September), harbor porpoises are concentrated in the northern Gulf of Maine, southern Bay of Fundy and around the southern tip of Nova Scotia, generally in waters less than 150 m deep (Gaskin 1977, Kraus *et al.* 1983, Palka 1995), with lower densities in the upper Bay of Fundy and on Georges Bank (Palka 2000). During fall (October–December) and spring (April–June), harbor porpoises are widely dispersed from New Jersey to Maine, with lower densities farther north and south. During winter (January to March), intermediate densities of harbor porpoises can be found in waters off New Jersey to North Carolina, and lower densities are found in waters off New York to New Brunswick, Canada. In non-summer months they have been seen from the coastline to deep waters (>1800 m; Westgate *et al.* 1998), although the majority are found over the continental shelf. Passive acoustic monitoring detected harbor porpoises regularly during the period January–May offshore of Maryland (Wingfield *et al.* 2017). There does not appear to be a temporally coordinated migration or a specific migratory route to and from the Bay of Fundy region. However, during the fall, several satellite-tagged harbor porpoises did favor the waters around the 92-m isobath, which is consistent with observations of high rates of incidental catches in this depth range (Read and Westgate 1997). There were two stranding records from Florida during the 1980s (Smithsonian strandings database) and one in 2003 (NE Regional Office/NMFS strandings and entanglement database).

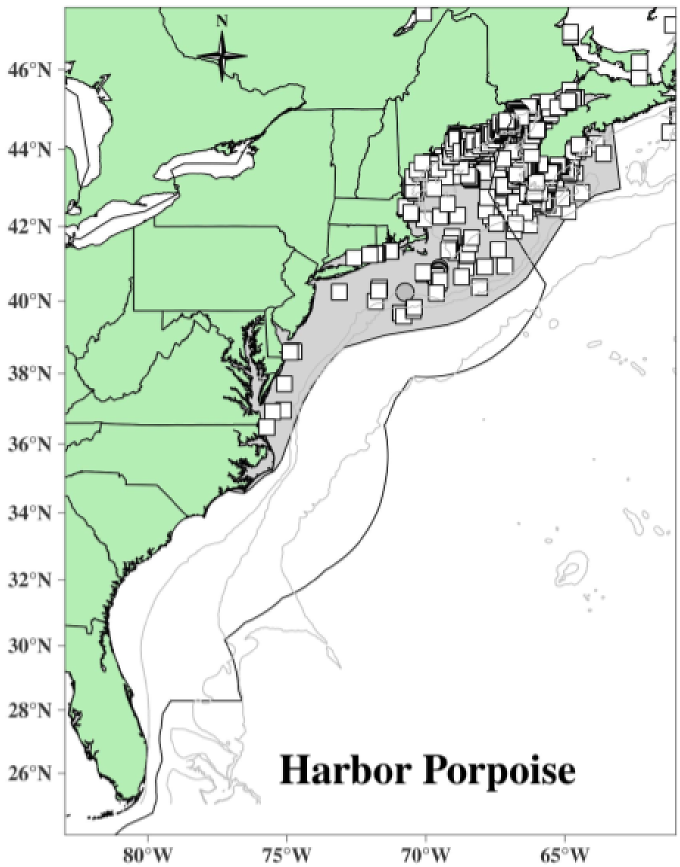


Figure 1. Distribution of harbor porpoises from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and portions of DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100m, 200m, 1000m, and 4000m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

Gaskin (1984, 1992) proposed that there were four separate populations in the western North Atlantic: the Gulf of Maine/Bay of Fundy, Gulf of St. Lawrence, Newfoundland and Greenland populations. Analyses involving mtDNA (Wang *et al.* 1996; Rosel *et al.* 1999a, 1999b), organochlorine contaminants (Westgate *et al.* 1997, Westgate and Tolley 1999), heavy metals (Johnston 1995), and life history parameters (Read and Hohn 1995) support Gaskin's proposal. Genetic studies using mitochondrial DNA (Rosel *et al.* 1999a) and contaminant studies using total PCBs

(Westgate and Tolley 1999) indicate that the Gulf of Maine/Bay of Fundy females were distinct from females from the other populations in the Northwest Atlantic. Gulf of Maine/Bay of Fundy males were distinct from Newfoundland and Greenland males, but not from Gulf of St. Lawrence males according to studies comparing mtDNA (Palka *et al.* 1996, Rosel *et al.* 1999a) and CHLORs, DDTs, PCBs and CHBs (Westgate and Tolley 1999). Nuclear microsatellite markers have also been applied to samples from these four populations, but this analysis failed to detect significant population sub-division in either sex (Rosel *et al.* 1999a). These patterns may be indicative of female philopatry coupled with dispersal of males. Both mitochondrial DNA and microsatellite analyses indicate that the Gulf of Maine/Bay of Fundy stock is not the sole contributor to the aggregation of porpoises found off the mid-Atlantic states during winter (Rosel *et al.* 1999a, Hiltunen 2006). Mixed-stock analyses using twelve microsatellite loci in both Bayesian and likelihood frameworks indicate that the Gulf of Maine/Bay of Fundy is the largest contributor (~60%), followed by Newfoundland (~25%) and then the Gulf of St. Lawrence (~12%), with Greenland making a small contribution (<3%). For Greenland, the lower confidence interval of the likelihood analysis includes zero. For the Bayesian analysis, the lower 2.5% posterior quantiles include zero for both Greenland and the Gulf of St. Lawrence. Intervals that reach zero provide the possibility that these populations contribute no animals to the mid-Atlantic aggregation.

This report follows Gaskin's hypothesis on harbor porpoise stock structure in the western North Atlantic, where the Gulf of Maine and Bay of Fundy harbor porpoises are recognized as a single management stock separate from harbor porpoise populations in the Gulf of St. Lawrence, Newfoundland, and Greenland. It is unlikely that the Gulf of Maine/Bay of Fundy harbor porpoise stock contains multiple demographically independent populations (Rosel *et al.* 1999a, Hiltunen 2006), but a comparison of samples from the Scotian shelf to the Gulf of Maine has not yet been made. There is currently an effort to conduct an integrated genetic analysis of harbor porpoise across the North Atlantic, including new samples collected recently in U.S. waters.

POPULATION SIZE

The best current abundance estimate of the Gulf of Maine/Bay of Fundy harbor porpoise stock is the sum of the 2016 NEFSC and Department of Fisheries and Oceans Canada (DFO) surveys: 95,543 (CV=0.31; Table 1). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CVs pooled using a delta method to produce a species abundance estimate for the stock area. A key uncertainty in the population size estimate is the precision and accuracy of the availability bias correction factor that was applied. More information on the spatio-temporal variability of the animals' dive profile is needed.

Earlier Abundance Estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. As recommended in the GAMMS II Workshop Report (Wade and Angliss 1997), estimates older than eight years are deemed unreliable for the determination of the current PBR.

Recent Surveys andAabundance Estimates

An abundance estimate of 75,079 (CV=0.38) harbor porpoises was generated from a U.S. shipboard and aerial survey conducted during 27 June–28 September 2016 (Table 1; Palka 2020) in a region covering 425,192 km^2. The aerial portion included 11,782 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, throughout the U.S. waters. The shipboard portion included 4,351 km of tracklines that were in waters offshore of central Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the outer limit of the U.S. EEZ). Both sighting platforms used a two-team data collection procedure, which allows estimation of abundance to correct for perception bias of the detected species (Laake and Borchers 2004). The estimates were also corrected for availability bias.

An abundance estimate of 20,464 (CV=0.39) harbor porpoises from the Canadian Bay of Fundy/Scotian shelf region was generated from an aerial survey conducted by the Department of Fisheries and Oceans, Canada (DFO). The entire survey covered Atlantic Canadian shelf and shelf break waters extending from the northern tip of Labrador to the U.S border off southern Nova Scotia in August and September of 2016 (Lawson and Gosselin 2018). A total of 29,123 km were flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf strata using two Cessna Skymaster 337s and 21,037 km were flown over the Newfound/Labrador strata using a DeHavilland Twin Otter. The harbor porpoise estimate was derived from the Skymaster data using single team multi-covariate distance sampling with left truncation (to accommodate the obscured area under the plane) where size-bias was also investigated. The Otter-based perception bias correction, which used double platform mark-recapture methods, was applied. An availability bias correction factor, which was based on published records of the cetaceans' surface intervals, was also applied.

Table 1. Summary of recent abundance estimates for the Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena) by month, year, and area covered during each abundance survey and the resulting abundance estimate (N_{est}) and coefficient of variation (CV). The estimate considered best in in bold font.

Month/Year	Area	N_{est}	CV
Jun–Sep 2016	Central Virginia to Maine	75,079	0.38
Aug–Sep 2016	Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf	20,464	0.39
Jun–Sep 2016	**Central Virginia to Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf — COMBINED**	**95,543**	**0.31**

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for harbor porpoises is 95,543 (CV=0.31). The minimum population estimate for the Gulf of Maine/Bay of Fundy harbor porpoise is 74,034.

Current Population Trend

A trend analysis has not been conducted for this stock. The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long survey interval. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV > 0.30) remains below 80% (alpha = 0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). There is current work to standardize the strata-specific previous abundance estimates to consistently represent the same regions and include appropriate corrections for perception and availability bias. These standardized abundance estimates will be used in state-space trend models that incorporate environmental factors that could potentially influence the process and observational errors for each stratum.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Several attempts have been made to estimate potential population growth rates. Barlow and Boveng (1991), who used a re-scaled human life table, estimated the upper bound of the annual potential growth rate to be 9.4%. Woodley and Read (1991) used a re-scaled Himalayan tahr life table to estimate a likely annual growth rate of 4%. In an attempt to estimate a potential population growth rate that incorporates many of the uncertainties in survivorship and reproduction, Caswell *et al.* (1998) used a Monte Carlo method to calculate a probability distribution of growth rates. The median potential annual rate of increase was approximately 10%, with a 90% confidence interval of 3–15%. This analysis underscored the considerable uncertainty that exists regarding the potential rate of increase in this population. Moore and Read (2008) conducted a Bayesian population modeling analysis to estimate the potential population growth of harbor porpoise in the absence of bycatch mortality. Their method used fertility data, in combination with age-at-death data from stranded animals and animals taken in gillnets, and was applied under two scenarios to correct for possible data bias associated with observed bycatch of calves. Demographic parameter estimates were ‘model averaged’ across these scenarios. The Bayesian posterior median estimate for potential natural growth rate was 0.046. This last, most recent, value will be the one used for the purpose of this assessment.

Key uncertainties in the estimate of the maximum net productivity rate for this stock were discussed in Moore and Read (2008), which included the assumption that the age structure is stable, and the lack of data to estimate the probability of survivorship to maximum age. The authors considered the effects of these uncertainties on the estimated potential natural growth rate to be minimal.

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 74,034. The maximum productivity rate is 0.046. The recovery factor is 0.5 because stock's status relative to OSP is unknown and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the Gulf of Maine/Bay of Fundy harbor porpoise is 851.

Table 2. Best and minimum abundance estimates for the Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
95,543	0.31	74,034	0.5	0.046	851

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual estimated average human-caused mortality and serious injury is 150 harbor porpoises per year (CV=0.15) from U.S. fisheries using observer data. Canadian bycatch information is not available.

Table 3. Total annual estimated average human-caused mortality and serious injury for the Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena).

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	150	0.14

A key uncertainty is the potential that the observer coverage in the mid-Atlantic gillnet may not be representative of the fishery during all times and places, since the observer coverage was relatively low for some times and areas, 0.02–0.10. The effect of this is unknown. Another key uncertainty is that mortalities and serious injuries in Canadian waters are largely unquantified. There are no major known sources of unquantifiable human-caused mortality or serious injury for the U.S. waters within the Gulf of Maine/Bay of Fundy harbor porpoise stock's habitat.

United States

Northeast Sink Gillnet

Harbor porpoise bycatch in the northern Gulf of Maine occurs primarily from June to September, while in the southern Gulf of Maine and south of New England, bycatch occurs from January to May and September to December. Annual bycatch is estimated using ratio estimator techniques that account for the use of pingers (Hatch and Orphanides 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

Harbor porpoise bycatch in mid-Atlantic waters occurs primarily from December to May in waters off New Jersey and less frequently in other waters ranging farther south, from New Jersey to North Carolina. Annual bycatch is estimated using ratio estimator techniques (Hatch and Orphanides 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Bottom Trawl

Since 1989, harbor porpoise mortalities have been observed in the northeast bottom trawl fishery, but many of these were not attributable to this fishery because decomposed animals are presumed to have been dead prior to being taken by the trawl. Those infrequently caught freshly dead harbor porpoises have been caught during January to April on Georges Bank or in the southern Gulf of Maine. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos *et al.* 2020). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Canada

No current estimates exist, but harbor porpoise interactions have been documented in the Bay of Fundy sink gillnet fishery and in herring weirs between the years 1998–2001 in the lower Bay of Fundy demersal gillnet fishery (Trippel and Shepherd 2004). That fishery has declined since 2001 and it is assumed bycatch is very small, if any (H. Stone, Department of Fisheries and Oceans Canada, pers. comm.).

Table 4. From observer program data, summary of the incidental mortality of Gulf of Maine/Bay of Fundy harbor porpoise (Phocoena phocoena phocoena) by commercial fishery including the years sampled, the type of data used, the annual observer coverage, the mortalities and serious injuries recorded by on-board observers, the estimated annual serious injury and mortality, the estimated CV of the annual mortality, and the mean annual combined mortality (CV in parentheses).

Fishery	Years	Data Type [a]	Observer Coverage [b]	Obs. Serious Injury[c]	Obs. Mortality	Est. Serious Injury[c]	Est. Mort.	Est. Combined Mortality	Est. CVs	Mean Combined Annual Mortality
Northeast Sink Gillnet	2014 2015 2016 2017 2018	Obs. Data, Trip Logbook, Allocated Dealer Data	0.18 0.14 0.10 0.12 0.11	0 0 0 1 0	28 23 11 18 9	0 0 0 7 0	128 177 125 129 92	128 177 125 136 92	0.27 0.28 0.34 0.28 0.52	132 (0.15)
Mid-Atlantic Gillnet	2014 2015 2016 2017 2018	Obs. Data, Weighout	0.05 0.06 0.08 0.09 0.09	0 0 0 0 0	1 2 2 1 0	0 0 0 0 0	22 33 23 9.1 0	22 33 23 9.1 0	1.03 1.16 0.64 0.95 0	17 (0.55)
Northeast Bottom Trawl	2014 2015 2016 2017 2018	Obs. Data, Weighout	0.19 0.19 0.12 0.12 0.12	0 0 0 0 0	4 0 0 0 0	0 0 0 0 0	5.5 0 0 0 0	5.5 0 0 0 0	0.86 0 0 0 0	1.1 (0.86)
Total										**150 (0.14)**

a Observer data (Obs. Data) are used to measure bycatch rates and the data are collected within the Northeast Fisheries Observer Program. NEFSC collects Weighout (Weighout) landings data that are used as a measure of total effort for the U.S. gillnet fisheries. Mandatory vessel trip report (VTR; Trip Logbook) data are used to determine the spatial distribution of fishing effort in the Northeast sink gillnet fishery.
b Observer coverage for the U.S. Northeast and mid-Atlantic coastal gillnet fisheries is based on tons of fish landed. Northeast bottom trawl fishery coverages are ratios based on trips.
c Serious injuries were evaluated for the period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2021).

Other Mortality

United States

There is evidence that harbor porpoises were harvested by natives in Maine and Canada before the 1960s, and the meat was used for human consumption, oil, and fish bait (NMFS 1992). The extent of these past harvests is unknown, though it is believed to have been small. Up until the early 1980s, small kills by native hunters (Passamaquoddy Indians) were reported. It was believed to have nearly stopped (Polacheck 1989) until media reports in September 1997 depicted a Passamaquoddy tribe member dressing out a harbor porpoise. Recent harbor porpoise strandings on the U.S. Atlantic coast are documented in Table 5 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 20 November 2019).

Stranding data underestimate the extent of fishery-related mortality and serious injury because all of the marine mammals that die or are seriously injured may not wash ashore, nor will all of those that do wash ashore necessarily show signs of entanglement or other fishery-interaction. Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of fishery interaction.

Table 5. Harbor porpoise (Phocoena phocoena phocoena) reported strandings along the U.S. and Canadian Atlantic coast, 2014–2018.

Area	2014	2015	2016	2017	2018	Total
Maine[a, b, c, e]	5	2	5	8	8	28
New Hampshire	1	0	1	2	2	6
Massachusetts[a, b, c, e, f]	22	18	8	29	13	90
Rhode Island[c, d, e]	0	2	2	0	0	4
New York[a, b, e]	1	3	1	12	2	19
New Jersey[a, b, e]	4	2	5	14	5	30
Delaware	0	0	0	6	0	6
Maryland	0	0	0	2	0	2
Virginia[b, d]	3	3	2	5	1	14
North Carolina[c]	11	14	1	1	3	30
TOTAL U.S.	47	44	25	79	34	229
Nova Scotia/Prince Edward Island[g]	9	13	16	22	20	81
Newfoundland and New Brunswick[h]	0	2	0	0	0	5
Total	**56**	**59**	**41**	**101**	**54**	**315**

a. In 2016, one animal in Maine and one animal in New Jersey were responded to and released alive. Ten animals were released alive in 2017, 6 of them in Massachusetts, 2 in Maine and 2 in New York.
b. Five total HI cases in 2014: 2 in Maine, 1 each in Massachusetts, New Jersey and Virginia. The Virginia case was recorded as a fishery interaction.
c. Two HI cases in 2015: 1 in Rhode Island and 1 in North Carolina
d. Two HI cases in 2016: 1 in Rhode Island and 1 in Virginia. The Virginia case was coded as a fishery interaction.
e. Seven HI cases in 2017: 2 in Maine were released alive and another was a neonate with an infected laceration that required euthanization. One dead HI animal in Massachusetts was coded as a fishery interaction and another HI animal was released alive. One HI animal in New York was released alive and one dead animal in New Jersey had evidence of vessel interaction.
f. Two HI cases in 2018; both in Massachusetts. One was coded as a fishery interaction.
g. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.). Not included in count for 2014 are at least 8 animals released alive from weirs. One of the 2015 animals a suspected fishery interaction.
h. (Ledwell and Huntington 2014, 2015, 2017, 2018)

Canada

Whales and dolphins stranded on the coast of Nova Scotia, New Brunswick and Prince Edward Island are recorded by the Marine Animal Response Society and the Nova Scotia Stranding Network. See Table 3 for details.

Harbor porpoises stranded on the coasts of Newfoundland and Labrador are reported by the Newfoundland and Labrador Whale Release and Strandings Program (Ledwell and Huntington 2014, 2015, 2017, 2018, 2019; Table 5).

HABITAT ISSUES

In U.S. waters, harbor porpoise are mostly found in nearshore areas and inland waters, including bays, tidal areas, and river mouths. As a result, in addition to fishery bycatch, harbor porpoise are vulnerable to contaminants, such as PCBs (Hall *et al.* 2006), ship traffic (Oakley *et al.* 2017, Terhune 2015) and physical modifications resulting from urban and industrial development activities such as construction of docks and other over-water structures, dredging (Todd *et al.* 2015), installation of offshore windfarms (Carstensen *et al.* 2006, Brandt *et al.* 2011, Teilmann and Carstensen 2012, Dähne *et al.* 2013, Benjamins *et al.* 2017), seismic surveys and other sources of anthropogenic noise (Lucke *et al.* 2009).

Climate-related changes in spatial distribution and abundance, including poleward and depth shifts, have been documented in and predicted for a range of plankton species and commercially important fish stocks (Nye *et al*. 2009, Head *et al*. 2010, Pinsky *et al*. 2013, Poloczanska *et al.* 2013, Hare *et al*. 2016, Grieve *et al.* 2017, Morley *et al.* 2018) and cetacean species (e.g., MacLeod 2009; Sousa *et al.* 2019). There is uncertainty in how, if at all, the distribution and population size of this species will respond to these changes and how the ecological shifts will affect human impacts to the species.

STATUS OF STOCK

Harbor porpoise in the Gulf of Maine/Bay of Fundy stock are not listed as threatened or endangered under the Endangered Species Act, and this stock is not considered strategic under the MMPA. The total U.S. fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The status of harbor porpoises, relative to OSP, in the U.S. Atlantic EEZ is unknown. Population trends for this species have not been investigated.

REFERENCES CITED

Barlow, J. and P. Boveng. 1991. Modeling age-specific mortality for marine mammal populations. Mar. Mamm. Sci. 7:50–65.

Benjamins, S., N. van Geel, G. Hastie, J. Elliott and B. Wilson. 2017. Harbour porpoise distribution can vary at small spatiotemporal scales in energetic habitats. Deep-Sea Res. II. 141:191–202. https://doi.org/10.1016/j.dsr2.2016.07.002

Brandt, M.J., A. Diederichs, K. Betke and G. Nehls. 2011. Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Mar. Ecol. Prog. Ser. 421:205–216. https://doi.org/10.3354/meps08888

Carstensen, J., O.D. Henriksen and J. Teilmann. 2006. Impacts of offshore wind farm construction on harbour porpoises: Acoustic monitoring of echolocation activity using porpoise detectors (T-PODs). Mar Ecol Prog Ser. 321:295–308.

Caswell, H., S. Brault, A.J. Read and T.D. Smith. 1998. Harbor porpoise and fisheries: An uncertainty analysis of incidental mortality. Ecol. Appl. 8(4):1226–1238.

Dähne, M., A. Gilles, K. Lucke, V. Peschko, S. Adler, K. Krugel, J. Sundermeyer and U. Siebert. 2013. Effects of pile-driving on harbour porpoises (*Phocoena phocoena*) at the first offshore wind farm in Germany. Environ. Res. Lett. 8:1–15.

Gaskin, D.E. 1977. Harbour porpoise, *Phocoena phocoena* (L.), in the western approaches to the Bay of Fundy 1969–75. Rep. Int. Whal. Comm. 27:487–492.

Gaskin, D.E. 1984. The harbor porpoise *Phocoena phocoena* (L.): Regional populations, status, and information on direct and indirect catches. Rep. Int. Whal. Comm. 34:569–586.

Gaskin, D.E. 1992. The status of the harbour porpoise. Can. Field-Nat. 106:36–54.

Grieve, B.D., J.A. Hare and V.S. Saba. 2017. Projecting the effects of climate change on *Calanus finmarchicus* distribution within the US Northeast continental shelf. Sci. Rep. 7:6264.

Hall, A.J., K. Hugunin, R. Deaville, R.J. Law, C.R. Allchin and P.D. Jepson. 2006. The risk of infection from polychlorinates biphenyl exposure in the harbor porpoise (*Phocoena phocoena*): A case-control approach. Environmental Health Perpectives. 114(5):704–711.

Hare, J.A., W.E. Morrison, M.W. Nelson, M.M. Stachura, E.J. Teeters, R.B. Griffis, M.A. Alexander, J.D. Scott, L. Alade, R.J. Bell, A.S. Chute, K.L. Curti, T.H. Curtis, D. Kurcheis, J.F. Kocik, S.M. Lucey, C.T. McCandless, L.M. Milke, D.E. Richardson, E. Robillard, H.J. Walsh, M.C. McManus, K.E. Maranick and C.A. Griswold. 2016. A vulnerability assessment of fish and invertebrates to climate change on the Northeast U.S. continental shelf.

Hatch, J.M. and C.D. Orphanides. 2016. Estimates of cetacean and pinniped bycatch in the 2014 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 16-05. 22pp.

Head, E.J.H. and P. Pepin. 2010. Spatial and inter-decadal variability in plankton abundance and composition in the Northwest Atlantic (1958–2006). J. Plankton Res. 32:1633–1648.

Hiltunen, K.H. 2006. Mixed-stock analysis of harbor porpoises (*Phocoena phocoena*) along the U.S. mid-Atlantic coast using microsatellite DNA markers. MS thesis. The College of Charleston, Charleston, SC. 92pp.

Johnston, D.W. 1995. Spatial and temporal differences in heavy metal concentrations in the tissues of harbour porpoises (*Phocoena phocoena* L.) from the western North Atlantic. M.S. thesis. University of Guelph, Guelph, Ontario, Canada. 152pp.

Josephson, E., F. Wenzel and M.C. Lyssikatos. 2021. Serious injury determinations for small cetaceans and pinnipeds caught in commercial fisheries off the northeast U.S. coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-04. 33pp.

Kraus, S.D., J.H. Prescott and G.S. Stone. 1983. Harbor porpoise, *Phocoena phocoena*, in the U.S. coastal waters off the Gulf of Maine: A survey to determine seasonal distribution and abundance. NMFS. NA82FAC00027. 22pp.

Kraus, S.D., A.J. Read, A. Solow, K. Baldwin, T. Spradlin, E. Anderson and J. Williamson. 1997. Acoustic alarms reduce porpoise mortality. Nature. 388(6642):525.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108-189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Lawson J. and J-F. Gosselin. 2018. Estimates of cetacean abundance from the 2016 NAISS aerial surveys of eastern Canadian waters, with a comparison to estimates from the 2007 TNASS. NAMMCO SC/25/AE/09.

Ledwell, W. and J. Huntington. 2014. Incidental entrapments and entanglements of cetaceans and leatherback sea turtles, strandings, ice entrapments reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2014. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 23pp.

Ledwell, W. and J. Huntington. 2015. Incidental entrapments and entanglements of cetaceans and leatherback sea turtles, strandings, ice entrapments reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2015. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 22pp.

Ledwell, W. and J. Huntington. 2017. Incidental entrapments and entanglements of cetaceans and leatherback sea turtles and strandings and harassments reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2016. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 23pp.

Ledwell, W. and J. Huntington. 2018. Incidental entrapments and entanglements of cetaceans and leatherback sea turtles, strandings, ice entrapments reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a summary of the Whale Release and Strandings program during 2017. Report to the Department of Fisheries and Oceans Canada, St. John's, Newfoundland, Canada. 24pp.

Lucke K., U. Siebert, P.A. Lepper and M-A. Blanchet. 2009. Temporary shift in masking hearing thresholds in a harbor porpoise (*Phocoena phocoena*) after exposure to seismic airgun stimuli. J. Acoust. Soc. Am. 125(6):4060–4070.

Lyssikatos, M.C., S. Chavez-Rosales and J. Hatch. 2020. Estimates of cetacean and pinniped bycatch in Northeast and mid-Atlantic bottom trawl fisheries, 2013–2017. Northeast Fish. Sci. Cent. Ref. Doc. 20-04. 11pp.

MacLeod, C.D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: A review and synthesis. Endang. Species Res. 7:125–136.

Moore, J.E. and A.J. Read. 2008. A Bayesian uncertainty analysis of cetacean demography and bycatch mortality using age-at-death data. Ecol. Appl. 18(8):1914–1931.

Morley, J.W., R.L. Selden, R.J. Latour, T.L. Frölicher, R.J. Seagraves and M.L. Pinsky. 2018. Projecting shifts in thermal habitat for 686 species on the North American continental shelf. PLoS ONE 13(5):e0196127. https://doi.org/10.1371/journal.pone.0196127

NMFS [National Marine Fisheries Service]. 1992. Harbor porpoise in Eastern North America: Status and Research Needs. Results of a scientific workshop held May 5–8, 1992 at NEFSC, Woods Hole, MA, USA. Northeast Fish. Sci. Cent. Ref. Doc. 92-06. National Marine Fisheries Service. 28pp.

Nye, J., J. Link, J. Hare and W. Overholtz. 2009. Changing spatial distribution of fish stocks in relation to climate and population size on the Northeast United States continental shelf. Mar. Ecol. Prog. Ser. 393:111–129.

Oakley, J.A., A.T. Williams and T. Thomas. 2017. Reactions of harbor porpoise (*Phocoena phocoena*) to vessel traffic in the coastal waters of South West Wales, UK. Ocean & Coastal Management 138:158–169.

Orphanides, C.D. 2019. Estimates of cetacean and pinniped bycatch during 2016 in the New England and mid-Atlantic sink gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 19-04. 12pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Orphanides, C.D. 2020. Estimates of cetacean and pinniped bycatch during 2017 in the New England and mid-Atlantic sink gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 20-03. 16pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Orphanides, C.D. 2021. Estimates of cetacean and pinniped bycatch during 2018 in the New England and mid-Atlantic sink gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 21-01.

Orphanides, C.D. and J. Hatch. 2017. Estimates of cetacean and pinniped bycatch during 2015 in the New England and mid-Atlantic sink gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 17-18. 21pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Palka, D. 2020. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2016 line transect surveys conducted by the Northeast Fisheries Science Center. Northeast Fish. Sci. Cent. Ref. Doc. 20-05.

Palka, D. 1995. Influences on spatial patterns of Gulf of Maine harbor porpoises. Pages 69–75 *in*: A.S. Blix, L. Walloe and O. Ulltang (Eds.). Whales, Seals, Fish and Man. Elsevier Science. Amsterdam.

Palka, D.L., A.J. Read, A.J. Westgate and D.W. Johnston. 1996. Summary of current knowledge of harbour porpoises in US and Canadian Atlantic waters. Rep. Int. Whal. Comm. 46:559–565.

Palka, D. 2000. Abundance of the Gulf of Maine/Bay of Fundy harbor porpoise based on shipboard and aerial surveys during 1999. Northeast Fish. Sci. Cent. Ref. Doc. 00-07. 29pp. Available from: https://repository.library.noaa.gov/view/noaa/3290

Palka, D.L. 2012. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2011 line transect survey. Northeast Fish. Sci. Cent. Ref. Doc. 12-29. 37pp. Available from: https://repository.library.noaa.gov/view/noaa/4312

Pinsky, M.L., B. Worm, M.J. Fogarty, J.L. Sarmiento and S.A. Levin. 2013. Marine taxa track local climate velocities. Science. 341:1239–1242.

Polacheck, T. 1989. Harbor porpoises and the gillnet fishery. Oceanus. 32(1):63–70.

Poloczanska, E.S., C.J. Brown, W.J. Sydeman, W. Kiessling, D.S. Schoeman, P.J. Moore, K. Brander, J.F. Bruno, L.B. Buckley, M.T. Burrows, C.M. Duarte, B.S. Halpern, J. Holding, C.V. Kappel, M.I. O'Connor, J.M. Pandolfi, C. Parmesan, F. Schwing, S.A. Thompson and A.J. Richardson. 2013. Global imprint of climate change on marine life. Nat. Clim. Change 3:919–925.

Read, A.J. and A.A. Hohn. 1995. Life in the fast lane: The life history of harbour porpoises from the Gulf of Maine. Mar. Mamm. Sci. 11(4):423–440.

Read, A.J. and A.J. Westgate. 1997. Monitoring the movements of harbour porpoises (*Phocoena phocoena*) with satellite telemetry. Mar. Biol. 130:315–22.

Rosel, P.E., S.C. France, J.Y. Wang and T.D. Kocher. 1999a. Genetic structure of harbour porpoise *Phocoena phocoena* populations in the northwest Atlantic based on mitochondrial and nuclear markers. Mol. Ecol. 8:S41–S54.

Rosel, P.E., R. Tiedemann and M. Walton. 1999b. Genetic evidence for limited trans-Atlantic movements of the harbor porpoise *Phocoena phocoena*. Mar. Biol. 133:583–591.

Sousa, A., F. Alves, A. Dinis, J. Bentz, M.J. Cruz and J.P. Nunes. 2019. How vulnerable are cetaceans to climate change? Developing and testing a new index. Ecol. Indic. 98:9–18.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Teilmann, J. and J. Carstensen. 2012. Negative long term impacts on harbor porpoises from a large scale offshore wind farm in the Baltic—evidence of a slow recovery. Environ, Res, Lett. 7:045101.

Terhune, J.M. 2015. Harbour porpoise presence near oil tankers. J. Canadian Ac. 43(3). 2pp.

Thomas L., J.L. Laake, E. Rexstad, S. Strindberg, F.F.C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, M.L. Burt, S.L. Hedley, J.H. Pollard, J.R.B. Bishop and T.A. Marques. 2009. Distance 6.0. Release 2. [Internet]. University of St. Andrews (UK): Research Unit for Wildlife Population Assessment. Available from: http://distancesampling.org/Distance/

Todd, V.L.G., I.B. Todd, J.C. Gardiner, E.C.N. Morrin, N.A. MacPherson, N.A. DiMarzio and F. Thomsen. 2015. A review of impacts of marine dredging activities on marine mammals. ICES J. of Mar. Sci. 72(2):328–340.

Trippel, E.A. and T.D. Shepherd. 2004. By-catch of harbour porpoise (*Phocoena phocoena*) in the Lower Bay of Fundy gillnet fishery from 1998–2001. Department of Fisheries and Oceans. Ottawa, Ontario. DFO Research Document 2004/2521 iv+33pp. Available from: http://www.fmap.ca/ramweb/papers-total/Trippel_Shepherd_2004.pdf

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp. Available from: https://repository.library.noaa.gov/view/noaa/15963

Wang, J.Y., D.E. Gaskin and B.N. White 1996. Mitochondrial DNA analysis of harbour porpoise, *Phocoena phocoena*, subpopulations in North American waters. Can. J. Fish. Aquat. Sci 53:1632–45.

Westgate, A.J., D.C.G. Muir, D.E. Gaskin and M.C.S. Kingsley. 1997. Concentrations and accumulation patterns of organochlorine contaminants in the blubber of harbour porpoises, *Phocoena phocoena*, from the coast of Newfoundland, the Gulf of St. Lawrence and the Bay of Fundy/Gulf of Maine. Envir. Pollut. 95:105–119.

Westgate, A.J., A.J. Read, T.M. Cox, T.D. Schofield, B.R. Whitaker and K.E. Anderson. 1998. Monitoring a rehabilitated harbor porpoise using satellite telemetry. Mar. Mamm. Sci. 14(3):599–604.

Westgate, A.J. and K.A. Tolley. 1999. Geographical differences in organochlorine contaminants in harbour porpoises *Phocoena phocoena* from the western North Atlantic. Mar. Ecol. Prog. Ser. 177:255–268.

Wingfield J.E., M. O'Brien, V. Lyubchich, J.J. Roberts, P.N. Halpin, A.N. Rice and H. Bailey. 2017. Year-round spatiotemporal distribution of harbor porpoises within and around the Maryland wind energy area. PLoS ONE 12(5):e0176653. https://doi.org/10.1371/journal.pone.0176653

Woodley, T.H. and A.J. Read. 1991. Potential rates of increase of a harbor porpoise (*Phocoena phocoena*) population subjected to incidental mortality in commercial fisheries. Can. J. Fish. Aquat. Sci 48:2429–35.

HARBOR SEAL (*Phoca vitulina vitulina*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The harbor seal (*Phoca vitulina*) is widespread in all nearshore waters of the North Atlantic and North Pacific Oceans and adjoining seas above about 30ºN (Burns 2009, Desportes *et al*. 2010).

Harbor seals are year-round inhabitants of the coastal waters of eastern Canada and Maine (Katona *et al*. 1993), and occur seasonally along the coasts from southern New England to Virginia from September through late May (Schneider and Payne 1983, Schroeder 2000, Rees *et al*. 2016, Toth *et al*. 2018). Scattered sightings and strandings have been recorded as far south as Florida (NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). A general southward movement from the Bay of Fundy to southern New England and mid-Atlantic waters occurs in autumn and early winter (Rosenfeld *et al.* 1988, Whitman and Payne 1990, Jacobs and Terhune 2000). A northward movement to Maine and eastern Canada occurs prior to the pupping season, which takes place from early May through early June primarily along the Maine coast (Gilbert *et al.* 2005, Skinner 2006).

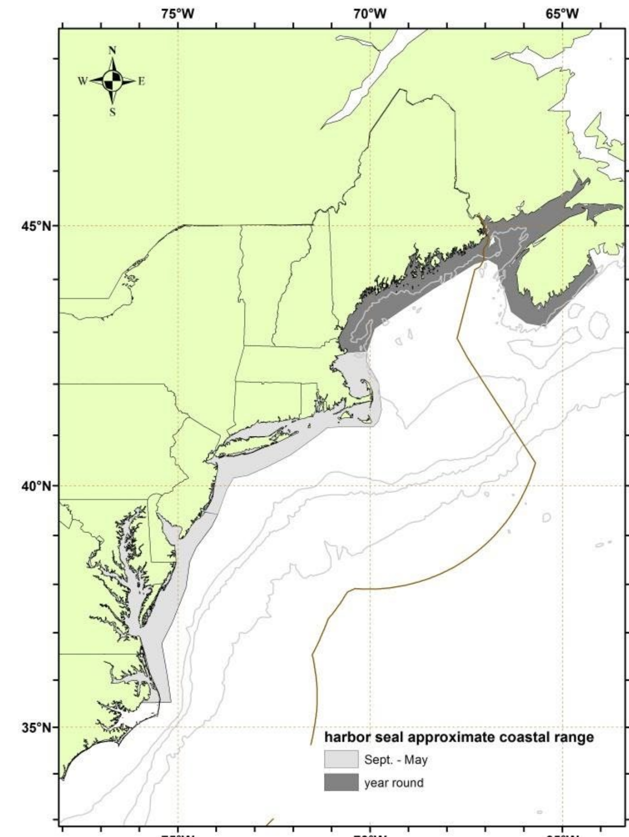


Figure 1. Approximate coastal range of harbor seals. Isobaths are the 100-m, 1000-m, and 4000-m depth contours.

Tagging studies of adult harbor seals demonstrate that adults can make long-distance migrations through the mid-Atlantic and Gulf of Maine (Waring *et al.* 2006, Jones *et al.* 2018). Prior to these studies, it was believed that the majority of seals moving into southern New England and mid-Atlantic waters were subadults and juveniles (Whitman and Payne 1990, Katona *et al.* 1993). The more recent studies demonstrate that various age classes utilize habitat along the eastern seaboard throughout the year. Reconnaissance flights for pupping south of Maine would help confirm the extent of the current pupping range.

Although the stock structure of western North Atlantic harbor seals is unknown, it is thought that harbor seals found along the eastern U.S. and Canadian coasts represent one population (Temte *et al.* 1991, Andersen and Olsen 2010). However, uncertainty in the single stock designation is suggested by multiple sources, both in this population and by inference from other populations. Stanley *et al.* (1996) demonstrated some genetic differentiation in Atlantic Canada harbor seal samples. Gilbert *et al.* (2005) noted regional differences in pup count trends along the coast of Maine. Goodman (1998) observed high degrees of philopatry in eastern North Atlantic populations. In addition, multiple lines of evidence have suggested fine-scaled sub-structure in Northeast Pacific harbor seals (Westlake and O'Corry-Crowe 2002, O'Corry-Crowe *et al.* 2003, Huber *et al*. 2010).

POPULATION SIZE

The best current abundance estimate of harbor seals is 75,834 (CV=0.15) which is from a 2012 survey (Waring *et al.* 2015). Aerial photographic surveys and radio tracking of harbor seals on ledges along the Maine coast were conducted during the pupping period in late May 2012. Twenty-nine harbor seals (20 adults and nine juveniles) were captured and radio-tagged prior to the aerial survey. Of these, 18 animals were available during the survey to develop a correction factor for the fraction of seals not observed. A key uncertainty is that the area from which the samples were drawn in 2012 may not have included the area the entire population occupied in late May and early June. Additionally, since the most current estimate dates from a survey done in 2012, the ability for that estimate to accurately represent the present population size has become increasingly uncertain. A population survey was conducted in 2018 to provide updated abundance estimates and these data are being analyzed.

Table 1. Summary of recent abundance estimates for the western North Atlantic harbor seal (Phoca vitulina vitulina) by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{est}) and coefficient of variation (CV).

Month/Year	Area	N_{est}	CV
May/June 2012	Maine coast	75,834	0.15

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). The best estimate of abundance for harbor seals is 75,834 (CV=0.15). The minimum population estimate is 66,884 based on corrected available counts along the Maine coast in 2012.

Current Population Trend

A trend analysis is currently underway using the 2018 survey data combined with historical data, but the results are not yet available. There are some lines of evidence that support an apparent decline in abundance and/or changing distributions. In 2001, the population was estimated to be 99,340 (95%CI: 83,118–121,397; Gilbert *et al.* 2005). While the estimated population size was lower in 2012, Waring *et al.* (2015) did not consider the population to be declining because the 2012 and 2001 estimates were not significantly different and there was uncertainty over whether some fraction of the population was not in the survey area. In southeastern Massachusetts, counts of harbor seals progressively declined after 2009 (Pace *et al.* 2019), and reduced population size has been hypothesized from declining rates of stranded and bycaught animals (Johnston *et al.* 2015). However, the occupancy patterns of harbor seals at haul-out sites has also changed through time in relation to the growth of the sympatric gray seal population (Pace *et al.* 2019), so inferences about abundance could reflect a sampling and monitoring plan that needs to be revisited. If juvenile seals are redistributing to new areas they may be missed during population surveys, designed around historical pupping habitat. This may have explained differences in the estimated size of the population between 2001 and 2012 (Waring *et al.* 2015).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.12. This value is based on theoretical modeling showing that pinniped populations may not grow at rates much greater than 12% given the constraints of their reproductive life history (Barlow *et al.* 1995). Key uncertainties about the maximum net productivity rate are due to the limited understanding of the stock-specific life history parameters; thus the default value was used.

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 66,884 animals. The maximum productivity rate is 0.12, the default value for pinnipeds. The recovery factor (F_r) is 0.5, the default value for stocks of unknown status relative to optimum sustainable population (OSP) and with the CV of the average mortality estimate less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic stock of harbor seals is 2,006.

Table 2. Best and minimum abundance estimates for the Western North Atlantic harbor seal (Phoca vitulina vitulina) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
75,834	0.15	66,883	0.5	0.12	2,006

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period from 2014–2018, the average annual estimated human-caused mortality and serious injury to harbor seals in the U.S. is 365 (Table 3). Mortality in U.S. fisheries is explained in further detail below.

Table 3. The total annual estimated average human-caused mortality and serious injury for the Western North Atlantic harbor seal (Phoca vitulina vitulina).

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	351	0.12
2014–2018	Non-fishery human interaction stranding mortalities	14.2	
2014–2018	Research mortalities	0	
Total		**365.2**	

Fishery Information

Detailed fishery information is given in Appendix III.

United States

Northeast Sink Gillnet

The Northeast sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2014–2018 was 51 animals, and the average annual total mortality was 319 (CV=0.13; Hatch and Orphanides 2015, 2016; Orphanides and Hatch 2017; Orphanides 2019*,* 2020, 2021; Josephson *et al.* 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery is a Category I fishery. The average annual observed mortality from 2014–2018 was two animals, and the average annual total mortality was 23 (CV=0.34; Hatch and Orphanides 2015, 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021; Josephson *et al.* 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and see Appendix V for historical bycatch information.

Northeast Bottom Trawl

The Northeast bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2014–2018 was <1 animal, and the average annual total mortality was four (CV=0.54; Lyssikatos *et al*. 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and see Appendix V for historical bycatch information.

Mid-Atlantic Bottom Trawl

The mid-Atlantic bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2014–2018 was <1 animal, and the average annual total mortality was five (CV=0.57; Lyssikatos *et al*. 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and see Appendix V for historical bycatch information.

Northeast Mid-water Trawl Fishery (Including Pair Trawl)

The Northeast mid-water and pair trawl fisheries are Category II fisheries. The average annual observed mortality from 2014–2018 was <1 animal. An expanded bycatch estimate has not been calculated for the current 5-year period. See Table 4 for observed mortality and serious injury during the current 5-year period, and Appendix V for historical bycatch information.

Gulf of Maine Atlantic Herring Purse Seine Fishery

No mortalities have been observed in this fishery, and no harbor seals were captured and released alive in 2014–2018.

Canada

Currently, scant data are available on bycatch in Atlantic Canada fisheries due to limited observer programs (Baird 2001). An unknown number of harbor seals have been taken in Newfoundland, Labrador, Gulf of St. Lawrence and Bay of Fundy groundfish gillnets; Atlantic Canada and Greenland salmon gillnets; Atlantic Canada cod traps; and in Bay of Fundy herring weirs (Read 1994, Cairns *et al.* 2000). Furthermore, some of these mortalities (e.g., seals trapped in herring weirs) are the result of direct shooting under nuisance permits.

Table 4. Summary of the incidental mortality of harbor seals (Phoca vitulina vitulina) by commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the mortalities recorded by on-board observers (Observed Mortality), the estimated annual mortality (Estimated Mortality), the estimated CV of the annual mortality (Estimated CVs) and the mean annual mortality (CV in parentheses).

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury[c]	Observed Mortality	Est. Serious Injury	Est. Mortality	Est. Comb. Mortality	Est. CVs	Mean Annual Combined Mortality
Northeast Sink Gillnet	2014	Obs. Data, Weighout, Logbook	0.18	0	59	0	390	390	0.39	319 (0.13)
	2015		0.14	0	87	0	474	474	0.17	
	2016		0.10	0	36	0	245	245	0.29	
	2017		0.12	0	63	0	298	298	0.18	
	2018		0.11	0	22	0	188	188	0.36	
Mid-Atlantic Gillnet	2014	Obs. Data, Weighout	0.05	0	1	0	19	19	1.06	23 (0.34)
	2015		0.06	0	5	0	48	48	0.52	
	2016		0.08	0	2	0	18	18	0.95	
	2017		0.09	0	1	0	3	3	0.62	
	2018		0.09	0	3	0	26	26	0.52	
Northeast Bottom Trawl	2014	Obs. Data, Weighout	0.19	0	4	0	19	19	0.63	3.8 (0.54)
	2015		0.19	0	4	0	23	23	0	
	2016		0.12	0	0	0	0	0	0	
	2017		0.12	0	2	0	16	16	0.96	
	2018		0.12	0	5	0	32	32	0.52	
Mid-Atlantic Bottom Trawl	2014	Obs. Data, Dealer	0.09	0	2	0	10	10	0.95	4.6 (0.57)
	2015		0.09	0	1	0	7	7	1	
	2016		0.10	0	0	0	0	0	0	
	2017		0.14	0	0	0	0	0	0	
	2018		0.12	0	1	0	6	6	0.94	
Northeast Mid-water Trawl – Incl. Pair Trawl	2014	Obs. Data, Weighout, Trip Logbook	0.42	0	1	0	na	na	na	0.8 (na)
	2015		0.08	0	2	0	na	na	na	
	2016		0.27	0	1	0	na	na	na	
	2017		0.16	0	0	0	0	0	0	
	2018		0.14	0	0	0	0	0	0	
Total										**351 (0.12)**

a. Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program (NEFOP). NEFSC collects landings data (Weighout), and total landings are used as a measure of total effort for the sink gillnet fishery. Mandatory logbook (Logbook) data are used to determine the spatial distribution of fishing effort in the northeast sink gillnet fishery.
b. The observer coverages for the northeast sink gillnet fishery and the mid-Atlantic gillnet fisheries are ratios based on tons of fish landed and coverages for the bottom and mid-water trawl fisheries are ratios based on trips. Total observer coverage reported for bottom trawl gear and gillnet gear in the years 2014–2018 includes samples collected from traditional fisheries observers in addition to fishery monitors through NEFOP.
c. Serious injuries were evaluated for the 2014–2018 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2021).

Other Mortality

United States

Historically, harbor seals were bounty-hunted in New England waters, which may have caused a severe decline of this stock in U.S. waters (Katona *et al.* 1993, Lelli *et al.* 2009). Bounty-hunting ended in the mid-1960s.

Harbor seals strand each year throughout their migratory range. Stranding data provide insight into some of these sources of mortality. Tables 5 and 6 present summaries of harbor seal stranding mortalities as reported to the NOAA National Marine Mammal Health and Stranding Response Database (accessed 20 November 2019). In an analysis of mortality causes of stranded marine mammals on Cape Cod and southeastern Massachusetts between 2000 and 2006, Bogomolni *et al.* (2010) reported that 13% of harbor seal stranding mortalities were attributed to human interaction.

A number of Unusual Mortality Events (UMEs) have affected harbor seals over the past decade. A UME was declared for harbor seals in northern Gulf of Maine waters in 2003 and continued into 2004. No consistent cause of death could be determined. The UME was declared over in spring 2005 (MMC 2006). NMFS declared another UME in the Gulf of Maine in autumn 2006 based on infectious disease. A UME was declared in November of 2011 that involved 567 harbor seal stranding mortalities between June 2011 and October 2012 in Maine, New Hampshire, and Massachusetts. The UME was declared closed in February 2013 (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events). Another UME was declared by NMFS beginning in July 2018 due to increased numbers of harbor and gray seal strandings along the U.S. coasts of Maine, New Hampshire, and Massachusetts. Strandings remained elevated over the summer and the UME area was expanded to include nine states from Maine to Virginia with strandings continuing into 2019. From July to December 2018, 1,100 harbor seals stranded predominantly in Maine, New Hampshire and Massachusetts. The preliminary cause of the UME was attributed to a phocine distemper outbreak (https://www.fisheries.noaa.gov/new-england-mid-atlantic/marine-life-distress/2018-2020-pinniped-unusual-mortality-event-along).

Stobo and Lucas (2000) have documented shark predation as an important source of natural mortality at Sable Island, Nova Scotia. They suggest that shark-inflicted mortality in pups, as a proportion of total production, was less than 10% in 1980–1993, approximately 25% in 1994–1995, and increased to 45% in 1996. Also, shark predation on adults was selective towards mature females. The decline in the Sable Island population appears to result from a combination of shark-inflicted mortality on both pups and adult females and inter-specific competition with the much more abundant gray seal for food resources (Stobo and Lucas 2000, Bowen *et al.* 2003).

Canada

Aquaculture operations in eastern Canada can be licensed to shoot nuisance seals, but the number of seals killed is unknown (Jacobs and Terhune 2000, Baird 2001). Small numbers of harbor seals are taken in subsistence hunting in northern Canada (DFO 2011).

Table 5. Harbor seal (Phoca vitulina vitulina) stranding mortalities along the U.S. Atlantic coast (2014–2018) with subtotals of animals recorded as pups in parentheses.

State	2014	2015	2016	2017	2018	Total
ME	127 (94)	73 (47)	76 (58)	120 (84)	819 (75)	1,215 (344)
NH	38 (22)	56 (43)	45 (27)	26 (20)	113 (60)	278 (171)
MA	58 (15)	81 (24)	55 (19)	78 (29)	204 (58)	476 (145)
RI	7 (1)	8 (0)	5 (1)	9 (3)	9 (0)	38 (5)
CT	0	2 (1)	1 (0)	2 (0)	2 (1)	7 (2)
NY	13 (4)	21 (0)	1 (0)	11 (0)	12 (1)	58 (5)
NJ	2 (1)	9 (4)	4 (0)	9 (3)	14 (2)	38 (10)
DE	3 (0)	1 (0)	1 (1)	1 (0)	2 (1)	8 (2)
MD	2 (0)	0	0	1 (0)	4 (0)	7 (0)
VA	2 (0)	1 (0)	1 (0)	2 (0)	1 (0)	7 (0)
NC	3 (1)	5 (2)	4 (2)	4 (4)	7 (2)	23 (11)
SC	1 (0)	0	0	0	0	1 (0)
Total	**256**	**257**	**193**	**263**	**1,187**	**2,156 (635)**
Unspecified seals (all states)	38	31	13	86	92	260

Table 6. Harbor seal (Phoca vitulina vitulina) human-interaction stranding mortalities along the U.S. Atlantic coast (2014–2018) by type of interaction.

Cause	2014	2015	2016	2017	2018	Total
Fishery Interaction	2	2	3	1	5	13
Boat Strike	2	1	5	3	2	13
Shot	1	0	0	0	0	1
Human Interaction — Other	6	15	8	6	22	57
Total	**11**	**18**	**16**	**10**	**29**	**84**

STATUS OF STOCK

Harbor seals are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The 2014–2018 average annual human-caused mortality and serious injury does not exceed PBR. The status of the western North Atlantic harbor seal stock, relative to OSP, in the U.S. Atlantic EEZ is unknown. Total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate.

REFERENCES CITED

Anderson, L.W. and M.T. Olsen. 2010. Distribution and population structure of North Atlantic harbour seals (*Phoca vitulina*). Pages 173–188 *in*: Harbour Seals of the North Atlantic and the Baltic. NAMMCO Scientific Publications 8.

Baird, R.W. 2001. Status of harbor seals, *Phoca vitulina*, in Canada. Can. Field-Nat. 115:663–675.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Bogomolni, A.L., K.R. Pugliares, S.M. Sharp, K. Patchett, C.T. Harry, J.M. LaRocque, K.M. Touhey and M. Moore. 2010. Mortality trends of stranded marine mammals on Cape Cod and southeastern Massachusetts, USA, 2000 to 2006. Dis. Aq. Org. 88:143–155.

Bowen, W.D., S.L. Ellis, S.J. Iverson and D.J. Boness. 2003. Maternal and newborn life-history traits during periods of contrasting population trends: Implications for explaining the decline of harbour seals (*Phoca vitulina*), on Sable Island. J. Zool. 261:155–163.

Burns, J.J. 2009. Harbor seal and spotted seal (*Phoca vitulina* and *P. largha*). Pages 533–542 *in*: W.F. Perrin, B. Wursig and J.G.M. Thewissen (Eds.). Encyclopedia of marine mammals, second edition. Academic Press, San Diego, CA.

Cairns, D.K., D.M. Keen, P-Y. Daoust, D.J. Gillis and M. Hammill. 2000. Conflicts between seals and fishing gear on Prince Edward Island. Can. Tech. Rep. of Fish. and Aq. Sci. 2333. 39pp.

Desportes G., A. Bjorge, A. Rosing-Asvid and G.T. Waring, eds. 2010. Harbour seals of the North Atlantic and the Baltic. NAMMCO Scientific Publications, vol. 8. North Atlantic Marine Mammal Commission, Tromsø, Norway. 377pp.

DFO [Dept of Fisheries and Oceans]. 2011. 2011–2015 Integrated fisheries management plan for Atlantic seals. Available from: http://www.dfo-mpo.gc.ca/fm-gp/seal-phoque/reports-rapports /mgtplan-planges20112015/mgtplan-planges20112015-eng.htm#c2

Gilbert, J.R., G.T. Waring, K.M. Wynne and N. Guldager. 2005. Changes in abundance and distribution of harbor seals in Maine, 1981–2001. Mar. Mamm. Sci. 21:519–535.

Goodman, S.J. 1998. Patterns of extensive genetic differentiation and variation among European harbor seals (*Phoca vitulina vitulina)* revealed using microsatellite DNA polymorphisms. Mol. Biol. Evol. 15:104–118.

Hatch, J.M. and C.D. Orphanides. 2015. Estimates of cetacean and pinniped bycatch in the 2013 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 15-15. 33pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Hatch, J.M. and C.D. Orphanides. 2016. Estimates of cetacean and pinniped bycatch in the 2014 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 16-05. 22pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Huber, H. R., S.J. Jeffries, D.M. Lambourn and B.R. Dickerson. 2010. Population substructure of harbor seals (*Phoca vitulina richardsi*) in Washington State using mtDNA. Can. J. Zool. 88:280–288.

Jacobs, S.R. and J.M. Terhune. 2000. Harbor seal (*Phoca vitulina*) numbers along the New Brunswick coast of the Bay of Fundy in autumn in relation to aquaculture. Northeast. Nat. 7:289–296.

Johnston, D.W., J. Frungillo, A. Smith, K. Moore, B. Sharp, J. Schuh and A.J. Read. 2015. Trends in stranding and by-catch rates of gray and harbor seals along the northeastern coast of the United States: Evidence of divergence in the abundance of two sympatric phocid species? PLoS ONE 10(7):e0131660. https://doi.org/10.1371/journal.pone.0131660

Jones, D.V, D.R. Rees and B.A. Bartlett. 2018. Haul-out counts and photo-identification of pinnipeds in Chesapeake Bay and Eastern Shore, Virginia: 2017/2018 Annual Progress Report. Final Report. Prepared for U.S. Fleet Forces Command, Norfolk, Virginia. 21 December 2018.

Josephson, E., F. Wenzel and M.C. Lyssikatos. 2021. Serious injury determinations for small cetaceans and pinnipeds caught in commercial fisheries off the northeast U.S. coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-04. 33pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Katona, S.K., V. Rough and D.T. Richardson. 1993. A field guide to whales, porpoises, and seals from Cape Cod to Newfoundland. Smithsonian Institution Press, Washington, DC. 316pp.

Lelli, B., D.E. Harri and A-M. Aboueissa. 2009. Seal bounties in Maine and Massachusetts, 1888 to 1962. Northeast. Nat. 16:239–254.

Lyle, J.M. and S.T. Willcox. 2008. Dolphin and seal interactions with mid-water trawling in the commonwealth small pelagic fishery, including an assessment of bycatch mitigation. Australian Fisheries Management Authority, Final Report Project R05/0996. 49p.

Lyssikatos, M., S. Chavez-Rosales and J. Hatch. 2021. Estimates of cetacean and pinniped bycatch in northeast and mid-Atlantic bottom trawl fisheries, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-02. 12pp.

MMC [Marine Mammal Commission]. 2006. US Marine Mammal Commission annual report to Congress, 2005. Bethesda, MD. vi+163pp. Available from: http://www.mmc.gov/reports/annual/pdf/2005annualreport.pdf

O'Corry-Crowe, G.M., K.K. Martien and B.L. Taylor. 2003. The analysis of population genetic structure in Alaskan harbor seals, *Phoca vitulina*, as a framework for the identification of management stocks. Southwest Fish. Sci. Cent. Admin. Rep. LJ-03-08. 66pp. Available from: https://alaskafisheries.noaa.gov/sites/default/files/geneticstructure_occrowe03.pdf

Orphanides, C.D. and J. Hatch. 2017. Estimates of cetacean and pinniped bycatch in the 2015 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 17-18. 21pp.

Orphanides, C.D. 2019. Estimates of cetacean and pinniped bycatch in the 2016 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 19-04. 17pp.

Orphanides, C.D. 2020. Estimates of cetacean and pinniped bycatch in the 2017 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 20-03. 16pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Orphanides, C.D. 2021. Estimates of cetacean and pinniped bycatch in the 2018 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 21-01. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Pace, R.M., E. Josephson, S. Wood and K. Murray. 2019. Trends and patterns of seal abundance at haul-out sites in a gray seal recolonization zone. Northeast Fish. Sci Center Tech Memo. NMFS-NE-251.

Read, A.J. 1994. Interactions between cetaceans and gillnet and trap fisheries in the northwest Atlantic. Rep. Int. Whal. Comm. (Special Issue) 15:133–147.

Rees, D.R., D.V. Jones and B.A. Bartlett. Haul-out counts and photo-identification of pinnipeds in Chesapeake Bay, Virginia: 2015/16 Annual Progress Report. Final Report. Prepared for U.S. Fleet Forces Command, Norfolk, Virginia. 15 November 2016.

Rosenfeld, M., M. George and J.M. Terhune. 1988. Evidence of autumnal harbour seal, *Phoca vitulina*, movement from Canada to the United States. Can. Field-Nat. 102:527–529.

Schneider, D.C. and P.M. Payne. 1983. Factors affecting haul-out of harbor seals at a site in southeastern Massachusetts. J. Mamm. 64:518–520.
Schroeder, C.L. 2000. Population status and distribution of the harbor seal in Rhode Island waters. M.S. thesis. University of Rhode Island, Kingston, RI. 197pp.
Skinner, J.P. 2006. Physical and behavioral development of nursing harbor seal (*Phoca vitulina*) pups in Maine. M.S. thesis. University of Maine, Orono, ME. 140pp.
Stanley, H.F., S. Casey, J.M. Carnahan, S. Goodman, J. Harwood and R.K. Wayne. 1996. Worldwide patterns of mitochondrial DNA differentiation in the harbor seal (*Phoca vitulina*). Mol. Biol. Evol. 13:368–382.
Stobo, W.T. and Z. Lucas. 2000. Shark-inflicted mortality on a population of harbour seals (*Phoca vitulina*) at Sable Island, Nova Scotia. J. Zool. 252:405–414.
Temte, J.L., M.A. Bigg and O. Wiig. 1991. Clines revisited: The timing of pupping in the harbour seal (*Phoca vitulina*). J. Zool. 224:617–632.
Toth, J., S. Evert, E. Zimmermann, M. Sullivan, L. Dotts, K.W. Able, R. Hagan and C. Slocum. 2018. Annual residency patterns and diet of *Phoca vitulina concolor* (Western Atlantic harbor seal) in a southern New Jersey estuary. Northeastern Naturalist. 25(4):611–626.
Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp. Available from: https://repository.library.noaa.gov/view/noaa/15963
Waring, G.T., J.R. Gilbert, J. Loftin and N. Cabana. 2006. Short-term movements of radio-tagged harbor seals in New England. Northeast. Nat. 13:1–14.
Waring, G.T., R.A. DiGiovanni Jr, E. Josephson, S. Wood and J.R. Gilbert. 2015. 2012 population estimate for the harbor seal (*Phoca vitulina concolor*) in New England waters. NOAA Tech. Memo. NMFS NE-235. 15pp.
Westlake R.L. and G.M. O'Corry-Crowe. 2002. Macrogeographic structure and patterns of genetic diversity in harbor seals (*Phoca vitulina*) from Alaska to Japan. J. Mamm. 83:111–1126.
Whitman, A.A. and P.M. Payne. 1990. Age of harbour seals, *Phoca vitulina concolor*, wintering in southern New England. Can. Field-Nat. 104:579–582.

GRAY SEAL (*Halichoerus grypus atlantica*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

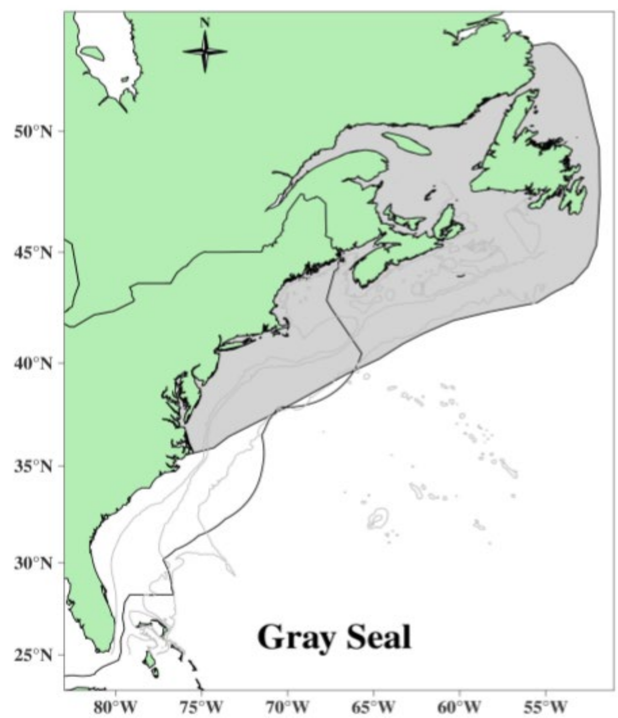


Figure 1: Approximate range of the Western North Atlantic stock of gray seals (Halichoerus grypus atlantica).

The gray seal (*Halichoerus grypus*) is found on both sides of the North Atlantic, with three major populations: Northeast Atlantic, Northwest Atlantic and the Baltic Sea (Haug *et al.* 2007). The Northeast Atlantic and the Northwest Atlantic populations are classified as the subspecies *H. g. atlantica* (Olsen *et al.* 2016). The Northwest Atlantic population includes the western North Atlantic stock ranges from New Jersey to Labrador (Figure 1; Davies 1957, Mansfield 1966, Katona *et al.* 1993, Lesage and Hammill 2001). This stock is separated from the northeastern Atlantic stocks by geography, differences in the breeding season, and mitochondrial and nuclear DNA variation (Bonner 1981, Boskovic *et al.* 1996, Lesage and Hammill 2001, Klimova *et al.* 2014). There are three breeding aggregations in eastern Canada: Sable Island, Gulf of St. Lawrence, and at sites along the coast of Nova Scotia (Laviguer and Hammill 1993). Animals from these aggregations mix outside the breeding season (Lavigueur and Hammill 1993; Harvey *et al.* 2008; Breed *et al.* 2006, 2009) and they are considered a single population based on genetic similarity (Boskovic *et al.* 1996, Wood *et al.* 2011).

After near extirpation due to bounties, which ended in the 1960s, small numbers of animals and pups were observed on several isolated islands along the Maine coast and in Nantucket-Vineyard Sound, Massachusetts (Katona *et al.* 1993, Rough 1995, Gilbert *et al*. 2005). In December 2001, NMFS initiated aerial surveys to monitor gray seal pup production on Muskeget Island and adjacent sites in Nantucket Sound, and Green and Seal Islands off the coast of Maine (Wood *et al.* 2007). Tissue samples collected from Canadian and U.S. populations were examined for genetic variation using mitochondrial and nuclear DNA (Wood *et al.* 2011). All individuals were identified as belonging to one population, confirming that the new U.S. population was recolonized by Canadian gray seals. The genetic evidence (Boskovic *et al.* 1996, Wood *et al.* 2011) provides a high degree of certainty that the western North Atlantic stock of gray seals comprise a single stock. Further supporting evidence comes from sightings of seals in the U.S. that had been branded on Sable Island, resights of tagged animals, and satellite tracks of tagged animals (Puryear *et al.* 2016). However, the percentage of time that individuals are resident in U.S. waters is unknown.

POPULATION SIZE

The size of the Northwest Atlantic gray seal population is estimated separately for the portion of the population in Canada versus the U.S., and mainly reflects the size of the breeding population in each respective country (Table 1). Currently there is a lack of information on the rate of exchange between animals in the U.S. and Canada, which influences seasonal changes in abundance throughout the range of this transboundary stock as well as life history parameters in population models. Total pup production in 2016 at breeding colonies in Canada was estimated to be 98,650 pups (CV=0.10; den Heyer 2017, DFO 2017). Production at Sable Island, Gulf of St. Lawrence, and Coastal Nova Scotia colonies accounted for 85%, 11% and 4%, respectively, of the estimated total number of pups born.

Population models, incorporating estimates of age-specific reproductive rates and removals, are fit to these pup production estimates to estimate total population levels in Canada. The total Canadian gray seal population in 2016 was estimated to be 424,300 (95%CI: 263,600–578,300; DFO 2017). Uncertainties in the population estimate derive from uncertainties in life history parameters such as mortality rates and sex ratios (DFO 2017).

In U.S. waters, the number of pupping sites has increased from one in 1988 to nine in 2019, and are located in Maine and Massachusetts (Wood *et al*. 2019). Although white-coated pups have stranded on eastern Long Island beaches in New York, no pupping colonies have been detected in that region. A minimum of 6,308 pups were born in 2016 at U.S. breeding colonies (Wood *et al*. 2019), approximately 6% of the total pup production over the entire range of the population (denHeyer *et al.* 2017). The percentage of pup production in the U.S. is considered a minimum because pup counts are single day counts that have not been adjusted to account for pups born after the survey, or that left the colony prior to the survey. Mean rates of increase in the number of pups born at various times since 1988 at four of the more frequently surveyed pupping sites (Muskeget, Monomoy, Seal, and Green Islands) ranged from -0.2% (95%CI: -2.3–1.9%) to 26.3% (95%CI: 21.6–31.4%; Wood *et al.* 2019). These high rates of increase provide further support that seals from other areas are continually supplementing the breeding population in U.S. waters.

The number of pups born at U.S. breeding colonies can be used to approximate the total size (pups and adults) of the gray seal population in U.S. waters, based on the ratio of total population size to pups in Canadian waters (4.3:1; den Heyer *et al.* 2017, DFO 2017). Although not yet measured for U.S. waters, this ratio falls within the range of other adult to pup ratios suggested for pinniped populations (Harwood and Prime 1978, Thomas *et al.* 2019). Using this approach, the population estimate in U.S. waters is 27,131 (CV=0.19; 95%: 18,768–39,221) animals. The CV and CI around this estimate are based on CVs and CIs from Canadian population estimates, rather than using a default CV when the variance is unknown (Wade and Angliss 1997). There is further uncertainty in this abundance level in the U.S. because life history parameters that influence the ratio of pups to total individuals in this portion of the population are unknown. It also does not reflect seasonal changes in stock abundance in the Northeast region for a transboundary stock. For example, roughly 24,000 seals were observed in southeastern Massachusetts alone in 2015 (Pace *et al.* 2019), yet 28,000–40,000 gray seals were estimated to be in this region in 2015 using correction factors applied to seal counts obtained from Google Earth imagery (Moxley *et al.* 2017).

Table 1. Summary of recent abundance estimates for the western North Atlantic gray seal (Halichoerus grypus atlantica) by year, and area covered, resulting total abundance estimate (N_{est}) and 95% confidence interval (95%CI).

Year	Area	N_{est} [a]	95%CI
2012[b]	Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island	331,000	263,000–458,000
2014[c]	Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island	505,000	329,000–682,000
2016[d]	Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island	424,300	263,600–578,300
2016	U.S	27,131[e]	18,768–39,221

a. These are model-based estimates derived from pup surveys.
b. DFO 2013
c. DFO 2014
d. DFO 2017
e. This is derived from total population size to pup ratios in Canada, applied to U.S. pup counts.

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997). Based on an estimated U.S. population in 2016 of 27,131 (CV=0.19), the minimum population estimate in U.S. waters is 23,153 (Table 2). Similar to the best abundance estimate, there is uncertainty in this minimum abundance level in the U.S. because life history parameters that influence the ratio of pups to total individuals in this population are unknown.

Current Population Trend

In the U.S., the mean rate of increase in the number of pups born differs across the pupping. From 1988–2019, the estimated mean rate of increase in the number of pups born was 12.8% on Muskeget Island, 26.3% on Monomoy Island, 11.5% on Seal Island, and -0.2% on Green Island (Wood *et al*. 2019). These rates only reflect new recruits to the population and do not reflect changes in total population growth resulting from Canadian seals migrating to the region.

The total population of gray seals in Canada was estimated to be increasing by 4.4% per year from 1960–2016 (Hammill *et al*. 2017), primarily due to increases at Sable Island. Pup production on Sable Island increased exponentially at a rate of 12.8% per year between the 1970s and 1997 (Bowen *et al.* 2003). Pupping also occurs on Hay Island off Nova Scotia, in colonies off southwestern Nova Scotia, and in the Gulf of St. Lawrence. Since 1997, the rate of increase has slowed (Bowen *et al.* 2011, den Heyer *et al.* 2017), supporting the hypothesis that density-dependent changes in vital rates may be limiting population growth. While slowing, pup production is still increasing on Sable Island and in southwest Nova Scotia, and stabilizing on Hay Island in the Gulf of St. Lawrence (DFO 2017, den Heyer *et al.* 2017). In the Gulf of St. Lawrence, the proportion of pups born on the ice has declined from 100% in 2004 to 1% in 2016 due to a decline in winter ice cover in the area, and seals have responded by pupping on nearby islands (DFO 2017).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For the purposes of this assessment, the maximum net productivity rate was assumed to be 0.12. This value is based on theoretical modeling showing that pinniped populations may not grow at rates much greater than 12% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362, Wade and Angliss 1997). The minimum population size for the portion of the stock residing in U.S. waters is 23,153. The maximum productivity rate is 0.12, the default value for pinnipeds. The recovery factor (F_r) for this stock is 1.0, the value for stocks of unknown status, but which are known to be increasing. PBR for the western North Atlantic stock of gray seals residing in U.S. waters is 1,389 animals (Table 2). Uncertainty in the PBR level arises from the same sources of uncertainty in calculating a minimum abundance estimate in U.S. waters.

Table 2. Best (N_{est}) and minimum abundance (N_{min}) estimates for the western North Atlantic gray seal (Halichoerus grypus atlantica) with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	Fr	R_{max}	PBR
27,131	0.19	23,153	1	0.12	1,389

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2014–2018, the average annual estimated human-caused mortality and serious injury to gray seals in the U.S. and Canada was 4,729 (953 U.S./3,776 Canada) per year. Mortality in U.S. fisheries is explained in further detail below.

Table 3. The total annual estimated average human-caused mortality and serious injury for the western North Atlantic gray seal (Halichoerus grypus atlantica).

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	946	0.11
2014–2018	U.S. non-fishery human interaction stranding mortalities	6.2	
2014–2018	U.S. research mortalities	1.2	
2014–2018	Canadian commercial harvest	636	
2014–2018	DFO Canada scientific collections	62	
2014–2018	Canadian removals of nuisance animals	3,078	
Total		**4,729**	

Some human-caused mortality or serious injury may not be able to be quantified. Observed serious injury rates are lower than would be expected from the anecdotally observed numbers of gray seals living with ongoing entanglements. Estimated rates of entanglement in gillnet gear, for example, may be biased low because 100% of observed animals are dead when they come aboard the vessel (Josephson *et al.* 2021); therefore, rates do not reflect the number of live animals that may have broken free of the gear and are living with entanglements. For example, mean prevalence of live entangled gray seals ranged from roughly 1 to 4% at haul-out sites in Massachusetts and Isle of Shoals (Iruzun Martins *et al.* 2019). Reports of seal shootings and other non-fishery-related human interactions are minimum counts.

Fishery Information

Detailed fishery information is given in Appendix III.

United States

Northeast Sink Gillnet

The Northeast sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2014–2018 was 199 animals, and the average annual estimated total mortality was 896 (CV=0.11; Hatch and Orphanides 2016; Orphanides and Hatch 2017; Orphanides 2019*,* 2020, 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Gillnet

The mid-Atlantic sink gillnet fishery is a Category I fishery. The average annual observed mortality from 2014–2018 was <1 animal, and the average annual total mortality was 8.8 (CV=0.67; Hatch and Orphanides 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020, 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Gulf of Maine Atlantic Herring Purse Seine Fishery

The Gulf of Maine Atlantic Herring Purse Seine Fishery is a Category III fishery. No mortalities have been observed in this fishery, during the current 5-year period, however, two gray seals were captured and released alive in 2014, zero in 2015, five in 2016, zero in 2017 and one in 2018. In addition, two seals of unknown species were captured and released alive in 2015 and one in 2016 (Josephson *et al.* 2021).

Northeast Bottom Trawl

The Northeast bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2014–2018 was three animals, and the average annual total mortality was 18 (CV=0.22; Lyssikatos *et al.* 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Mid-Atlantic Bottom Trawl

The mid-Atlantic bottom trawl fishery is a Category II fishery. The average annual observed mortality from 2014–2018 was two animals, and the average annual total mortality was 23 (CV=0.33; Lyssikatos *et al.* 2021). See Table 4 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Mid-Water Trawl and Pair Trawl

The Northeast mid-water trawl and pair trawl fisheries are Category II fisheries. Only one gray seal was observed in these fisheries from 2014–2018 and an expanded bycatch estimate has not been generated. See Table 4 for observed mortality and serious injury for during the current 5-year period, and Appendix V for historical bycatch information.

Canada

There is limited information on Canadian fishery bycatch (DFO 2017). Historically, an unknown number of gray seals have been taken in Newfoundland and Labrador, Gulf of St. Lawrence, and Bay of Fundy groundfish gillnets; Atlantic Canada and Greenland salmon gillnets; Atlantic Canada cod traps, and Bay of Fundy herring weirs (Read 1994).

Table 4. Summary of the incidental serious injury and mortality of gray seals (Halichoerus grypus atlantica) by commercial fishery including the years sampled, the type of data used (Data Type), the annual observer coverage (Observer Coverage), the mortalities recorded by on-board observers (Observed Mortality), the estimated annual mortality (Estimated Mortality), the estimated CV of the annual mortality (Estimated CVs) and the mean annual combined mortality (CV in parentheses).

Fishery	Years	Data Type [a]	Observer Coverage[b]	Observed Serious Injury[c]	Observed Mortality	Est. Serious Injury	Est. Mortality	Est. Comb. Mortality	Est. CVs	Mean Annual Combined Mortality
Northeast Sink Gillnet	2014 2015 2016 2017 2018	Obs. Data, Weighout, Logbook	0.18 0.14 0.10 0.12 0.11	0 0 0 0 0	159 131 43 158 103	0 0 0 0 0	917 1021 498 930 1113	917 1021 498 930 1113	0.14 0.25 0.33 0.16 0.32	896 (0.11)
Mid-Atlantic Gillnet	2014 2015 2016 2017 2018	Obs. Data, Logbook, Allocated Dealer Data	0.05 0.06 0.08 0.09 0.09	0 0 0 0 0	1 1 1 0 0	0 0 0 0 0	22 15 7 0 0	22 15 7 0 0	1.09 1.04 0.93 0 0	8.8 (0.67)
Northeast Bottom Trawl	2014 2015 2016 2017 2018	Obs. Data, Logbook	0.19 0.19 0.12 0.12 0.12	0 0 0 0 0	4 4 0 2 5	0 0 0 0 0	19 23 0 16 32	19 23 0 16 32	0.45 0.46 0 0.24 0.42	18 (0.22)
Mid-Atlantic Bottom Trawl	2014 2015 2016 2017 2018	Obs. Data, Logbook	0.09 0.09 0.10 0.14 0.12	0 0 0 0 0	1 0 3 5 7	0 0 0 0 0	7 0 26 26 56	7 0 26 26 56	0.96 0 0.57 0.40 0.58	23 (0.33)
Northeast Mid-water Trawl – Incl. Pair Trawl	2014 2015 2016 2017 2018	Obs. Data, Logbook	0.42 0.08 0.27 0.16 0.14	0 0 0 0 0	0 0 0 0 1	0 0 0 0 0	0 0 0 0 na	0 0 0 0 na	0 0 0 0 na	0.2 (na) [d]
Total										**946 (0.11)**

a. Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. The Northeast Fisheries Observer Program collects landings data (Weighout), and total landings are used as a measure of total effort for the sink gillnet fishery. Mandatory logbook (Logbook) data are used to determine the spatial distribution of fishing effort in the Northeast multispecies sink gillnet fishery.

b. The observer coverages for the northeast sink gillnet fishery and the mid-Atlantic gillnet fisheries are ratios based on tons of fish landed. North Atlantic bottom trawl, mid-Atlantic bottom trawl, and mid-Atlantic mid-water trawl fishery coverages are ratios based on trips. Total observer coverage reported for bottom trawl gear and gillnet gear includes traditional fisheries observers in addition to fishery monitors through the Northeast Fisheries Observer Program (NEFOP).

c. Serious injuries were evaluated for the 2014–2018 period (Josephson *et al.* 2021).

d. Unextrapolated number from observed data.

Other Mortality

United States

Gray seals, like harbor seals, were hunted for bounty in New England waters until the late 1960s (Katona *et al.* 1993, Lelli *et al*. 2009). This hunt may have severely depleted this stock in U.S. waters (Rough 1995, Lelli *et al*. 2009). Other sources of mortality include human interactions, storms, abandonment by the mother, disease, and shark predation. Mortalities caused by human interactions include research mortalities, boat strikes, fishing gear interactions, power plant entrainment, oil spill/exposure, harassment, and shooting. Seals entangled in netting are common at haul-out sites in the Gulf of Maine and Southeastern Massachusetts.

Tables 5 and 6 present summaries of gray seal strandings as reported to the NOAA National Marine Mammal Health and Stranding Response Databaseaccessed 20 November 2019). Most stranding mortalities were in Massachusetts, which is the center of gray seal abundance in U.S. waters. In an analysis of mortality causes of stranded marine mammals on Cape Cod and southeastern Massachusetts between 2000 and 2006, Bogomolni *et al.* (2010) reported that 45% of gray seal stranding mortalities were attributed to human interaction.

A UME was declared in November of 2011 that involved at least 137 gray seal stranding mortalities between June 2011 and October 2012 in Maine, New Hampshire, and Massachusetts. The UME was declared closed in February 2013 (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events).

Canada

Between 2014 and 2018, the average annual human-caused mortality and serious injury to gray seals in Canadian waters from commercial harvest is 636 individuals, though up to 60,000 seals/year are permitted (http://www.dfo-mpo.gc.ca/decisions/fm-2015-gp/atl-001-eng.htm). This included: 82 in 2014, 1,381 in 2015, 1,588 in 2016, 64 in 2017, and 66 in 2018 (DFO 2017, Courtney D'Aoust pers. comm.). In addition, between 2014 and 2018, an average of 3,078 nuisance animals per year were killed. This included 3,732 annually in 2014–2017 (DFO 2017) and 461 in 2018 based on the total number of licenses that were issued (Courtney D'Aoust pers. comm). Lastly, DFO took 83 animals in 2014, 42 animals in 2015, 30 animals in 2016, 60 animals in 2017, and 96 animals in 2018 for scientific collections, for an annual average of 62 animals (DFO 2017, Samuel Mongrain pers. comm).

Table 5. Gray seal (Halichoerus grypus atlantica) stranding mortalities along the U.S. Atlantic coast (2014–2018) with subtotals of animals recorded as pups in parentheses.

State	2014	2015	2016	2017	2018	Total
ME	3 (1)	5	6(0)	14 (1)	25 (0)	53
NH	3 (2)	2	0	3 (0)	9 (3)	17
MA	62 (6)	77 (3)	54(0)	135 (21)	261 (29)	589
RI	8 (1)	7 (1)	4(0)	16 (5)	20 (3)	55
CT	0	0	0	3 (0)	1(0)	4
NY	12 (4)	10	1 (1)	57 (0)	25 (1)	105
NJ	7 (6)	7 (6)	3 (1)	4 (3)	14 (10)	35
DE	3 (3)	3 (3)	0	1 (0)	4 (2)	11
MD	1 (0)	0	0	0	1 (1)	2
VA	0	3	0	0	1 (1)	4
NC	2 (2)	0	0	0	5 (2)	7
Total	**101 (25)**	**114 (13)**	**68 (2)**	**192 (30)**	**346 (48)**	**882**
Unspecified seals (all states)	38	31	13	86	92	193

Table 6. Documented gray seal (Halichoerus grypus atlantica) human-interaction related stranding mortalities along the U.S. Atlantic coast (2014–2018) by type of interaction. "Fishery interactions" are subsumed in the total estimated mortality calculated from observer data.

Cause	2014	2015	2016	2017	2018	Total
Fishery Interaction	2	14	0	10	10	36
Boat Strike	3	3	0	4	2	12
Shot	0	1	1	0	0	2
Human Interaction - Other	3	2	0	3	9	17
Total	**8**	**20**	**1**	**17**	**21**	**67**

STATUS OF STOCK

Gray seals are not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the Marine Mammal Protection Act. The average annual human-caused mortality and serious injury during 2014–2018 in U.S. waters does not exceed the PBR of the U.S. portion of the stocks. The status of the gray seal population relative to OSP in U.S. Atlantic EEZ waters is unknown, but the stock's abundance appears to be increasing in Canadian and U.S. waters. Total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching the zero mortality and serious injury rate. Uncertainties described in the above sections could have an effect on the designation of the status of this stock in U.S. waters.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Bogomolni, A.L., K.R. Pugliares, S.M. Sharp, K. Patchett, C.T. Harry, J.M. LaRocque, K.M. Touhey and M. Moore. 2010. Mortality trends of stranded marine mammals on Cape Cod and southeastern Massachusetts, USA, 2000 to 2006. Dis. Aq. Org. 88:143–155.

Bonner, W.N. 1981. Grey seal (*Halichoerus grypus*) Fabricus, 1791. Pages 111-144 *in*: S.H. Ridgway and R.J. Harrison (Eds.). Handbook of marine mammals, Vol. 2: Seals. Academic Press, London, UK.

Boskovic, R., K.M. Kovacs, M.O. Hammill and B.N. White. 1996. Geographic distribution of mitochondrial DNA haplotypes in grey seals (*Halichoerus grypus*). Can. J. Zool. 74:1787–1796.

Bowen, W.D., J.I. McMillan and W. Blanchard. 2007. Reduced population growth of gray seals at Sable Island: Evidence from pup production and age of primiparity. Mar. Mamm. Sci. 23:48–64.

Bowen, W.D., C. den Heyer, J.I. McMillan and M.O. Hammill. 2011. Pup production at Scotian Shelf grey seal (*Halichoerus grypus*) colonies in 2010. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/066: vi+25pp.

Breed, G.A., W.D. Bowen, J.I. McMillan and M.L. Leonard. 2006. Sexual segregation of seasonal foraging habitats in a non-migratory marine mammal. Proc. Royal Soc. B 273:2319–2326.

Breed, G.A., I.D. Jonsen, R.A. Myers, W.D. Bowen and M.L. Leonard. 2009. Sex-specific, seasonal foraging tactics of adult grey seals (*Halichoerus grypus*) revealed by state-space analysis. Ecology. 90:3209–3221.

Cammen, K.M., S.Vincze, A.S. Heller, B.A. McLeod, S.A. Wood, W.D. Bowen, M.O. Hammill, W.B. Puryear, J. Runstadler, F.W. Wenzel, M. Kinnison and T.R. Frasier. 2018. Genetic diversity from pre-bottleneck to recovery in two sympatric pinniped species in the Northwest Atlantic. Cons. Gen. 19:555–569.

Davies, J.L. 1957. The geography of the gray seal. J. Mamm. 38:297–310.

den Heyer, C.E., S.L.C. Lang, W.D. Bowen and M.O. Hammill. 2017. Pup production at Scotian Shelf grey seal (*Halichoerus grypus*) colonies in 2016. DFO Can. Sci. Advis. Sec. Res. Doc. 2017/056. v+34 p.

DFO [Department of Fisheries and Oceans]. 2011. Stock assessment of Northwest Atlantic grey seals (*Halichoerus grypus*). Can. Sci. Advis. Sec. Advis. Rep. 2010/091. 12pp.

DFO [Department of Fisheries and Oceans]. 2013. Stock assessment of Northwest Atlantic grey seals (*Halichoerus grypus*). DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2013/008. 11pp.

DFO [Department of Fisheries and Oceans]. 2014. Stock assessment of Canadian grey seals (*Halichoerus grypus*). DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2014/010. 13pp.

DFO [Dept. of Fisheries and Oceans]. 2017. Stock assessment of Canadian Northwest Atlantic grey seals (*Halichoerus grypus*). DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2017/045. 13pp.

Gilbert, J.R., G.T. Waring, K.M. Wynne and N. Guldager. 2005. Changes in abundance and distribution of harbor seals in Maine, 1981–2001. Mar. Mamm. Sci. 21:519–535.

Hammill, M.O., C.E. den Heyer, W.D. Bowen and S.L.C. Lang. 2017. Grey seal population trends in Canadian waters, 1960–2016 and harvest advice. DFO Can. Sci. Advis. Sec. Res. Doc. 2017/052. v+30p.

Harvey, V., S.D. Côté and M.O. Hammill. 2008. The ecology of 3-D space use in a sexually dimorphic mammal. Ecography. 31:371–380.

Harwood, J. and J.H. Prime. 1978. Some factors affecting the size of British grey seal populations. J. Appl. Ecol. 15:401–411.

Hatch, J.M. and C.D. Orphanides. 2016. Estimates of cetacean and pinniped bycatch in the 2014 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish Sci Cent Ref Doc. 16-05. 22pp.

Haug, T., M.O. Hammill and D. Ólafsdóttir (Eds.). 2007. Grey seals in the North Atlantic and the Baltic. NAMMCO Scientific Publications. 6:7–12.

Iruzun Martins, M.C., L. Sette, E. Josephson, A. Bogomolni, K. Rose, S. Sharp, M. Niemeyer and M. Moore. 2019. Unoccupied aerial system assessment of entanglement in Northwest Atlantic gray seals (*Halichoerus grypus*). Mar. Mamm. Sci. 35:1613–1624. https://doi.org/10.1111/mms.12590

Josephson, E., F. Wenzel and M.C. Lyssikatos. 2021. Serious injury determinations for small cetaceans and pinnipeds caught in commercial fisheries off the northeast U.S. coast, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 20-04. 33pp.

Katona, S.K., V. Rough and D.T. Richardson. 1993. A field guide to whales, porpoises, and seals from Cape Cod to Newfoundland. Smithsonian Institution Press, Washington, DC. 316pp.

Klimova, A., C.D. Phillips, K. Fietz, M.T. Olsen, J. Harwood, W. Amos and J.I. Hoffman. 2014. Global population structure and demographic history of the grey seal. Mol. Ecol. 23:3999–4017.

Laviguer, L. and M.O. Hammill. 1993. Distribution and seasonal movements of grey seals, *Halichoerus grypus*, born in the Gulf of St. Lawrence and eastern Nova Scotia shore. Can. Field-Nat. 107:329–340.

Lesage, V. and M.O. Hammill. 2001. The status of the grey seal, *Halichoerus grypus*, in the Northwest Atlantic. Can. Field-Nat. 115:653–662.

Lelli, B., D.E. Harris and A.M. Aboueissa. 2009. Seal bounties in Maine and Massachusetts, 1888 to 1962. Northeast. Nat. 16:239–254.

Lyssikatos, M.C., S. Chavez-Rosales and J. Hatch. 2021. Estimates of cetacean and pinniped bycatch in Northeast and mid-Atlantic bottom trawl fisheries, 2014–2018. Northeast Fish. Sci. Cent. Ref. Doc. 21-02. 12pp.

Mansfield, A.W. 1966. The grey seal in eastern Canadian waters. Can. Audubon Mag. 28:161–166.

Mohn, R. and W.D. Bowen. 1996. Grey seal predation on the eastern Scotian Shelf: Modeling the impact on Atlantic cod. Can. J. Fish. Aquat. Sci. 53:2722–2738.

Moxley, J.H., A. Bogomolni, M.O. Hammill, K.M.T. Moore, M.J. Polito, L. Sette, B. Sharp, G.T. Waring, J.R. Gilbert, P.N. Halpin and D.W. Johnston. 2017. Google Haul Out: Earth observation imagery and digital aerial surveys in coastal wildlife management and abundance estimation. BioScience. 67:760–768.

Olsen, M.T., A. Galatius, V. Biard, K. Gregersen and C.C. Kinze. 2016. The forgotten type specimen of the grey seal [*Halichoerus grypus* (Fabricius, 1791)] from the island of Amager, Denmark. Zoo. J. Linnean Soc. 178:713–720.

Orphanides, C.D. and J. Hatch. 2017. Estimates of cetacean and pinniped bycatch in the 2015 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish Sci Cent Ref Doc. 17-18. 21pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Orphanides, C.D. 2019. Estimates of cetacean and pinniped bycatch in the 2016 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish Sci Cent Ref Doc. 19-04. 12pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Orphanides, C.D. 2020. Estimates of cetacean and pinniped bycatch in the 2017 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 20-03. 16pp. Available from: https://www.fisheries.noaa.gov/resource/publication-database/marine-mammal-mortality-and-serious-injury-reports

Orphanides, C.D. 2021. Estimates of cetacean and pinniped bycatch in the 2018 New England sink and mid-Atlantic gillnet fisheries. Northeast Fish. Sci. Cent. Ref. Doc. 21-01.

Pace, R.M., E. Josephson, S. Wood and K. Murray. 2019. Trends and patterns of seal abundance at haul-out sites in a gray seal recolonization zone. Northeast Fish. Sci Center Technical Memorandum. NMFS-NE-251. 23pp.

Puryear, W.B., M. Keogh, N. Hill, J. Moxley, E. Josephson, K.R. Davis, C. Bandoro, D. Lidgard, A. Bogomolni, M. Levin, S. Lang, M. Hammill, D. Bowen, D.W. Johnston, T. Romano, G. Waring and J. Runstadler. 2016. Prevalence of influenza A virus in live-captured North Atlantic gray seals: A possible wild reservoir. Emerg. Microbes & Infect. 5:1–9. https://doi.org/10.1038/emi.2016.77

Read, A.J. 1994. Interactions between cetaceans and gillnet and trap fisheries in the northwest Atlantic. Rep. Int. Whal. Comm. (Special Issue) 15:133–147.

Rough, V. 1995. Gray seals in Nantucket Sound, Massachusetts, winter and spring, 1994. Final report to Marine Mammal Commission. Contract T10155615. 28pp.

Thomas, L., D.J.F. Russell, C.D. Duck, C.D. Morris, M. Lonergan, F. Empacher, D. Thompson and J. Harwood. 2019. Modelling the population size and dynamics of the British grey seal. Aquatic Conserv: Mar. Freshw. Ecosyst. 29(S1):6–23.

Thomas, L., M.O. Hammill and W.D. Bowen. 2011. Estimated size of the Northwest Atlantic grey seal population. Can. Sci. Advis. Sec. Res. Doc. 2011/017. 23pp.

Trzcinski, M.K., R. Mohn and W.D. Bowen. 2005. Estimation of grey seal population size and trends at Sable Island. DFO Research Document 2005/067. Canadian Department of Fisheries and Oceans. Ottawa, Ontario, Canada. 10pp. Available from: http://www.dfo-mpo.gc.ca/csas/Csas/Publications/ResDocsDocRech/2005/2005067e.htm

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp. Available from: https://repository.library.noaa.gov/view/noaa/15963

Wood, S.A., K.T. Murray, E. Josephson and J.R. Gilbert. 2019. Rates of increase in gray seal (*Halichoerus grypus atlantica*) pupping at recolonized sites in the United States, 1988–2019. J. Mammal. 101:121–128. https://doi.org/10.1093/jmammal/gyz184

Wood, S.A., S. Brault and J.R. Gilbert. 2007. 2002 aerial survey of grey seals in the northeastern United States. Pages 117–121 *in*: T. Haug, M. Hammill and D. Ólafsdóttir (Eds.). Grey seals in the North Atlantic and Baltic. NAMMCO Sci. Pub. 6, Tromsø, Norway.

Wood, S.A., T.R. Frasier, B.A. McLeod, J.R. Gilbert, B.N. White, W.D. Bowen, M.O. Hammill, G.T. Waring and S. Brault. 2011. The genetics of recolonization: An analysis of the stock structure of grey seals (*Halichoerus grypus*) in the Northwest Atlantic. Can. J. Zool. 89:490–497.

SPERM WHALE (*Physeter macrocephalus*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Sperm whales are found throughout the world's oceans in deep waters from the tropics to the edge of the ice at both poles (Leatherwood and Reeves 1983, Rice 1989, Whitehead 2002). Sperm whales were commercially hunted in the Gulf of Mexico by American whalers from sailing vessels until the early 1900s (Townsend 1935, Reeves *et al.* 2011). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), systematic aerial and ship surveys indicate that sperm whales inhabit continental slope and oceanic waters where they are widely distributed (Figure 1; Fulling *et al.* 2003, Mullin and Fulling 2004, Mullin *et al.* 2004, Maze-Foley and Mullin 2006, Mullin 2007, Garrison and Aichinger Dias 2020). Seasonal aerial surveys confirm that sperm whales are present in the northern Gulf of Mexico in all seasons (Mullin *et al.* 1994, Hansen *et al.* 1996, Mullin and Hoggard 2000).

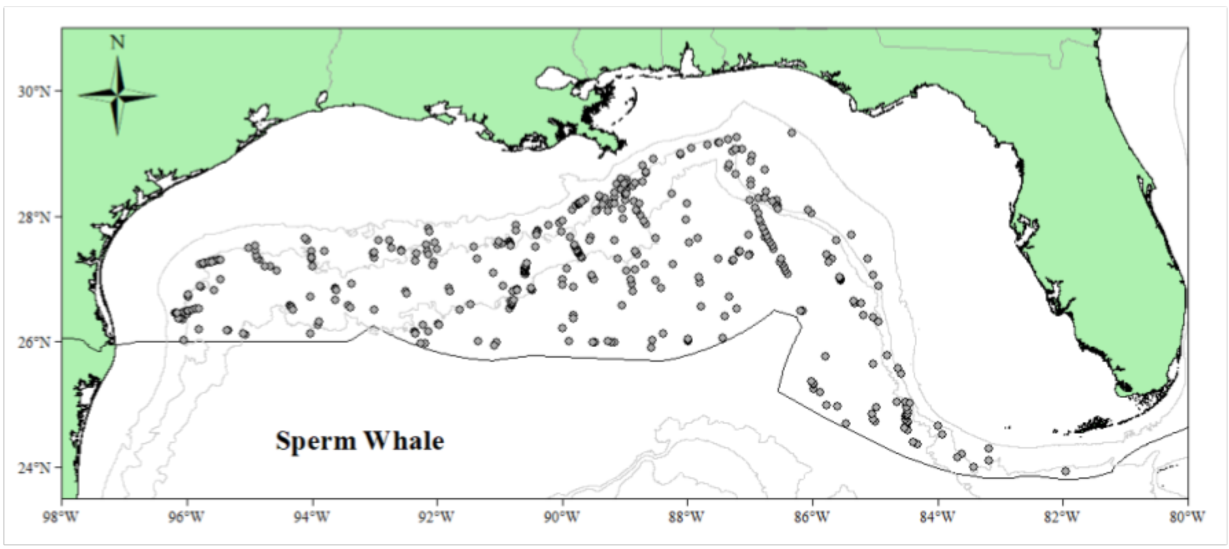


Figure 1. Distribution of sperm whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al*. 2005; Jefferson *et al*. 2008; Vázquez Castán *et al.* 2009; Whitt *et al*. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Sperm whales throughout the world exhibit a geographic social structure where females and juveniles of both sexes occur in mixed groups and inhabit tropical and subtropical waters. Males, as they mature, initially form bachelor groups but eventually become more socially isolated and more wide-ranging, inhabiting temperate and polar waters as well (Whitehead 2003). While this pattern also applies to the Gulf of Mexico, results of multi-disciplinary research conducted in the Gulf since 2000 confirms speculation by Schmidly (1981) and indicates clearly that Gulf of Mexico sperm whales constitute a stock that is distinct from other Atlantic Ocean stocks (Mullin *et al.* 2003, Jaquet 2006, Jochens *et al.* 2008). Measurements of the total length of Gulf of Mexico sperm whales indicate that they are 1.5–2.0 m smaller on average compared to whales measured in other areas (Jochens *et al.* 2008). Female/immature group size in the Gulf is about one-third to one-fourth that found in the Pacific Ocean but more similar to group sizes in the Caribbean (Richter *et al.* 2008, Jaquet and Gendron 2009). Tracks from 39 whales satellite tagged in the northern Gulf were monitored for up to 607 days. No discernable seasonal migrations were made, but Gulf-wide movements primarily along the northern Gulf slope did occur. The tracks showed that whales exhibit a range of movement patterns within the Gulf, including movement into the southern Gulf in a few cases, but that only one whale (a male) left the Gulf of Mexico (Jochens *et al.* 2008). This animal moved into the North Atlantic and then back into the Gulf after about two months. Additionally, no matches were found when 285 individual whales photo-identified from the Gulf and about 2500 from the North Atlantic and Mediterranean Sea were compared (Jochens *et al.* 2008).

Gero *et al.* (2007) also suggested that movements of sperm whales between the adjacent areas of the Caribbean Sea, Gulf of Mexico and Atlantic may not be common. No matches were made from animals photo-identified in the eastern Caribbean Sea (islands of Dominica, Guadeloupe, Grenada, St. Lucia and Martinique) with either animals from the Sargasso Sea or the Gulf of Mexico. Engelhaupt *et al.* (2009) conducted an analysis of matrilineally inherited mitochondrial DNA and found significant genetic differentiation between animals from the northern Gulf of Mexico and those from the western North Atlantic Ocean, North Sea and Mediterranean Sea. Analysis of biparentally inherited nuclear DNA showed no significant difference between whales sampled in the Gulf and those from the other areas of the North Atlantic, suggesting that while females show strong philopatry to the Gulf, male-mediated gene flow between the Gulf and North Atlantic Ocean may be occurring (Engelhaupt *et al.* 2009).

Sperm whales make vocalizations called “codas” that have distinct patterns and are apparently culturally transmitted (Watkins and Schevill 1977, Whitehead and Weilgart 1991, Rendell and Whitehead 2001), and based on degree of social affiliation, mixed groups of sperm whales (mixed-sex groups of females/immatures) worldwide can be placed in recognizable acoustic clans (Rendell and Whitehead 2003). Recordings from mixed groups in the Gulf of Mexico compared to those from other areas of the Atlantic indicated that Gulf sperm whales constitute a distinct acoustic clan that is rarely encountered outside of the Gulf. It is assumed from this that groups from other clans enter the northern Gulf only infrequently (Gordon *et al.* 2008). Antunes (2009) used additional data to further examine variation in sperm whale coda repertoires in the North Atlantic Ocean, and found that variation in the North Atlantic is mostly geographically structured as coda patterns were unique to certain regions and a significant negative correlation was found between coda repertoire similarities and geographic distance. His work also suggested sperm whale codas differed between the Gulf of Mexico and the North Atlantic.

Thus, there are now multiple lines of evidence supporting delimitation of separate Gulf of Mexico and western North Atlantic stocks of sperm whales. However, there are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (N_{est}) for the northern Gulf of Mexico sperm whale is 1,180 (CV=0.22; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of sperm whale abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=2,542 (CV=0.34); 2004, N=1,686 (CV=0.41); and 2009, N=2,096 (CV=0.55).

Recent Surveys and Abundance Estimates

An abundance estimate for sperm whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21, Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified large whales that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al*. 2020). The 2017 and 2018 estimates were N=1,078 (CV=0.29) and N=1,307 (CV=0.33), respectively. The inverse variance weighted mean abundance estimate for sperm whales in oceanic waters during 2017 and 2018 was 1,180 (CV=0.22; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline. There may be a portion of the detection probability that is not accounted for due to long dive times.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico sperm whales in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	1,180	0.22

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for sperm whales is 1,180 (CV=0.22). The minimum population estimate for the northern Gulf of Mexico sperm whale stock is 983 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June–August), 2004 (April–June), and 2009 (July–August; see above), and the 2017 (July–August) and 2018 (August–October) estimates, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 983. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.1 because the sperm whale is an endangered species. PBR for the northern Gulf of Mexico sperm whale is 2.0 (Table 2).

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico sperm whale with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
1,180	0.22	983	0.1	0.04	2.0

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The estimated mean annual fishery-related mortality and serious injury for this stock during 2014–2018 was 0.2 sperm whales (CV=1.00) due to interactions with the large pelagics longline fishery (see Fisheries Information sections below; Tables 3–4). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 9.4. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 9.6.

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico sperm whale.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0.2	1.00

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively.

There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of sperm whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During the second quarter of 2015, one sperm whale was observed to be seriously injured (Garrison and Stokes 2017). The average annual serious injury and mortality in the Gulf of Mexico pelagic longline fishery for the five-year period from 2014 to 2018 is 0.2 (CV=1.00; Table 4; Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

During the first and second quarters of 2014–2018, observer coverage in the Gulf of Mexico pelagic longline fishery was greatly enhanced to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2014–2018 primarily reflect high coverage rates during the first and second quarters of each year. During these quarters, this elevated coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rates observed during the first and second quarters are representative of that which occurs throughout the year.

A commercial fishery for sperm whales operated in the Gulf of Mexico during the late 1700s to the late 1800s (Reeves *et al.* 2011), but the exact number of whales taken is not known (Townsend 1935, Lowery 1974). Reeves *et al.* (2011) estimated the number of sperm whales removed from the Gulf during the 1780s–1870s as 1,179 (SE=224).

Table 4. Summary of the incidental mortality and serious injury of sperm whales by the pelagic longline commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the annual observed serious injury and mortality recorded by on-board observers, the annual estimated serious injury and mortality, the combined annual estimates of serious injury and mortality (Est. Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs), the mean of the combined annual mortality estimates, and the CV of the mean combined annual mortality estimate (CV of Mean).

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury[c]	Est. Mort.	Est. Combined Mortality	Est. CVs	Mean Combined Annual Mortality	CV of Mean
Pelagic Longline	2014	Obs. Data, Trip Logbook	0.18	0	0	0	0	0	-	0.2	1.00
	2015		0.19	1	0	0.94	0	0.94	1		
	2016		0.23	0	0	0	0	0	-		
	2017		0.13	0	0	0	0	0	-		
	2018		0.20	0	0	0	0	0	-		
Total										**0.2**	**1.00**

[a] Number of vessels in the fishery is based on vessels reporting effort to the pelagic longline logbook.
[b] Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. Mandatory logbook data were used to measure total effort for the longline fishery. These data are collected at the Southeast Fisheries Science Center (SEFSC). Observer coverage in the GOM is dominated by very high coverage rates during April–June associated with efforts to improve estimates of bluefin tuna bycatch.
[c] Proportion of sets observed.

Other Mortality

There were seven sperm whale strandings in the northern Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). There was evidence of human interaction for one stranding (healed scarring). No evidence of human interaction was detected for one stranding, and for the remaining five strandings it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012, Wells *et al*. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). Six sperm whale strandings during 2010–2013 were considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 7% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 94 sperm whales died during 2010–2013 (four year annual average of 24) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 47 sperm whales died due to elevated mortality associated with oil exposure. The population model used to predict sperm whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for sperm whales occupying waters outside of the Gulf of Mexico. Proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Also, there was no estimation of uncertainty in model parameters or outputs.

Table 5. Sperm whale strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of sperm whales in Mississippi.

State	2014	2015	2016	2017	2018	Total
Alabama	0	0	1	0	0	1
Florida	0	0	1	0	0	1
Louisiana	0	0	2	1	0	3
Texas	1	0	1	0	0	2
Total	**1**	**0**	**5**	**1**	**0**	**7**

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 16% (95%CI: 11–23) of sperm whales in the Gulf were exposed to oil, that 7% (95%CI: 3–10) of females suffered from reproductive failure, and 6% (95%CI: 2–9) of sperm whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 7% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). Seismic vessel operations in the Gulf of Mexico (commercial and academic) now operate with marine mammal observers as part of required mitigation measures. There have been no reported seismic-related or industry ship-related mortalities or injuries to sperm whales. However, disturbance by anthropogenic noise may prove to be an important habitat issue in some areas of this population's range, notably in areas of oil and gas activities and/or where shipping activity is high. Results from very limited studies of northern Gulf of Mexico sperm whale responses to seismic exploration indicate that sperm whales do not appear to exhibit horizontal avoidance of seismic survey activities (Miller *et al.* 2009, Winsor *et al.* 2017). Data did suggest there may be some decrease in foraging effort during exposure to full-array airgun firing, at least for some individuals. Further study is needed as sample sizes are insufficient at this time (Miller *et al.* 2009). Farmer *et al.* (2018a) developed a bio-energetics model to examine the consequences of frequent disruptions to foraging on sperm whales. The simulations suggested that frequent and severe disruptions could lead to terminal starvation. A follow-up study examined the population level effects of acoustic disturbance in combination with the impacts of the DWH oil spill and suggested that acoustic disturbance could have significant population effects, though terminal starvation and fetal abortions were unlikely (Farmer *et al.* 2018b). Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

Vessel strikes to whales occur world-wide and are a source of injury and mortality. No vessel strikes have been documented in recent years (2014–2018) for sperm whales in the Gulf of Mexico. Historically, one possible sperm whale mortality due to a vessel strike was documented for the Gulf of Mexico. The incident occurred in 1990 in the vicinity of Grande Isle, Louisiana. Deep cuts on the dorsal surface of the whale indicated the vessel strike was probably pre-mortem (Jensen and Silber 2004).

STATUS OF STOCK

The sperm whale is listed as endangered under the Endangered Species Act, and therefore the northern Gulf of Mexico stock is considered strategic under the MMPA. In addition, the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR for this stock. Total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The status of sperm whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock in the northern Gulf of Mexico.

REFERENCES CITED

Antunes, R. 2009. Variation in sperm whale (*Physeter macrocephalus*) coda vocalizations and social structure in the North Atlantic Ocean. Ph.D. dissertation from University of St. Andrews, U.K. 123pp.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Engelhaupt, D., A.R. Hoelzel, C. Nicholson, A. Frantzis, S. Mesnick, S. Gero, H. Whitehead, L. Rendell, P. Miller, R. De Stefanis, A. Cañadas, S. Airoldi and A.A. Mignucci-Giannoni. 2009. Female philopatry in coastal basins and male dispersion across the North Atlantic in a highly mobile marine species, the sperm whale (*Physeter macrocephalus*). Mol. Ecol. 18:4193–4205.

Farmer, N.A., D.P. Noren, E.M. Fougeres, A. Machernis and K. Baker. 2018a. Resilience of the endangered sperm whale *Physeter macrocephalus* to foraging disturbance in the Gulf of Mexico, USA: A bioenergetic approach. Mar. Ecol. Prog. Ser. 589:241–261.

Farmer, N.A., K. Baker, D.G. Zeddies, S.L. Denes, D.P. Noren, L.P. Garrison, A. Machernis, E.M. Fougères and M. Zykov. 2018b. Population consequences of disturbance by offshore oil and gas activity for endangered sperm whales (*Physeter macrocephalus*). Biol. Conserv. 227:189–204.

Fulling, G.L., K.D. Mullin and C.W. Hubard. 2003. Abundance and distribution of cetaceans in outer continental shelf waters of the U.S. Gulf of Mexico. Fish. Bull. 101:923–932.

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gero, S., J. Gordon, C. Carlson, P. Evans and H. Whitehead. 2007. Population estimate and inter-island movement of sperm whales, *Physeter macrocephalus*, in the Eastern Caribbean Sea. J. Cetacean Res. Manage. 9(2):143–150.

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Gordon, J., L. Rendell, R. Antunes, N. Jaquet, C. Richter and B. Würsig. 2008. Analysis of codas from the Gulf of Mexico and implications for management. Pages 201–213 *in:* Jochens, A., D. Biggs, K. Benoit-Bird, D. Engelhaupt, J. Gordon, C. Hu, N. Jaquet, M. Johnson, R. Leben, B. Mate, P. Miller, J. Ortega-Ortiz, A. Thode, P. Tyack and B Würsig. Sperm whale seismic study in the Gulf of Mexico: Synthesis report. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 2008-006. 341pp.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in*: R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jaquet, N. 2006. A simple photogrammetric technique to measure sperm whales at sea. Mar. Mamm. Sci. 22(4):862–879.

Jaquet, N. and D. Gendron. 2009. The social organization of sperm whales in the Gulf of California and comparisons with other populations. J. Mar. Biol. Assoc. U.K. 89(5):975–983.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Jensen, A. S. and G. K. Silber. 2004. Large whale ship strike database. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-OPR-25. 37pp.

Jochens, A., D. Biggs, K. Benoit-Bird, D. Engelhaupt, J. Gordon, C. Hu, N. Jaquet, M. Johnson, R. Leben, B. Mate, P. Miller, J. Ortega-Ortiz, A. Thode, P. Tyack and B. Würsig. 2008. Sperm whale seismic study in the Gulf of Mexico: Synthesis report. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana. OCS Study MMS 2008-006. 341pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009):

Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112:161–175.
Lowery, G.H., Jr. 1974. The mammals of Louisiana and its adjacent waters. Louisiana State University Press, Baton Rouge, Louisiana. 565pp.
Maze-Foley, K. and K.D. Mullin 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.
Miller, P.J.O., M.P. Johnson, P.T. Madsen, N. Biassoni, M. Quero and P. L. Tyack. 2009. Using at-sea experiments to study the effects of airguns on the foraging behavior of sperm whales in the Gulf of Mexico. Deep-Sea Res. I 56:1168–1181.
Mullin, K.D. 2007. Abundance of cetaceans in the oceanic northern Gulf of Mexico from 2003 and 2004 ship surveys. NOAA Southeast Fisheries Science Center, 3209 Frederic Street, Pascagoula, Mississippi 39567. PRBD Contribution #PRBD-2016-03. 27pp.
Mullin, K.D. and G.L. Fulling 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.
Mullin, K.D. and W. Hoggard 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in*: R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana. OCS Study MMS 96-0027.
Mullin, K.D., D. Engelhaupt, C.E. Cates and N.B. Barros 2003. Sperm whale research in the Gulf of Mexico. International Whaling Commission Working Paper SC/55/O15. Available from: NMFS, Southeast Fisheries Science Center, P.O. Drawer 1207, Pascagoula, MS 39568.
Mullin, K.D., W. Hoggard and L.J. Hansen 2004. Abundance and seasonal occurrence of cetaceans in outer continental shelf and slope waters of the north-central and northwestern Gulf of Mexico. Gulf of Mexico Science 2004(1):62–73.
Mullin, K., W. Hoggard, C. Roden, R. Lohoefener, C. Rogers and B. Taggart 1994. Cetaceans on the upper continental slope in the north-central Gulf of Mexico. Fish. Bull. 92:773–786.
Mullin, K.D. and G.L. Fulling 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.
NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892
Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: Time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.
Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.
Ortega Ortiz, J.G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. thesis. Texas A&M University. 170pp.
Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.
Reeves, R.R., J.N. Lund, T.D. Smith and E.A. Josephson. 2011. Insights from whaling logbooks on whales, dolphins, and whaling in the Gulf of Mexico. Gulf Mex. Sci. 29(1):41–67.
Rendell, L. E. and H. Whitehead. 2001. Culture in whales and dolphins. Behav. Brain Sci. 24:309–382.
Rendell, L. and H. Whitehead. 2003. Vocal clans in sperm whales (*Physeter macrocephalus*). Proc. R. Soc. Lond. (Biol) 270:225–231.
Rice, D.W. 1989. Sperm whale, *Physeter macrocephalus* Linnaeus, 1758. Pages 177-233 *in*: S. H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 4: River dolphins and the larger toothed whales. Academic Press, London.
Richter, C., J. Gordon, N. Jaquet and B. Würsig. 2008. Social structure of sperm whales in the northern Gulf of Mexico. Gulf Mex. Sci. 26(2):118–123.
Schmidly, D.J. 1981. Marine mammals of the southeastern United States and the Gulf of Mexico. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. FWS/OBS-80/41. 165pp.
Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman

and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.
Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.
Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.
Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.
Townsend, C.H. 1935. The distribution of certain whales as shown by logbook records of American whale ships. Zoologica 19:1–50.
Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.
Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.
Wade, P.R. and R.P. Angliss 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.
Watkins, W. A. and W. E. Schevill. 1977. Sperm whale codas. J. Acoust. Soc. Am. 62:1486–1490.
Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.
Whitehead, H. 2002. Estimates of the current global population size and historical trajectory for sperm whales. Mar. Ecol. Prog. Ser. 242:295–304.
Whitehead, H. 2003. Sperm whales: Social evolution in the ocean. The University of Chicago Press, Chicago, Illinois. 431pp.
Whitehead, H. and L. Weilgart. 1991. Patterns of visually observable behaviour and vocalizations in groups of female sperm whales. Behaviour. 118:275–296.
Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.
Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.
Winsor, M.H., L.M. Irvine and B.R. Mate. 2017. Analysis of the spatial distribution of satellite-tagged sperm whales (*Physeter macrocephalus*) in close proximity to seismic surveys in the Gulf of Mexico. Aquat. Mamm. 43(4):439–446.

BRYDE'S WHALE (*Balaenoptera edeni*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Bryde's whales are distributed worldwide in tropical and sub-tropical waters, but the taxonomy and number of species and/or subspecies of Bryde's whales in the world is currently a topic of debate (Kato and Perrin 2008, Rosel and Wilcox 2014). In the western Atlantic Ocean, Bryde's whales are reported from the Gulf of Mexico and the southern West Indies to Cabo Frio, Brazil (Leatherwood and Reeves 1983). Sighting records and acoustic detections of Bryde's whales in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur almost exclusively in the northeastern Gulf in the De Soto Canyon area, along the continental shelf break between 100 m and 400 m depth, with a single sighting at 408 m (Figure 1; Hansen *et al.* 1996, Mullin and Hoggard 2000, Mullin and Fulling 2004, Maze-

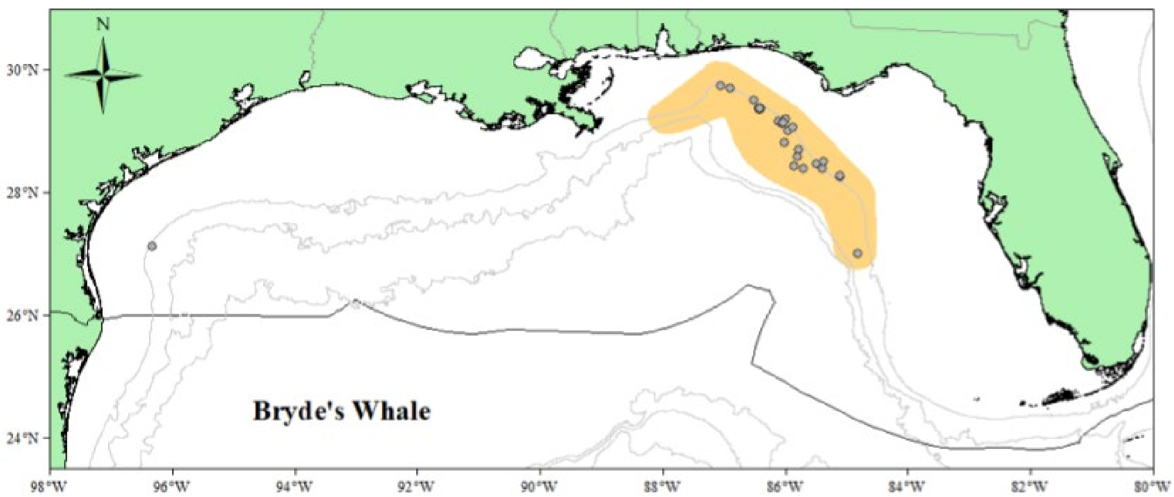


Figure 1. Distribution of all Bryde's whale sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ. The shaded area indicates the core habitat.

Foley and Mullin 2006, Rice *et al.* 2014; Rosel and Wilcox 2014; Širović *et al.* 2014; Rosel *et al*. 2016; Soldevilla *et al.* 2017). Bryde's whales have been sighted in all seasons within the De Soto Canyon area (Mullin and Hoggard 2000, Maze-Foley and Mullin 2006, Mullin 2007, DWH MMIQT 2015). Genetic analysis suggests that Bryde's whales from the northern Gulf of Mexico represent a unique evolutionary lineage distinct from other recognized Bryde's whale subspecies, including those found in the southern Caribbean and southwestern Atlantic off Brazil (Rosel and Wilcox 2014). The geographic distribution of this Bryde's whale form has not yet been fully identified. Two strandings from the southeastern U.S. Atlantic coast share the same genetic characteristics with those from the northern Gulf of Mexico (Rosel and Wilcox 2014), but it is unclear whether these are extralimital strays (Mead 1977) or whether they indicate the population extends from the northeastern Gulf of Mexico to the Atlantic coast of the southern U.S. (Rosel and Wilcox 2014). There have been no confirmed sightings of Bryde's whales along the U.S. east coast during NMFS cetacean surveys (Rosel *et al*. 2016).

Historical whaling records from the 1800s suggest Bryde's whales may have been more common in the U.S. waters of the north central Gulf of Mexico and in the southern Gulf of Mexico in the Bay of Campeche (Reeves *et al.* 2011). How regularly they currently use U.S. waters of the western Gulf of Mexico is unknown. There has been only one confirmed sighting of a Gulf of Mexico Bryde's whale in this region, a whale observed during a 2017 NMFS vessel survey off Texas, despite substantial NMFS survey effort in the north central and western Gulf dating back to the early 1990s (e.g., Hansen *et al.* 1996; Mullin and Hoggard 2000; Mullin and Fulling 2004; Maze-Foley and Mullin 2006). A compilation of available records of cetacean sightings, strandings, and captures in Mexican waters of the southern Gulf of Mexico identified no Bryde's whales (Ortega-Ortiz 2002). There are insufficient data to determine whether it is plausible the stock contains multiple demographically independent populations that should be separate stocks.

POPULATION SIZE

The best abundance estimate available for Bryde's whales in the northern Gulf of Mexico is 51 (CV=0.50; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of Bryde's whale abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than "closing" mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=0 (CV=NA); 2004, N=64 (CV=0.88); and 2009, N=100 (CV=1.03).

Recent Surveys and Abundance Estimates

An abundance estimate for Bryde's whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Due to the restricted habitat range of Gulf of Mexico Bryde's whales, survey effort was re-stratified to include only effort within their core habitat area (Figure 1; https://www.fisheries.noaa.gov/resource/map/gulf-mexico-brydes-whale-core-distribution-area-map-gis-data) including 941 km of effort in 2017 and 848 km of effort in 2018. In addition, there was an insufficient number of Bryde's whale sightings during these surveys to develop an appropriate detection probability function. Therefore, a detection function was derived based on 91 sightings of Bryde's whale groups observed during SEFSC large vessel surveys between 2003 and 2019. The abundance estimates include unidentified large whales and baleen whales observed within the Bryde's whale habitat. However, the estimate does not include the sighting of a confirmed Bryde's whale in the western Gulf of Mexico in 2017. It is not possible to extrapolate estimated density beyond the core area since little is known about habitat use and distribution outside of this area. Estimates of abundance were derived using MCDS distance sampling methods that account for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. The 2017 and 2018 estimates were N=84 (CV=0.92) and N=40 (CV=0.55), respectively. The inverse variance weighted mean abundance for Bryde's whales in oceanic waters during 2017 and 2018 was 51 (CV=0.50; Table 1; Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline because there was only one resighting and few sightings overall of Bryde's whales during the two-team surveys.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of Bryde's whales in northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	51	0.50

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Bryde's whales is 51 (CV=0.50). The minimum population estimate for the northern Gulf of Mexico Bryde's whale is 34 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

All verified Bryde's whale sightings, with one exception, have occurred in a very restricted area of the northeastern Gulf (Figure 1) during surveys that uniformly sampled the entire oceanic northern Gulf. Because the population size is small, in order to effectively monitor trends in Bryde's whale abundance in the future, other methods need to be used.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations likely do not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995). Between 1988 and 2018, there have been two documented strandings of calves (total length <700 cm) in the northern Gulf of Mexico (SEUS Historical Stranding Database unpublished data; NOAA National Marine Mammal Health and Stranding Response Database unpublished data).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997; Wade 1998). The minimum population size is 34. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.1 because the stock is listed as endangered. PBR for the northern Gulf of Mexico Bryde's whale is 0.1 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico Bryde's whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
51	0.50	34	0.1	0.04	0.1

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual estimated fishery-related mortality and serious injury for the Gulf of Mexico Bryde's whale stock during 2014–2018 is unknown. There was no documented fishery-caused mortality or serious injury for this stock during 2014–2018 (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 0.5. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2015 was, therefore, 0.5. This is considered a minimum mortality estimate as some fisheries with which the stock could interact have limited observer coverage. In addition, the likelihood is low that a whale killed at sea due to a fishery interaction or vessel-strike will be recovered (Williams *et al.* 2011).

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Bryde's whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	Unknown	-

Fisheries Information

There are three commercial fisheries that overlap geographically and potentially could interact with this stock in the Gulf of Mexico. These include the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery, and two Category III fisheries, the Southeastern U.S. Atlantic, Gulf of Mexico shark bottom longline/hook-and-line fishery and the Southeastern U.S. Atlantic, Gulf of Mexico, and Caribbean snapper-grouper and other reef fish bottom longline/hook-and-line fishery. See Appendix III for detailed fishery information. All three of these fisheries have observer programs, however observer coverage is limited for the two Category III fisheries.

Pelagic swordfish, tunas, and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to Bryde's whales by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b). Percent observer coverage (percentage of sets observed) for this longline fishery for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. For the two category III bottom longline/hook-and-line fisheries, the target species are large and small coastal sharks and reef fishes such as snapper, grouper, and tilefish. There has been no reported fishery-related mortality or serious injury of a Bryde's whale by either of these fisheries (e.g., Scott-Denton *et al.* 2011; Gulak *et al.* 2013, 2014; Enzenauer *et al.* 2015, 2016; Mathers *et al.* 2017, 2018, 2020). Within the Gulf of Mexico, observer coverage for the snapper-grouper and other reef fish bottom longline fishery is ~1% or less annually, and for the shark bottom longline fishery coverage is 1–2% annually. Usually bottom longline gear is thought to pose less of a risk for cetaceans to become entangled than pelagic longline gear. However, if cetaceans forage along the seafloor, as is suspected for the Bryde's whale (Soldevilla *et al.* 2017), then there is an opportunity for these whales to become entangled in the mainline as well as in the vertical buoy lines (Rosel *et al.* 2016).

Two other commercial fisheries that overlap to a small degree with the primary Bryde's whale habitat in the northeastern Gulf of Mexico are the Category III Gulf of Mexico butterfish trawl fishery and Category II Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl fishery (Rosel *et al.* 2016). No interactions with Bryde's whales have been documented for either of these fisheries. There is no observer coverage for the butterfish trawl fishery. The shrimp trawl fishery has ~2% observer coverage annually.

Other Mortality

There were no reported strandings of Bryde's whales in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012, Wells *et al*. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014;

http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). Two Bryde's whale strandings in 2012 were considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Based on the population model, it was projected that 2.3 Bryde’s whales died during 2014–2018 (see Appendix VI) due to elevated mortality associated with oil exposure and that the stock experienced a 22% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The DWH Marine Mammal Injury Quantification Team cautioned that the capability of Bryde's whales to recover from the DWH oil spill is unknown because the population models do not account for stochastic processes and genetic effects (DWH MMIQT 2015), to which small populations are highly susceptible (Shaffer 1981; Rosel and Reeves 2000). The population model used to predict Bryde's whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for Bryde's whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The *DWH* MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days, ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the NRDA process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 48% of Bryde's whales in the Gulf were exposed to oil, that 22% (95%CI: 10–31) of females suffered from reproductive failure, and 18% (95%CI: 7–28) of the population suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 22% reduction in population size (see Other Mortality section above).

Vessel strikes also pose a threat to this stock (Soldevilla *et al.* 2017). In 2009, a Bryde’s whale was found floating in the Port of Tampa, Tampa Bay, Florida. The whale had evidence of pre-mortem and post-mortem blunt trauma, and was determined to have been struck by a vessel, draped across the bow, and carried into port.

Anthropogenic sound in the world’s oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

The Bryde's whale is listed as endangered under the Endangered Species Act, and therefore the northern Gulf of Mexico stock is considered strategic under the MMPA. The stock is very small and exhibits very low genetic diversity, which places the stock at great risk of demographic stochasticity. The stock’s restricted range also places it at risk of environmental stochasticity. In addition, the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR for this stock. The status of Bryde’s whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan.

Enzenauer, M.P., B.M. Deacy and J.K. Carlson. 2015. Characterization of the shark bottom longline fishery, 2014. NOAA Tech. Memo. NMFS-SEFSC-677. 24pp.

Enzenauer, M.P., B.M. Deacy and J.K. Carlson. 2016. Characterization of the shark bottom longline fishery, 2015. NOAA Tech. Memo. NMFS-SEFSC-689. 23pp.

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp.

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Gulak, S.J.B., M.P. Enzenauer and J.K. Carlson. 2013. Characterization of the shark and reef fish bottom longline fisheries: 2012. NOAA Tech. Memo. NMFS-SEFSC-652. 42pp.

Gulak, S.J.B., M.P. Enzenauer and J.K. Carlson. 2014. Characterization of the shark and reef fish bottom longline fisheries: 2013. NOAA Tech. Memo. NMFS-SEFSC-658. 22pp.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Kato, H. and W.F. Perrin. 2008. Bryde's whales *Balaenoptera edeni/brydei*. Pages 158–163 *in:* W.F. Perrin, B. Würsig and J.G.M. Thewissen (Eds.). Encyclopedia of Marine Mammals, Second Edition. Academic Press, San Diego, California. 1352pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108-189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake, and L. Thomas. Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112:161–175.

Mathers, A.N., B.M. Deacy, M.P. Enzenauer and J.K. Carlson. 2017. Characterization of the shark bottom longline fishery, 2016. NOAA Tech. Memo. NMFS-SEFSC-714. 23pp.

Mathers, A.N., B.M. Deacy, H.E. Moncrief-Cox and J.K. Carlson. 2018. Characterization of the shark bottom longline fishery, 2017. NOAA Tech. Memo. NMFS-SEFSC-727. 21pp.

Mathers, A.N., B.M. Deacy, H.E. Moncrief-Cox and J.K. Carlson. 2020. Catch and bycatch in U.S. Southeast gillnet fisheries, 2018. NOAA Tech. Memo. NMFS-SEFSC-743. 15pp.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Mead, J.G. 1977. Records of Sei and Bryde's whales from the Atlantic Coast of the United States, the Gulf of Mexico, and the Caribbean. Rep. Int. Whal. Comm. (Special Issue) 1:113–116.

Mullin, K.D. 2007. Abundance of cetaceans in the oceanic northern Gulf of Mexico from 2003 and 2004 ship surveys. NOAA Southeast Fisheries Science Center, 3209 Frederic Street, Pascagoula, MS 39567. PRBD Contribution #PRBD-2016-03. 27pp.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Mullin, K.D. and G.L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: Time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Ortiz, J. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Reeves, R.R., J.N. Lund, T.D. Smith and E.A. Josephson. 2011. Insights from whaling logbooks on whales, dolphins, and whaling in the Gulf of Mexico. G. Mex. Sci. 29(1):41–67.

Rice, A.N., J.T. Tielens, K.J. Palmer, A. Muirhead and C.W. Clark. 2014. Potential Bryde's whale calls (*Balaenoptera edeni*) recorded in the northern Gulf of Mexico. J. Acoust. Soc. Amer. 135:3066–3076.

Rosel, P.E. and R.R. Reeves. 2000. Genetic and demographic considerations for the conservation of Asian river cetaceans. Pages 144–152 *in:* R.R. Reeves, B.D. Smith and T. Kasuya (Eds.). Biology and conservation of freshwater cetaceans in Asia. IUCN, Gland, Switzerland and Cambridge, UK.

Rosel, P.E. and L.A. Wilcox. 2014. Genetic evidence reveals a unique lineage of Bryde's whales in the northern Gulf of Mexico. Endang. Species Res. 25:19–34.

Rosel, P.E., P. Corkeron, L. Engleby, D. Epperson, K.D. Mullin, M.S. Soldevilla and B.L. Taylor. 2016. Status review of Bryde's whales (*Balaenoptera edeni*) in the Gulf of Mexico under the Endangered Species Act. NOAA Tech. Memo. NMFS-SEFSC-692. 133pp.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Scott-Denton, E., P.F. Cryer, J.P. Gocke, M.R. Harrelson, D.L. Kinsella, J.R. Pulver, R.C. Smith and J.A. Williams. 2011. Descriptions of the U.S. Gulf of Mexico reef fish bottom longline and vertical line fisheries based on observer data. Mar. Fish. Rev. 73(2):1–26.

Shaffer, M.L. 1981. Minimum population sizes for species conservation. BioScience 31:131–134.

Širović, A., H.R. Bassett, S.C. Johnson, S.M. Wiggins and J.A. Hildebrand. 2014. Bryde's whale calls recorded in the Gulf of Mexico. Mar. Mamm. Sci. 30(1):399–409.

Soldevilla, M.S., J.A. Hildebrand, K.E. Frasier, L.A. Dias, A. Martinez, K.D. Mullin, P.E. Rosel and L.P. Garrison. 2017. Spatial distribution and dive behavior of Gulf of Mexico Bryde's whales: Potential risk of vessel strikes and fisheries interactions. Endang. Species Res. 32:533–550.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23:157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Mar. Mamm. Sci. 14(1):1–37.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

CUVIER'S BEAKED WHALE (*Ziphius cavirostris*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Cuvier's beaked whales are distributed throughout the world's oceans except for the polar regions (Leatherwood and Reeves 1983, Heyning 1989). Strandings have occurred in all months along the east coast of the U.S. (Schmidly 1981) and throughout the year in the Gulf of Mexico (Würsig *et al*. 2000). In the northern Gulf of Mexico, Cuvier's beaked whales are seen primarily in waters ≥1,000 m (Figure 1) and have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Hansen *et al*. 1996, Mullin and Hoggard 2000, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). Beaked whale sightings made during spring and summer vessel surveys have been widely distributed in waters >500 m deep (Maze-Foley and Mullin 2006).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g.,

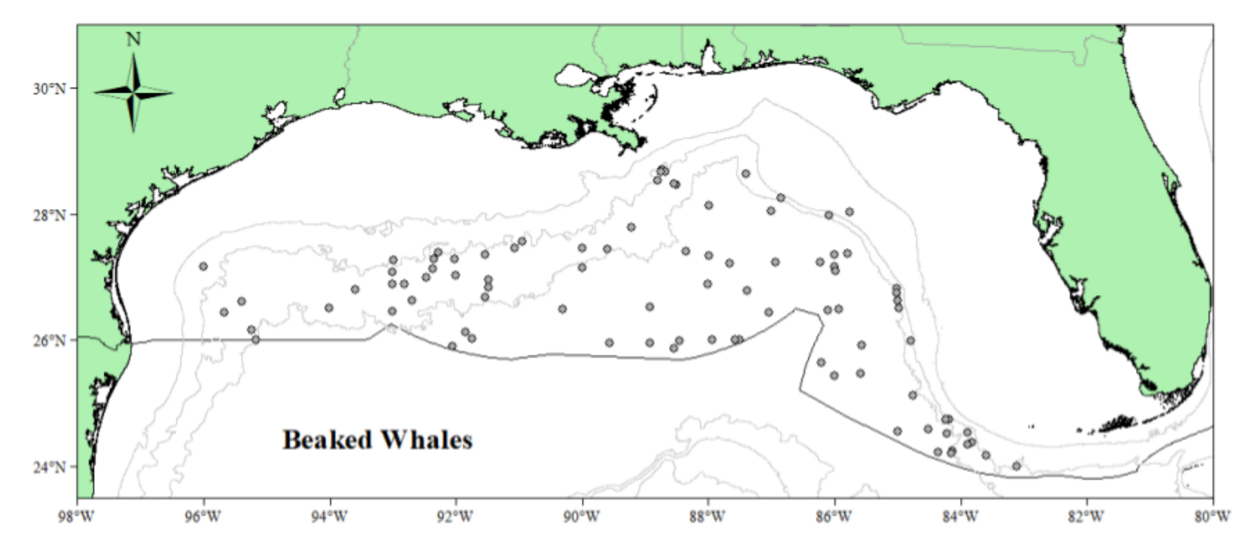


Figure 1. Distribution of beaked whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al*. 2005; Jefferson *et al*. 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Cuvier's beaked whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding *et al.* 2007,Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for Cuvier's beaked whales in the northern Gulf of Mexico is 18 (CV=0.75; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The estimate for the same time period for unidentified Ziphiids was 181 (CV=0.31), which may have also included an unknown number of Cuvier's beaked whales.

Earlier Abundance Estimates

Five point estimates of all Ziphiid (i.e., unidentified Ziphiids, *Mesoplodon* spp., Cuvier's beaked whale, and Gervais' beaked whale combined) abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than "closing" mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences. This resulted in revised abundance estimates for all Ziphiids of: 2003, N=573 (CV=0.44); 2004, N=55 (CV=0.72); and 2009, N=276 (CV=0.59; Garrison *et al.* 2020).

Recent Surveys and Abundance Estimates

An abundance estimate for Cuvier's beaked whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of this species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21, Laake *et al.* 2020) in the R statistical programming language. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The 2017 and 2018 estimates were N=12 (CV=1.01) and N=24 (CV=1.01), respectively. The inverse variance weighted mean abundance estimate for Cuvier's beaked whales in oceanic waters during 2017 and 2018 was 18 (CV=0.75; Table 1; Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The 2017 and 2018 estimates for unidentified Ziphiids were N=165 (CV=0.47) and N=193 (0.65), respectively, and the inverse variance weighted mean abundance estimate for unidentified Ziphiids was 181 (CV=0.31), which may have also included an unknown number of Cuvier's beaked whales. The 2017 and 2018 estimates for all Ziphiids (i.e., unidentified Ziphiids, *Mesoplodon* spp., Cuvier's beaked whale, and Gervais' beaked whale combined) were N=303 (CV=0.28) and N=322 (CV=0.34), respectively.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of Cuvier's beaked whales in the northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	18	0.75

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Cuvier's beaked whales is 18 (CV=0.75). The minimum population estimate for the northern Gulf of Mexico Cuvier's beaked whale is 10 (Table 2).

Current Population Trend

Using revised abundance estimates for all Ziphiids for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates for all Ziphiids, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2004 estimates (p.adjusted=0.012) and the 2004 and 2018 estimates (p.adjusted=0.067; Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for the Cuvier's beaked whale is 10. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor for this stock is 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico Cuvier's beaked whale is 0.1 (Table 2).

Table 2. Best and minimum abundance estimates for Gulf of Mexico Cuvier's beaked whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
18	0.75	10	0.5	0.04	0.1

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Cuvier's beaked whales or unidentified beaked whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for all beaked whales due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 5.2. The minimum total mean annual human-caused mortality and serious injury for beaked whales during 2014–2018 was, therefore, 5.2. This is a combined estimate for Blainville's, Gervais', and Cuvier's beaked whales. The minimum total mean annual human-caused mortality and serious injury for Cuvier's beaked whale is unknown.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Cuvier's beaked whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of beaked whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the pelagic longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to Cuvier's or other beaked whales by this fishery during 2014–2018 (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There was one reported stranding of a Cuvier's beaked whale in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). The whale stranded in 2014 in Florida, and it could not be determined if there was evidence of human interaction. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no Cuvier's beaked whale strandings recovered within the spatial and temporal boundaries of this UME. A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, this model estimated that the stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 51 beaked whales died during 2010–2013 (four year annual average of 13) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 26 beaked whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict beaked whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for beaked whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 12% (95%CI: 2–22) of beaked whales in the Gulf, which included Blainville's, Cuvier's and Gervais' beaked whales, were exposed to oil, that 5% (95%CI: 3–8) of females suffered from reproductive failure, and 4% (95%CI: 2–7) of the beaked whale populations suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). Anthropogenic noise, particularly from military sonar, shipping, and seismic testing, is an increasing habitat concern for beaked whales (Aguilar de Soto *et al.* 2006; Cox *et al.* 2006; McCarthy *et al.* 2011; Tyack *et al.* 2011; Joyce *et al.* 2020). Several mass strandings of beaked whales throughout their worldwide range have been associated with naval activities (D'Amico *et al.* 2009; Filadelfo *et al.* 2009). In March 2000, 14 beaked whales live stranded in the Bahamas. Six of the whales (5 Cuvier's and 1 Blainville's) died and necropsy revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand (Balcomb and Claridge 2001; NMFS 2001; Cox *et al*. 2006). Fourteen beaked whales (mostly Cuvier's beaked whales but also including Gervais' and Blainville's beaked whales) stranded in the Canary Islands in 2002 (Cox *et al.* 2006, Fernandez *et al.* 2005; Martin *et al.* 2004). Gas bubble-associated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez *et al.* 2005). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown. Finally, ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach contents of stranded beaked whales.

STATUS OF STOCK

Cuvier's beaked whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all beaked whale species combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Cuvier's beaked whales in the northern Gulf of Mexico, relative to OSP, is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED

Aguilar de Soto, N., M. Johnson, P.T. Madsen, P.L. Tyack, A. Bocconcelli and J. Fabrizio Borsani. 2006. Does intense ship noise disrupt foraging in deep-diving Cuvier's beaked whales (*Ziphius cavirostris*)? Mar. Mamm. Sci. 22(3):690–699.

Balcomb, K.C.I. and D.E. Claridge. 2001. A mass stranding of cetaceans caused by naval sonar in the Bahamas. Bahamas J. Sci. 2:2–12.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

Cox, T.M., T.J. Ragen, A.J. Read, E. Vos, R.W. Baird, K. Balcomb, J. Barlow, J. Caldwell, T. Cranford, L. Crum, A. D'Amico, G. D'Spain, A. Fernandez, J. Finneran, R. Gentry, W. Gerth, F. Gulland, J. Hildebrand, D. Houser, T. Hullar, P.D. Jepson, D. Ketten, C.D. MacLeod, P. Miller, S. Moore, D.C. Mountain, D. Palka, P. Ponganis, S. Rommel, T. Rowles, B. Taylor, P. Tyack, D. Wartzok, R. Gisiner, J. Mead and L. Benner. 2006. Understanding the impacts of anthropogenic sound on beaked whales. J. Cetacean Res. Manage. 7(3):177–187.

D'Amico, A., R.C. Gisiner, D.R. Ketten, J.A. Hammock, C. Johnson, P.L. Tyack and J. Mead. 2009. Beaked whale strandings and naval exercises. Aquat. Mamm. 35(4):452–472.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Fernandez, A., J.F. Edwards, F. Rodriguez, A.E. de los Monteros, P. Herraez, P. Castro, J.R. Jaber, V. Martin and M. Arbelo. 2005. "Gas and Fat Embolic Syndrome" involving a mass stranding of beaked whales (Family Ziphiidae) exposed to anthropogenic sonar signals. Vet. Path. 42(4):446–457.

Filadelfo, R., J. Mintz, E. Michlovich, A. D'Amico, P.L. Tyack and D.R. Ketten. 2009. Correlating military sonar use with beaked whale mass strandings: What do the historical data show? Aquat. Mamm. 35(4):435–444.

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA.

Heyning, J.E. 1989. Cuvier's beaked whale *Ziphius cavirostris* G. Cuvier, 1923. Pages 289–308 *in*: S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 4: River dolphins and the larger toothed whales. Academic Press, London. 442pp.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1): 27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Joyce, T.W., J.W. Durban, D.E. Claridge, C.A. Dunn, L.S. Hickmott, H. Fearnbach, K. Dolan and D. Moretti. 2020. Behavioral responses of satellite tracked Blainville's beaked whales (*Mesoplodon densirostris*) to mid-frequency active sonar. Mar. Mamm. Sci. 36(1):29–46.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake, and L. Thomas. Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Martín, V., A. Servidio and S. García 2004. Mass strandings of beaked whales in the Canary Islands. ECS Newsletter 42:33–6.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

McCarthy, E., D. Moretti, L. Thomas, N. DiMarzio, R. Morrissey, S. Jarvis, J. Ward, A. Izzi and A. Dilley. 2011. Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (*Mesoplodon densirostris*) during multiship exercises with mid-frequency sonar. Mar. Mamm. Sci. 27:E206–E226.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Mullin, K.D. and G.L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indic. 18:278–290.

Schmidly, D.J. 1981. Marine mammals of the southeastern United States and the Gulf of Mexico. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC, FWS/OBS-80/41.165pp.

Schwacke , L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Tyack, P.L., W.M. Zimmer, D. Moretti, B.L. Southall, D.E. Claridge, J.W. Durban, C.W. Clark, A. D'Amico, N. DiMarzio, S. Jarvis and E. McCarthy. 2011. Beaked whales respond to simulated and actual navy sonar. PloS One 6:e17009.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

Würsig, B., T.A. Jefferson and D.J. Schmidly. 2000. The marine mammals of the Gulf of Mexico. Texas A&M University Press, College Station, Texas.

BLAINVILLE'S BEAKED WHALE (*Mesoplodon densirostris*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Three species of *Mesoplodon* are known to occur in the Gulf of Mexico, based on stranding or sighting data (Hansen *et al.* 1995, Würsig *et al.* 2000). These are Blainville's beaked whale (*M. densirostris*), Gervais' beaked whale (*M. europaeus*) and Sowerby's beaked whale (*M. bidens*). Sowerby's beaked whale in the Gulf of Mexico is considered extralimital because there is only one known stranding of this species (Bonde and O'Shea 1989) and because it normally occurs in northern temperate waters of the North Atlantic (Mead 1989). The possibility of another unknown species of *Mesoplodon* inhabiting the Gulf has been suggested based on passive acoustic recordings (Hildebrand *et al.* 2015).

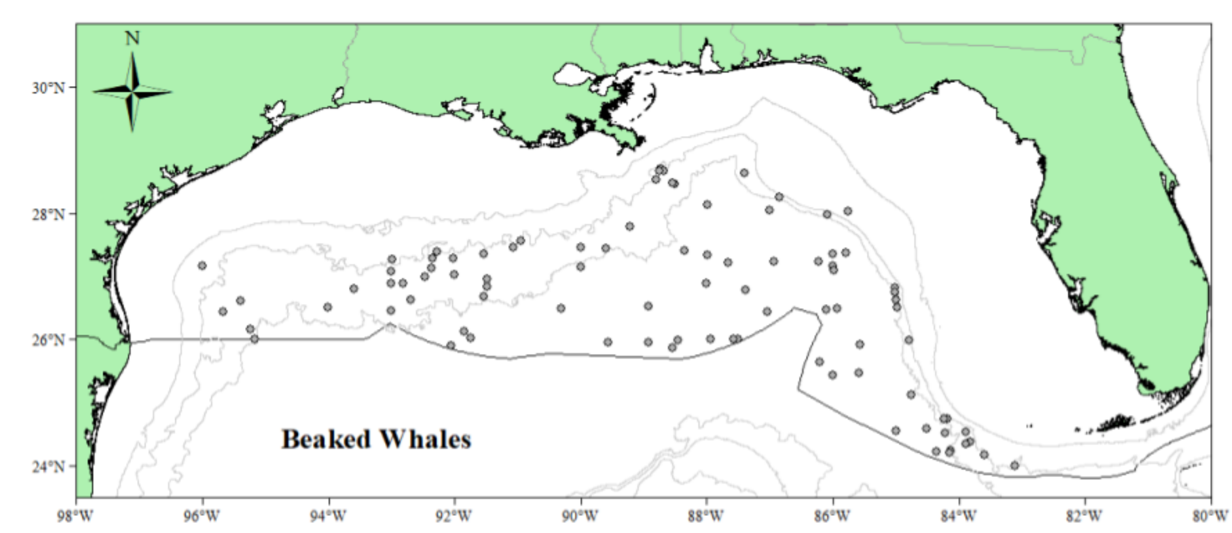


Figure 1. Distribution of beaked whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

Blainville's beaked whales appear to be widely but sparsely distributed in temperate and tropical waters of the world's oceans (Leatherwood *et al.* 1976, Leatherwood and Reeves 1983). Strandings have occurred along the northwestern Atlantic coast from Florida to Nova Scotia (Schmidly 1981), and in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Würsig *et al.* 2000). Because at-sea identification of *Mesoplodon* to species in the Gulf of Mexico is very difficult, sightings of beaked whales (Family Ziphiidae) made during visual surveys are often identified only as *Mesoplodon* sp. or unidentified Ziphiids, and are referred to more generically as 'beaked whales.' In the northern Gulf of Mexico, beaked whales are sighted most commonly in waters ≥1,000 m and they have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Hansen *et al.* 1996, Mullin and Hoggard 2000, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020; Figure 1). There are several confirmed sightings of Blainville's beaked whales, two in the western Gulf and one off the Florida shelf (Garrison and Aichinger Dias 2020).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011).

This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Blainville's beaked whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The total number of Blainville's beaked whales in the northern Gulf of Mexico is unknown. The best abundance estimate (Nest) is for *Mesoplodon* spp., and is a combined estimate for Blainville's beaked whale and Gervais' beaked whale. The estimate of abundance for *Mesoplodon* spp. from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ is 98 (CV=0.46; Table 1; Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The estimate for the same time period for unidentified Ziphiids was 181 (CV=0.31), which may have also included an unknown number of Blainville's beaked whales.

Earlier Abundance Estimates

Five point estimates of all Ziphiid (i.e., unidentified Ziphiids, *Mesoplodon* spp., Cuvier's beaked whale, and Gervais' beaked whale combined) abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than "closing" mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences. This resulted in revised abundance estimates for all Ziphiids of: 2003, N=573 (CV=0.44); 2004, N=55 (CV=0.72); and 2009, N=276 (CV=0.59; Garrison *et al.* 2020).

Recent Surveys and Abundance Estimates

An abundance estimate for *Mesoplodon* spp. was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200 m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of these species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21, Laake *et al.* 2020) in the R statistical programming language. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The 2017 and 2018 estimates for *Mesoplodon* spp. were N=127 (CV=0.61) and N=65 (CV=0.65), respectively. The inverse variance weighted mean abundance estimate for *Mesoplodon* spp. in oceanic waters during 2017 and 2018 was 98 (CV=0.46; Table 1; Garrison *et al.* 2020). This was a combined estimate for Blainville's and Gervais' beaked whales. This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of these species. The 2017 and 2018 estimates for unidentified Ziphiids were N=165 (CV=0.47) and N=193 (0.65), respectively, and the inverse variance weighted mean abundance estimate for the same time period for unidentified Ziphiids was 181 (CV=0.31), which may have also included an unknown number of Blainville's beaked whales. The 2017 and 2018 estimates for all Ziphiids (i.e., unidentified Ziphiids, *Mesoplodon* spp., Cuvier's beaked whale, and Gervais' beaked whale combined) were N=303 (CV=0.28) and N=322 (CV=0.34), respectively.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of Mesoplodon spp. in the northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	U.S. Gulf of Mexico	98	0.46

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for *Mesoplodon* spp. in the northern Gulf of Mexico is 98 (CV=0.46). The minimum population estimate for *Mesoplodon* spp. in the northern Gulf of Mexico is 68 (Table 2).

Current Population Trend

Using revised abundance estimates for all Ziphiids for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates for all Ziphiids, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2004 estimates (p.adjusted=0.012) and the 2004 and 2018 estimates (p.adjusted=0.067; Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for *Mesoplodon* spp. is 68. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico *Mesoplodon* spp. is 0.7 (Table 2). It is not possible to determine the PBR for only Blainville's beaked whales.

Table 2. Best and minimum abundance estimates for Gulf of Mexico Mesoplodon spp. with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
98	0.46	68	0.5	0.04	0.7

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Blainville's beaked whales or unidentified beaked whales from U.S. fisheries in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for all beaked whales due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 5.2. The minimum total mean annual human-caused mortality and serious injury for beaked whales during 2014–2018 was, therefore, 5.2. This is a combined estimate for Blainville's, Gervais', and Cuvier's beaked whales. The minimum total mean annual human-caused mortality and serious injury for Blainville's beaked whale is unknown.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Blainville's beaked whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of beaked whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to Blainville's or other beaked whales by this fishery during 2014–2018 (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There was one stranding of a Blainville's beaked whale during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). This stranding occurred in Florida in 2014, and it could not be determined if there was evidence of human interaction. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 13 bottlenose dolphin die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico, and one of these included a few Blainville's beaked whales. Between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with *K. brevis* blooms and fish kills in the Florida Panhandle (additional strandings included three Atlantic spotted dolphins, *Stenella frontalis*, one Risso's dolphin, *Grampus griseus*, two Blainville's beaked whales, and four unidentified dolphins). Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012, Litz *et al.* 2014). An UME was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). The 2014 stranding of a Blainville's beaked whale was considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, this model estimated that the stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 51 beaked whales died during 2010–2013 (four year annual average of 13) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 26 beaked whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict beaked whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for beaked whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 12% (95%CI: 2–22) of beaked whales in the Gulf, which included Blainville's, Cuvier's and Gervais' beaked whales, were exposed to oil, that 5% (95%CI: 3–8) of females suffered from reproductive failure, and 4% (95%CI: 2–7) of the beaked whale populations suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). Anthropogenic noise, particularly from military sonar, shipping, and seismic testing, is an increasing habitat concern for beaked whales (Aguilar de Soto *et al.* 2006, Cox *et al.* 2006, McCarthy *et al.* 2011, Tyack *et al.* 2011, Joyce *et al.* 2020). Seven Blainville's beaked whales were satellite tagged and tracked in the Bahamas prior to and during naval sonar exercises (Joyce *et al.* 2020). Following exposure to mid-frequency active sonar, five of the whales were displaced 28–68 km from the source, and did not return for two to four days after military exercises ceased. Data also suggested the whales spent less time in deep dives during the early periods of sonar exposure (Joyce *et al.* 2020). In addition, several mass strandings of beaked whales throughout their worldwide range have been associated with naval activities (D'Amico *et al.* 2009, Filadelfo *et al.* 2009). In March 2000, 14 beaked whales live stranded in the Bahamas. Six of the whales (5 Cuvier's and 1 Blainville's) died and necropsy revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand (Balcomb and Claridge 2001, NMFS 2001, Cox *et al.* 2006). Fourteen beaked whales (mostly Cuvier's beaked whales but also including Gervais' and Blainville's beaked whales) stranded in the Canary Islands in 2002 (Cox *et al.* 2006, Fernandez *et al.* 2005, Martin *et al.* 2004). Gas bubble-associated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez *et al.* 2005). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

Finally, ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach contents of stranded beaked whales.

STATUS OF STOCK

Blainville's beaked whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all beaked whale species combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Blainville's beaked whales in the northern Gulf of Mexico, relative to OSP, is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED

Aguilar Soto, N., M. Johnson, P.T. Madsen, P.L. Tyack, A. Bocconcelli and J. Fabrizio Borsani. 2006. Does intense ship noise disrupt foraging in deep-diving Cuvier's beaked whales (*Ziphius cavirostris*)? Mar. Mamm. Sci. 22(3):690–699.

Balcomb, K.C.I. and D.E. Claridge. 2001. A mass stranding of cetaceans caused by naval sonar in the Bahamas. Bahamas J. Sci. 2:2–12.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Bonde, R.K. and T.J. O'Shea. 1989. Sowerby's beaked whale (*Mesoplodon bidens*) in the Gulf of Mexico. J. Mammal. 70:447–449.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning and B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

Cox, T.M., T.J. Ragen, A.J. Read, E. Vos, R.W. Baird, K. Balcomb, J. Barlow, J. Caldwell, T. Cranford, L. Crum, A. D'Amico, G. D'Spain, A. Fernandez, J. Finneran, R. Gentry, W. Gerth, F. Gulland, J. Hildebrand, D. Houser, T. Hullar, P.D. Jepson, D. Ketten, C.D. MacLeod, P. Miller, S. Moore, D.C. Mountain, D. Palka, P. Ponganis, S. Rommel, T. Rowles, B. Taylor, P. Tyack, D. Wartzok, R. Gisiner, J. Mead and L. Benner. 2006. Understanding the impacts of anthropogenic sound on beaked whales. J. Cetacean Res. Manage. 7(3):177–-187.

D'Amico, A., R.C. Gisiner, D.R. Ketten, J.A. Hammock, C. Johnson, P.L. Tyack and J. Mead. 2009. Beaked whale strandings and naval exercises. Aquat. Mamm. 35(4):452–472.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Fernandez, A., J.F. Edwards, F. Rodriguez, A.E. de los Monteros, P. Herraez, P. Castro, J.R. Jaber, V. Martin and M. Arbelo. 2005. "Gas and Fat Embolic Syndrome" involving a mass stranding of beaked whales (Family Ziphiidae) exposed to anthropogenic sonar signals. Vet. Path. 42(4):446–457.

Filadelfo, R., J. Mintz, E. Michlovich, A. D'Amico, P.L. Tyack and D.R. Ketten. 2009. Correlating military sonar use with beaked whale mass strandings: What do the historical data show? Aquat. Mamm. 35(4):435–444.

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin and C.L. Roden. 1995. Estimates of cetacean abundance in the northern Gulf of Mexico from vessel surveys. Southeast Fisheries Science Center, Miami Laboratory, Contribution No. MIA-94/95-25. 9pp.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Hildebrand, J.A., S. Baumann-Pickering, K.E. Frasier, J.S. Trickey, K.P. Merkens, S.M. Wiggins, M.A. McDonald, L.P. Garrison, D. Harris, T.A. Marques and L. Thomas. 2015. Passive acoustic monitoring of beaked whale densities in the Gulf of Mexico. Sci. Rep. 5:16343.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Joyce, T.W., J.W. Durban, D.E. Claridge, C.A. Dunn, L.S. Hickmott, H. Fearnbach, K. Dolan and D. Moretti. 2020. Behavioral responses of satellite tracked Blainville's beaked whales (*Mesoplodon densirostris*) to mid-frequency active sonar. Mar. Mamm. Sci. 36(1):29–46.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108-189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S., D. K. Caldwell and H. E. Winn. 1976. Whales, dolphins, and porpoises of the western North Atlantic. NOAA Tech. Rep. NMFS CIRC-396. 176pp.

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Martín, V., A. Servidio and S. García 2004. Mass strandings of beaked whales in the Canary Islands. ECS Newsletter 42:33–36.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

McCarthy, E., D. Moretti, L. Thomas, N. DiMarzio, R. Morrissey, S. Jarvis, J. Ward, A. Izzi and A. Dilley. 2011. Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (*Mesoplodon densirostris*) during multiship exercises with mid-frequency sonar. Mar. Mamm. Sci. 27:E206–E226.

Mead, J.G. 1989. Beaked whales of the genus *Mesoplodon*. Pages 349–430 *in:* S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 4: River dolphins and the larger toothed whales. Academic Press, London. 442pp.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 2000-003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2001. Joint interim report on the Bahamas marine mammal stranding event of 15–16 March 2000 (December 2001). NOAA unpublished report. 55pp. Available from: http://www.nmfs.noaa.gov/pr/pdfs/health/stranding_bahamas2000.pdf

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: Time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J.G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station, Texas. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indic. 18:278–290.

Schmidly, D.J. 1981. Marine mammals of the southeastern United States and the Gulf of Mexico. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC, FWS/OBS-80/41. 165pp.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Twiner, M.J., L.J. Flewelling, S.E. Fire, S.R. Bowen-Stevens, J.K. Gaydos, C.K. Johnson, J.H. Landsberg, T.A. Leighfield, B. Mase-Guthrie, L. Schwacke, F.M. Van Dolah, Z. Wang and T.K. Rowles. 2012. Comparative analysis of three brevetoxin-associated bottlenose dolphin (*Tursiops truncatus*) mortality events in the Florida panhandle region (USA). PLoS ONE 7(8):e42974.

Tyack, P.L., W.M. Zimmer, D. Moretti, B.L. Southall, D.E. Claridge, J.W. Durban, C.W. Clark, A. D'Amico, N. DiMarzio, S. Jarvis and E. McCarthy. 2011. Beaked whales respond to simulated and actual navy sonar. PloS One 6:e17009.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

Würsig, B., T.A. Jefferson and D.J. Schmidly. 2000. The marine mammals of the Gulf of Mexico. Texas A&M University Press, College Station, Texas.

GERVAIS' BEAKED WHALE (*Mesoplodon europaeus*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Three species of *Mesoplodon* are known to occur in the Gulf of Mexico, based on stranding or sighting data (Hansen *et al.* 1995, Würsig *et al.* 2000). These are Gervais' beaked whale (*M. europaeus*), Blainville's beaked whale (*M. densirostris*), and Sowerby's beaked whale (*M. bidens*). Sowerby's beaked whale in the Gulf of Mexico is considered extralimital because there is only one known stranding of this species (Bonde and O'Shea 1989) and because it normally occurs in northern temperate waters of the North Atlantic (Mead 1989). The possibility of another unknown species of *Mesoplodon* inhabiting the Gulf has been suggested based on passive acoustic recordings (Hildebrand *et al.* 2015).

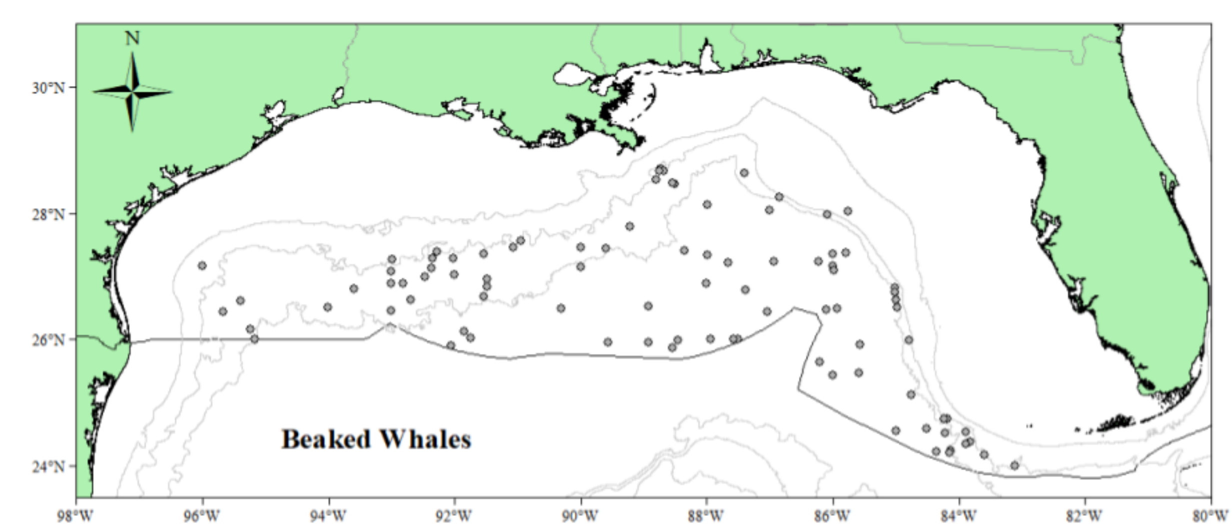


Figure 1. Distribution of beaked whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

Gervais' beaked whales appear to be widely but sparsely distributed in temperate and tropical waters of the world's oceans (Leatherwood *et al.* 1976, Leatherwood and Reeves 1983). Strandings have occurred along the northwestern Atlantic coast from Florida to Nova Scotia (Schmidly 1981) and in the northern Gulf of Mexico (Würsig *et al.* 2000). Because at-sea identification of *Mesoplodon* to species in the Gulf of Mexico is very difficult, sightings of beaked whales (Family Ziphiidae) made during visual surveys are often identified only as *Mesoplodon* sp. or unidentified Ziphiids, and are referred to more generically as 'beaked whales.' In the northern Gulf of Mexico, beaked whales are sighted most commonly in waters ≥500 m and they have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Hansen *et al.* 1996, Mullin and Hoggard 2000, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020; Figure 1).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et* *al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Gervais' beaked whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for Gervais' beaked whales in the northern Gulf of Mexico is 20 (CV=0.98; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The estimate for the same time period for *Mesoplodon* spp. (Blainville's and Gervais' beaked whales) was 98 (CV=0.46), and that for unidentified Ziphiids was 181 (CV=0.31). The *Mesoplodon* spp. and unidentified Ziphiids may have also included an unknown number of Gervais' beaked whales.

Earlier Abundance Estimates

Five point estimates of all Ziphiids (i.e., unidentified Ziphiids, *Mesoplodon* spp., Cuvier's beaked whale, and Gervais' beaked whale combined) abundance have been made based on data from surveys during: 2003 (June–August), 2004 (April–June), 2009 (July–August), 2017 (July–August), and 2018 (August–October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than "closing" mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences. This resulted in revised abundance estimates for all Ziphiids of: 2003, N=573 (CV=0.44); 2004, N=55 (CV=0.72); and 2009, N=276 (CV=0.59; Garrison *et al.* 2020).

Recent Surveys and Abundance Estimates

An abundance estimate for Gervais' beaked whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of this species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21, Laake *et al.* 2020) in the R statistical programming language. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The 2017 and 2018 estimates for Gervais' beaked whale were N=0 (CV=NA) and N=40 (CV=0.98), respectively. The inverse variance weighted mean abundance estimate for Gervais' beaked whales in oceanic waters during 2017 and 2018 was 20 (CV=0.98; Table 1; Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of this species. The 2017 and 2018 estimates for *Mesoplodon* spp. (Blainville's and Gervais' beaked whales) were N=127 (CV=0.61) and N=65 (CV=0.65), respectively, and the inverse variance weighted mean abundance estimate for *Mesoplodon* spp. was 98 (CV=0.46). The 2017 and 2018 estimates for unidentified Ziphiids were N=165 (CV=0.47) and N=193 (0.65), respectively, and the inverse variance weighted mean abundance estimate for unidentified Ziphiids was 181 (CV=0.31). The *Mesoplodon* spp. and unidentified Ziphiids may have also included an unknown number of Gervais' beaked whales. The 2017 and 2018 estimates for all Ziphiids (i.e., unidentified Ziphiids, *Mesoplodon* spp., Cuvier's beaked whale, and Gervais' beaked whale combined) were N=303 (CV=0.28) and N=322 (CV=0.34), respectively.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of Gervais' beaked whales in the northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	U.S. Gulf of Mexico	20	0.98

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Gervais' beaked whale is 20 (CV=0.98). The minimum population estimate for the northern Gulf of Mexico Gervais' beaked whale is 10 (Table 2).

Current Population Trend

Using revised abundance estimates for all Ziphiids for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates for all Ziphiids, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2004 estimates (p.adjusted=0.012) and the 2004 and 2018 estimates (p.adjusted=0.067; Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for Gervais' beaked whale is 10. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico Gervais' beaked whale is 0.1 (Table 2).

Table 2. Best and minimum abundance estimates for Gulf of Mexico Gervais' beaked whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
20	0.98	10	0.5	0.04	0.1

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Gervais' beaked whales or unidentified beaked whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for all beaked whales due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 5.2. The minimum total mean annual human-caused mortality and serious injury for beaked whales during 2014–2018 was, therefore, 5.2. This is a combined estimate for Blainville's, Gervais', and Cuvier's beaked whales. The minimum total mean annual human-caused mortality and serious injury for Gervais' beaked whale is unknown.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Gervais' beaked whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of beaked whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the pelagic longline fishery operating in the northern Gulf of Mexico. There were no reports of mortality or serious injury to Gervais' or other beaked whales by this fishery during 2014–2018 (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There were four strandings of Gervais' beaked whales during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). All four strandings occurred in Florida. For one stranding, there was evidence of human interaction (the interaction being the animal was pushed out to sea by the public). For the remaining three strandings, it could not be determined whether there was evidence of human interaction. Stranding data underestimate the extent of human and fishery-related mortality and serious injury because not all of the marine mammals that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon (DWH)* oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no Gervais' beaked whale strandings recovered within the spatial and temporal boundaries of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, this model estimated that the stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 51 beaked whales died during 2010–2013 (four year annual average of 13) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 26 beaked whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict beaked whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for beaked whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 12% (95%CI: 2–22) of beaked whales in the Gulf, which included Blainville's, Cuvier's and Gervais' beaked whales, were exposed to oil, that 5% (95%CI: 3–8) of females suffered from reproductive failure, and 4% (95%CI: 2–7) of the beaked whale populations suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). Anthropogenic noise, particularly from military sonar, shipping, and seismic testing, is an increasing habitat concern for beaked whales (Aguilar de Soto *et al.* 2006, Cox *et al.* 2006, McCarthy *et al.* 2011, Tyack *et al.* 2011, Joyce *et al.* 2020). Several mass strandings of beaked whales throughout their worldwide range have been associated with naval activities (D'Amico *et al.* 2009, Filadelfo *et al.* 2009). In March 2000, 14 beaked whales live stranded in the Bahamas. Six of the whales (5 Cuvier's and 1 Blainville's) died and necropsy revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand (Balcomb and Claridge 2001, NMFS 2001, Cox *et al.* 2006). Fourteen beaked whales (mostly Cuvier's beaked whales but also including Gervais' and Blainville's beaked whales) stranded in the Canary Islands in 2002 (Cox *et al.* 2006, Fernandez *et al.* 2005, Martin *et al.* 2004). Gas bubble-associated lesions and fat embolism were found in necropsied animals from this event, leading researchers to link nitrogen supersaturation with sonar exposure (Fernandez *et al.* 2005). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown. Finally, ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach contents of stranded beaked whales.

STATUS OF STOCK

Gervais' beaked whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all beaked whale species combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Gervais' beaked whales in the northern Gulf of Mexico, relative to OSP, is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED

Aguilar de Soto, N., M. Johnson, P.T. Madsen, P.L. Tyack, A. Bocconcelli and J. Fabrizio Borsani. 2006. Does intense ship noise disrupt foraging in deep-diving Cuvier's beaked whales (*Ziphius cavirostris*)? Mar. Mamm. Sci. 22(3):690–699.

Balcomb, K.C. III and D.E. Claridge. 2001. A mass stranding of cetaceans caused by naval sonar in the Bahamas. Bahamas J. Sci. 2:2–12.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Bonde, R.K. and T.J. O'Shea. 1989. Sowerby's beaked whale (*Mesoplodon bidens*) in the Gulf of Mexico. J. Mammal. 70:447–449.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

Cox, T.M., T.J. Ragen, A.J. Read, E. Vos, R.W. Baird, K. Balcomb, J. Barlow, J. Caldwell, T. Cranford, L. Crum, A. D'Amico, G. D'Spain, A. Fernandez, J. Finneran, R. Gentry, W. Gerth, F. Gulland, J. Hildebrand, D. Houser, T. Hullar, P.D. Jepson, D. Ketten, C.D. MacLeod, P. Miller, S. Moore, D.C. Mountain, D. Palka, P. Ponganis, S. Rommel, T. Rowles, B. Taylor, P. Tyack, D. Wartzok, R. Gisiner, J. Mead and L. Benner. 2006. Understanding the impacts of anthropogenic sound on beaked whales. J. Cetacean Res. Manage. 7(3):177–187.

D'Amico, A., R.C. Gisiner, D.R. Ketten, J.A. Hammock, C. Johnson, P.L. Tyack and J. Mead. 2009. Beaked whale strandings and naval exercises. Aquat. Mamm. 35(4):452–472.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan.

Fernandez, A., J.F. Edwards, F. Rodriguez, A.E. de los Monteros, P. Herraez, P. Castro, J.R. Jaber, V. Martin and M. Arbelo. 2005. "Gas and Fat Embolic Syndrome" involving a mass stranding of beaked whales (Family Ziphiidae) exposed to anthropogenic sonar signals. Vet. Path. 42(4):446–457.

Filadelfo, R., J. Mintz, E. Michlovich, A. D'Amico, P.L. Tyack and D.R. Ketten. 2009. Correlating military sonar use with beaked whale mass strandings: What do the historical data show? Aquat. Mamm. 35(4):435–444.

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin and C.L. Roden. 1995. Estimates of cetacean abundance in the northern Gulf of Mexico from vessel surveys. Southeast Fisheries Science Center, Miami Laboratory, Contribution No. MIA-94/95-25. 9pp. Available from: NMFS, Southeast Fisheries Science Center, 75 Virginia Beach Dr., Miami, Florida 33149.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA.

Hildebrand, J.A., S. Baumann-Pickering, K.E. Frasier, J.S. Trickey, K.P. Merkens, S.M. Wiggins, M.A. McDonald, L.P. Garrison, D. Harris, T.A. Marques and L. Thomas. 2015. Passive acoustic monitoring of beaked whale densities in the Gulf of Mexico. Sci. Rep. 5:16343.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Joyce, T.W., J.W. Durban, D.E. Claridge, C.A. Dunn, L.S. Hickmott, H. Fearnbach, K. Dolan and D. Moretti. 2020. Behavioral responses of satellite tracked Blainville's beaked whales (*Mesoplodon densirostris*) to mid-frequency active sonar. Mar. Mamm. Sci. 36(1):29–46.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108-189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S., D. K. Caldwell and H. E. Winn. 1976. Whales, dolphins, and porpoises of the western North Atlantic. NOAA Tech. Rep. NMFS CIRC-396. 176 pp.

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Martín, V., A. Servidio and S. García. 2004. Mass strandings of beaked whales in the Canary Islands. ECS Newsletter 42:33–6.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

McCarthy, E., D. Moretti, L. Thomas, N. DiMarzio, R. Morrissey, S. Jarvis, J. Ward, A. Izzi and A. Dilley. 2011. Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (*Mesoplodon densirostris*) during multiship exercises with mid-frequency sonar. Mar. Mamm. Sci. 27:E206–E226.

Mead, J. G. 1989. Beaked whales of the genus *Mesoplodon*. Pages 349–430 *in:* S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 4: River dolphins and the larger toothed whales. Academic Press, London. 442pp.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2001. Joint interim report on the Bahamas marine mammal stranding event of 15–16 March 2000 (December 2001). NOAA unpublished report 55pp. Available from: http://www.nmfs.noaa.gov/pr/pdfs/health/stranding_bahamas2000.pdf

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: Time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indic. 18:278–290.

Schmidly, D.J. 1981. Marine mammals of the southeastern United States and the Gulf of Mexico. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC, FWS/OBS-80/41. 165pp.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Tyack, P.L., W.M. Zimmer, D. Moretti, B.L. Southall, D.E. Claridge, J.W. Durban, C.W. Clark, A. D'Amico, N. DiMarzio, S. Jarvis and E. McCarthy. 2011. Beaked whales respond to simulated and actual navy sonar. PloS One 6:e17009.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

Würsig, B., T.A. Jefferson and D.J. Schmidly. 2000. The marine mammals of the Gulf of Mexico. Texas A&M University Press, College Station, Texas.

COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Northern Gulf of Mexico Oceanic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Thirty-six common bottlenose dolphin stocks have been designated in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico; Waring *et al.* 2016). Northern Gulf of Mexico inshore habitats have been separated into 31 bay, sound and estuary stocks. Three northern Gulf of Mexico coastal stocks inhabit coastal waters from the shore to the 20-m isobath. The northern Gulf of Mexico Continental Shelf Stock inhabits waters from 20 to 200 m deep. The northern Gulf of Mexico Oceanic Stock inhabits the waters from the 200-m isobath to the seaward extent of the U.S. Exclusive Economic Zone (EEZ; Figure 1).

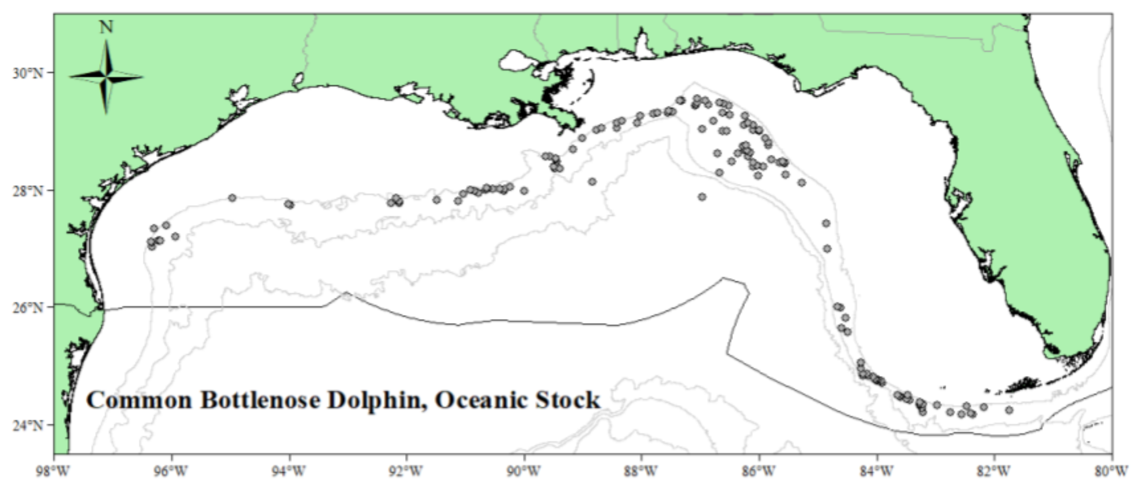


Figure 1. Distribution of common bottlenose dolphin on-effort sightings in oceanic waters from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

Both "coastal" and "offshore" ecotypes of common bottlenose dolphins (Mead and Potter 1995) occur in the Gulf of Mexico (Vollmer 2011, Vollmer and Rosel 2013), but the distribution of each is not well defined. The offshore and coastal ecotypes are genetically distinct based on both mitochondrial and nuclear markers (Hoelzel *et al.* 1998, Vollmer 2011). Ongoing research is aimed at better defining stock boundaries in coastal, continental shelf and oceanic waters of the Gulf of Mexico. Although the boundaries are not certain, all 141 *Tursiops* samples collected during 1994−2008 in waters greater than 200 m were of the offshore ecotype (Vollmer 2011), and so the Oceanic Stock as currently defined is thought to be composed entirely of bottlenose dolphins of the offshore ecotype.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with both Cuba and Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

The northern Gulf of Mexico Oceanic Stock of common bottlenose dolphins is managed separately from the western North Atlantic Offshore Stock of common bottlenose dolphins. One line of evidence to support this decision comes from Baron *et al.* (2008), who found that Gulf of Mexico common bottlenose dolphin whistles (collected from oceanic waters) were significantly different from those in the western North Atlantic Ocean (collected from continental shelf and oceanic waters) in duration, number of inflection points and number of steps. Coupled with evidence for population structure in other areas and the fact that the western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding *et al.* 2007), designation of the two stocks is reasonable and consistent with maintaining stocks as functioning elements of their ecosystems. Restricted genetic exchange has been documented among offshore populations within the Gulf of Mexico, suggesting multiple demographically-independent populations of the offshore morphotype exist (Vollmer and Rosel 2017).

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico Oceanic Stock of common bottlenose dolphins is 7,462 (CV=0.31; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of abundance for the oceanic stock of common bottlenose dolphins have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=21,350 (CV=0.47); 2004, N=8,864 (CV=0.50); and 2009, N=9,640 (CV=0.66).

Recent Surveys and Abundance Estimates

An abundance estimate for the oceanic stock of common bottlenose dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200 m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 5,104 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 5,205 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al*. 2020). The 2017 and 2018 estimates were N=8,756 (CV=0.41) and N=5,833 (CV=0.46), respectively. The inverse variance weighted mean abundance estimate for common bottlenose dolphins in oceanic waters during 2017 and 2018 was 7,462 (CV=0.31; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico common bottlenose dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	7,462	0.31

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for common bottlenose dolphins is 7,462 (CV=0.31). The minimum population estimate for the northern Gulf of Mexico oceanic stock of common bottlenose dolphin is 5,769 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum productivity rates are unknown for this stock. For purposes of this assessment, the maximum productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of minimum population size, one-half the maximum productivity rate and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 5,769. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the Gulf of Mexico oceanic common bottlenose dolphin is 58 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico oceanic common bottlenose dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
7,462	0.31	5,769	0.5	0.04	58

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to oceanic bottlenose dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 32. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 32.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico oceanic common bottlenose dolphins.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are three commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico: the Category I Atlantic Highly Migratory Species (high seas) longline fishery and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery; and the Category III Gulf of Mexico butterfish trawl fishery (Appendix III).

Percent observer coverage (percentage of sets observed) for the two Category I longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of common bottlenose dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico and during 2014–2018 there were no observed mortalities or serious injuries to common bottlenose dolphins by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

The Category III Gulf of Mexico butterfish trawl fishery may also interact with this stock (Appendix III). A trawl fishery for butterfish was monitored by NMFS observers for a short period in the 1980s with no records of incidental take of marine mammals (Burn and Scott 1988, NMFS unpublished data), although an experimental set by NMFS resulted in the death of two common bottlenose dolphins (Burn and Scott 1988). There are no other data available with regard to this fishery.

Other Mortality

A total of 1,764 common bottlenose dolphins were found stranded in the northern Gulf of Mexico from 2014 through 2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Of these, 177 showed evidence of human interaction (e.g., gear entanglement, mutilation, gunshot wounds). The vast majority of stranded common bottlenose dolphins are assumed to belong to one of the coastal stocks or to bay, sound and estuary stocks. Nevertheless, it is possible that some of the stranded common bottlenose dolphins belonged to the continental shelf or oceanic stock and that they were among those strandings with evidence of human interactions. Strandings do occur for other cetacean species whose primary range in the Gulf of Mexico is outer continental shelf or oceanic waters, but oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011).

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014, https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). During 2014, 126 common bottlenose dolphins were considered to be part of the UME. The vast majority of stranded common bottlenose dolphins are assumed to come from stocks that live nearest to land, namely the bay, sound and estuary stocks and the three coastal stocks. Nevertheless, it is possible that some of the stranded common bottlenose dolphins considered part of the UME belonged to the continental shelf or oceanic stock, given the overlap in distribution between the spill and distribution of this population.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 4% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 308 oceanic common bottlenose dolphins died during 2010–2013 (four year annual average of 77) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model, estimated 160 oceanic common bottlenose dolphins died due to elevated mortality associated with oil exposure. The population model used to predict oceanic common bottlenose dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for common bottlenose dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016).

Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 10% (95%CI: 5–10) of oceanic common bottlenose dolphins in the Gulf were exposed to oil, that 5% (95%CI: 2–6) of females suffered from reproductive failure, and 4% (95%CI: 1–6) of oceanic common bottlenose dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 4% maximum reduction in population size (see Other Mortality section above).

The use of explosives to remove oil rigs in portions of the continental shelf in the western Gulf of Mexico has the potential to cause serious injury or mortality to marine mammals. These activities have been closely monitored by NMFS observers since 1987 (Gitschlag and Herczeg 1994). There have been no reports of either serious injury or mortality to common bottlenose dolphins in the oceanic Gulf of Mexico associated with these activities.

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico Oceanic Stock is not considered strategic under the MMPA. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of bottlenose dolphins, relative to OSP, in the northern Gulf of Mexico oceanic waters is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade 1995. U.S. marine mammal stock assessment: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6, 73pp.

Baron, S.C., A. Martinez, L.P. Garrison and E.O. Keith. 2008. Differences in acoustic signals from Delphinids in the western North Atlantic and northern Gulf of Mexico. Mar. Mamm. Sci. 24(1):42–56.

Burn, D. and G.P. Scott. 1988. Synopsis of available information on marine mammal-fisheries interactions in the southeastern United States: Preliminary report. Contribution ML-CRG-87/88-26, National Marine Fisheries Service, Miami, Florida. 37pp.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning and B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gitschlag, G.R. and B.A. Herczeg. 1994. Sea turtle observations at explosive removals of energy structures. Mar. Fish. Rev. 56(2):1–8.

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hoelzel, A.R., C.W. Potter and P.B. Best. 1998. Genetic differentiation between parapatric 'nearshore' and 'offshore' populations of the bottlenose dolphin. Proc. R. Soc. Lond. B 265:1177–1183.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Mead, J.G. and C.W. Potter. 1995. Recognizing two populations of the bottlenose dolphin (*Tursiops truncatus*) off the Atlantic coast of North America: Morphological and ecological considerations. IBI Reports. 5:31–44.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: Time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J.G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE. 10(5):e0126538.

Vollmer, N.L. 2011. Population structure of common bottlenose dolphins in coastal and offshore waters of the Gulf of Mexico revealed by genetic and environmental analyses. Ph.D. Dissertation from University of Louisiana at Lafayette. 420pp.

Vollmer, N.L. and P.E. Rosel. 2013. A review of common bottlenose dolphins (*Tursiops truncatus truncatus*) in the northern Gulf of Mexico: Population biology, potential threats, and management. Southeast. Nat. 12 (Monograph 6):1–43.

Vollmer, N.L. and P.E. Rosel. 2017. Fine-scale population structure of common bottlenose dolphins (*Tursiops truncatus*) in offshore and coastal waters of the US Gulf of Mexico. Mar. Biol. 164:160.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS workshop April 3–5, 1996, Seattle, WA. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Waring, G.T., E. Josephson, K. Maze-Foley and P.E. Rosel. 2016. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2015. NOAA Tech. Memo. NMFS-NE-238. 512pp.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

PANTROPICAL SPOTTED DOLPHIN (*Stenella attenuata attenuata*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pantropical spotted dolphin is distributed worldwide in tropical and some subtropical oceans (Perrin *et al.* 1987, Perrin and Hohn 1994). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species are common during visual surveys in oceanic waters >200 m and in all seasons (Figure 1; Hansen *et al.* 1996, Mullin and Hoggard 2000, Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and

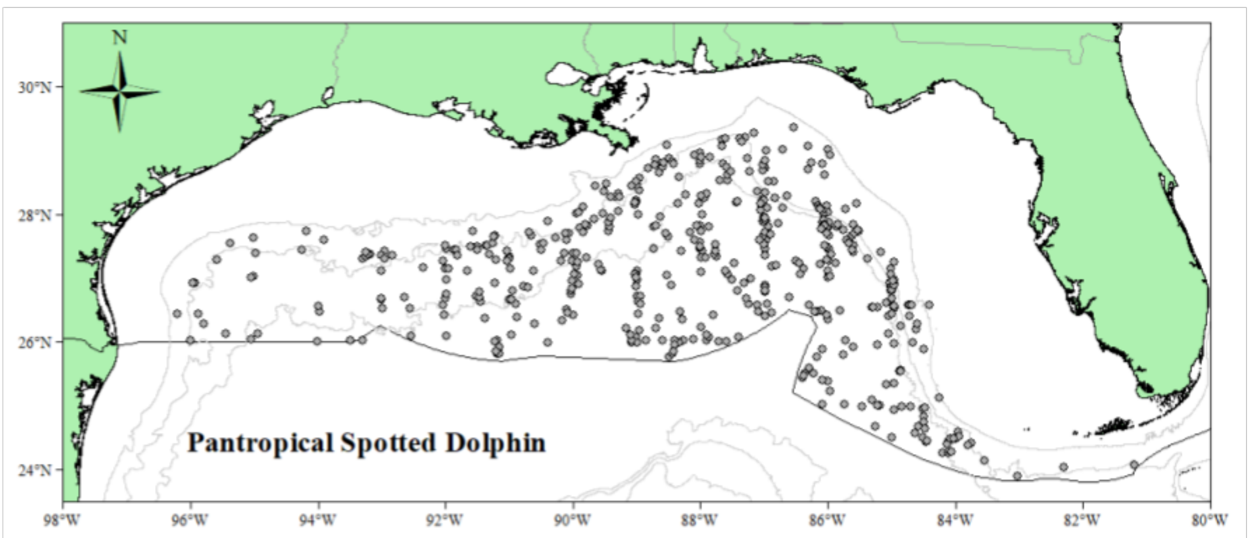


Figure 1. Distribution of pantropical spotted dolphin on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Pantropical spotted dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, such separation is consistent with evidence for population structure in other areas, including more pelagic waters of the eastern tropical Pacific (Leslie and Morin 2016), and is further supported because the two stocks occupy distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico pantropical spotted dolphin is 37,195 (CV=0.24; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of pantropical spotted dolphin abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=72,901 (CV=0.20); 2004, N=78,879 (CV=0.41); and 2009, N=84,047 (CV=0.36).

Recent Surveys and Abundance Estimates

An abundance estimate for pantropical spotted dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). The 2017 and 2018 estimates were N=27,362 (CV=0.27) and N=58,725 (CV=0.41), respectively. The inverse variance weighted mean abundance estimate for pantropical spotted dolphins in oceanic waters during 2017 and 2018 was 37,195 (CV=0.24; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico pantropical spotted dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	37,195	0.24

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for pantropical spotted dolphins is 37,195 (CV=0.24). The minimum population estimate for the northern Gulf of Mexico pantropical spotted dolphin is 30,377 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2017 estimates (p.adjusted=0.016) and the 2009 and 2017 estimates (p.adjusted=0.051; Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate, and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 30,377. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico pantropical spotted dolphin is 304 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico pantropical spotted dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
37,195	0.24	30,377	0.5	0.04	304

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to pantropical spotted dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 241. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 241.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico pantropical spotted dolphins.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas longline) fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the high seas longline fishery, and no takes of pantropical spotted dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagic longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to pantropical spotted dolphins by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

Five pantropical spotted dolphins were reported stranded in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Three strandings were documented in 2014, all in Florida, and two strandings were documented in 2018, one each in Alabama and in Texas. No evidence of human interaction was detected for one stranded animal, and for the remaining four animals, it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012, Wells *et al*. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). Three pantropical spotted dolphin strandings during 2011 were considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 9% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 2,367 pantropical spotted dolphins died during 2010–2013 (four year annual average of 592) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 1,203 pantropical spotted dolphins died due to elevated mortality associated with oil exposure. The population model used to predict pantropical spotted dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for pantropical spotted dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 20% (95%CI: 15–26) of pantropical spotted dolphins in the Gulf were exposed to oil, that 9% (95%CI: 4–13) of females suffered from reproductive failure, and 7% (95%CI: 3–11) of pantropical spotted dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 9% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Pantropical spotted dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of pantropical spotted dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. The population trend for this stock is also unknown.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning and B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leslie, M.S. and P.A. Morin. 2016. Using genome-wide SNPs to detect structure in high-diversity and low-divergence populations of severely impacted Eastern Tropical Pacific spinner (*Stenella longirostris*) and pantropical spotted dolphins (*S. attenuata*). Front. Mar. Sci. 3:253.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112:161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G.L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 2000-003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: Time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J.G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Perrin, W.F., E.D. Mitchell, J.G. Mead, D.K. Caldwell, M.C. Caldwell, P.J.H. van Bree and W.H. Dawbin. 1987. Revision of the spotted dolphins, *Stenella* spp. Mar. Mamm. Sci. 3(2):99–170.

Perrin, W.F. and. A.A. Hohn. 1994. Pantropical spotted dolphin *Stenella attenuata* (Meyen, 1833). Pages 129-159 *in:* S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 5: The first book of dolphins. Academic Press, London. 416pp.

Schwacke , L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12, 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

STRIPED DOLPHIN (*Stenella coeruleoalba*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The striped dolphin is distributed worldwide in tropical to temperate oceanic waters (Leatherwood and Reeves 1983, Perrin *et al.* 1994). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species occur in waters >200 m deep, with most observations in waters ≥1,000 m deep (Figure 1; Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). Like spinner dolphins, the majority of sightings are east of the Mississippi River, but striped dolphins are also observed over the continental slope in the western Gulf and out in the deeper central basin. Striped dolphins have been seen in all seasons during NMFS visual surveys (Hansen *et al.* 1996, Mullin and Hoggard 2000).

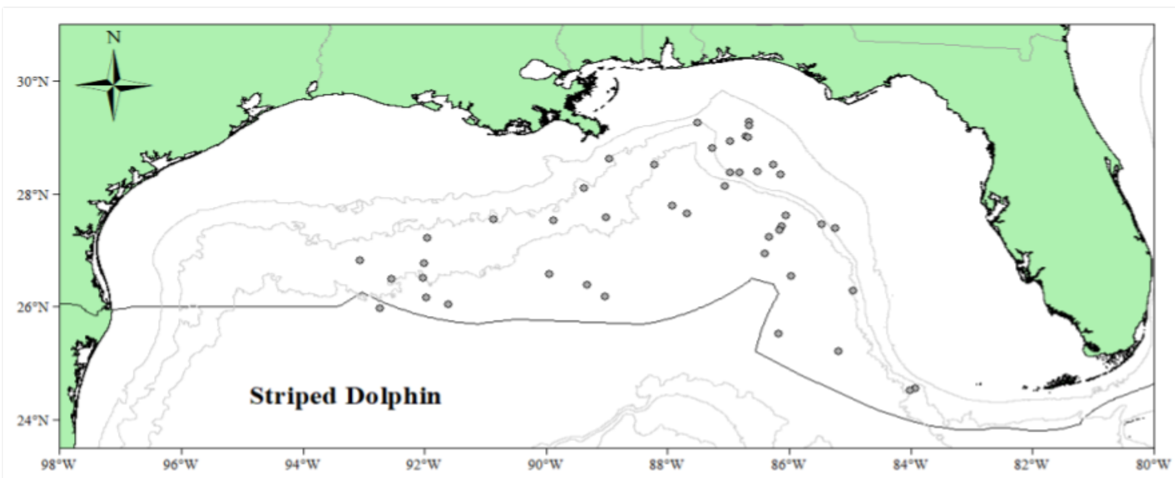


Figure 1. Distribution of striped dolphin on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Striped dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico striped dolphin is 1,817 (CV=0.56; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of striped dolphin abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=5,494 (CV=0.43); 2004, N=10,764 (CV=0.51); and 2009, N=3,060 (CV=0.73).

Recent Surveys and Abundance Estimates

An abundance estimate for striped dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010), while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). There were sightings of striped dolphins in 2018 but there were none in 2017. The 2017 and 2018 estimates were N=0 (CV=NA) and N=3,633 (CV=0.56), respectively. The inverse variance weighted mean abundance estimate for striped dolphins in oceanic waters during 2017 and 2018 was 1,817 (CV=0.56; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico striped dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	1,817	0.56

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for striped dolphins is 1,817 (CV=0.56). The minimum population estimate for the northern Gulf of Mexico striped dolphin is 1,172 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,172. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico striped dolphin is 12 (Table 2).

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico striped dolphin with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
1,817	0.56	1,172	0.5	0.04	12

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to striped dolphins in the northern Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 13. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 13.

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico striped dolphin.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of striped dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to striped dolphins by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There was one reported stranding of a striped dolphin in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). This animal stranded during 2015 in Florida, and it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no striped dolphin strandings recovered within the spatial and temporal boundaries of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 124 striped dolphins died during 2010–2013 (four year annual average of 31) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 63 striped dolphins died due to elevated mortality associated with oil exposure. The population model used to predict striped dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for striped dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 13% (95%CI: 8–22) of striped dolphins in the Gulf were exposed to oil, that 6% (95%CI: 3–9) of females suffered from reproductive failure, and 5% (95%CI: 2–8) of striped dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Striped dolphins are not listed as threatened or endangered under the Endangered Species Act, but the northern Gulf of Mexico stock is considered strategic under the MMPA because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of striped dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning and B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.
Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.
Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/
Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.
Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112:161–175.
Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.
Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.
Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.
Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 2000-003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.
NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892
Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.
Ortega Ortiz, J.G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.
Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.
Perrin, W.F., C.E. Wilson and F.I. Archer II. 1994. Striped dolphin *Stenella coeruleoalba* (Meyen, 1833). Pages 129–159 *in:* S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 5: The first book of dolphins. Academic Press, London. 416pp.
Schwacke , L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.
Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.
Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.
Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.
Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.
Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

SPINNER DOLPHIN (*Stenella longirostris longirostris*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The spinner dolphin is distributed worldwide in tropical to temperate oceanic and coastal waters (Leatherwood and Reeves 1983, Perrin and Gilpatrick 1994). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species occur in waters >200 m and are concentrated over the continental slope, particularly east of the Mississippi River (Figure 1; Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). Spinner dolphins have been seen in all seasons during NMFS visual surveys of the northern Gulf of Mexico (Hansen *et al.* 1996, Mullin and Hoggard 2000).

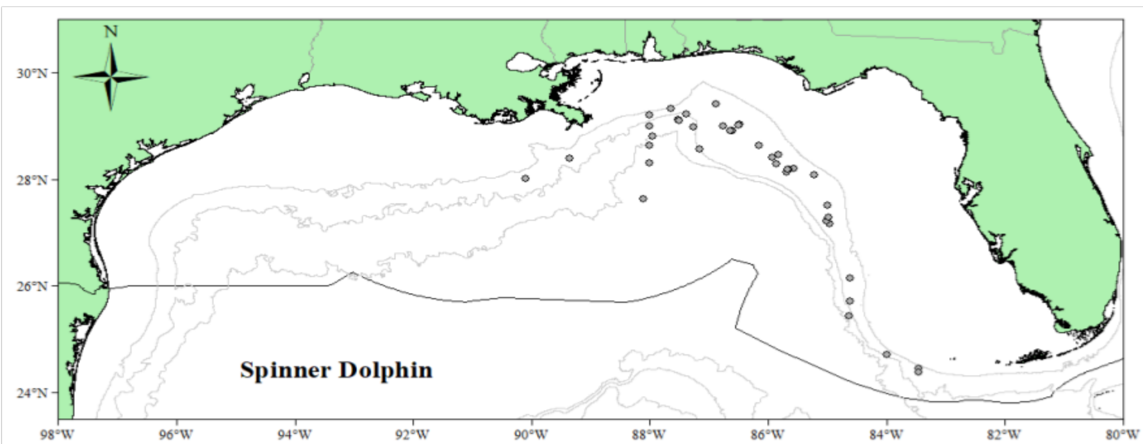


Figure 1. Distribution of spinner dolphin on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. Spinner dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with evidence for population structure in other areas, including more pelagic waters of the eastern tropical Pacific (Leslie and Morin 2016), and is further supported because the two stocks occupy distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) available for the northern Gulf of Mexico spinner dolphin is 2,991 (CV=0.54; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of spinner dolphin abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than "closing" mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=5,160 (CV=0.55); 2004, N=24,536 (CV=0.58); and 2009, N=19,678 (CV=0.53).

Recent Surveys and Abundance Estimates

An abundance estimate for spinner dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010), while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). There were sightings of spinner dolphins in 2017 but there were none in 2018. The 2017 and 2018 estimates were N=5,982 (CV=0.54) and N=0 (CV=NA), respectively. The inverse variance weighted mean abundance estimate for spinner dolphins in oceanic waters during 2017 and 2018 was 2,991 (CV=0.54; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico spinner dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	2,991	0.54

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for spinner dolphins is 2,991 (CV=0.54). The minimum population estimate for the northern Gulf of Mexico spinner dolphin is 1,954 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,954. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico spinner dolphin is 20 (Table 2).

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico spinner dolphin with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
2,991	0.54	1,954	0.5	0.04	20

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to spinner dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 113. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 113.

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico spinner dolphin.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

The commercial fishery which potentially could interact with this stock in the Gulf of Mexico is the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for this fishery for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to spinner dolphins by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There were 13 reported strandings of spinner dolphins in the Gulf of Mexico during 2014–2018, including one mass stranding of 11 individuals in Florida during 2016 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). No evidence of human interaction was detected for five stranded spinner dolphins, and for the remaining eight spinner dolphins, it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). Twelve spinner dolphin strandings were considered to be part of this UME, one of which occurred during 2014.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 23% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 1114 spinner dolphins died during 2010–2013 (four year annual average of 278) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 566 spinner dolphins died due to elevated mortality associated with oil exposure. The population model used to predict spinner dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for spinner dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Spinner dolphin strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of spinner dolphins in Alabama or Mississippi.

State	2014	2015	2016	2017	2018	Total
Florida	0	0	11[b]	0	0	11
Louisiana	1[a]	0	0	0	0	1
Texas	0	1	0	0	0	1
Total	**1**	**1**	**11**	**0**	**0**	**13**

a. This stranding was part of the Northern Gulf of Mexico UME.
b. This was a mass strandings of 11 animals.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals.

These studies estimated that 47% (95%CI: 24–91) of spinner dolphins in the Gulf were exposed to oil, that 21% (95%CI: 10–30) of females suffered from reproductive failure, and 17% (95%CI: 6–27) of spinner dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 23% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Spinner dolphins are not listed as threatened or endangered under the Endangered Species Act, but the northern Gulf of Mexico stock is considered strategic under the MMPA because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of spinner dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning and B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1): 27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling.S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Leslie, M.S. and P.A. Morin. 2016. Using genome-wide SNPs to detect structure in high-diversity and low-divergence populations of severely impacted Eastern Tropical Pacific spinner (*Stenella longirostris*) and pantropical spotted dolphins (*S. attenuata*). Front. Mar. Sci. 3:253.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112:161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 2000-003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station, Texas. 170 pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Perrin, W.F. and J.W. Gilpatrick, Jr. 1994. Spinner dolphin *Stenella longirostris* (Gray, 1828). Pages 99–128 *in:* S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 5: The first book of dolphins. Academic Press, London. 416pp.

Schwacke , L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

ROUGH-TOOTHED DOLPHIN (*Steno bredanensis*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Rough-toothed dolphins (*Steno bredanensis*) are distributed worldwide in the Atlantic, Pacific, and Indian Oceans, generally in warm temperate, subtropical, or tropical waters. They are commonly reported in a wide range of water depths, from shallow, nearshore waters to oceanic waters (West *et al.* 2011). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), rough-toothed dolphins occur in oceanic and to a lesser extent continental shelf waters (Figure 1; Fulling *et al.* 2003, Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). They have been observed in all seasons during NMFS visual surveys in the Gulf of Mexico (Hansen *et al.* 1996, Mullin and Hoggard 2000) but are not seen every survey year attesting to their low density in this region.

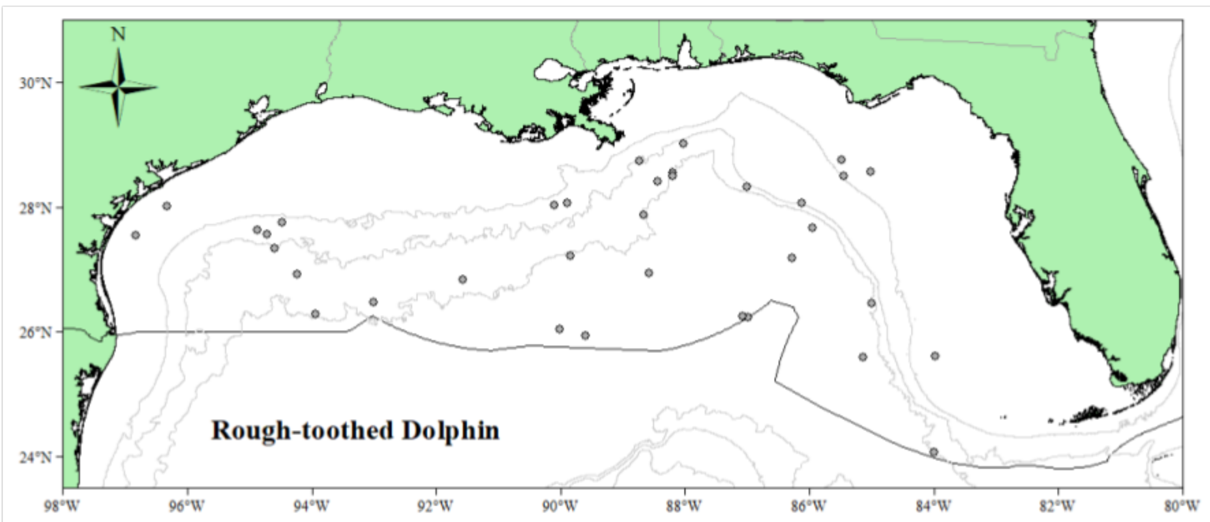


Figure 1. Distribution of rough-toothed dolphin on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Rough-toothed dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Several lines of evidence support this distinction. Four dolphins from a mass stranding of 62 animals in the Florida Panhandle in December 1997 were rehabilitated and released in 1998, and satellite-linked transmitters on three of these were tracked for up to 112 days. A report after five months indicated that the animals returned to, and remained in, northeastern Gulf waters (Wells *et al.* 2008), providing evidence for fidelity to the Gulf. In addition, analyses of worldwide genetic differentiation in Steno indicate animals in the western Atlantic Ocean are strongly differentiated from those in the Pacific and Indian Oceans (Albertson 2014, da Silva *et al.* 2015). Albertson (2014) illustrated that this species may exhibit fine-scale population structure and da Silva *et al.* (2015) provided evidence for multiple populations in the western South Atlantic. Finally, the separation of Atlantic and Gulf of Mexico stocks is consistent with the fact that the two areas belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011).

There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico rough-toothed dolphin is unknown (Table 1) since no sightings of this species were made during the summer 2017 or summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Estimates of rough-toothed dolphin abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=9,253 (CV=0.78); 2004, N=0 (CV=NA); and 2009, N=3,509 (CV=0.67).

Recent Surveys and Abundance Estimates

Two vessel surveys were conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. No sightings of rough-toothed dolphins were made during these two vessel surveys; therefore, the abundance estimate for rough-toothed dolphins is unknown.

Table 1. Summary of recent abundance estimates for rough-toothed dolphins in the northern Gulf of Mexico oceanic waters (200 m to the offshore extent of the EEZ) by month, year, and area covered during each abundance survey and the resulting abundance estimate (N_{est}) and coefficient of variation (CV).

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	Unknown	-

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best and minimum estimates of abundance for rough-toothed dolphins are unknown.

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), pairwise comparisons of the non-zero log-transformed means were conducted, and significant differences were assessed at alpha=0.10. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is currently undetermined. PBR is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is unknown. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.40 because the CV of the average mortality estimate is greater than 0.8 (Wade and Angliss 1997; Table 2).

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico rough-toothed dolphin stock with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
Unknown	-	Unknown	0.40	0.04	Undetermined

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The estimated mean annual fishery-related mortality and serious injury for this stock during 2014–2018 was 0.8 rough-toothed dolphins (CV=1.00) due to interactions with the large pelagics longline fishery and 0.2 rough-toothed dolphins due to an interaction with the hook and line fishery (see Fisheries Information sections below; Tables 3–4). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 38. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 39 (Table 5).

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico rough-toothed dolphin stock.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	1.0	1.00

Fisheries Information

There are three commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These include two Category I fisheries, the Atlantic Highly Migratory Species (high seas) longline fishery, and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery, and one Category III fishery, the Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery (Appendix III).

Longline

There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of rough-toothed dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. For the five-year period 2014–2018, the estimated annual combined serious injury and mortality attributable to the large pelagics longline fishery in the northern Gulf of Mexico was 0.8 (CV=1.00) rough-toothed dolphins (Table 4; Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b). During the second quarter of 2014, one serious injury was observed (Garrison and Stokes 2016). Percent observer coverage (percentage of sets observed) for the two longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. During the first and second quarters of 2014–2018, observer coverage in the Gulf of Mexico large pelagics longline fishery was greatly enhanced to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2014–2018 (Table 4) primarily reflect high coverage rates during the first and second quarters of each year. During these quarters, this elevated coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rates observed during the first and second quarters are representative of that which occurs throughout the year.

Table 4. Summary of the incidental mortality and serious injury of rough-toothed dolphins by the pelagic longline commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the annual observed serious injury and mortality recorded by on-board observers, the annual estimated serious injury and mortality, the combined annual estimates of serious injury and mortality (Est. Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs), the mean of the combined annual mortality estimates, and the CV of the mean combined annual mortality estimate (CV of Mean).

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury[c]	Est. Mort.	Est. Combined Mortality	Est. CVs	Mean Combined Annual Mortality	CV of Mean
Pelagic Longline	2014	Obs. Data, Trip Logbook	0.18	1	0	4.2	0	4.2	1	0.8	1.00
	2015		0.19	0	0	0	0	0	-		
	2016		0.23	0	0	0	0	0	-		
	2017		0.13	0	0	0	0	0	-		
	2018		0.20	0	0	0	0	0	-		
Total										**0.8**	**1.00**

[a] Number of vessels in the fishery is based on vessels reporting effort to the pelagic longline logbook.
[b] Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. Mandatory logbook data were used to measure total effort for the longline fishery. These data are collected at the Southeast Fisheries Science Center (SEFSC). Observer coverage in the GOM is dominated by very high coverage rates during April–June associated with efforts to improve estimates of bluefin tuna bycatch.
[c] Proportion of sets observed.

Other Mortality

Hook and Line (Rod and Reel)

During 2014–2018, stranding data included one mortality and one serious injury for which hook and line gear entanglement or ingestion were documented. For the mortality, the stranding data suggested the hook and line gear interaction was not a contributing factor to cause of death. Therefore, only the serious injury (Maze-Foley and Garrison 2020) was included in the annual human-caused mortality and serious injury total for this stock (Table 5). Both cases occurred in 2018 and were included in the stranding database and are included in the stranding totals presented in Table 6 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019).

It should be noted that, in general, it cannot be determined if rod and reel hook and line gear originated from a commercial (i.e., commercial fisherman, charter boat, or headboat) or recreational angler because the gear type used by both sources is typically the same. Also, it is not possible to estimate the total number of interactions with hook and line gear because there is no systematic observer program. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Table 5. Summary of the incidental mortality and serious injury of rough-toothed dolphins during 2014–2018 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and other sources.

Mean Annual Mortality due to the observed commercial large pelagics longline fishery (2014–2018) (Table 4)	0.8
Mean Annual Mortality due to the unobserved hook and line fishery (2014–2018)	0.2
Mean Annual Mortality due to Other Human-Caused Sources (DWH oil spill) (2014–2018)	38
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2014–2018)	**39**

Other Mortality

There were six stranded rough-toothed dolphins in the northern Gulf of Mexico during 2014–2018 (Table 6; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Evidence of human interaction was detected for two of the stranded animals, both of which were classified as fishery interactions. No evidence of human interaction was detected for one stranded animal, and for the remaining three, it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012, Wells *et al*. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME), involving primarily bottlenose dolphins, was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see “Habitat Issues” below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). One stranding of a rough-toothed dolphin in 2013 was considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 17% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 362 rough-toothed dolphins died during 2010–2013 (four year annual average of 91) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 188 rough-toothed dolphins died due to elevated mortality associated with oil exposure. The population model used to predict rough-toothed dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for rough-toothed dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 6. Rough-toothed dolphin strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of rough-toothed dolphins in Alabama or Texas.

State	2014	2015	2016	2017	2018	Total
Florida	0	3	0	1	0	4
Louisiana	0	0	0	0	1[a]	0

Mississippi	0	0	0	0	1[a]	0
Total	**0**	**3**	**0**	**1**	**0**	**4**

a. Both 2018 animals were classified as fishery interactions.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 41% (95%CI: 16–100) of rough-toothed dolphins in the Gulf were exposed to oil, that 19% (95%CI: 9–26) of females suffered from reproductive failure, and 15% (95%CI: 6–23) of rough-toothed dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 17% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Rough-toothed dolphins are not listed as threatened or endangered under the Endangered Species Act. The most recent abundance surveys (2017–2018) observed no rough-toothed dolphins, rendering PBR undetermined. The northern Gulf of Mexico stock is therefore not considered strategic under the MMPA. However, the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill (38 animals) greatly exceeds the previous, but expired, estimate of PBR for this stock (2.5) based on 2009 surveys. Total fishery-related mortality and serious injury for this stock was 0.8, which is not less than 10% of the previously calculated PBR, and therefore it is likely that fishery-related mortality and serious injury cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The status of rough-toothed dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Albertson, G.R. 2014. Worldwide phylogeography and local population structure of the rough-toothed dolphin (*Steno bredanensis*). Ph.D. dissertation, Oregon State University. 179pp.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

da Silva, D.M.P., A.F. Azevedo, E.R. Secchi, L.A. Barbosa, P.A.C. Flores, R.R. Carvalho, T.L. Bisi, J. Lailson-Brito and H.A. Cunha. 2015. Molecular taxonomy and population structure of the rough-toothed dolphin *Steno bredanensis* (Cetartiodactyla: Delphinidae). Zool. J. Linn. Soc. 175:949–962.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Fulling, G.L., K.D. Mullin and C.W. Hubard. 2003. Abundance and distribution of cetaceans in outer continental shelf waters of the U.S. Gulf of Mexico. Fish. Bull. 101:923–932.

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake, and L. Thomas (Eds.). Oxford University Press, New York.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112:161–175.

Maze-Foley, K. and L.P. Garrison. 2020. Serious injury determinations for small cetaceans off the southeast U.S. coast, 2014–2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2020-06. 43pp.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 2000-003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: Time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station, Texas. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indic. 18:278–290.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., G.A. Early, J.G. Gannon, R.G. Lingenfelser and P. Sweeney. 2008. Tagging and tracking of rough-toothed dolphins (*Steno bredanensis*) from the March 2005 mass stranding in the Florida Keys. NOAA Tech. Memo. NMFS-SEFSC-574. 44pp.

West, K.L., J.G. Mead and W. White. 2011. *Steno bredanensis* (Cetacea: Delphinidae). Mammalian Species 43(886):177–189.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

CLYMENE DOLPHIN (*Stenella clymene*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The Clymene dolphin is endemic to tropical and subtropical waters of the Atlantic (Leatherwood and Reeves 1983, Perrin and Mead 1994). Sightings of these animals in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur primarily over the deeper waters off the continental shelf and primarily west of the Mississippi River (Mullin *et al.* 1994, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020; Figure 1). Clymene dolphins were seen in the winter, spring and summer during GulfCet aerial surveys of the northern Gulf of Mexico during 1992 to 1998 (Hansen *et al.* 1996, Mullin and Hoggard 2000).

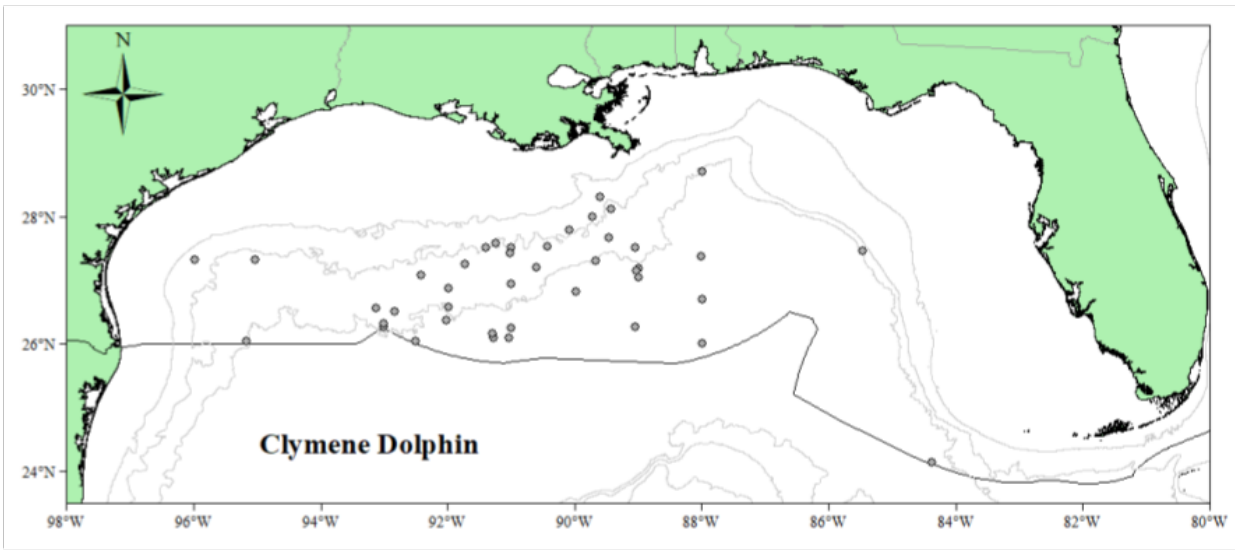


Figure 1. Distribution of Clymene dolphin on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Nara *et al.* (2017) analyzed mitochondrial DNA sequence data from Clymene dolphin samples collected in the western North Atlantic, Gulf of Mexico, and western South Atlantic and found significant genetic differentiation among all three regions, supporting delimitation of separate western North Atlantic and Gulf of Mexico stocks. There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for northern Gulf of Mexico Clymene dolphins is 513 (CV=1.03; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of Clymene dolphin abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=10,900 (CV=0.42); 2004, N=13,257 (CV=0.81); and 2009, N=1,319 (CV=0.78).

Recent Surveys and Abundance Estimates

An abundance estimate for Clymene dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, this species was observed during tracklines when only one survey team was on effort. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. The estimated detection probability on the trackline for similar species was then applied to develop the final abundance estimate. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among species based on their relative density within the survey strata (Garrison *et al.* 2020). The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. There were sightings of Clymene dolphins in 2017 but there were none in 2018. The 2017 and 2018 estimates were N=1,026 (CV=1.03) and N=0 (CV=NA), respectively. The inverse variance weighted mean abundance estimate for Clymene dolphins in oceanic waters during 2017 and 2018 was 513 (CV=1.03; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

***Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico Clymene dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys*.**

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	513	1.03

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Clymene dolphins is 513 (CV=1.03). The minimum population estimate for the northern Gulf of Mexico stock of Clymene dolphins is 250 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. Pairwise comparisons indicated significant differences between the 2003 estimate and both the 2009 (p.adjusted=0.024) and 2017 (p.adjusted=0.030) estimates, and between the 2004 estimate and both the 2009 (p.adjusted=0.039) and 2017 (p.adjusted=0.039) estimates (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 250. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico Clymene dolphin is 2.5 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico Clymene dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
513	1.03	250	0.5	0.04	2.5

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Clymene dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 8.4. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 8.4.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Clymene dolphins.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of Clymene dolphins within high-seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico and there were no observed mortalities or serious injuries to Clymene dolphins by this fishery during 2014–2018 (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There were 16 reported strandings of Clymene dolphins in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019; Table 4). For one stranding, there was evidence of human interaction (healed scars). No evidence of human interaction was detected for two strandings, and for the remaining 13 strandings, it could not be determined whether there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). One stranding of a Clymene dolphin in 2010 was considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 3% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 83 Clymene dolphins died during 2010–2013 (four year annual average of 21) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 42 Clymene dolphins died due to elevated mortality associated with oil exposure. The population model used to predict Clymene dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for Clymene dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Clymene dolphin strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. No strandings of Clymene dolphins occurred in Alabama, Louisiana, or Mississippi.

Area	2014	2015	2016	2017	2018	Total
Florida	0	0	1	13	1	15
Texas	0	0	0	1	0	1
Total	**0**	**0**	**1**	**14**	**1**	**16**

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 7% (95%CI: 3–15) of Clymene dolphins in the Gulf were exposed to oil, that 3% (95%CI: 2–5) of females suffered from reproductive failure, and 3% (95%CI: 1–4) of Clymene dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated that the stock experienced a 3% maximum reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Clymene dolphins are not listed as threatened or endangered under the Endangered Species Act, but the northern Gulf of Mexico stock is considered strategic under the MMPA because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill exceeds PBR. No fishery-related mortality or serious injury has been observed; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Clymene dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. The population trend for this stock is also unknown.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96- 0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 180–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake, and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D., L.V. Higgins, T.A. Jefferson and L.J. Hansen. 1994. Sightings of the Clymene dolphin (*Stenella clymene*) in the Gulf of Mexico. Mar. Mamm. Sci. 10(4):464–470.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 2000-003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Nara, L., A.C.O. de Meirelles, L.R.A. Souto, J.M. Silva, Jr. and A.P.C. Farro. 2017. An initial population structure and genetic diversity analysis for *Stenella clymene* (Gray, 1850): Evidence of differentiation between the North and South Atlantic Ocean. Aquat. Mamm. 43:507–516.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Perrin, W.F. and J.G. Mead. 1994. Clymene dolphin *Stenella clymene* (Gray, 1846). Pages 161–171 *in:* S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 5: The first book of dolphins. Academic Press, London. 416pp.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

FRASER'S DOLPHIN (*Lagenodelphis hosei*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Fraser's dolphins are distributed worldwide in tropical waters (Perrin *et al.* 1994), and they have more recently been reported from temperate and subtropical areas of the North Atlantic (Gomes-Pereira *et al.* 2013). They are generally oceanic in distribution but may be seen closer to shore where deep water can be found near the shore, such as in the Lesser Antilles of the Caribbean Sea (Dolar 2009). Sightings occur only sporadically during vessel surveys in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) and are generally confined to oceanic waters (>200 m) (Figure 1; Maze-Foley and Mullin 2006) and they have been observed in all seasons (Leatherwood *et al.* 1993, Hansen *et al.* 1996, Mullin and Hoggard 2000).

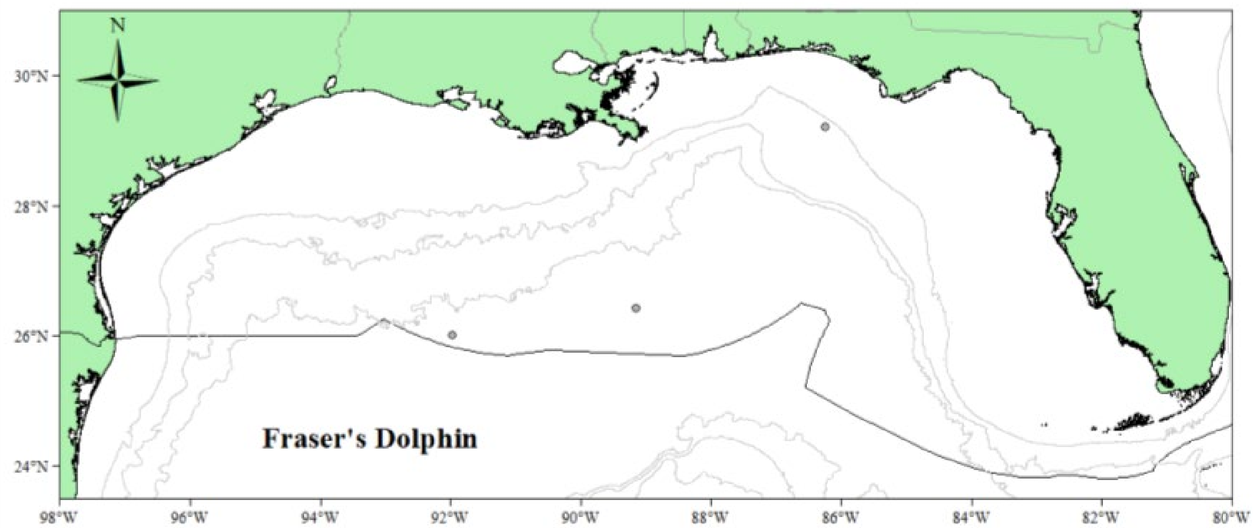


Figure 1. Distribution of Fraser's dolphin on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Fraser's dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). Due to the paucity of sightings in the northern Gulf of Mexico, there are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for northern Gulf of Mexico Fraser's dolphins is 213 (CV=1.03; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

During surveys conducted in 2003, 2004, and 2009, there were no sightings of Fraser's dolphins.

Recent Surveys and Abundance Estimates

An abundance estimate for Fraser's dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, this species was observed during tracklines when only one survey team was on effort. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. The estimated detection probability on the trackline for similar species was then applied to develop the final abundance estimate. The abundance estimate for this stock included sightings of unidentified dolphins that were apportioned among species based on their relative density within the survey strata (Garrison *et al.* 2020). The surveys were conducted in "passing mode" (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in "closing mode." Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. There were sightings of Fraser's dolphins in 2017 but there were none in 2018. The 2017 and 2018 estimates were N=427 (CV=1.03) and N=0 (CV=NA), respectively. The inverse variance weighted mean abundance for Fraser's dolphins in oceanic waters during 2017 and 2018 was 213 (CV=1.03; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico Fraser's dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	213	1.03

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Fraser's dolphins is 213 (CV=1.03). The minimum population estimate for the northern Gulf of Mexico Fraser's dolphin is 104 (Table 2).

Current Population Trend

There are insufficient data to determine the population trends for this species. No Fraser's dolphins were sighted during 2003, 2004, 2009, and 2018 surveys. The fluctuations in total abundance of Fraser's dolphins are probably due to a number of factors. Fraser's dolphin is most certainly a resident species in the Gulf of Mexico but probably occurs in low numbers and the survey effort is not sufficient to estimate the abundance of uncommon or rare species with precision. Also, because this is likely a transboundary stock, the temporal changes in abundance estimates are difficult to interpret without a Gulf of Mexico-wide understanding of Fraser's dolphin distribution and abundance.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 104. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico Fraser's dolphin is 1.0 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico Fraser's dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
213	1.03	104	0.5	0.04	1.0

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Fraser's dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (e.g., the *Deepwater Horizon* oil spill) was unknown (see Habitat Issues section). The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, unknown.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Fraser's dolphins.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for this fishery for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of Fraser's dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to Fraser's dolphins by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There was one mass stranding of five Fraser's dolphins in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). The mass stranding occurred off Florida in July 2017, and it could not be determined if there was evidence of human interaction for any of the dolphins. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no Fraser's dolphin strandings recovered within the spatial and temporal boundaries of this UME.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies did not include Fraser's dolphins regarding impacts of the spill due to insufficient data to determine the overlap of the DWH oil spill footprint and the range of Fraser's dolphins (DWH MMIQT 2015). The impact of the spill on Fraser's dolphins is unknown.

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Fraser's dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of Fraser's dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

Dolar, M.L.L. 2009. Fraser's dolphin: *Lagenodelphis hosei*. Pages 469–471 *in*: W.F. Perrin, B. Würsig, J.G.M. Thewissen (Eds). The encyclopedia of marine mammals, second edition. Academic Press, San Diego, CA.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomes-Pereira, J.N., R. Marques, M.J. Cruz and A. Martins. 2013. The little-known Fraser's dolphin *Lagenodelphis hosei* in the North Atlantic: New records and a review of distribution. Mar. Biodiv. 43:321–332.

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake, and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S., T.A. Jefferson, J.C. Norris, W.E. Stevens, L.J. Hansen and K.D. Mullin. 1993. Occurrence and sounds of Fraser's dolphin in the Gulf of Mexico. Texas J. Sci. 45(4):349–354.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Perrin, W.F., S. Leatherwood and A. Collet. 1994. Fraser's dolphin *Lagenodelphis hosei* (Fraser 1956). Pages 225–240 *in:* S.H. Ridgway and R. Harrison (Eds.). *Handbook* of marine mammals, Vol. 5: The first book of dolphins. Academic Press, London. 416pp.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

KILLER WHALE (*Orcinus orca*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The killer whale is distributed worldwide from tropical to polar regions (Leatherwood and Reeves 1983). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings occur only sporadically during visual surveys and are generally confined to slope and basin waters >700 m (Hansen *et al*. 1996, O'Sullivan and Mullin 1997, Mullin and Hoggard 2000, Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020; Figure 1).

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al*. 2005; Jefferson *et al*. 2008; Vázquez Castán *et al*. 2009; Whitt *et al*. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

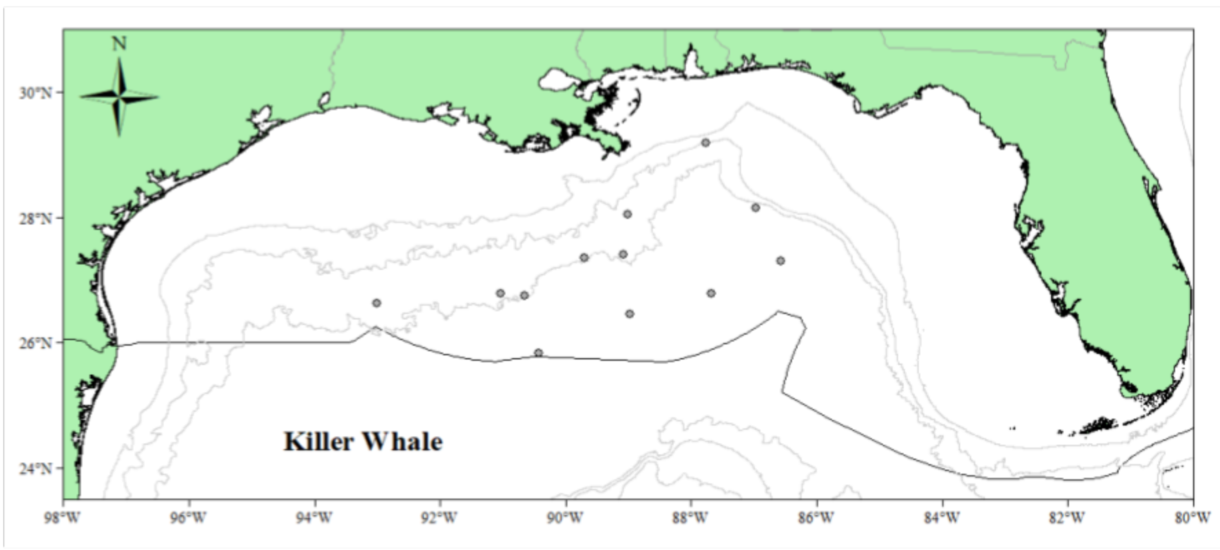


Figure 1. Distribution of killer whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

Killer whales exhibit significant variation in genetic diversity, color pattern, feeding behavior, body size and vocalizations worldwide and several different ecotypes have been identified (Bigg *et al.* 1990, Pitman *et al.* 2007, Foote *et al.* 2009, Parsons *et al.* 2009). Morin *et al.* (2010) analyzed whole mitogenomes and concluded that several ecotypes should be elevated to full species. A single sample from the Gulf of Mexico was included in this study and it grouped most closely with killer whales from the Antarctic to the exclusion of samples collected in the eastern North Atlantic, and a single sample collected in the western North Atlantic (Morin *et al.* 2010). Further work is needed to determine where killer whales in the Gulf of Mexico fit in the global picture of killer whale taxonomy.

Killer whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there is currently no information to differentiate the stocks, such separation is consistent with the fact that the two areas belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011) and the photo-identification data suggest some degree of long-term site fidelity to the Gulf of Mexico. Thirty-two individual killer whales have been photographically identified to date in the northern Gulf of Mexico, with one individual having been sighted over a 20-year period, four whales resighted over 15 years, and three whales resighted over 10 years. Due to the paucity of sightings in the northern Gulf of Mexico, there are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico killer whale is 267 (CV=0.75; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of killer whale abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=0 (CV=NA); 2004, N=198 (CV=1.00); and 2009, N=51 (CV=0.97).

Recent Surveys and Abundance Estimates

An abundance estimate for killer whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). There were sightings of killer whales in 2017 but there were none in 2018. Unidentified small whales observed during the 2018 survey were apportioned by the relative density from the summer 2017 survey to develop an abundance estimate for killer whales in 2018. The 2017 and 2018 estimates were N=86 (CV=0.87) and N=450 (CV=0.88), respectively. The inverse variance weighted mean abundance estimate for killer whales in oceanic waters during 2017 and 2018 was 267 (CV=0.75; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico killer whales in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	267	0.75

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for killer whales is 267 (CV=0.75). The minimum population estimate for the northern Gulf of Mexico killer whale is 152 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August) (see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 152. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico killer whale is 1.5 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico killer whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
267	0.75	152	0.5	0.04	1.5

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to killer whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (e.g., the *Deepwater Horizon* oil spill) was unknown (see Habitat Issues section). The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, unknown.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico killer whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of killer whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to killer whales by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There were no reported strandings of killer whales in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no killer whale strandings recovered within the spatial and temporal boundaries of this UME.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies did not include killer whales regarding impacts of the spill due to insufficient data to determine the overlap of the DWH oil spill footprint and the range of killer whales (DWH MMIQT 2015). The impact of the spill on killer whales is unknown.

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

The northern Gulf of Mexico stock of killer whales is not listed as threatened or endangered under the Endangered Species Act, nor is it considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of killer whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Bigg, M.A., P.F. Olesiuk, G.M. Ellis, J.K.B. Ford and K.C. Balcomb 1990. Social organization and genealogy of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State. Pages 383–405 *in*: P. S. Hammond, S. A. Mizroch and G. P. Donovan (Eds.). Individual recognition of cetaceans: Use of photoidentification and other techniques to estimate population parameters. Rep. Int. Whal. Comm., Cambridge. Special Issue 12.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning and B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Foote, A.D., J. Newton, S.B. Piertney, E. Willerslev and M.T.P. Gilbert. 2009. Ecological, morphological and genetic divergence of sympatric North Atlantic killer whale populations. Mol. Ecol. 18:5207–5217.

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and

Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in*: R. W. Davis and G. S. Fargion. (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling, edited by S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake, and L. Thomas. Oxford University Press, New York.

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Maze-Foley, K. and K.D. Mullin 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Morin, P.A., F.I. Archer, A.D. Foote, J.Vilstrup, E.E. Allen, P. Wade, J. Durban, K. Parson, R. Pitman, L. Li, P. Bouffard, S.C. Abel Nielsen, M. Rasmussen, E. Willerslev, M.T.P. Gilbert and T. Harkins. 2010. Complete mitochondrial genome phylogeographic analysis of killer whales (*Orcinus orca*) indicates multiple species. Genome Res. 20:908–916.

Mullin, K.D. and G.L. Fulling 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in*: R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana. OCS Study MMS 96-0027.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59, 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

O'Sullivan, S. and K.D. Mullin 1997. Killer whales (*Orcinus orca*) in the northern Gulf of Mexico. Mar. Mamm. Sci. 13(1):141–147.

Parsons, K., K.C. Balcomb, J.K.B. Ford and J.W. Durban. 2009. The social dynamics of southern resident killer whales and conservation implications for this endangered population. Anim. Behav. 77:963–971.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Pitman, R.L., W.L. Perryman, D. LeRoi and E. Eilers. 2007. A dwarf form of killer whale in the Antarctic. J. Mammal. 88:43–48.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

FALSE KILLER WHALE (*Pseudorca crassidens*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The false killer whale is distributed worldwide throughout warm temperate and tropical oceans (Leatherwood and Reeves 1983). Sightings of this species in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur in oceanic waters, primarily in the eastern Gulf (Figure 1; Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). They are sporadically seen during vessel and aerial surveys of the northern Gulf of Mexico (Hansen *et al.* 1996, Mullin and Hoggard 2000, Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020).

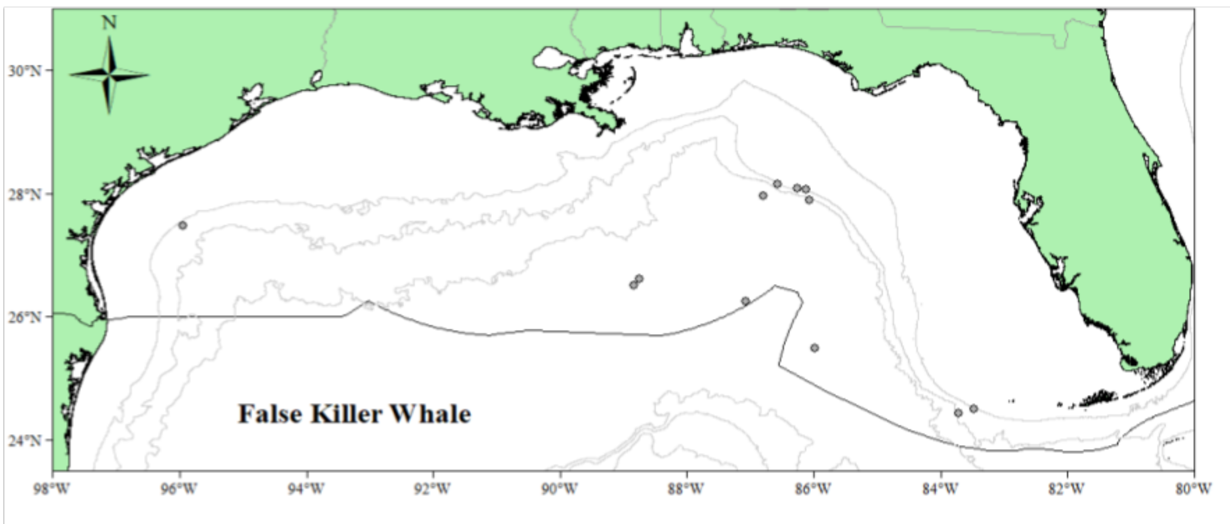


Figure 1. Distribution of false killer whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Genetic analyses (Chivers *et al.* 2007, Martien *et al.* 2014) indicate false killer whales exhibit significant population structuring in the Pacific, with restricted gene flow among whales sampled near the main Hawaiian Islands, the Northwestern Hawaiian Islands, and pelagic waters of the eastern and the central North Pacific. Martien *et al.* (2014) also found their two Atlantic samples to be genetically divergent from those in the Pacific. False killer whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there is currently no information to differentiate the two stocks, such separation is consistent with evidence for strong population structuring in other areas (Martien *et al*. 2014) and further supported because the two stocks occupy distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico false killer whale is 494 (CV=0.79; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of false killer whale abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=1,293 (CV=0.63); 2004, N=0 (CV=NA); and 2009, N=0 (CV=NA).

Recent Surveys and Abundance Estimates

An abundance estimate for false killer whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). There were sightings of false killer whales in 2017 but there were none in 2018. Unidentified small whales observed during the 2018 survey were apportioned by the relative density from the summer 2017 survey to develop an abundance estimate for false killer whales in 2018. The 2017 and 2018 estimates were N=1,069 (CV=0.97) and N=162 (CV=0.74), respectively. The inverse variance weighted mean abundance estimate for false killer whales in oceanic waters during 2017 and 2018 was 494 (CV=0.79; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico false killer whales in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	494	0.79

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for false killer whales is 494 (CV=0.79). The minimum population estimate for the northern Gulf of Mexico false killer whale is 276 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2018 estimates (p.adjusted=0.027) and the 2017 and 2018 estimates (p.adjusted=0.072; Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 276. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico false killer whale is 2.8 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico false killer whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
494	0.79	276	0.5	0.04	2.8

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to false killer whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 2.2. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 2.2.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico false killer whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of false killer whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to false killer whales by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There was one mass stranding of 99 false killer whales in the Gulf of Mexico during 2014–2018 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). The mass stranding occurred within the Everglades National Park in Florida in 2017. Evidence of human interaction was detected for one of the stranded whales (ingested plastic debris), and for the remaining strandings, it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no false killer whale strandings recovered within the spatial and temporal boundaries of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 9% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 21 false killer whales died during 2010–2013 (four year annual average of 5.3) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 11 false killer whales died due to elevated mortality associated with oil exposure. The population model used to predict false killer whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for false killer whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 18% (95%CI: 7–48) of false whales in the Gulf were exposed to oil, that 8% (95%CI: 4–12) of females suffered from reproductive failure, and 7% (95%CI: 3–11) of false killer whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 9% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

False killer whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of false killer whales in the northern Gulf of Mexico, relative to OSP, is unknown. The population trend for this stock is also unknown.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Chivers, S.J., R.W. Baird, D.J. McSweeney, D.L. Webster, N.M. Hedrick and J.C. Salinas. 2007. Genetic variation and evidence for population structure in eastern North Pacific false killer whales (*Pseudorca crassidens*). Can. J. Zool. 85:783–794.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero, Pages 108–189 *in:* Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake, and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Martien, K.K., S.J. Chivers, R.W. Baird, F.I. Archer, A.M. Gorgone, B.L. Hancock-Hanser, D. Mattila, D.J. McSweeney, E.M. Oleson, C. Palmer, V.L. Pease, K.M. Robertson, G.S. Schorr, M.B. Schultz, D.L. Webster and B.L. Taylor. 2014. Nuclear and mitochondrial patterns of population structure in North Pacific false killer whales (*Pseudorca crassidens*). J. Heredity 105:611–626.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J.G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station, Texas. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. Ecol. Indicators. 18:278–290.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

PYGMY KILLER WHALE (*Feresa attenuata*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pygmy killer whale is distributed worldwide in tropical and subtropical waters (Ross and Leatherwood 1994). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species during visual surveys are sporadic and occur primarily in waters >1000 m (Figure 1; Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). Pygmy killer whales have been documented in all seasons (Hansen *et al.* 1996, Mullin and Hoggard 2000).

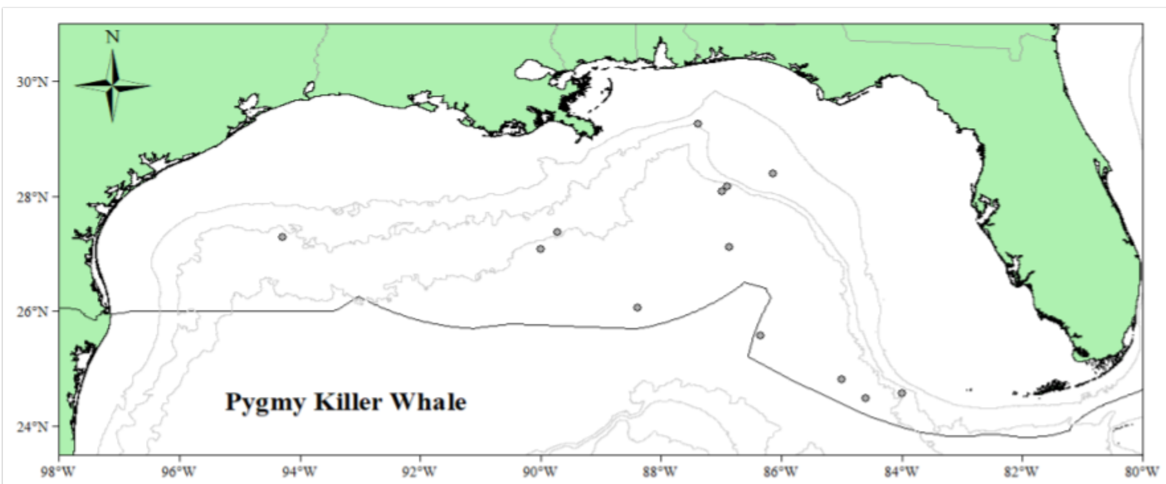


Figure 1. Distribution of pygmy killer whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Pygmy killer whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, such separation is consistent with evidence for population structure in other areas (Baird 2018) and is further supported because the two stocks occupy distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). In addition, two pygmy killer whales that stranded in Mississippi were rehabilitated, tagged with a satellite-linked transmitter, released, and tracked for 15 and 88 days (Pulis *et al.* 2018). Nearly all the tracked locations occurred over continental slope waters ranging from 200 to 1,200 m in depth in the northern Gulf of Mexico. As Wells *et al.* (2009) note, it is difficult to determine the effects of stranding and rehabilitation on post-release behavior, so it is unknown whether these movements were representative of pygmy killer whale ranging patterns in the northern Gulf of Mexico. Due to the paucity of sightings, there are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico pygmy killer whale is 613 (CV=1.15; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of pygmy killer whale abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=501 (CV=0.74); 2004, N=490 (CV=0.87); and 2009, N=359 (CV=0.95).

Recent Surveys and Abundance Estimates

An abundance estimate for pygmy killer whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, this species was observed during tracklines when only one survey team was on effort. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. The estimated detection probability on the trackline for similar species was then applied to develop the final abundance estimate. The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among species based on their relative density within the survey strata (Garrison *et al.* 2020). The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. There were sightings of pygmy killer whales in 2017 but there were none in 2018. The 2017 and 2018 estimates were N=1,227 (CV=1.15) and N=0 (CV=NA), respectively. The inverse variance weighted mean abundance estimate for pygmy killer whales in oceanic waters during 2017 and 2018 was 613 (CV=1.15; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico pygmy killer whales in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	Nest	CV Nest
2017, 2018	Gulf of Mexico	613	1.15

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for pygmy killer whales is 613 (CV=1.15). The minimum population estimate for the northern Gulf of Mexico pygmy killer whale is 283 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the non-zero log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 283. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico pygmy killer whale is 2.8 (Table 2).

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico pygmy killer whale with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
613	1.15	283	0.5	0.04	2.8

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to pygmy killer whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 1.6. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 1.6.

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico pygmy killer whale.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the high seas longline fishery, and no takes of pygmy killer whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagic longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to pygmy killer whales by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b). There has historically been some take of this species in small cetacean fisheries in the Caribbean (Caldwell and Caldwell 1971).

Other Mortality

There were seven reported strandings of pygmy killer whales in the Gulf of Mexico during 2014–2018 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). No evidence of human interaction was detected for three stranded animals, and for the remaining four stranded animals, it could not be determined if there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). However, there were no pygmy killer whale strandings recovered within the spatial and temporal boundaries of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 7% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 16 pygmy killer whales died during 2010–2013 (four year annual average of 3.9) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 8.1 pygmy killer whales died due to elevated mortality associated with oil exposure. The population model used to predict pygmy killer whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for pygmy killer whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Pygmy killer whale strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of pygmy killer whales in Alabama, Louisiana, or Texas.

Area	2014	2015	2016	2017	2018	Total
Florida	0	1	1	0	3	5
Mississippi	0	2	0	0	0	2
Total	**0**	**3**	**1**	**0**	**3**	**7**

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 15% (95%CI: 7–33) of pygmy killer whales in the Gulf were exposed to oil, that 7% (95%CI: 3–10) of females suffered from reproductive failure, and 6% (95%CI: 2–9) of pygmy killer whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 7% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Pygmy killer whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of pygmy killer whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Baird, R.W. 2018. Pygmy killer whale, *Feresa attenuata*. Pages 788–790 *in*: B. Würsig, J.G.M. Thewissen, and K M. Kovacs (Eds.). Encyclopedia of marine mammals, 3rd ed. Academic Press/Elsevier, San Diego, California.

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Caldwell, D. K. and M. C. Caldwell. 1971. The pygmy killer whale, *Feresa attenuata*, in the western Atlantic, with a summary of world records. J. Mamm. 52:206–209.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans.

Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1): 27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick, eds. 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern

Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 2000-003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Pulis, E.E., R.S. Wells, G.S. Schorr, D.C. Douglas, M.M. Samuelson and M. Solangi. 2018. Movements and dive patterns of pygmy killer whales (*Feresa attenuata*) released in the Gulf of Mexico following rehabilitation. Aquat. Mamm. 44(5):555–567.

Ross, G.J.B. and S. Leatherwood. 1994. Pygmy killer whale *Feresa attenuata* (Gray, 1874). Pages 387-404 *in:* S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 5: The first book of dolphins. Academic Press, London. 416pp.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola. 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., C.A. Manire, L. Byrd, D.R. Smith, J.G. Gannon, D. Fauquier and K.D. Mullin. 2009. Movements and dive patterns of a rehabilitated Risso's dolphin, Grampus griseus, in the Gulf of Mexico and Atlantic Ocean. Mar. Mamm. Sci. 25(2):420–429.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

DWARF SPERM WHALE (*Kogia sima*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The dwarf sperm whale is distributed worldwide in temperate to tropical waters (Caldwell and Caldwell 1989). Sightings of these animals in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur primarily in oceanic waters (Figure 1; Mullin *et al.* 1991, Mullin and Fulling 2004, Maze-Foley and Mullin 2006). Dwarf sperm whales and pygmy sperm whales (*Kogia breviceps*) are often difficult to differentiate at sea (Caldwell and Caldwell 1989, Bloodworth and Odell 2008, McAlpine 2009) unless sighting conditions are ideal, and sightings of either species are usually categorized as *Kogia* spp. In addition, the acoustic signals of dwarf and pygmy sperm whales also cannot be distinguished from each other at this time (Merkens *et al.* 2018) adding to the difficulties of identification at sea.

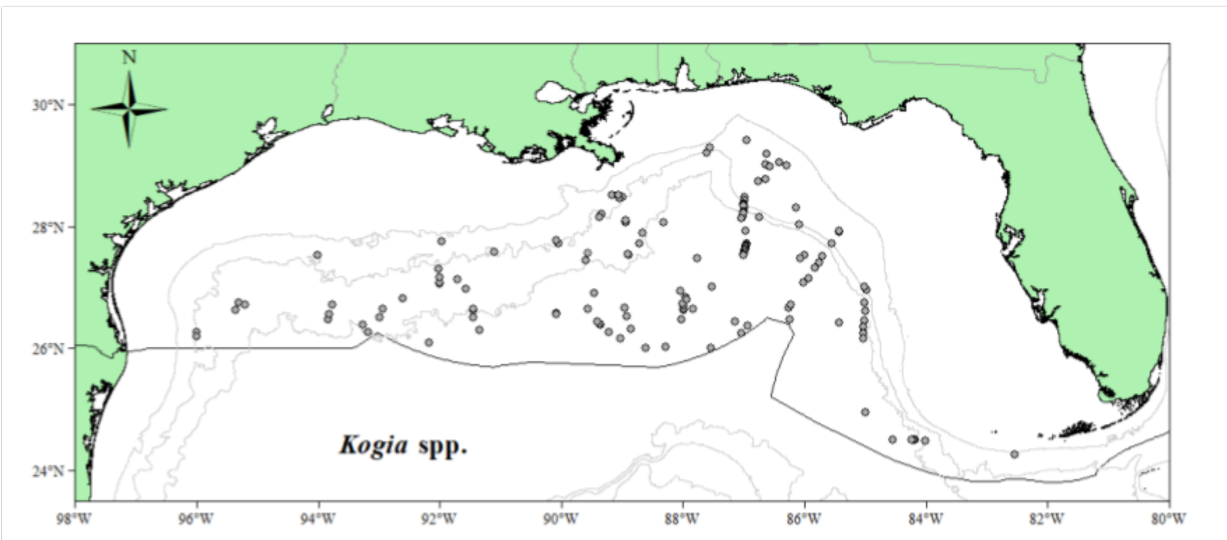


Figure 1. Distribution of* Kogia *spp. on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

In the northern Gulf of Mexico, *Kogia* spp. are sighted in waters >200 m, over the continental slope and deep basin. They have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (Hansen *et al.* 1996, Mullin and Hoggard 2000, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020; Figure 1). All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al*. 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Dwarf sperm whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for northern Gulf of Mexico dwarf and pygmy sperm whales combined is 336 (CV=0.35; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate because *Kogia* spp. are often difficult to see, present little of themselves at the surface, do not fluke when they dive, and have long dive times. In addition, they exhibit avoidance behavior towards ships and changes in behavior towards approaching survey aircraft (Würsig *et al.* 1998).

Earlier Abundance Estimates

Five point estimates of dwarf and pygmy sperm whale combined abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences. This resulted in revised abundance estimates for dwarf and pygmy sperm whales combined of: 2003, N=441 (CV=0.42); 2004, N=38 (CV=0.71); and 2009, N=124 (CV=0.60; Garrison *et al.* 2020).

Recent Surveys and Abundance Estimates

An abundance estimate for dwarf and pygmy sperm whales combined was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of these species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21, Laake *et al.* 2020) in the R statistical programming language. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The 2017 and 2018 estimates were N=293 (CV=0.59) and N=359 (CV=0.42), respectively. The inverse variance weighted mean abundance estimate for dwarf and pygmy sperm whales in oceanic waters during 2017 and 2018 was 336 (CV=0.35; Table 1; Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of these species.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico dwarf and pygmy sperm whales in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	336	0.35

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for dwarf and pygmy sperm whales is 336 (CV=0.35). It is not possible to determine the minimum population estimate for only dwarf sperm whales. The minimum population estimate for the northern Gulf of Mexico dwarf and pygmy sperm whales is 253 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2004 estimates (p.adjusted=0.006) and between the 2004 estimate and both the 2017 (p.adjusted=0.063) and 2018 (p.adjusted=0.014) estimates (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for dwarf and pygmy sperm whales is 253. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico dwarf and pygmy sperm whales is 2.5 (Table 2). It is not possible to determine the PBR for only dwarf sperm whales.

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico dwarf and pygmy sperm whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
336	0.35	253	0.5	0.04	2.5

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to dwarf or pygmy sperm whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for dwarf and pygmy sperm whales due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 31. The minimum total mean annual human-caused mortality and serious injury for dwarf and pygmy sperm whales during 2014–2018 was, therefore, 31. The minimum total mean annual human-caused mortality and serious injury for dwarf sperm whales is unknown.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico dwarf and pygmy sperm whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of pygmy or dwarf sperm whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to dwarf or pygmy sperm whales by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

At least nine dwarf sperm whale strandings were documented in the northern Gulf of Mexico during 2014–2018 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). No evidence of human interaction was detected for one animal; for the remaining eight animals, it could not be determined if there was evidence of human interaction. An additional 10 *Kogia* spp. stranded during 2014–2018. No evidence of human interaction was detected for one of the *Kogia* spp. strandings; it could not be determined if there was evidence of human interaction for the remaining nine *Kogia* spp. strandings. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). Four dwarf sperm whale strandings were considered to be part of this UME, one of which occurred during 2014.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that dwarf and pygmy sperm whale stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 340 dwarf and pygmy sperm whale died during 2010–2013 (four year annual average of 85) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 154 dwarf and pygmy sperm whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict dwarf/pygmy sperm whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for dwarf/pygmy sperm whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Dwarf and pygmy sperm whale (Kogia sima (Ks), Kogia breviceps (Kb) and Kogia spp. (Sp)) strandings along the northern Gulf of Mexico coast, 2014–2018. Strandings that were not reported to species have been reported as Kogia spp. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded Kogia whales to species, reports to specific species should be viewed with caution.

State	2014	2014	2014	2015	2015	2015	2016	2016	2016	2017	2017	2017	2018	2018	2018	Total	Total	Total
	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp
Alabama	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0
Florida	2	0	1	0	2	1	0	3	1	2	1	1	2	1	0	6	7	4
Louisiana	1	0	2	0	0	0	0	0	0	0	1	0	0	1	0	1	2	2
Mississippi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Texas	0	2	1	0	2	1	1	0	0	1	1	2	0	4	0	2	9	4
Total	**3**	**2**	**4**	**0**	**4**	**2**	**1**	**3**	**1**	**3**	**3**	**3**	**2**	**7**	**0**	**9**	**19**	**10**

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 15% (95%CI: 8–29) of dwarf/pygmy sperm whales in the Gulf were exposed to oil, that 7% (95%CI: 3–10) of females suffered from reproductive failure, and 6% (95%CI: 2–9) of dwarf/pygmy sperm whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Dwarf sperm whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all dwarf and pygmy sperm whales combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of dwarf sperm whales in the northern Gulf of Mexico, relative to OSP, is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Bloodworth, B.E. and D.K. Odell. 2008. *Kogia breviceps* (Cetacea: Kogiidae). Mamm. Species 819:1–12.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Caldwell, D.K. and M.C. Caldwell. 1989. Pygmy sperm whale *Kogia breviceps* (de Blainville, 1838): Dwarf sperm whale *Kogia sima* Owen, 1866. Pages 235–260 *in*: S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 4: River dolphins and the larger toothed whales. Academic Press, San Diego, California.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1): 27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108-189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

McAlpine, D.F. 2009. Pygmy and dwarf sperm whales. Pages 936–938 *in*: W.F. Perrin, B. Würsig, and J.G.M. Thewissen (Eds.). Encyclopedia of marine mammals, Second edition. Academic Press, San Diego, California. 1352pp.

Merkens, K., D. Mann, V.M. Janik, D. Claridge, M. Hill and E. Oleson. 2018. Clicks of dwarf sperm whales (*Kogia sima*). Mar. Mamm. Sci. 34(4):963–978.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Mullin, K., W. Hoggard, C. Roden, R. Lohoefener, C. Rogers and B. Taggart. 1991. Cetaceans on the upper continental slope in the north-central Gulf of Mexico. OCS Study/MMS 91-0027. U.S. Dep. Interior, Minerals Management Service, Gulf of Mexico OCS Regional Office, New Orleans, Louisiana. 108pp.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J.G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

Würsig, B., S.K. Lynn, T.A. Jefferson and K.D. Mullin. 1998. Behavior of cetaceans in the northern Gulf of Mexico relative to survey ships and aircraft. Aquat. Mamm. 24:41–50.

PYGMY SPERM WHALE (*Kogia breviceps*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pygmy sperm whale is distributed worldwide in temperate to tropical waters (Caldwell and Caldwell 1989, Bloodworth and Odell 2008). Sightings of these animals in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur primarily in oceanic waters (Figure 1; Mullin *et al.* 1991, Mullin and Fulling 2004, Maze-Foley and Mullin 2006). Pygmy sperm whales and dwarf sperm whales (*Kogia sima*) are often difficult to differentiate at sea (Caldwell and Caldwell 1989, Bloodworth and Odell 2008, McAlpine 2009) unless sighting conditions are ideal, and sightings of either species are often categorized as *Kogia* spp. In addition, the acoustic signals of dwarf and pygmy sperm whales also cannot be distinguished from each other at this time (Merkens *et al.* 2018) adding to the difficulties of identification at sea.

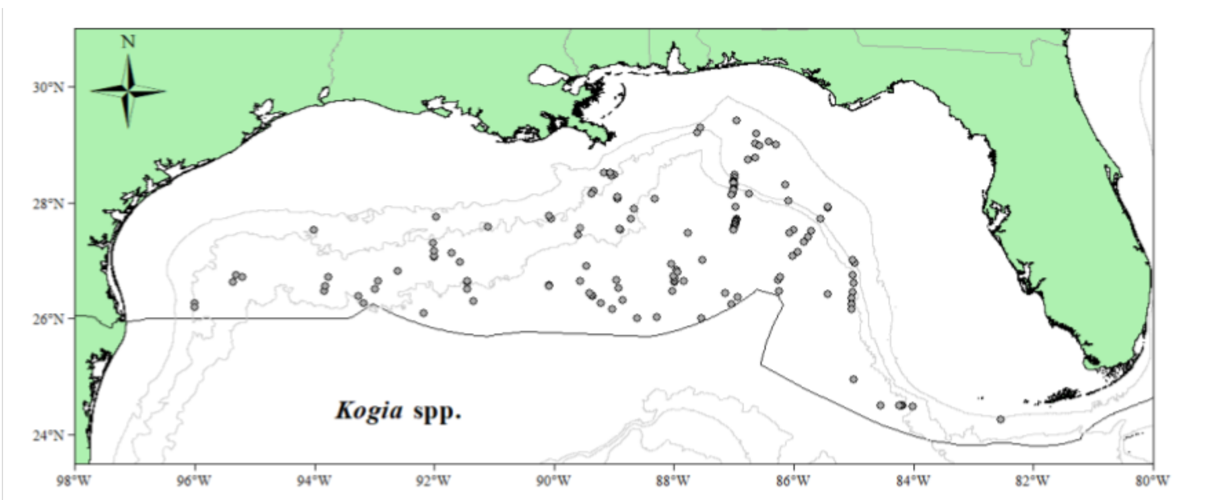


Figure 1. Distribution of* Kogia *spp. on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

In the northern Gulf of Mexico, *Kogia* spp. are sighted in waters >200 m, over the continental slope and deep basin. They have been seen in all seasons during aerial and vessel surveys of the northern Gulf of Mexico (Hansen *et al.* 1996, Mullin and Hoggard 2000, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020; Figure 1). All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Pygmy sperm whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with the fact that the Gulf of Mexico and western North Atlantic belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for northern Gulf of Mexico pygmy and dwarf sperm whales combined is 336 (CV=0.35; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate because *Kogia* spp. are often difficult to see, present little of themselves at the surface, do not fluke when they dive, and have long dive times. In addition, they exhibit avoidance behavior towards ships and changes in behavior towards approaching survey aircraft (Würsig *et al.* 1998).

Earlier Abundance Estimates

Five point estimates of dwarf and pygmy sperm whale combined abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than "closing" mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences. This resulted in revised abundance estimates for dwarf and pygmy sperm whales combined of: 2003, N=441 (CV=0.42); 2004, N=38 (CV=0.71); and 2009, N=124 (CV=0.60; Garrison *et al.* 2020).

Recent Surveys and Abundance Estimates

An abundance estimate for pygmy and dwarf sperm whales combined was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure to allow estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). However, there were too few sightings and too few resightings of these species to allow estimation of detection probability on the trackline. Therefore, abundance estimates were derived using MCDS distance sampling methods that accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip (Thomas *et al.* 2010) implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The 2017 and 2018 estimates were N=293 (CV=0.59) and N=359 (CV=0.42), respectively. The inverse variance weighted mean abundance estimate for dwarf and pygmy sperm whales in oceanic waters during 2017 and 2018 was 336 (CV=0.35; Table 1; Garrison *et al.* 2020). This estimate was not corrected for the probability of detection on the trackline, and is likely a severe underestimate due to the long dive times of these species.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico pygmy and dwarf sperm whales in oceanic waters (200m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	336	0.35

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for pygmy and dwarf sperm whales is 336 (CV=0.35). It is not possible to determine the minimum population estimate for only pygmy sperm whales. The minimum population estimate for the northern Gulf of Mexico pygmy and dwarf sperm whales is 253 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2004 estimates (p.adjusted=0.006) and between the 2004 estimate and both the 2017 (p.adjusted=0.063) and 2018 (p.adjusted=0.014) estimates (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size for pygmy and dwarf sperm whales is 253. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico pygmy and dwarf sperm whales is 2.5 (Table 2). It is not possible to determine the PBR for only pygmy sperm whales.

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico pygmy and dwarf sperm whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
336	0.35	253	0.5	0.04	2.5

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to pygmy or dwarf sperm whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 for pygmy and dwarf sperm whales due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 31. The minimum total mean annual human-caused mortality and serious injury for pygmy and dwarf sperm whales during 2014–2018 was, therefore, 31. The minimum total mean annual human-caused mortality and serious injury for pygmy sperm whales is unknown.

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico pygmy and dwarf sperm whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery, and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of pygmy or dwarf sperm whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to dwarf or pygmy sperm whales by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

At least 19 pygmy sperm whale strandings were documented in the northern Gulf of Mexico during 2014–2018 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Evidence of human interaction was detected for one of the stranded animals, and this involved the ingestion of plastic debris. For four of the strandings, no evidence of human interaction was detected, and for the remaining 14, it could not be determined if there was evidence of human interaction. An additional 10 *Kogia* sp. stranded during 2014–2018. No evidence of human interaction was detected for one of the *Kogia* sp. strandings; it could not be determined if there was evidence of human interaction for the remaining nine *Kogia* sp. strandings. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). Four pygmy sperm whale strandings (from 2011, 2012 and 2013) were considered to be part of this UME. A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that dwarf and pygmy sperm whale stocks experienced a 6% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 340 dwarf and pygmy sperm whale died during 2010–2013 (four year annual average of 85) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 154 dwarf and pygmy sperm whales died due to elevated mortality associated with oil exposure. However, this mortality estimate is not comparable to the current abundance estimate derived from visual surveys because the population model included a correction factor for detection probability derived from acoustic density estimates (DWH MMIQT 2015). The population model used to predict dwarf/pygmy sperm whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for dwarf/pygmy sperm whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Dwarf and pygmy sperm whale (Kogia sima (Ks), Kogia breviceps (Kb) and Kogia sp. (Sp)) strandings along the northern Gulf of Mexico coast, 2014–2018. Strandings that were not reported to species have been reported as Kogia spp. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded Kogia whales to species, reports to specific species should be viewed with caution.

	2014	2014	2014	2015	2015	2015	2016	2016	2016	2017	2017	2017	2018	2018	2018	Total	Total	Total
State	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp
Alabama	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0
Florida	2	0	1	0	2	1	0	3	1	2	1	1	2	1	0	6	7	4
Louisiana	1	0	2	0	0	0	0	0	0	0	1	0	0	1	0	1	2	2
Mississippi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Texas	0	2	1	0	2	1	1	0	0	1	1	2	0	4	0	2	9	4
Total	**3**	**2**	**4**	**0**	**4**	**2**	**1**	**3**	**1**	**3**	**3**	**3**	**2**	**7**	**0**	**9**	**19**	**10**

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 15% (95%CI: 8–29) of dwarf/pygmy sperm whales in the Gulf were exposed to oil, that 7% (95%CI: 3–10) of females suffered from reproductive failure, and 6% (95%CI: 2–9) of dwarf/pygmy sperm whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stocks experienced a maximum 6% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Pygmy sperm whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA because PBR is likely a severe underestimate due to the long dive times of this species and because the mean modeled annual human-caused mortality and serious injury due to the DWH oil spill is based on all dwarf and pygmy sperm whales combined and cannot be apportioned to individual species. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of pygmy sperm whales in the northern Gulf of Mexico, relative to OSP, is unknown. There are insufficient data to determine the population trends for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Bloodworth, B.E. and D.K. Odell. 2008. *Kogia breviceps* (Cetacea: Kogiidae). Mammalian Species 819:1–12.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Caldwell, D.K. and M.C. Caldwell. 1989. Pygmy sperm whale *Kogia breviceps* (de Blainville, 1838): Dwarf sperm whale *Kogia sima* Owen, 1866. Pages 235–260 *in*: S.H. Ridgway and R. Harrison (Eds.). Handbook of marine mammals, Vol. 4: River dolphins and the larger toothed whales. Academic Press, San Diego, California.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning and B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1): 27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

McAlpine, D.F. 2009. Pygmy and dwarf sperm whales. Pages 936–938 *in:* W.F. Perrin, B. Würsig, and J.G.M. Thewissen (Eds.). Encyclopedia of marine mammals, Second edition. Academic Press, San Diego, California. 1352pp.

Merkens, K., D. Mann, V.M. Janik, D. Claridge, M. Hill and E. Oleson. 2018. Clicks of dwarf sperm whales (*Kogia sima*). Mar. Mamm. Sci. 34(4):963–978.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Mullin, K., W. Hoggard, C. Roden, R. Lohoefener, C. Rogers and B. Taggart. 1991. Cetaceans on the upper continental slope in the north-central Gulf of Mexico. OCS Study/MMS 91-0027. U.S. Dep. Interior, Minerals Management Service, Gulf of Mexico OCS Regional Office, New Orleans, Louisiana. 108pp.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station, Texas. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

Würsig, B., S.K. Lynn, T.A. Jefferson and K.D. Mullin. 1998. Behavior of cetaceans in the northern Gulf of Mexico relative to survey ships and aircraft. Aquat. Mamm. 24:41–50.

MELON-HEADED WHALE (*Peponocephala electra*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The melon-headed whale is distributed worldwide in tropical to subtropical waters (Jefferson *et al.* 2008). Sightings in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) have generally occurred in water depths >800 m and west of Mobile Bay, Alabama (Figure 1; Mullin *et al.* 1994, Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020) and have been documented in all seasons (Hansen *et al.* 1996, Mullin and Hoggard 2000).

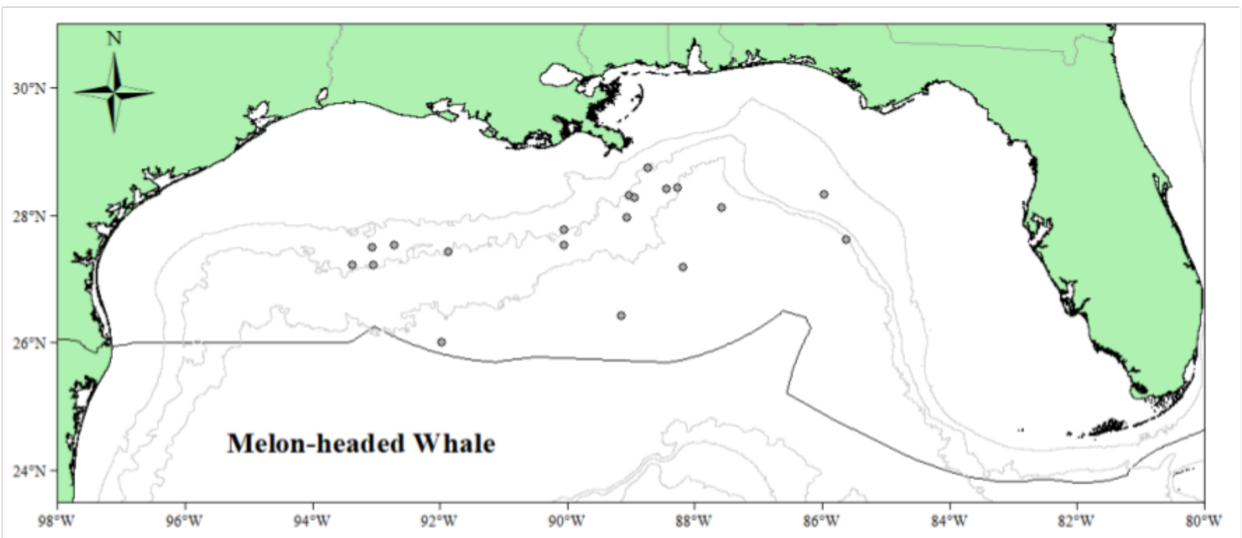


Figure 1. Distribution of melon-headed whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless, there are records for most oceanic species in the southern Gulf (e.g., Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Melon-headed whales in the Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with evidence for population structuring in other areas (Martien *et al.* 2017) and is further supported because the two stocks occupy distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico melon-headed whale is 1,749 (CV=0.68; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of melon-headed whale abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than "closing" mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=1,502 (CV=0.96); 2004, N=7,351 (CV=0.87); and 2009, N=4,188 (CV=0.76).

Recent Surveys and Abundance Estimates

An abundance estimate for melon-headed whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). The 2017 and 2018 estimates were N=2,694 (CV=0.76) and N=454 (CV=0.89), respectively. The inverse variance weighted mean abundance estimate for melon-headed whales in oceanic waters during 2017 and 2018 was 1,749 (CV=0.68; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico melon-headed whales in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	1,749	0.68

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for melon-headed whales is 1,749 (CV=0.68). The minimum population estimate for the northern Gulf of Mexico melon-headed whale is 1,039 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There was a significant difference between the 2004 and 2018 estimates (p.adjusted=0.047; Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,039. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico melon-headed whale is 10 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico melon-headed whales with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
1,749	0.68	1,039	0.5	0.04	10

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to melon-headed whales in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 9.5. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 9.5.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico melon-headed whales.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of melon-headed whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to melon-headed whales by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b). There has historically been some take of this species in small cetacean fisheries in the Caribbean (Caldwell *et al.* 1976).

Other Mortality

There were 12 reported strandings of melon-headed whales in the Gulf of Mexico during 2014–2018 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). Evidence of human interaction was detected for one stranding (the interaction being the animal was pushed back into the water by the public). No evidence of human interaction was detected for three strandings, and for the remaining eight strandings, it could not be determined whether there was evidence of human interaction. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al.* 2012, Wells *et al.* 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al.* 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). Ten melon-headed whale strandings were considered to be part of this UME, one of which occurred during 2014.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 7% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 93 melon-headed whales died during 2010–2013 (four year annual average of 23) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 47 melon-headed whales died due to elevated mortality associated with oil exposure. The population model used to predict melon-headed whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for melon-headed whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Melon-headed whale strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019.

Area	2014	2015	2016	2017	2018	Total
Alabama	0	1	0	2	0	3
Florida	1[a]	0	0	2	3	6
Louisiana	0	0	1	0	0	1
Mississippi	0	0	0	0	0	0
Texas	1	0	1	0	0	2
Total	**2**	**1**	**2**	**4**	**3**	**12**

a. This stranding was part of the Northern Gulf of Mexico UME.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil and gas were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 15% (95%CI: 6–36) of melon-headed whales in the Gulf were exposed to oil, that 7% (95%CI: 3–10) of females suffered from reproductive failure, and 6% (95%CI: 2–9) of melon-headed whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 7% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Melon-headed whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the Marine Mammal Protection Act. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered insignificant and approaching the zero mortality and serious injury rate. The status of melon-headed whales in the northern Gulf of Mexico, relative to OSP, is unknown. The population trend for this stock is also unknown.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Caldwell, D.K., M.C. Caldwell and R.V. Walker. 1976. First records for Fraser's dolphin (*Lagenodelphis hosei*) in the Atlantic and the melon-headed whale (*Peponocephala electra*) in the western Atlantic. Cetology 25:1–4.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in*: Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake, and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112(2):161–175.

Martien, K.K., B.L. Hancock-Hanser, R.W. Baird, J.J. Kiszka, J.M. Aschettino, M. Oremus and M.C. Hill. 2017. Unexpected patterns of global population structure in melon-headed whales, *Peponocephala electra*. Mar. Ecol. Prog. Ser. 577:205–220.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G.L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Mullin, K.D., T.A. Jefferson, L.J. Hansen and W. Hoggard. 1994. First sightings of melon-headed whales (*Peponocephala electra*) in the Gulf of Mexico. Mar. Mamm. Sci. 10(3):342–348.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170 pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3-5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

RISSO'S DOLPHIN (*Grampus griseus*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Risso's dolphins are distributed worldwide in tropical to warm temperate waters (Leatherwood and Reeves 1983, Jefferson *et al.* 2014). Risso's dolphins in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur throughout oceanic waters but are concentrated in continental slope waters (Figure 1; Baumgartner 1997, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). This species has been observed in all seasons in the northern Gulf of Mexico (Hansen *et al.* 1996, Mullin and Hoggard 2000).

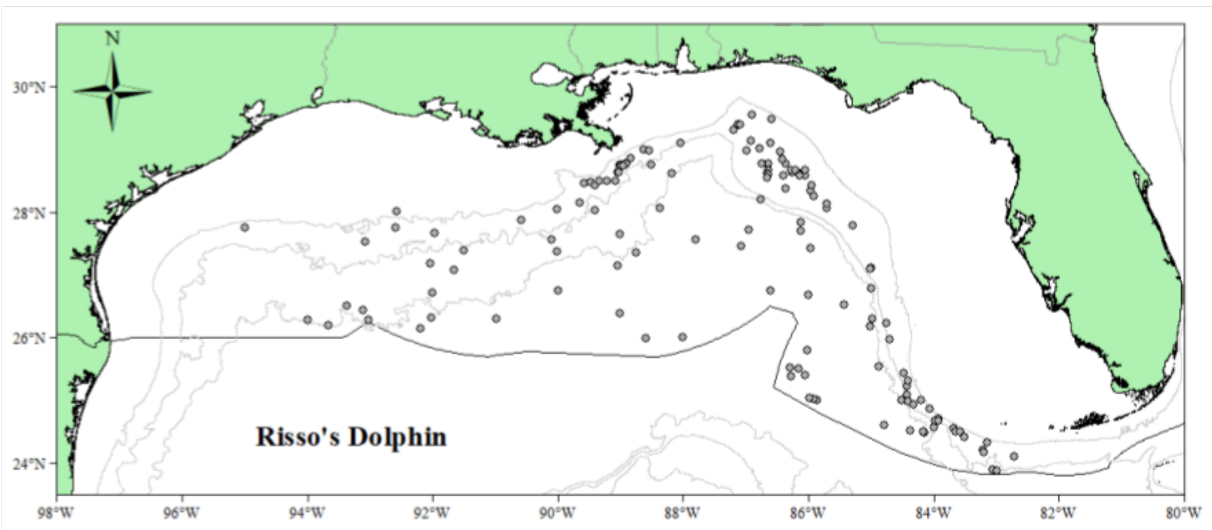


Figure 1. Distribution of Risso's dolphin on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known. Risso's dolphins in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, this management structure is consistent with acoustic evidence. The frequency values of spectral peaks in Risso's dolphin echolocation clicks differ between the western North Atlantic and Gulf of Mexico stocks (Soldevilla *et al.* 2017). In addition, these two stocks occupy distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011) and biogeographic endemism has been identified for Risso's dolphins in the North Pacific (Chen *et al.* 2018). However, a stranded, rehabilitated Risso's dolphin that was released and tagged with a satellite-linked transmitter moved from the Gulf release site near Tampa, Florida, into the Atlantic Ocean and north to just off of Delaware over a 23 day period (Wells *et al.* 2009), suggesting the possibility of connectivity between the two basins. As Wells *et al.* (2009) note, it is difficult to determine the effects of stranding and rehabilitation on post-release behavior, so it is unknown whether these movements were representative of Risso's dolphin ranging patterns in either the Gulf of Mexico or Atlantic Ocean. There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico Risso's dolphin is 1,974 (CV=0.46; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of Risso's dolphin abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than "closing" mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=4,471 (CV=0.47); 2004, N=4,641 (CV=0.86); and 2009, N=7,788 (CV=0.67).

Recent Surveys and Abundance Estimates

An abundance estimate for Risso's dolphins was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Table 1; Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). The 2017 and 2018 estimates were N=2,998 (CV=0.52) and N=632 (CV=0.60), respectively. The inverse variance weighted mean abundance estimate for Risso's dolphins in oceanic waters during 2017 and 2018 was 1,974 (CV=0.46; Table 1; Garrison *et al.* 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico Risso's dolphins in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	1,974	0.46

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Risso's dolphins is 1,974 (CV=0.46). The minimum population estimate for the northern Gulf of Mexico Risso's dolphin is 1,368 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were significant differences between the 2003 and 2018 estimates (p.adjusted=0.026) and the 2009 and 2018 estimates (p.adjusted=0.011; Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,368. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because the stock is of unknown status. PBR for the northern Gulf of Mexico Risso's dolphin is 14 (Table 2).

Table 2. Best and minimum abundance estimates for northern Gulf of Mexico Risso's dolphins with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
1,974	0.46	1,368	0.5	0.04	14

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2014–2018 was presumed to be zero, as there were no reports of mortalities or serious injuries to Risso's dolphins in the Gulf of Mexico (Table 3). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 5.3. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 5.3.

Table 3. Total annual estimated fishery-related mortality and serious injury for northern Gulf of Mexico Risso's dolphins.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0	-

Fisheries Information

There are two commercial fisheries that interact, or that could potentially interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the high seas longline fishery, and no takes of Risso's dolphins within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. During 2014–2018 there were no observed mortalities or serious injuries to Risso's dolphins by this fishery (Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b).

Other Mortality

There were five reported strandings of Risso's dolphins in the Gulf of Mexico during 2014–2018 (Table 4; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). No evidence of human interaction was detected for one of the stranded animals, and it could not be determined if there was evidence of human interaction for the remaining four stranded animals. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012, Wells *et al*. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Since 1990, there have been 13 common bottlenose dolphin or cetacean die-offs or Unusual Mortality Events (UMEs) in the northern Gulf of Mexico, and two of these included a Risso's dolphin. Between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with K. brevis blooms and fish kills in the Florida Panhandle (additional strandings included three Atlantic spotted dolphins, *Stenella frontalis*, one Risso's dolphin, two Blainville's beaked whales, *Mesoplodon densirostris*, and four unidentified dolphins. Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012, Litz *et al.* 2014). A UME was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). One Risso's dolphin stranding from 2012 was considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 3% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 52 Risso's dolphins died during 2010–2013 (four year annual average of 13) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 27 Risso's dolphins died due to elevated mortality associated with oil exposure. The population model used to predict Risso's dolphin mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for Risso's dolphins occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 4. Risso's dolphin strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of Risso's dolphins in Alabama, Mississippi, or Texas.

State	2014	2015	2016	2017	2018	Total
Florida	0	1	3	0	0	4
Louisiana	0	0	0	1	0	1
Total	**0**	**1**	**3**	**1**	**0**	**5**

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 8% (95%CI: 5–13) of Risso's dolphins in the Gulf were exposed to oil, that 3% (95%CI: 2–5) of females suffered from reproductive failure, and 3% (95%CI: 1–4) of Risso's dolphins suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 3% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Risso's dolphins are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. No fishery-related mortality or serious injury has been observed in recent years; therefore, total fishery-related mortality and serious injury can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of Risso's dolphins in the northern Gulf of Mexico, relative to OSP, is unknown. The population trend for this stock is also unknown.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Baumgartner, M.F. 1997. The distribution of Risso's dolphin (*Grampus griseus*) with respect to physiography in the northern Gulf of Mexico. Mar. Mamm. Sci. 13:614–638.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Chen, I., S. Nishida, L.S. Chou, Y. Tajima, W.C. Yang, T. Isobe, T.K. Yamada, K. Hartman and A.R. Hoelzel. 2018. Concordance between genetic diversity and marine biogeography in a highly mobile marine mammal, the Risso's dolphin. J. Biogeogr. 45(9):2092–2103.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gomez, C., J.W. Lawson, A.J. Wright, A.D. Buren, D. Tollit and V. Lesage. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Can. J. Zool. 94:801–819.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in*: R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1): 27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Jefferson, T.A., C.R. Weir, R.C. Anderson, L.T. Ballance, J. Barlow, R.D. Kenney and J.J. Kiszka. 2014. Global distribution of Risso's dolphin *Grampus griseus*: A review and critical evaluation. Mamm. Rev. 44:56–68.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108-189 *in:* Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas. Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112:161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in*: R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana. OCS Study MMS 96-0027.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J.G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators. 18:278–290.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Soldevilla, M.S., S. Baumann-Pickering, D. Cholewiak, L.E.W. Hodge, E.M. Oleson and S. Rankin. 2017. Geographic variation in Risso's dolphin echolocation click spectra. J. Acoust. Soc. Am. 142(2):599–617.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Twiner, M.J., L.J. Flewelling, S.E. Fire, S.R. Bowen-Stevens, J.K. Gaydos, C.K. Johnson, J.H. Landsberg, T.A. Leighfield, B. Mase-Guthrie, L. Schwacke, F.M. Van Dolah, Z. Wang and T.K. Rowles. 2012. Comparative analysis of three brevetoxin-associated bottlenose dolphin (*Tursiops truncatus*) mortality events in the Florida panhandle region (USA). PLoS ONE 7(8):e42974.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, Washington. NOAA Tech. Memo. NMFS-OPR-12. 93pp.

Wells, R.S., C.A. Manire, L. Byrd, D.R. Smith, J.G. Gannon, D. Fauquier and K.D. Mullin. 2009. Movements and dive patterns of a rehabilitated Risso's dolphin, *Grampus griseus*, in the Gulf of Mexico and Atlantic Ocean. Mar. Mamm. Sci. 25(2):420–429.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

SHORT-FINNED PILOT WHALE (*Globicephala macrorhynchus*): Northern Gulf of Mexico Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The short-finned pilot whale is distributed worldwide in tropical to temperate waters (Leatherwood and Reeves 1983). In the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico), sightings of this species occur primarily on the continental slope west of 89˚W (Figure 1; Mullin and Fulling 2004, Maze-Foley and Mullin 2006, Garrison and Aichinger Dias 2020). Short-finned pilot whales have been seen in all seasons during NMFS visual surveys of the northern Gulf of Mexico (Hansen *et al.* 1996, Mullin and Hoggard 2000).

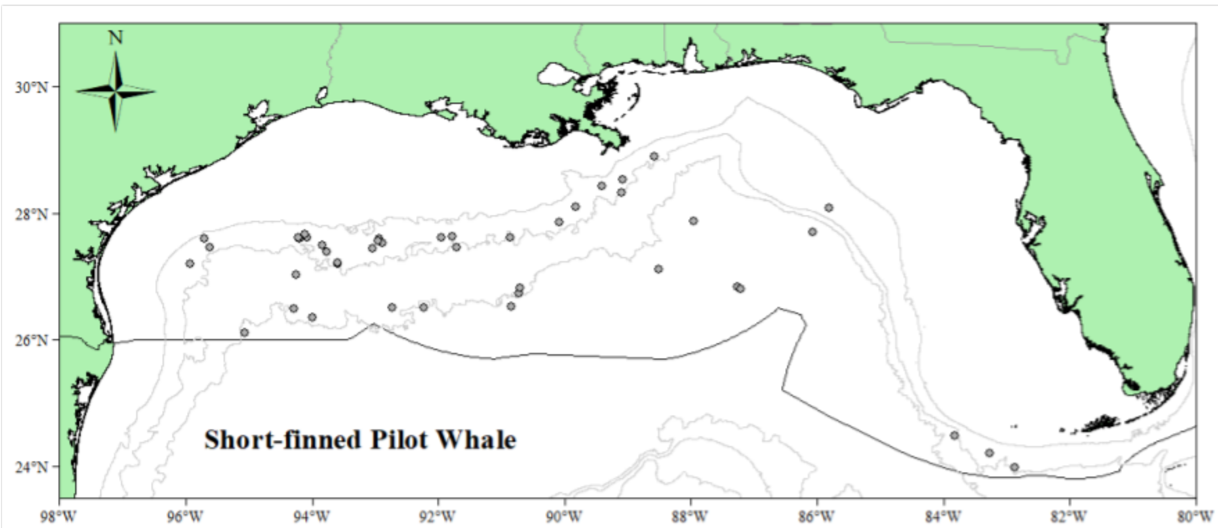


Figure 1. Distribution of short-finned pilot whale on-effort sightings from SEFSC vessel surveys during spring 1996–2001, summer 2003, spring 2004, summer 2009, summer 2017, and summer/fall 2018. Isobaths are the 200-m, 1,000-m, and 2,000-m depth contours. The darker line indicates the U.S. EEZ.

All the cetacean species found in the oceanic northern Gulf of Mexico almost certainly occur in similar habitat beyond U.S. boundaries in the southern Gulf. There are fewer cetacean sighting and stranding records in the southern Gulf due to more limited effort. Nevertheless there are records for most oceanic species in the southern Gulf (e.g., Jefferson and Schiro 1997; Ortega Ortiz 2002; Ortega-Argueta *et al.* 2005; Jefferson *et al.* 2008; Vázquez Castán *et al.* 2009; Whitt *et al.* 2011). This is therefore likely a transboundary stock with Cuba and/or Mexico. Because U.S. waters only comprise about 40% of the entire Gulf of Mexico and 35% of the oceanic (i.e., >200 m) Gulf of Mexico (Mullin and Fulling 2004), abundance and stock boundaries of oceanic species are poorly known.

Short-finned pilot whales in the northern Gulf of Mexico are managed separately from those in the western North Atlantic. Although there have been no directed studies of the degree of demographic independence between the two areas, such separation is consistent with the fact that the two areas belong to distinct marine ecoregions (Spalding *et al.* 2007, Moore and Merrick 2011). However, there is some evidence to suggest there may be connectivity between the Gulf of Mexico and Atlantic, or at least between the eastern Gulf of Mexico and the Atlantic. A May 2011 mass stranding of 23 short-finned pilot whales in the Florida Keys was considered to be composed of northern Gulf of Mexico stock whales based on the stranding location, but two tagged and released individuals from this stranding travelled directly into the Atlantic (Wells *et al.* 2013). As Wells *et al.* (2009) note, it is difficult to determine the effects of stranding and rehabilitation on post-release behavior, so it is unknown whether these movements were representative of short-finned pilot whale ranging patterns in either the Gulf of Mexico or Atlantic Ocean. There are insufficient data to determine whether the northern Gulf of Mexico stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the Gulf of Mexico and across the broader geographic area.

POPULATION SIZE

The best abundance estimate (Nest) for the northern Gulf of Mexico short-finned pilot whale is 1,321 (CV=0.43; Table 1). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020).

Earlier Abundance Estimates

Five point estimates of short-finned pilot whale abundance have been made based on data from surveys during: 2003 (June−August), 2004 (April−June), 2009 (July−August), 2017 (July−August), and 2018 (August−October). Each of these surveys had a similar design and was conducted using the same vessel or a vessel with a similar observation platform. Surveys in 2003, 2004, and 2009 employed a single survey team while the 2017 and 2018 surveys employed two survey teams. In addition, the 2017 and 2018 surveys were conducted in "passing" mode rather than “closing” mode. Passing mode eliminates the problems of fragmented tracklines associated with using closing mode in areas with high densities of animals. When using the closing mode with the two-team method, both teams must be allowed the opportunity to see a mammal group and allow it to pass behind the ship before turning to close on it, making it difficult to reacquire the group and resulting in long periods spent chasing the group, with the increased potential for off-effort sightings. For passive acoustics, in closing mode the vessel often turns before the acoustic team is able to achieve a good localization. This is especially important for deep-diving species where visual surveys are less optimal for abundance estimates. However, passing mode can result in increased numbers of unidentified sightings and may have affected group size estimation for distant groups of dolphins and small whales. Comparisons of the survey results over the years 2003 through 2009 required adjustments for these differences, including apportioning unidentified species among identified taxa to address the first issue, applying the model for detection probability on the trackline from the summer 2017 survey to the abundance estimates from the 2003, 2004, and 2009 surveys, and examining relationships between sighting distance and estimated group size (Garrison *et al.* 2020). This resulted in revised abundance estimates of: 2003, N=2,740 (CV=0.52); 2004, N=587 (CV=0.88); and 2009, N=4,788 (CV=0.74).

Recent Surveys and Abundance Estimates

An abundance estimate for short-finned pilot whales was generated from vessel surveys conducted in the northern Gulf of Mexico from the continental shelf edge (~200-m isobath) to the seaward extent of the U.S. EEZ (Garrison *et al.* 2020). One survey was conducted from 2 July to 25 August 2017 and consisted of 7,302 km of on-effort trackline, and the second survey was conducted from 11 August to 6 October 2018 and consisted of 6,473 km of on-effort trackline within the surveyed strata. Both surveys used a double-platform data-collection procedure, which allowed estimation of the detection probability on the trackline using the independent observer approach assuming point independence (Laake and Borchers 2004). Abundance was calculated using mark-recapture distance sampling implemented in package mrds (version 2.21; Laake *et al.* 2020) in the R statistical programming language. This approach accounted for the effects of covariates (e.g., sea state, glare) on detection probability within the surveyed strip. The surveys were conducted in passing mode (e.g., Schwarz *et al.* 2010) while all prior surveys in the Gulf of Mexico have been conducted in closing mode. The abundance estimate for this stock included sightings of unidentified small whales that were apportioned among identified species based on their relative density within the survey strata (Garrison *et al.* 2020). The 2017 and 2018 estimates were N=1,274 (CV=0.54) and N=1,402 (CV=0.71), respectively. The inverse variance weighted mean abundance estimate for short-finned pilot whales in oceanic waters during 2017 and 2018 was 1,321 (CV=0.43; Table 1; Garrison *et al*. 2020). Unlike previous abundance estimates, this estimate was corrected for the probability of detection on the trackline.

Table 1. Most recent abundance estimate (N_{est}) and coefficient of variation (CV) of northern Gulf of Mexico short-finned pilot whales in oceanic waters (200 m to the offshore extent of the EEZ) based on the inverse variance weighted mean from summer 2017 and summer/fall 2018 vessel surveys.

Years	Area	N_{est}	CV
2017, 2018	Gulf of Mexico	1,321	0.43

Minimum Population Estimate

The minimum population estimate (Nmin) is the lower limit of the two-tailed 60% confidence interval of the log-normal distributed abundance estimate. This is equivalent to the 20th percentile of the log-normal distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for short-finned pilot whales is 1,321 (CV=0.43). The minimum population estimate for the northern Gulf of Mexico short-finned pilot whale is 934 (Table 2).

Current Population Trend

Using revised abundance estimates for surveys conducted in 2003 (June−August), 2004 (April−June), and 2009 (July−August; see above), and the 2017 (July−August) and 2018 (August−October) estimates, pairwise comparisons of the log-transformed means were conducted between years, and significant differences were assessed at alpha=0.10. P-values were adjusted for multiple comparisons. There were no significant differences between survey years (Garrison *et al.* 2020).

However, the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 934. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.4 because the CV of the average mortality estimate is greater than 0.8 (Wade and Angliss 1997). PBR for the northern Gulf of Mexico short-finned pilot whale is 7.5 (Table 2).

Table 2. Best and minimum abundance estimates for the northern Gulf of Mexico short-finned pilot whale stock with Maximum Productivity Rate (R_{max}), Recovery Factor (F_r) and PBR.

N_{est}	CV	N_{min}	F_r	R_{max}	PBR
1,321	0.43	934	0.4	0.04	7.5

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The estimated mean annual fishery-related mortality and serious injury for this stock during 2014–2018 was 0.4 short-finned pilot whales (CV=1.00) due to interactions with the large pelagics longline fishery (see Fisheries Information sections below; Tables 3–4). Mean annual mortality and serious injury during 2014–2018 due to other human-caused actions (the *Deepwater Horizon* oil spill) was predicted to be 3.5. The minimum total mean annual human-caused mortality and serious injury for this stock during 2014–2018 was, therefore, 3.9.

Table 3. Total annual estimated fishery-related mortality and serious injury for the northern Gulf of Mexico short-finned pilot whale stock.

Years	Source	Annual Avg.	CV
2014–2018	U.S. fisheries using observer data	0.4	1.00

Fisheries Information

There are two commercial fisheries that interact, or that potentially could interact, with this stock in the Gulf of Mexico. These are the Category I Atlantic Highly Migratory Species (high seas) longline fishery and the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2014–2018 was 18, 19, 23, 13 and 20, respectively. There is very little effort within the Gulf of Mexico by the Atlantic Highly Migratory Species (high seas) longline fishery, and no takes of short-finned pilot whales within high seas waters of the Gulf of Mexico have been observed or reported thus far. Pelagic swordfish, tunas and billfish are the targets of the longline fishery operating in the northern Gulf of Mexico. The average annual serious injury and mortality in the Gulf of Mexico pelagic longline fishery for the five-year period from 2014 to 2018 is 0.44 (CV=1.00; Table 4; Garrison and Stokes 2016, 2017, 2019, 2020a, 2020b). During the first quarter of 2016, one short-finned pilot whale was observed to be seriously injured (Garrison and Stokes 2019).

During the first and second quarters of 2014–2018, observer coverage in the Gulf of Mexico pelagic longline fishery was greatly enhanced to collect more robust information on the interactions between pelagic longline vessels and spawning bluefin tuna. Therefore, the high annual observer coverage rates during 2014–2018 (Table 4) primarily reflect high coverage rates during the first and second quarters of each year. During these quarters, this elevated coverage results in an increased probability that relatively rare interactions will be detected. Species within the oceanic Gulf of Mexico are presumed to be resident year-round; however, it is unknown if the bycatch rates observed during the first and second quarters are representative of that which occurs throughout the year.

Table 4. Summary of the incidental mortality and serious injury of short-finned pilot whales by the pelagic longline commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the annual observed serious injury and mortality recorded by on-board observers, the annual estimated serious injury and mortality, the combined annual estimates of serious injury and mortality (Est. Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs), the mean of the combined annual mortality estimates, and the CV of the mean combined annual mortality estimate (CV of Mean).

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury[c]	Est. Mort.	Est. Combined Mortality	Est. CVs	Mean Combined Annual Mortality	CV of Mean
Pelagic Longline	2014	Obs. Data, Trip Logbook	0.18	0	0	0	0	0	-	0.4	1.00
	2015		0.19	0	0	0	0	0	-		
	2016		0.23	1	0	2.2	0	2.2	1		
	2017		0.13	0	0	0	0	0	-		
	2018		0.20	0	0	0	0	0	-		
Total										**0.4**	**1.00**

[a] Number of vessels in the fishery is based on vessels reporting effort to the pelagic longline logbook.
[b] Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. Mandatory logbook data were used to measure total effort for the longline fishery. These data are collected at the Southeast Fisheries Science Center (SEFSC). Observer coverage in the GOM is dominated by very high coverage rates during April–June associated with efforts to improve estimates of bluefin tuna bycatch.
[c] Proportion of sets observed.

Other Mortality

There were 93 reported strandings, including five mass strandings plus individual strandings, of short-finned pilot whales in the Gulf of Mexico during 2014–2018 (Table 5; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019). All strandings occurred in the state of Florida. During 2014 there were three mass stranding events, and there was one mass stranding each during 2016 and 2017 (Table 5). There was evidence of human interaction for four whales, including three whales with evidence of fishery interaction (longline scars) and one animal that was pushed out to sea by a member of the public without authorization. It could not be determined if there was evidence of human interaction for the remaining 89 stranded whales. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the whales that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered

(Peltier *et al*. 2012, Wells *et al*. 2015). In particular, oceanic stocks in the Gulf of Mexico are less likely to strand than nearshore coastal stocks or shelf stocks (Williams *et al.* 2011). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; https://www.fisheries.noaa.gov/national/marine-life-distress/2010-2014-cetacean-unusual-mortality-event-northern-gulf-mexico). It included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). One short-finned pilot whale stranding from 2013 in Florida was considered to be part of the UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Overall, the model estimated that this stock experienced a 3% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The mortality projected for the years 2010–2013 due to the spill has not been reported previously. Based on the population model, it was projected that 34 short-finned pilot whales died during 2010–2013 (four year annual average of 8.6) due to elevated mortality associated with oil exposure (see Appendix VI). For the 2014–2018 reporting period of this SAR, the population model estimated 17 short-finned pilot whales died due to elevated mortality associated with oil exposure. The population model used to predict short-finned pilot whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for short-finned pilot whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Table 5. Short-finned pilot whale strandings along the northern Gulf of Mexico coast, 2014–2018. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 21 May 2019. There were no strandings of short-finned pilot whales in Alabama, Louisiana, Mississippi, or Texas.

State	2014	2015	2016	2017	2018	Total
Florida	44[a]	0	35[b]	11[c]	3	93

a. This included three mass strandings: one mass stranding of 4 animals; one mass stranding of 14 animals (6 of the estimated 14 animals were examined or handled by NMFS and included in the database); and one mass stranding of 39 animals (33 of the estimated 39 animals were examined or handled by NMFS and included in the database).
b. This includes one mass stranding of 35 animals.
c. This includes one mass stranding of 10 animals.

HABITAT ISSUES

The DWH MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1,500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 6% (95%CI: 4–9) of short-finned pilot whales in the Gulf were exposed to oil, that 3% (95%CI: 1–4) of females suffered from reproductive failure, and 2% (95%CI: 1–3) of short-finned pilot whales suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 3% reduction in population size (see Other Mortality section above).

Anthropogenic sound in the world's oceans has been shown to affect marine mammals, with vessel traffic, seismic surveys, and active naval sonars being the main anthropogenic contributors to low- and mid-frequency noise in oceanic waters (e.g., Nowacek *et al.* 2015; Gomez *et al.* 2016; NMFS 2018). The long-term and population consequences of these impacts are less well-documented and likely vary by species and other factors. Impacts on marine mammal prey from sound are also possible (Carroll *et al.* 2017), but the duration and severity of any such prey effects on marine mammals are unknown.

STATUS OF STOCK

Short-finned pilot whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. Total fishery-related mortality and serious injury for this stock is less than 10% of PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of short-finned pilot whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Garrison, L.P. and L. Stokes. 2014. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2013. NOAA Tech. Memo. NMFS-SEFSC-667. 61pp.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

STATUS OF STOCK

Short-finned pilot whales are not listed as threatened or endangered under the Endangered Species Act, and the northern Gulf of Mexico stock is not considered strategic under the MMPA. Total fishery-related mortality and serious injury for this stock is less than 10% of PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of short-finned pilot whales in the northern Gulf of Mexico, relative to OSP, is unknown. There was no statistically significant trend in population size for this stock.

REFERENCES CITED

Barlow, J., S.L. Swartz, T.C. Eagle and P.R. Wade. 1995. U.S. Marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Byrd, B.L., A.A. Hohn, G.N. Lovewell, K.M. Altman, S.G. Barco, A. Friedlaender, C.A. Harms, W.A. McLellan, K.T. Moore, P.E. Rosel and V.G. Thayer. 2014. Strandings illustrate marine mammal biodiversity and human impacts off the coast of North Carolina, USA. Fish. Bull. 112:1–23.

Carroll, A.G., R. Przeslawski, A. Duncan, M. Gunning, B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish & invertebrates. Mar. Pollut. Bull. 114:9–24.

Colegrove, K.M., S. Venn-Watson, J. Litz, M.J. Kinsel, K.A. Terio, E. Fougeres, R. Ewing, D.A. Pabst, W.A. McLellan, S. Raverty, J. Saliki, S. Fire, G. Rappucci, S. Bowen-Stevens, L. Noble, A. Costidis, M. Barbieri, C. Field, S. Smith, R.H. Carmichael, C. Chevis, W. Hatchett, D. Shannon, M. Tumlin, G. Lovewell, W. McFee and T.K. Rowles. 2016. Fetal distress and *in utero* pneumonia in perinatal dolphins during the Northern Gulf of Mexico unusual mortality event. Dis. Aquat. Org. 119(1):1–16.

DWH MMIQT. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

DWH NRDAT [*Deepwater Horizon* Natural Resource Damage Assessment Trustees]. 2016. *Deepwater Horizon* oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Available from: http://www.gulfspillrestoration.noaa.gov/restoration-planning/gulf-plan

Garrison, L.P. and L. Aichinger Dias. 2020. Distribution and abundance of cetaceans in the northern Gulf of Mexico. NOAA Tech. Memo. NMFS-SEFSC-747. 40pp. Available from: https://repository.library.noaa.gov/view/noaa/25568

Garrison, L.P. and L. Stokes. 2016. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2014. NOAA Tech. Memo. NMFS-SEFSC-696. 62pp. Available from: https://repository.library.noaa.gov/view/noaa/14390

Garrison, L.P. and L. Stokes. 2017. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2015. NOAA Tech. Memo. NMFS-SEFSC-709. 61pp.

Garrison, L.P. and L. Stokes. 2019. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2019-01. 62pp.

Garrison, L.P. and L. Stokes. 2020a. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2017. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-05. 61pp.

Garrison, L.P. and L. Stokes. 2020b. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRD Contribution # PRD-2020-08. 56pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Garrison, L.P. and L. Stokes. 2014. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2013. NOAA Tech. Memo. NMFS-SEFSC-667. 61pp.

Hansen, L.J., K.D. Mullin, T.A. Jefferson and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. Pages 55–132 *in:* R.W. Davis and G.S. Fargion (Eds.). Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report. Volume II: Technical report. OCS Study MMS 96-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Jefferson, T.A. and A.J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Rev. 27(1):27–50.

Jefferson, T.A., M.A. Webber and R.L. Pitman. 2008. Marine mammals of the world. Academic Press, London. 573pp.

Laake, J.L. and D.L. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in:* Advanced distance sampling. S.T. Buckland, D.R. Andersen, K.P. Burnham, J.L. Laake and L. Thomas (Eds.). Oxford University Press, New York.

Laake, J., D. Borchers, L. Thomas, D. Miller and J. Bishop. 2020. Package 'mrds': Mark-recapture distance sampling. Version 2.2.3. Available from: http://github.com/DistanceDevelopment/mrds/

Leatherwood, S. and R.R. Reeves. 1983. The Sierra Club handbook of whales and dolphins. Sierra Club Books, San Francisco, California. 302pp.

Litz, J.A., M.A. Baran, S.R. Bowen-Stevens, R.H. Carmichael, K.M. Colegrove, L.P. Garrison, S.E. Fire, E.M. Fougeres, R. Hardy, S. Holmes, W. Jones, B.E. Mase-Guthrie, D.K. Odell, P.E. Rosel, J.T. Saliki, D.K. Shannon, S.F. Shippee, S.M. Smith, E.M. Stratton, M.C. Tumlin, H.R. Whitehead, G.A.J. Worthy and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990–2009): Providing context for the complex and long-lasting northern Gulf of Mexico cetacean UME. Dis. Aquat. Organ. 112:161–175.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. J. Cetacean Res. Manage. 8(2):203–213.

Moore, J.E. and R. Merrick (Eds.). 2011. Guidelines for assessing marine mammal stocks: Report of the GAMMS III workshop, February 15–18, 2011, La Jolla, California. Dept. of Commerce, NOAA Tech. Memo. NMFS-OPR-47.

Mullin, K.D. and G. L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4):787–807.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. Pages 111–172 *in:* R.W. Davis, W.E. Evans and B. Würsig (Eds.). Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations. Volume II: Technical report. OCS Study MMS 2000-003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

NMFS [National Marine Fisheries Service]. 2018. 2018 Revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-OPR-59. 167pp. Available from: https://repository.library.noaa.gov/view/noaa/17892

Nowacek, D.P., C.W. Clark, D. Mann, P.J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning. Front. Ecol. Environ. 13:378–386.

Ortega-Argueta, A., C.E. Perez-Sanchez, G. Gordillo-Morales, O.G. Gordillo, D.G. Perez and H. Alafita. 2005. Cetacean strandings on the southwestern coast of the Gulf of Mexico. Gulf Mex. Sci. 2005(2):179–185.

Ortega Ortiz, J. G. 2002. Multiscale analysis of cetacean distribution in the Gulf of Mexico. Ph.D. dissertation. Texas A&M University, College Station, Texas. 170pp.

Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecol. Indicators 18:278–290.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman and T.K. Rowles. 2014. Health of bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. Environ. Sci. Technol. 48(1):93–103.

Schwarz, L.K., T. Gerrodette and F.I. Archer. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. J. Cetacean Res. Manage. 11(3):253–265.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia and J. Robertson. 2007. Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. BioScience 57:573–583.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance in marine mammals. Mar. Mamm. Sci. 23(1):157–175.

Thomas, L., S.T. Buckland, E.A. Rexstad, J.L. Laake, S. Streindberg, S.L. Hedley, J.R.B. Bishop, T.A. Marques and K.P. Burnham. 2010. Distance software: Design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47(1):5–14.

Vázquez Castán, L., A. Serrano and J.A. Galindo. 2009. Estudio preliminar sobre la diversidad, distribución y abundancia de cetáceos en aguas profundas del Golfo de México. Revista UDO Agrícola 9(4):992–997.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5):e0126538.

Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3–5, 1996, Seattle, WA. NOAA Tech Memo. NMFS-OPR-12. 93pp.

Wells, R.S., J.B. Allen, G. Lovewell, J. Gorzelany, R.E. Delynn, D.A. Fauquier and N.B. Barros. 2015. Carcass-recovery rates for resident bottlenose dolphins in Sarasota Bay, Florida. Mar. Mamm. Sci. 31(1):355–368.

Wells, R.S., E.M. Fougeres, A.G. Cooper, R.O. Stevens, M.Brodsky, R. Lingenfelser, C. Dold, and D.C. Douglas 2013. Movements and dive patterns of short-finned pilot whales (*Globicephala macrorhynchus*) released from a mass stranding in the Florida Keys. Aq. Mamm. 39(1):61–72.

Wells, R.S., C.A. Manire, L. Byrd, D.R. Smith, J.G. Gannon, D. Fauquier and K.D. Mullin. 2009. Movements and dive patterns of a rehabilitated Risso's dolphin, *Grampus griseus*, in the Gulf of Mexico and Atlantic Ocean. Mar. Mamm. Sci. 25(2): 420–429.

Whitt, A.D., T.A. Jefferson, M. Blanco, D. Fertl and D. Rees. 2011. A review of marine mammal records of Cuba. Lat. Am. J. Aquat. Mamm. 9(2):65–122.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the *Deepwater Horizon*/BP incident. Conserv. Lett. 4:228–233.

Appendix I: Estimated mortality and serious injury (M/SI) of Western North Atlantic marine mammals listed by U.S. observed fisheries. Marine mammal species with zero (0) observed M/SI are not shown in this table. (unk = unknown)

Category, Fishery, Species	Years Observed	Observer Coverage	Est. SI by Year (CV)	Est. Mortality by Year (CV)	Mean Annual Mortality (CV)	PBR
CATEGORY I						
Gillnet Fisheries: Northeast Gillnet						
Harbor Porpoise	2014-2018	.18, .14, .10, .12, .11	0, 0, 0, 7, 9	128 (.27), 177 (.28), 125 (.34), 129 (.28), 92 (.52)	132 (.15)	851
Atlantic White-sided Dolphin	2013-2017	.18, .14, .10, .12, .11	0, 0, 0, 0, 0	4 (1.03), 10 (.66), 0, 0, 0	2.8 (.56)	544
Common Dolphin	2014-2018	.18, .14, .10, .12, .11	0, 0, 0, 0, 0	111 (.47), 55 (.54), 80 (.38), 133 (.28), 93 (.45)	94 (.19)	1,452
Risso's Dolphin	2013-2017	.18, .14, .10, .12, .11	0, 0, 0, 0, 0	23 (1.0), 0, 0, 0, 0	5.8 (.79)	303
Bottlenose Dolphin, Offshore	2013-2017	.18, .14, .10, .12, .11	0, 0, 0, 0, 0	26 (.95), 0, 0, 0, 8 (.92)	7.0 (.76)	561
Harbor Seal	2014-2018	.18, .14, .10, .12, .11	0, 0, 0, 0, 0	390 (.39), 474 (.17), 245 (.29), 298 (.18), 188 (.36)	319 (.13)	2,006
Gray Seal	2014-2018	.18, .14, .10, .12, .11	0, 0, 0, 0, 0	917 (.14), 1021 (.25), 498 (.33), 930 (.16), 1113 (.32)	896 (.11)	1,389
Harp Seal	2013-2017	.11, .18, .14, .10, .12	0, 0, 0, 0, 0	22 (.75), 57 (.42), 119 (.34), 85 (.50), 44 (.37)	65 (.21)	unk
Gillnet Fisheries: US Mid-Atlantic Gillnet						
Harbor Porpoise	2014-2018	.05, .06, .08, .09, .09	0, 0, 0, 0, 0	22 (1.03), 33 (1.16), 23 (.64), 9 (.95), 0	17 (.55)	851
Common Dolphin	2014-2018	.05, .06, .08, .09, .09	0, 0, 0, 11, 0	17 (.86), 30 (.55), 7 (.97), 11 (.71), 8 (.91)	17 (.34)	1,452
Harbor Seal	2014-2018	.05, .06, .08, .09, .09	0, 0, 0, 0, 0	19 (1.06), 48 (.52), 18 (.95), 3 (.18), 26 (.52)	23 (.34)	2,006
Gray Seal	2014-2018	.05, .06, .08, .09, .09	0, 0, 0, 0, 0	22 (1.09), 15 (1.04), 7 (.93), 0, 0	8.8 (.67)	1,389
Minke Whale	2014-2018	.05, .06, .08, .09, .09	0, 0, 0, 0, 0	0, 0, 1, 0, 0	0.2	14
Longline Fisheries: Pelagic Longline (Excluding NED-E)						
Risso's Dolphin	2013-2017	.09, .10, .12, .15, .12	1.9 (1.0), 7.7 (1.0), 8.4 (.71), 10.5 (.69), 0.2 (1)	0, 0, 0, 5.6 (1), 0	6.9 (.39)	303
Short-finned Pilot Whale	2013-2017	.09, .10, .12, .15, .12	124 (.32), 233 (.24), 200 (.24), 106 (.31), 133 (.29)	0, 0. 0, 5.1 (1.9), 0	160 (.12)	236
Long-finned Pilot Whale	2013-2017	.09, .10, .12, .15, .12	0, 9.6 (.43), 2.2 (.49), 1.1 (.6), 3.3 (.98)	0, 0, 0, 0, 0	3.2 (.33)	306
Common Dolphin	2014-2018	.10, .12, .15, .12, .10	0, 9.05 (1), 0, 4.92 (1), 1.44 (1)	0, 0, 0, 0, 0	3.1 (.67)	1,452

Category, Fishery, Species	Years Observed	Observer Coverage	Est. SI by Year (CV)	Est. Mortality by Year (CV)	Mean Annual Mortality (CV)	PBR
CATEGORY II						
Trawl Fisheries: Northeast Bottom Trawl						
Harp Seal	2013-2017	.15, .17, .19, .12, .16	0, 0, 0, 0, 0	2.9 (.81), 0, 0, 0, 0	0.6 (.81)	unk
Harbor Seal	2014-2018	.17, .19, .12, .12, .12	0, 0, 0, 0, 0	4 (.96), 11 (.63), 0, 0, 0	3 (.52)	2,006
Gray Seal	2014-2018	.17, .19, .12, .12, .12	0, 0, 0, 0, 0	19 (.45), 23 (.46), 0, 16 (.24), 32 (.42)	18 (.22)	1,389
Risso's Dolphin	2013-2017	.15, .17, .19, .12, .16	0, 0, 0, 0, 0	0, 4.2 (.91), 0, 17 (.88), 0	4.2 (.73)	303
Bottlenose Dolphin, Offshore	2013-2017	.15, .17, .19, .12, .16	0, 0, 0, 0, 0	0, 0, 19 (.65), 34 (.89)	10.4 (.62)	519
Long-finned Pilot Whale	2013-2017	.15, .17, .19, .12, .16	0, 6, 0, 0, 0	16 (.42), 25 (.44), 0, 29 (.58), 0	15 (.30)	306
Common Dolphin	2014-2018	.17, .19, .12, .12, .12	0, 0, 0, 0, 0	17(.53), 22(.45), 16(.46), 0, 28(.54)	17 (.26)	1,452
Atlantic White-sided Dolphin	2013-2017	.15, .17, .19, .12, .16	0, 0, 0, 0, 7.4	33 (.31), 16 (.5), 15 (.52), 28 (.46), 7.4 (.64)	21 (.21)	544
Harbor Porpoise	2014-2018	.17, .19, .12, .12, .12	0, 0, 0, 0, 0	5.5 (.86), 0, 0, 0, 0	1.1 (.86)	851
Mid-Atlantic Bottom Trawl						
Common Dolphin	2014-2018	.09, .09, .10, .14, .12	24, 0, 0, 0, 5	305 (.29), 250 (.32), 177 (.33), 380 (.23), 200 (.54)	268 (.13)	1,452
Atlantic White-sided Dolphin	2013-2017	.06, .08, .09, .10, .10	0, 0, 0, 0, 0	0, 9.7 (.94), 0, 0, 0	1.9 (.94)	544
Risso's Dolphin	2013-2017	.06, .08, .09, .10, .10	0, 0, 27, 0, 12	42 (.71), 21 (.93), 13 (.63), 39 (.56), 31 (.51)	37 (.29)	303
Bottlenose Dolphin, Offshore	2013-2017	.06, .08, .09, .10, .10	0, 0, 0, 0, 0	0, 25 (.66), 0, 7.3 (.93), 22 (.66)	11 (.42)	561
Harbor Seal	2014-2018	.09, .09, .10, .14, .12	0, 0, 0, 0, 0	10 (.95), 7, 0, 0, 6 (.94)	4.6 (0.57)	2,006
Gray Seal	2014-2018	.09, .09, .10, .14, .12	0, 0, 0, 0, 0	7 (.96), 0, 26 (.57), 26 (.40), 56 (.58)	23 (.33)	1,389
Northeast Mid-water Trawl (Including Pair Trawl)						
Long-finned Pilot Whale	2013-2017	.37, .42, .08, .27, .16	0, 0, 0, 0, 0	3, 4, 0, 3, 0	2.0 (na)	306
Harbor Seal	2014-2018	.42, .08, .27, .16, .14	0, 0, 0, 0, 0	na, na, na, 0, 0	0.8 (na)	2,006
Gray Seal	2014-2018	.42, .08, .27, .16, .14	0, 0, 0, 0, 0	0, 0, 0, 0, na	0.2 (na)	1,389

Appendix II: Summary of the confirmed anecdotal human-caused mortality and serious injury (M/SI) events involving baleen whale stocks along the Gulf of Mexico Coast, U.S. East Coast, and adjacent Canadian Maritimes, 2014–2018, with number of events attributed to entanglements or vessel collisions by year.

Stock	Mean Annual M/SI rate (PBR[1] for reference)	Entanglements Annual Rate (U.S. waters, Canadian waters, unknown first sighted in U.S., unknown first sighted in Canada)	Entanglements Confirmed Mortalities (2014, 2015, 2016, 2017, 2018)	Entanglements Injury Value Against PBR (2014, 2015, 2016, 2017, 2018)	Vessel Collisions Annual Rate (U.S. waters, Canadian waters, unknown first sighted in U.S., unknown first sighted in Canada)	Vessel Collisions Confirmed Mortalities (2014, 2015, 2016, 2017, 2018)	Vessel Collisions Injury Value Against PBR (2014, 2015, 2016, 2017, 2018)
Western North Atlantic Right Whale ***(Eubalaena glacialis)***	8.15 (0.8)	6.85 (0.2, 1.55, 3.25, 1.85)	(2, 0, 2, 4, 3)	(6, 3.5, 7.5, 2, 4.25)	1.3 (0.5, 0.8, 0, 0)	(0, 0, 1, 5, 0)	(0.52, 0, 0, 0, 0)
Gulf of Maine Humpback Whale ***(Megaptera novaeangliae)*[2]**	15.25 (22)	9.45 (2.05, 0.75, 6.3, 0.35)	(2, 1, 3, 2, 3)	(5.5, 7.5, 8, 6, 9.25)	5.8 (5.0, 0, 0.8, 0)	(0, 4, 5, 8, 7)	(0, 0, 2, 1, 2)
Western North Atlantic Fin Whale ***(Balaenoptera physalus)***	2.35 (11)	1.55 (0, 0.6, 0.95, 0)	(1, 0, 0, 1, 1)	(1.5, 1, 2.25, 0, 0)	0.8 (0.8, 0, 0, 0)	(2, 0, 0, 1, 1)	0
Nova Scotian Sei Whale ***(B. borealis)***	1.2 (6.2)	0.4 (0, 0, 0.4, 0)	(0, 0, 0, 0, 1)	(0, 0, 0, 1, 0)	0.8 (0.8, 0, 0, 0)	(3, 0, 1, 0, 0)	0
Canadian East Coast Minke Whale ***(B. acutorostrata)***	10.15 (170)	8.95 (3.15, 2.85, 2.05, 0.9)	(2, 7, 3, 12, 11)	(1.75, 2.5, 1.75, 1.5, 2.25)	1.2 (0.8, 0.4, 0, 0)	(2, 1, 0, 2, 1)	0

[1] Potential Biological Removal (PBR)
[2] Humpback SAR not updated in 2020– values reported here are published in Henry *et al* 2021

Appendix III: Fishery Descriptions

This appendix is broken into two parts: Part A describes commercial fisheries that have documented interactions with marine mammals in the Atlantic Ocean; and Part B describes commercial fisheries that have documented interactions with marine mammals in the Gulf of Mexico. A complete list of all known fisheries for both oceanic regions, the List of Fisheries, is published in the *Federal Register* annually. Each part of this appendix contains three sections: (I) data sources used to document marine mammal mortality/entanglements and commercial fishing effort trip locations, (II) links to fishery descriptions for Category I, II and some category III fisheries that have documented interactions with marine mammals and their historical level of observer coverage, and (III) historical fishery descriptions.

Part A. Description of U.S. Atlantic Commercial Fisheries

I. Data Sources

Items 1–5 describe sources of marine mammal mortality, serious injury or entanglement data; items 6–9 describe the sources of commercial fishing effort data used to summarize different components of each fishery (i.e. active number of permit holders, total effort, temporal and spatial distribution) and generate maps depicting the location and amount of fishing effort.

1. Northeast Region Fisheries Observer Program (NEFOP)

In 1989, a Fisheries Observer Program was implemented in the Northeast Region (Maine–Rhode Island) to document incidental bycatch of marine mammals in the Northeast Region Multi-species Gillnet Fishery. In 1993, sampling was expanded to observe bycatch of marine mammals in Gillnet Fisheries in the Mid-Atlantic Region (New York–North Carolina). The Northeast Fisheries Observer Program (NEFOP) has since been expanded to sample multiple gear types in both the Northeast and Mid-Atlantic Regions for documenting and monitoring interactions of marine mammals, sea turtles and finfish bycatch attributed to commercial fishing operations. At-sea observers placed onboard commercial fishing vessels collect data on fishing operations, gear and vessel characteristics, kept and discarded catch composition, bycatch of protected species, animal biology, and habitat (NMFS-NEFSC 2020).

2. Southeast Region Fishery Observer Programs

Three Fishery Observer Programs are managed by the Southeast Fisheries Science Center (SEFSC) that observe commercial fishery activity in U.S. Atlantic waters. The Pelagic Longline Observer Program (POP) administers a mandatory observer program for the U.S. Atlantic Large Pelagics Longline Fishery. The program has been in place since 1992 and randomly allocates observer effort by eleven geographic fishing areas proportional to total reported effort in each area and quarter. Observer coverage levels are mandated under the Highly Migratory Species Fisheries Management Plan (HMS FMP, 50 CFR Part 635). The second program is the Shark Gillnet Observer Program that observes the Southeastern U.S. Atlantic Shark Gillnet Fishery. The Observer Program is mandated under the HMS FMP, the Atlantic Large Whale Take Reduction Plan (ALWTRP; 50 CFR Part 229.32), and the Biological Opinion under Section 7 of the Endangered Species Act. Observers are deployed on any active fishing vessel reporting shark drift gillnet effort. In 2005, this program also began to observe sink gillnet fishing for sharks along the southeastern U.S. coast. The observed fleet includes vessels with an active directed shark permit and fish with sink gillnet gear (Carlson and Bethea 2007). The third program is the Southeastern Shrimp Otter Trawl Fishery Observer Program. Prior to 2007, this was a voluntary program administered by SEFSC in cooperation with the Gulf and South Atlantic Fisheries Foundation. The program was funding and project dependent, therefore observer coverage is not necessarily randomly allocated across the fishery. In 2007, the observer program was expanded, and it became mandatory for fishing vessels to take an observer, if selected. The program now includes more systematic sampling of the fleet based upon reported landings and effort patterns. The total level of observer coverage for this program is approximately 1% of the total fishery effort. In each Observer Program, the observers record information on the total target species catch, the number and type of interactions with protected species (including both marine mammals and sea turtles), and biological information on species caught.

3. Regional Marine Mammal Stranding Networks

The Northeast and Southeast Region Stranding Networks are components of the Marine Mammal Health and Stranding Response Program (MMHSRP). The goals of the MMHSRP are to facilitate collection and dissemination of data, assess health trends in marine mammals, correlate health with other biological and environmental parameters, and coordinate effective responses to unusual mortality events (Becker *et al.* 1994). Since 1997, the Northeast Region Marine Mammal Stranding Network has been collecting and storing data on marine mammal strandings and entanglements that occur from Maine through Virginia. The Southeast Region Strandings Program is responsible for data collection and stranding response coordination along the Atlantic coast from North Carolina to Florida, along the U.S. Gulf of Mexico coast from Florida through Texas, and in the U.S. Virgin Islands and Puerto Rico. Prior to 1997, stranding and entanglement data were maintained by the New England Aquarium and the National Museum of Natural History, Washington, D.C. Volunteer participants, acting under a letter of agreement, collect data on stranded animals that include: species; event date and location; details of the event (i.e., signs of human interaction) and determination on cause of death; animal disposition; morphology; and biological samples. Collected data are reported to the appropriate Regional Stranding Network Coordinator and are maintained in regional and national databases.

4. Marine Mammal Authorization Program

Commercial fishing vessels engaging in Category I or II fisheries are automatically registered under the Marine Mammal Authorization Program (MMAP) in order to lawfully take a non-endangered/threatened marine mammal incidental to fishing operations. These fishermen are required to carry an Authorization Certificate onboard while participating in the listed fishery, must be prepared to carry a fisheries observer if selected, and must comply with all applicable take reduction plan regulations. All vessel owners, regardless

of the category of fishery they are operating in, are required to report, within 48 hours of the incident and even if an observer has recorded the take, all incidental injuries and mortalities of marine mammals that have occurred as a result of fishing operations (NMFS-OPR 2019). Events are reported by fishermen on the Marine Mammal Mortality/Injury forms then submitted to and maintained by the NMFS Office of Protected Resources. The data reported include: captain and vessel demographics; gear type and target species; date, time and location of event; type of interaction; animal species; mortality or injury code; and number of interactions. Reporting can be done online at:
https://docs.google.com/a/noaa.gov/forms/d/e/1FAIpQLSfKe0moEVK24x1Jbly33A0MRAa2ljZgmAcCVO1hEXghtB3SYA/viewform

5. Other Data Sources for Protected Species Interactions/Entanglements/Ship Strikes

In addition to the above, data on fishery interactions/entanglements and vessel collisions with large cetaceans are reported from a variety of other sources including the New England Aquarium (Boston, Massachusetts); Provincetown Center for Coastal Studies (Provincetown, Massachusetts); U.S. Coast Guard; whale watch vessels; Canadian Department of Fisheries and Oceans (DFO); and members of the Atlantic Large Whale Disentanglement Network. These data, photographs, etc. are maintained by the Protected Species Division at the Greater Atlantic Regional Fisheries Office (GARFO), the Protected Species Branch at the Northeast Fisheries Science Center (NEFSC) and the Southeast Fisheries Science Center (SEFSC).

6. Northeast Region Vessel Trip Reports

The Northeast Region Vessel Trip Report Data Collection System is a mandatory, but self-reported, commercial fishing effort database (Wigley *et al.* 1998). The data collected include: species kept and discarded, gear types used, trip location, trip departure and landing dates, port, and vessel and gear characteristics. The reporting of these data is mandatory only for vessels fishing under a federal permit. Vessels fishing under a federal permit are required to report in the Vessel Trip Report even when they are fishing within state waters.

7. Southeast Region Fisheries Logbook System

The Fisheries Logbook System (FLS) is maintained at the SEFSC and manages data submitted from mandatory Fishing Vessel Logbook Programs under several FMPs. In 1986, a comprehensive logbook program was initiated for the Large Pelagics Longline Fishery and this reporting became mandatory in 1992. Logbook reporting has also been initiated since the 1990s for a number of other fisheries including: Reef Fish Fisheries, Snapper-Grouper Complex Fisheries, federally managed Shark Fisheries, and King and Spanish Mackerel Fisheries. In each case, vessel captains are required to submit information on the fishing location, the amount and type of fishing gear used, the total amount of fishing effort (e.g., gear sets) during a given trip, the total weight and composition of the catch, and the disposition of the catch during each unit of effort (e.g., kept, released alive, released dead). FLS data are used to estimate the total amount of fishing effort in the fishery and thus expand bycatch rate estimates from observer data to estimates of the total incidental take of marine mammal species in a given fishery. More information is available at: https://www.fisheries.noaa.gov/southeast/resources-fishing/southeast-fisheries-permits

8. Northeast Region Dealer Reported Data

The Northeast Region Dealer Database houses trip level fishery statistics on fish species landed by market category, vessel ID, permit number, port location and date of landing, and gear type utilized. The data are collected by both federally permitted seafood dealers and NMFS port agents. Data are considered to represent a census of both vessels actively fishing with a federal permit and total fish landings. It also includes vessels that fish with a state permit (excluding the state of North Carolina) that land a federally managed species. Some states submit the same trip level data to the Northeast Region, but contrary to the data submitted by federally permitted seafood dealers, the trip level data reported by individual states does not include unique vessel and permit information. Therefore, the estimated number of active permit holders reported within this appendix should be considered a minimum estimate. It is important to note that dealers were previously required to report weekly in a dealer call-in system. However, in recent years the NER regional dealer reporting system has instituted a daily electronic reporting system. Although the initial reports generated from this new system did experience some initial reporting problems, these problems have been addressed and the new daily electronic reporting system is providing better real time information to managers.

9. Northeast At-Sea Monitoring Program

At-sea monitors collect scientific, management, compliance, and other fisheries data onboard commercial fishing vessels through interviews of vessel captains and crew, observations of fishing operations, photographing catch, and measurements of selected portions of the catch and fishing gear. At-sea monitoring requirements are detailed under Amendment 16 to the NE Multispecies Fishery Management Plan with a planned implementation date of May 1st, 2010. At-sea monitoring coverage is an integral part of catch monitoring to ensure that Annual Catch Limits are not exceeded. At-sea monitors collect accurate information on catch composition and the data are used to estimate total discards by sectors (and common pool), gear type, and stock area. Coverage levels are expected around 30%.

II. Marine Mammal Protection Act's List of Fisheries

The List of Fisheries (LOF) classifies U.S. commercial fisheries into one of three Categories according to the level of incidental mortality or serious injury of marine mammals:

Category I: Frequent incidental mortality or serious injury of marine mammals
Category II: Occasional incidental mortality or serious injury of marine mammals
Category III: Remote likelihood of/no known incidental mortality or serious injury of marine mammals

The Marine Mammal Protection Act (MMPA) mandates that each fishery be classified by the level of mortality or serious injury and mortality of marine mammals that occurs incidental to each fishery as reported in the annual Marine Mammal Stock Assessment Reports for each stock. A fishery may qualify as one Category for one marine mammal stock and another Category for a different marine mammal stock. A fishery is typically categorized on the LOF according to its highest level of classification (e.g., a fishery that qualifies for Category III for one marine mammal stock and Category II for another marine mammal stock will be listed under Category II). The fisheries listed below are linked to classification based on the most current LOF published in the *Federal Register*.

III. U.S Atlantic Commercial Fisheries

Please see the List of Fisheries for more information on the following fisheries: Northeast Sink Gillnet, Northeast Anchored Float Gillnet Fishery, Northeast Drift Gillnet Fishery, Mid-Atlantic Gillnet, Mid-Atlantic Bottom Trawl, Northeast Bottom Trawl, Northeast Mid-Water Trawl Fishery (includes pair trawls), Mid-Atlantic Mid-Water Trawl Fishery (includes pair trawls), Bay of Fundy Herring Weir, Gulf of Maine Atlantic Herring Purse Seine Fishery, Northeast/Mid-Atlantic American Lobster Trap/Pot, Atlantic Mixed Species Trap/Pot Fishery, Atlantic Ocean/Caribbean/Gulf of Mexico Large Pelagics Longline, Southeast Atlantic Gillnet, Southeastern U.S. Atlantic Shark Gillnet Fishery, Atlantic Blue Crab Trap/Pot, Mid-Atlantic Haul/Beach Seine, North Carolina Inshore Gillnet Fishery, North Carolina Long Haul Seine, North Carolina Roe Mullet Stop Net, Virginia Pound Net, Mid-Atlantic Menhaden Purse Seine, Southeastern U.S. Atlantic/Gulf of Mexico Shrimp Trawl, and Southeastern U.S. Atlantic/Gulf of Mexico Stone Crab Trap/Pot Fishery.

IV. Historical Fishery Descriptions

Atlantic Foreign Mackerel

Prior to 1977, there was no documentation of marine mammal bycatch in Distant-Water Fishing (DWF) activities off the Northeast coast of the U.S. In 1977, with implementation of the Magnuson Fisheries Conservation and Management Act (MFCMA), an Observer Program was established which recorded fishery data and information on incidental bycatch of marine mammals. DWF effort in the U.S. Atlantic Exclusive Economic Zone (EEZ) under MFCMA had been directed primarily towards Atlantic mackerel and squid. From 1977 through 1982, an average mean of 120 different foreign vessels per year (range 102–161) operated within the U.S. Atlantic EEZ. In 1982, there were 112 different foreign vessels; 16%, or 18 vessels, were Japanese tuna longline vessels operating along the U.S. east coast. This was the first year that the Northeast Regional Observer Program assumed responsibility for observer coverage of the longline vessels. Between 1983 and 1991, the numbers of foreign vessels operating within the U.S. Atlantic EEZ each year were 67, 52, 62, 33, 27, 26, 14, 13, and 9, respectively. Between 1983 and 1988, the numbers of DWF Japanese longline vessels included 3, 5, 7, 6, 8, and 8, respectively. Observer coverage on DWF vessels was 25-35% during 1977-1982, and increased to 58%, 86%, 95% and 98%, respectively, in 1983–1986. One hundred percent observer coverage was maintained during 1987–1991. Foreign fishing operations for squid ceased at the end of the 1986 fishing season and for mackerel at the end of the 1991 season. Documented interactions with white-sided dolphins were reported in this fishery.

Pelagic Drift Gillnet

In 1996 and 1997, NMFS issued management regulations which prohibited the operation of this fishery in 1997. The fishery operated during 1998. Then, in January 1999 NMFS issued a Final Rule to prohibit the use of drift net gear in the North Atlantic Swordfish Fishery (50 CFR Part 630). In 1986, NMFS established a mandatory self-reported fisheries information system for Large Pelagic Fisheries. Data files are maintained at the SEFSC. The estimated total number of hauls in the Atlantic Pelagic Drift Gillnet Fishery increased from 714 in 1989 to 1,144 in 1990; thereafter, with the introduction of quotas, effort was severely reduced. The estimated number of hauls from 1991 to 1996 was 233, 243, 232, 197, 164, and 149, respectively. Fifty-nine different vessels participated in this fishery at one time or another between 1989 and 1993. In 1994 to 1998 there were 11, 12, 10, 0, and 11 vessels, respectively, in the fishery. Observer coverage, expressed as percent of sets observed, was 8% in 1989, 6% in 1990, 20% in 1991, 40% in 1992, 42% in 1993, 87% in 1994, 99% in 1995, 64% in 1996, no fishery in 1997, and 99% coverage during 1998. Observer coverage dropped during 1996 because some vessels were deemed too small or unsafe by the contractor that provided observer coverage to NMFS. Fishing effort was concentrated along the southern edge of Georges Bank and off Cape Hatteras, North Carolina. Examination of the species composition of the catch and locations of the fishery throughout the year suggest that the Drift Gillnet Fishery was stratified into two strata: (1) a southern, or winter, stratum and (2) a northern, or summer, stratum. Documented interactions with North Atlantic right whales, humpback whales, sperm whales, pilot whale spp., *Mesoplodon* spp., Risso's dolphins, common dolphins, striped dolphins and white-sided dolphins were reported in this fishery.

Atlantic Tuna Purse Seine

The Tuna Purse Seine Fishery occurring between the Gulf of Maine and Cape Hatteras, North Carolina is directed at large medium and giant bluefin tuna (BFT). Spotter aircraft are typically used to locate fish schools. The official start date, set by regulation, is 15 July of each year. Individual Vessel Quotas (IVQs) and a limited access system prevent a derby fishery situation. Catch rates for large medium, and giant tuna can be high and consequently, the season can last only a few weeks, however, over the last number of years, effort expended by this sector of the BFT fishery has diminished dramatically due to the unavailability of BFT on the fishing grounds.

The regulations allocate approximately 18.6% of the U.S. BFT quota to this sector of the fishery (five IVQs) with a tolerance limit established for large medium BFT (15% by weight of the total amount of giant BFT landed).

Limited observer data is available for the Atlantic Tuna Purse Seine Fishery. Out of 45 total trips made in 1996, 43 trips (95.6%) were observed. Forty-four sets were made on the 43 observed trips and all sets were observed. A total of 136 days were covered. No trips were observed during 1997 through 1999. Two trips (seven hauls) were observed in October 2000 in the Great South Channel Region. Four trips were observed in September 2001. No marine mammals were observed taken during these trips. Documented interactions with pilot whale spp. were reported in this fishery.

Atlantic Tuna Pelagic Pair Trawl

The Pelagic Pair Trawl Fishery operated as an experimental fishery from 1991 to 1995, with an estimated 171 hauls in 1991, 536 in 1992, 586 in 1993, 407 in 1994, and 440 in 1995. This fishery ceased operations in 1996 when NMFS rejected a petition to consider pair trawl gear as an authorized gear type in the Atlantic Tuna Fishery. The fishery operated from August to November in 1991, from June to November in 1992, from June to October in 1993 (Northridge 1996), and from mid-summer to December in 1994 and 1995. Sea sampling began in October of 1992 (Gerrior *et al.* 1994) where 48 sets (9% of the total) were sampled. In 1993, 102 hauls (17% of the total) were sampled. In 1994 and 1995, 52% (212) and 55% (238), respectively, of the sets were observed. Nineteen vessels have operated in this fishery. The fishery operated in the area between 35°N to 41°N and 69°W to 72°W. Approximately 50% of the total effort was within a one degree square at 39°N, 72°W, around Hudson Canyon, from 1991 to 1993. Examination of the 1991–1993 locations and species composition of the bycatch, showed little seasonal change for the six months of operation and did not warrant any seasonal or areal stratification of this fishery (Northridge 1996). During the 1994 and 1995 Experimental Pelagic Pair Trawl Fishing Seasons, fishing gear experiments were conducted to collect data on environmental parameters, gear behavior, and gear handling practices to evaluate factors affecting catch and bycatch (Goudey 1995, 1996), but the results were inconclusive. Documented interactions with pilot whale spp., Risso's dolphin and common dolphins were reported in this fishery.

Part B. Description of U.S. Gulf of Mexico Fisheries

I. Data Sources

Items 1 and 2 describe sources of marine mammal mortality, serious injury or entanglement data, and item 3 describes the source of commercial fishing effort data used to generate maps depicting the location and amount of fishing effort and the numbers of active permit holders. In general, commercial fisheries in the Gulf of Mexico have had little directed observer coverage and the level of fishing effort for most fisheries that may interact with marine mammals is either not reported or highly uncertain.

1. Southeast Region Fishery Observer Programs

Two fishery observer programs are managed by the SEFSC that observe commercial fishery activity in the U.S. Gulf of Mexico. The Pelagic Longline Observer Program (POP) administers a mandatory observer program for the U.S. Atlantic Large Pelagics Longline Fishery. The program has been in place since 1992, and randomly allocates observer effort by eleven geographic fishing areas proportional to total reported effort in each area and quarter. Observer coverage levels are mandated under the Highly Migratory Species FMP (HMS FMP, 50 CFR Part 635). The second is the Southeastern Shrimp Otter Trawl Fishery Observer Program. Prior to 2007, this was a voluntary program administered by SEFSC in cooperation with the Gulf and South Atlantic Fisheries Foundation. The program was funding and project dependent, therefore observer coverage is not necessarily randomly allocated across the fishery. In 2007, the observer program was expanded, and it became mandatory for fishing vessels to take an observer if selected. The program now includes more systematic sampling of the fleet based upon reported landings and effort patterns. The total level of observer coverage for this program is ~1% of the total fishery effort. In each Observer Program, the observers record information on the total target species catch, the number and type of interactions with protected species (including both marine mammals and sea turtles), and biological information on species caught.

2. Regional Marine Mammal Stranding Networks

The Southeast Regional Stranding Network is a component of the Marine Mammal Health and Stranding Response Program (MMHSRP). The goals of the MMHSRP are to facilitate collection and dissemination of data, assess health trends in marine mammals, correlate health with other biological and environmental parameters, and coordinate effective responses to unusual mortality events (Becker *et al.* 1994). The Southeast Region Strandings Program is responsible for data collection and stranding response coordination along the U.S. Gulf of Mexico coast from Florida through Texas. Prior to 1997, stranding and entanglement data were maintained by the New England Aquarium and the National Museum of Natural History. Volunteer participants, acting under a letter of agreement with NOAA Fisheries, collect data on stranded animals that include: species, event date and location, details of the event including evidence of human interactions, determinations of the cause of death, animal disposition, morphology, and biological samples. Collected data are reported to the appropriate Regional Stranding Network Coordinator and are maintained in regional and national databases.

3. Southeast Region Fisheries Logbook System (FLS)

The FLS is maintained at the SEFSC and manages data submitted from mandatory fishing vessel logbook programs under several FMPs. In 1986, a comprehensive logbook program was initiated for the Large Pelagics Longline Fisheries, and this reporting became mandatory in 1992. Logbook reporting has also been initiated since the early 1990s for a number of other fisheries including: Reef Fish Fisheries, Snapper-Grouper Complex Fisheries, federally managed Shark Fisheries, and King and Spanish Mackerel Fisheries. In each case, vessel captains are required to submit information on the fishing location, the amount and type of fishing gear used, the total amount of fishing effort (e.g., gear sets) during a given trip, the total weight and composition of the catch, and the disposition of the catch during each unit of effort (e.g., kept, released alive, released dead). FLS data are used to estimate the total amount of fishing effort in the fishery and thus expand bycatch rate estimates from observer data to estimate the total incidental take of marine mammal species in a given fishery.

4. Marine Mammal Authorization Program

Commercial fishing vessels engaging in Category I or II fisheries are automatically registered under the Marine Mammal Authorization Program (MMAP) in order to lawfully take a non-endangered/threatened marine mammal incidental to fishing operations. These fishermen are required to carry an Authorization Certificate onboard while participating in the listed fishery, must be prepared to carry a fisheries observer if selected, and must comply with all applicable take reduction plan regulations. All vessel owners, regardless of the category of fishery they are operating in, are required to report within 48 hours of the incident, even if an observer has recorded the take, all incidental injuries and mortalities of marine mammals that have occurred as a result of fishing operations (NMFS-OPR 2019). Events are reported by fishermen on the Marine Mammal Mortality/Injury forms then submitted to and maintained by the NMFS Office of Protected Resources. The data reported include: captain and vessel demographics; gear type and target species; date, time and location of event; type of interaction; animal species; mortality or injury code; and number of interactions. Reporting can be done online at: https://docs.google.com/a/noaa.gov/forms/d/e/1FAIpQLSfKe0moEVK24x1Jbly33A0MRAa2ljZgmAcCVO1hEXghtB3SYA/viewform

II. Gulf of Mexico Commercial Fisheries

Please see the List of Fisheries for more information on the following fisheries: Spiny Lobster Trap/Pot Fishery, Southeastern U.S. Atlantic/Gulf of Mexico Stone Crab Trap/Pot Fishery, Gulf of Mexico Menhaden Purse Seine Fishery, Gulf of Mexico Gillnet Fishery.

References Cited

Becker, P.R., D. Wilkinson and T. Lillestolen. 1994. Marine Mammal Health and Stranding Response Program: Program Development Plan. NOAA Tech. Memo. NMFS-OPR-94-2. 35pp.

Carlson, J.K. and D.M. Bethea. 2007. Catch and bycatch in the shark gillnet fishery: 2005-2006. NOAA Tech. Memo. NMFS-SEFSC-552. 26pp.

Gerrior, P., A.S. Williams and D.J. Christensen. 1994. Observations of the 1992 U.S. pelagic pair trawl fishery in the Northwest Atlantic. Mar. Fish. Rev. 56(3): 24–27.

Goudey, C.A. 1995. The 1994 experimental pair trawl fishery for tuna in the Northwest Atlantic. Massachusetts Institute of Technology, Sea Grant, Cambridge, MA. MITSG 95-6. 10pp.

Goudey, C.A. 1996. The 1995 experimental pair trawl fishery for tuna in the Northwest Atlantic. Massachusetts Institute of Technology, Sea Grant, Cambridge, MA. MITSG 96-17. 13pp.

NMFS-NEFSC [National Marine Fisheries Service - Northeast Fisheries Science Center]. 2020. Northeast Fisheries Observer Program. Retrieved May 13, 2020. Available from: https://www.fisheries.noaa.gov/new-england-mid-atlantic/fisheries-observers/northeast-fisheries-observer-program

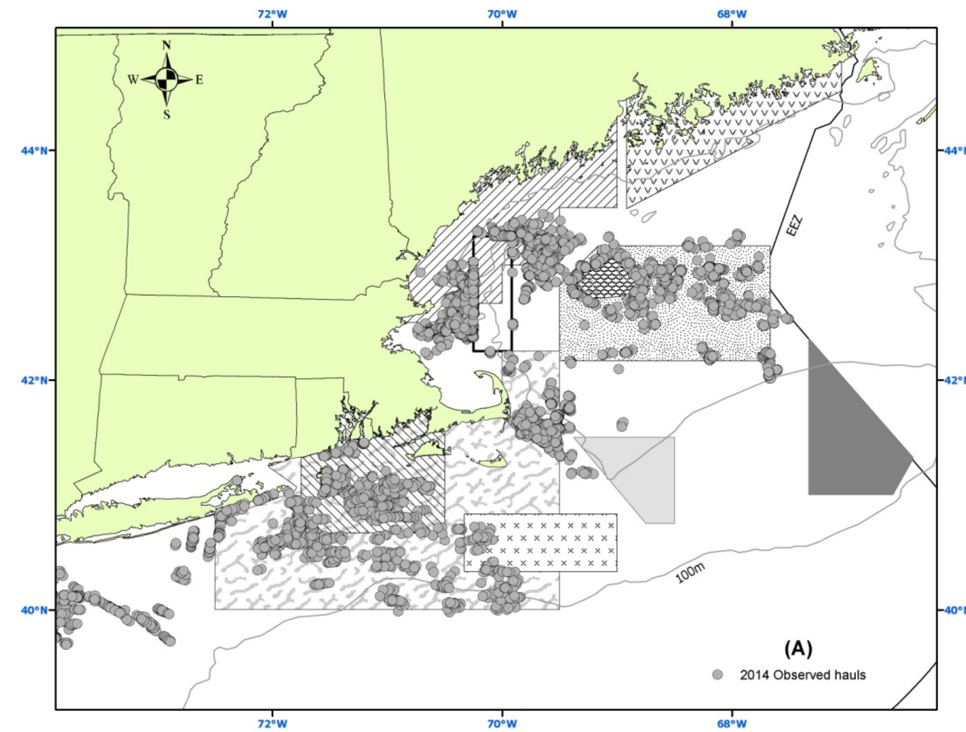
NMFS-OPR [National Marine Fisheries Service - Office of Protected Resources]. 2019. Marine Mammal Authorization Program. Retrieved February 14, 2019. Available from: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-authorization-program

Northridge, S. 1996. Estimation of cetacean mortality in the U.S. Atlantic swordfish and tuna driftnet and pair trawl fisheries. NMFS 40ENNF500160. 21pp.

Wigley, S., M. Terceiro, A. Delong and K. Sosebee. 1998. Proration of 1994-96 USA commercial landings of Atlantic cod, haddock and yellowtail flounder to unit stock areas. Northeast Fish. Sci. Cent. Ref. Doc. 98-02.

Appendix III: Fishery Descriptions - List of Figures

Figure 1. 2014 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan Year-round Closures:

Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closure

Harbor Porpoise Take Reduction Plan Management Areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cape Cod South Closure | Cashes Ledge Closure

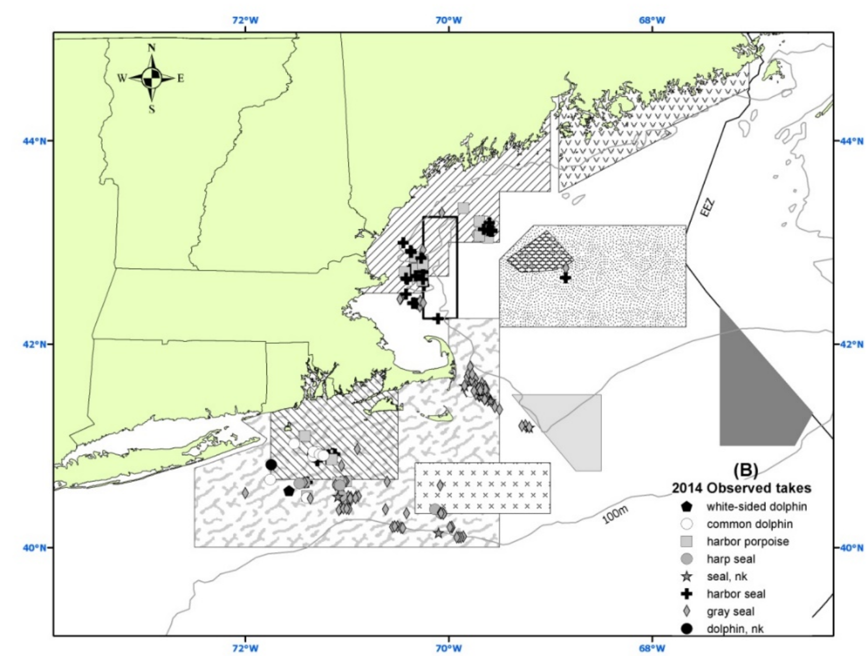
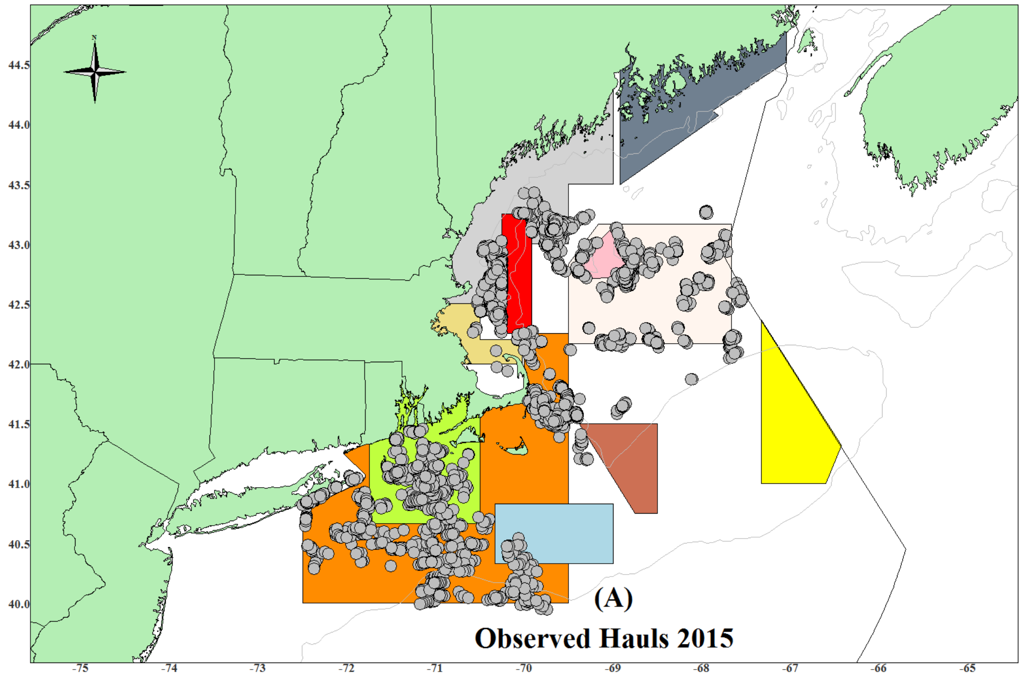


Figure 2. 2015 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan Year-round Closures:

Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Harbor Porpoise Take Reduction Plan Management Areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cod South Closure | Cashes Ledge Closed Area

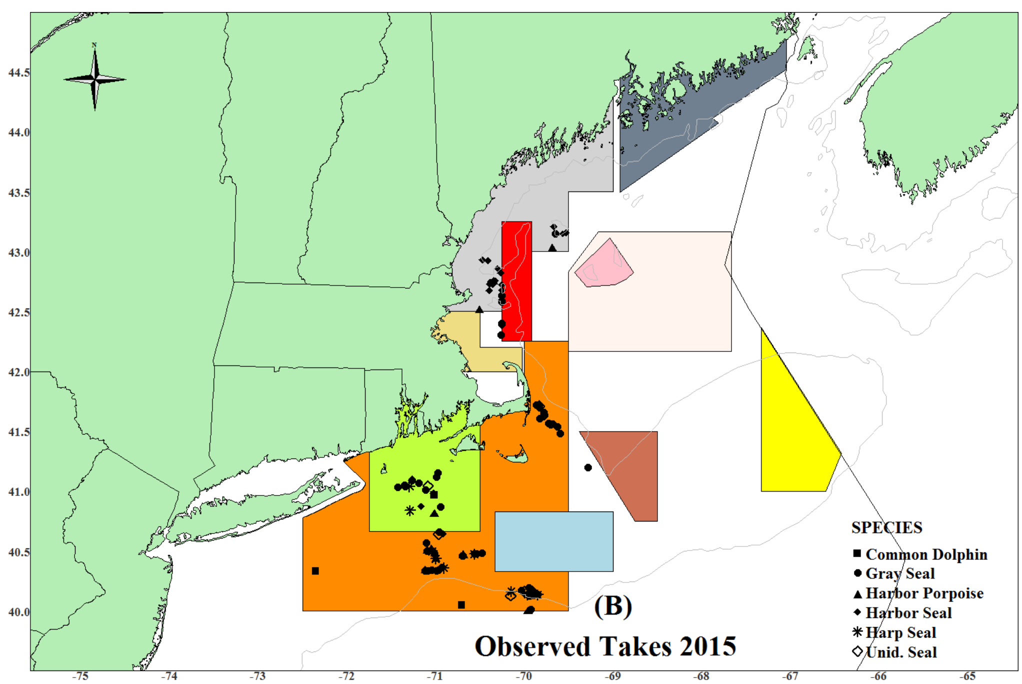


Figure 3. 2016 Northeast sink gillnet observed hauls (A) and observed takes (B).

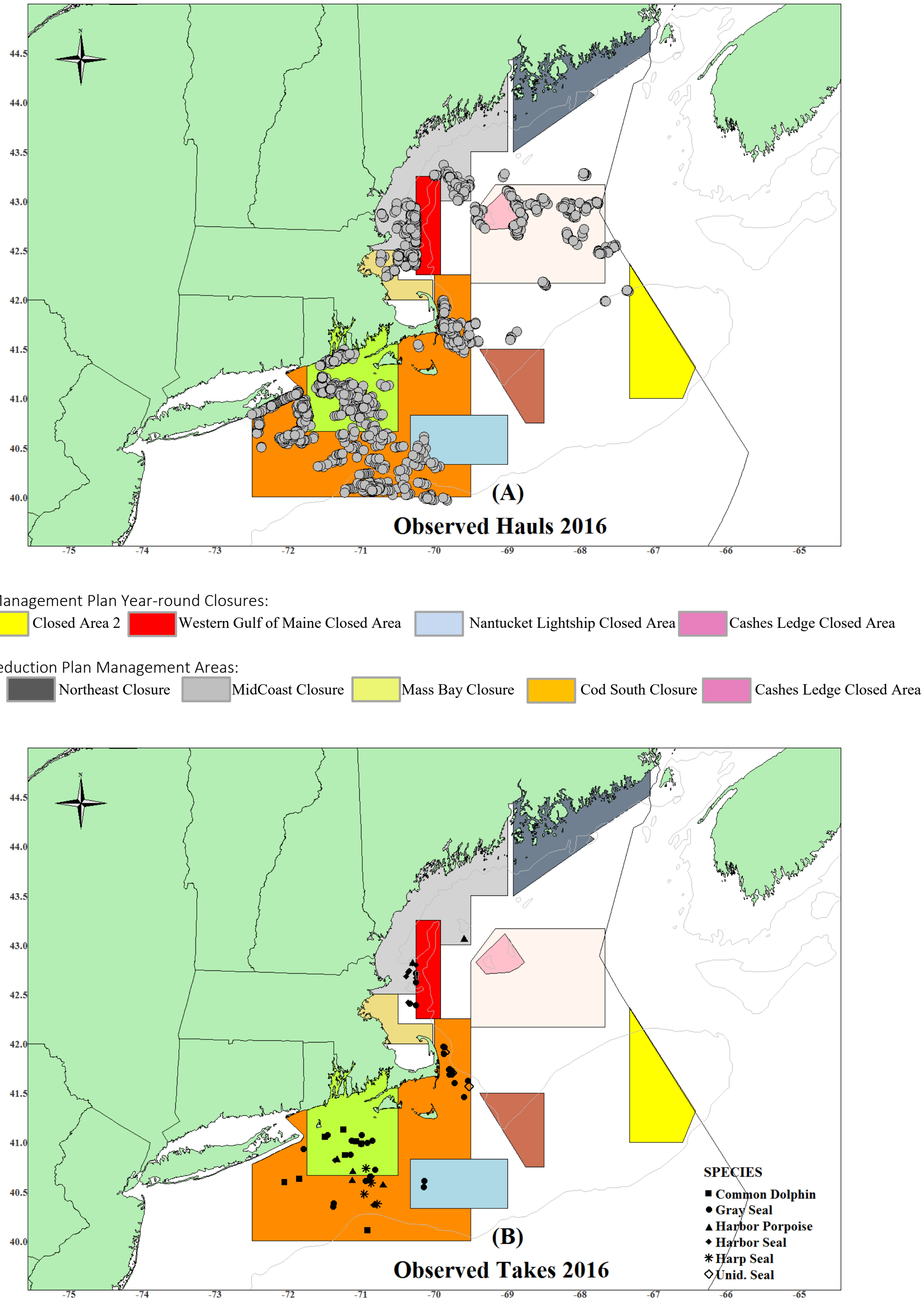
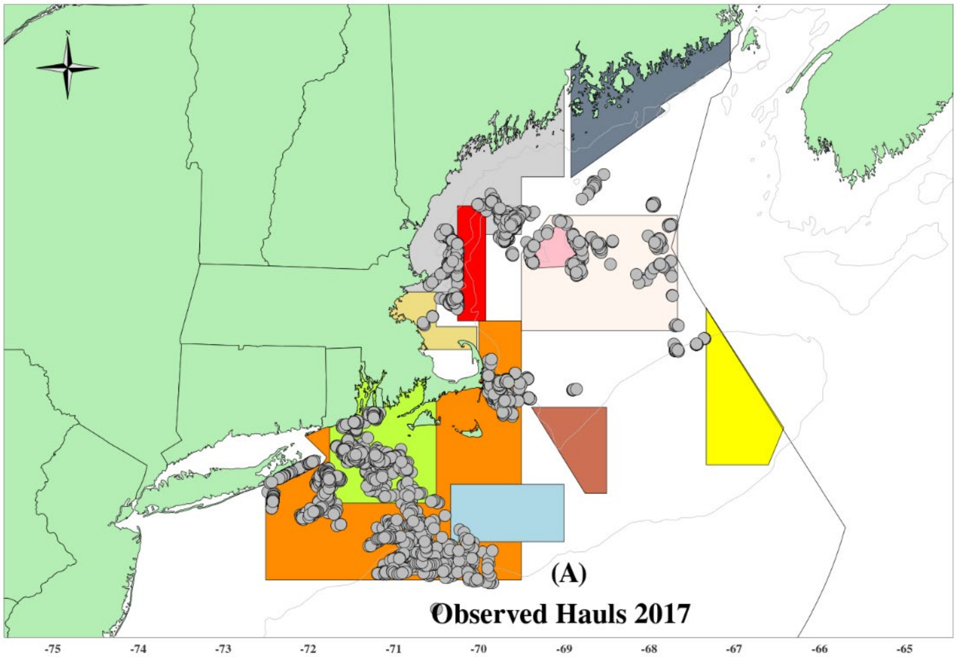


Figure 4. 2017 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan Year-round Closures:

Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Harbor Porpoise Take Reduction Plan Management Areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cod South Closure | Cashes Ledge Closed Area

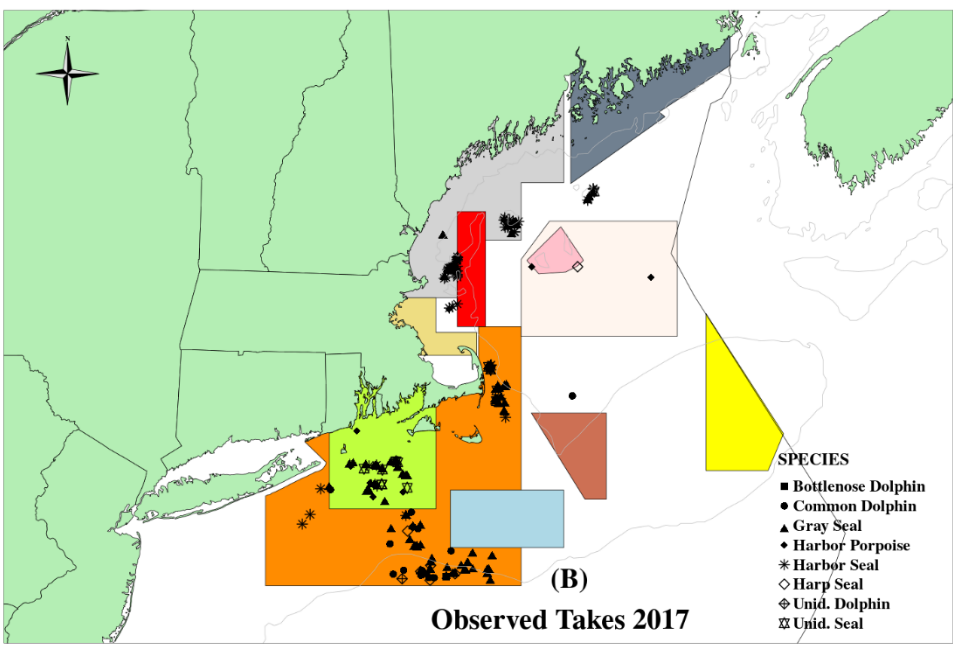
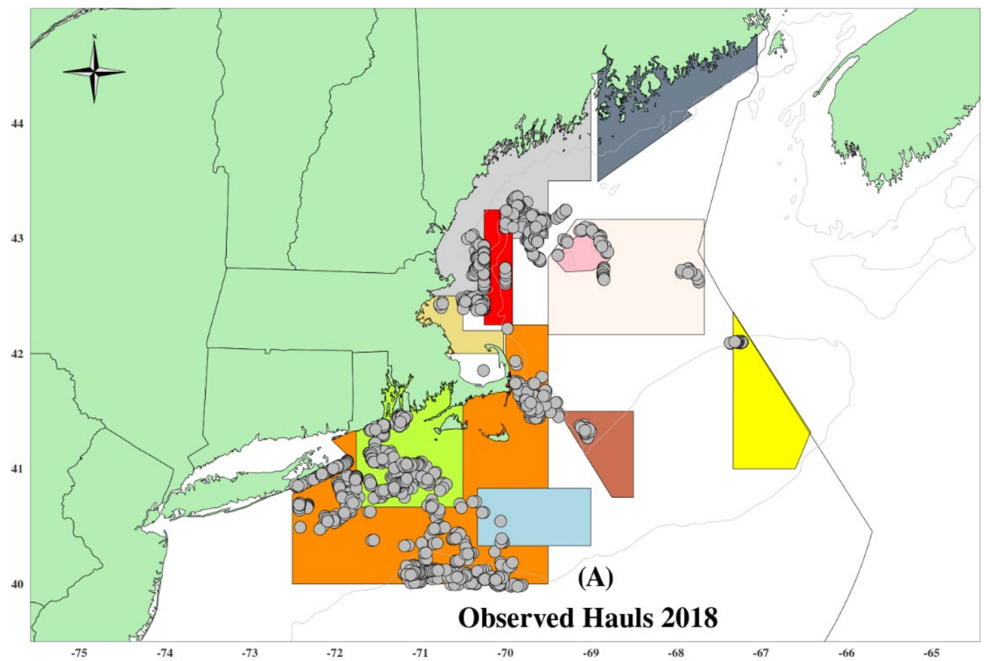


Figure 5. 2018 Northeast sink gillnet observed hauls (A) and observed takes (B).



Multispecies Fisheries Management Plan Year-round Closures:

Closed Area 1 | Closed Area 2 | Western Gulf of Maine Closed Area | Nantucket Lightship Closed Area | Cashes Ledge Closed Area

Harbor Porpoise Take Reduction Plan Management Areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cod South Closure | Cashes Ledge Closed Area

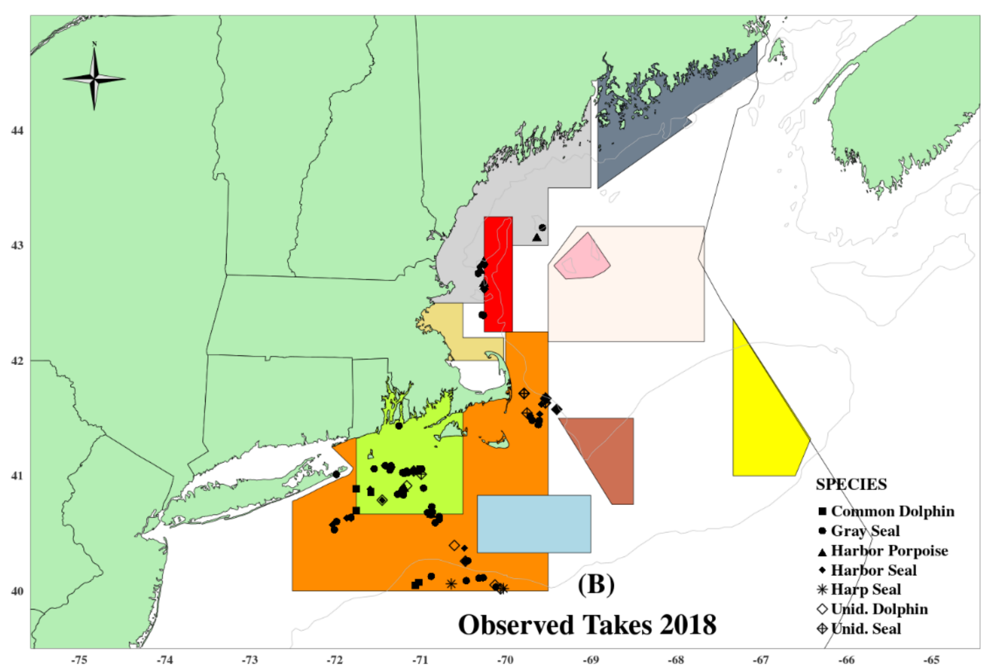
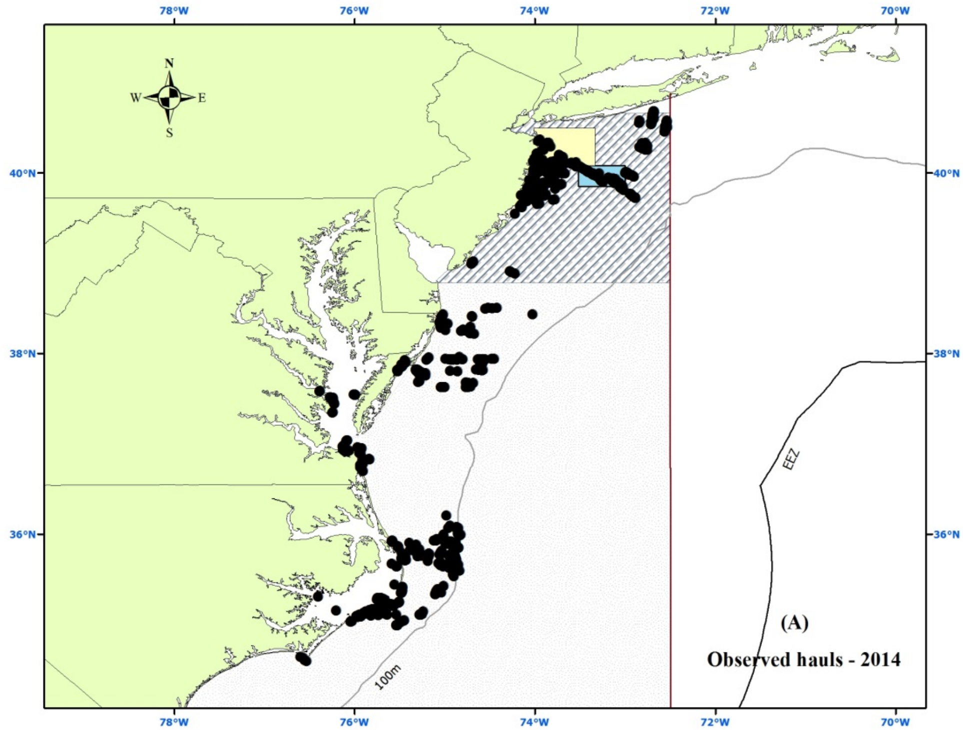


Figure 6. 2014 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor Porpoise Take Reduction Plan Management Areas:

Southern mid-Atlantic waters   New Jersey Mudhole   waters off New Jersey

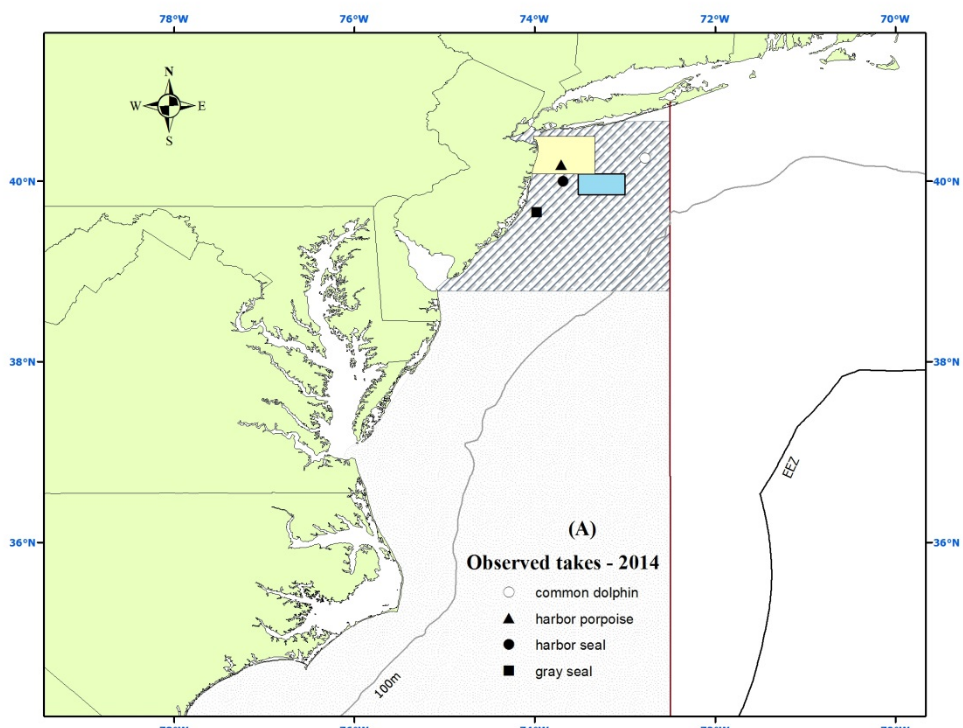
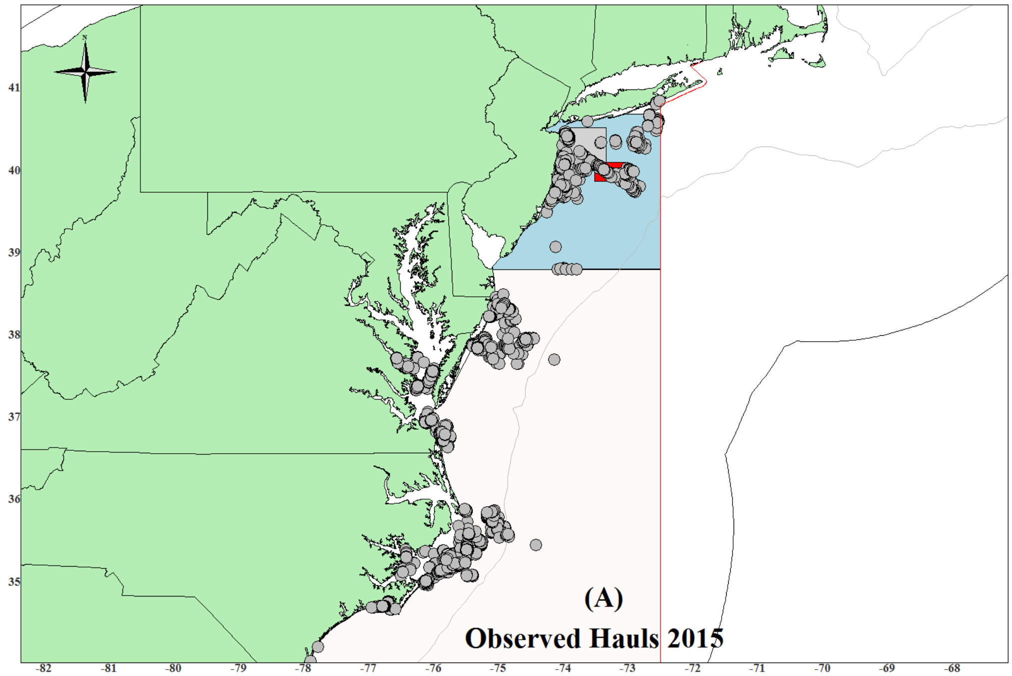


Figure 7. 2015 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor Porpoise Take Reduction Plan Management Areas:

Southern Mid-Atlantic Waters | New Jersey Mudhole | Mudhole South | Waters off New Jersey

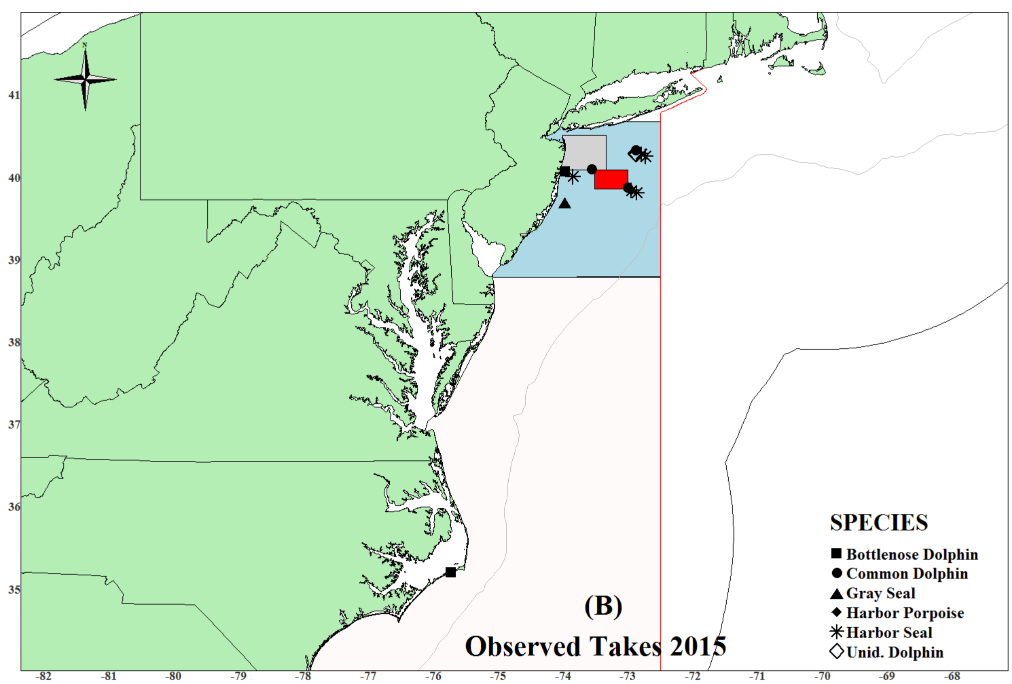
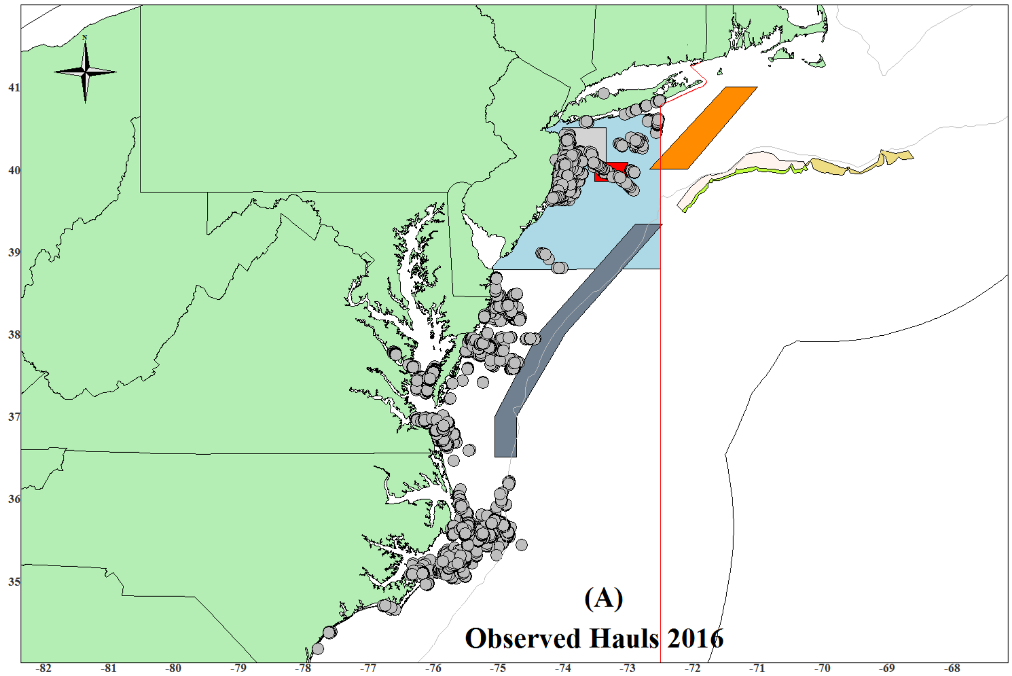


Figure 8. 2016 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor Porpoise Take Reduction Plan Management Areas:

Southern Mid-Atlantic Waters | New Jersey Mudhole | Mudhole South | Waters off New Jersey

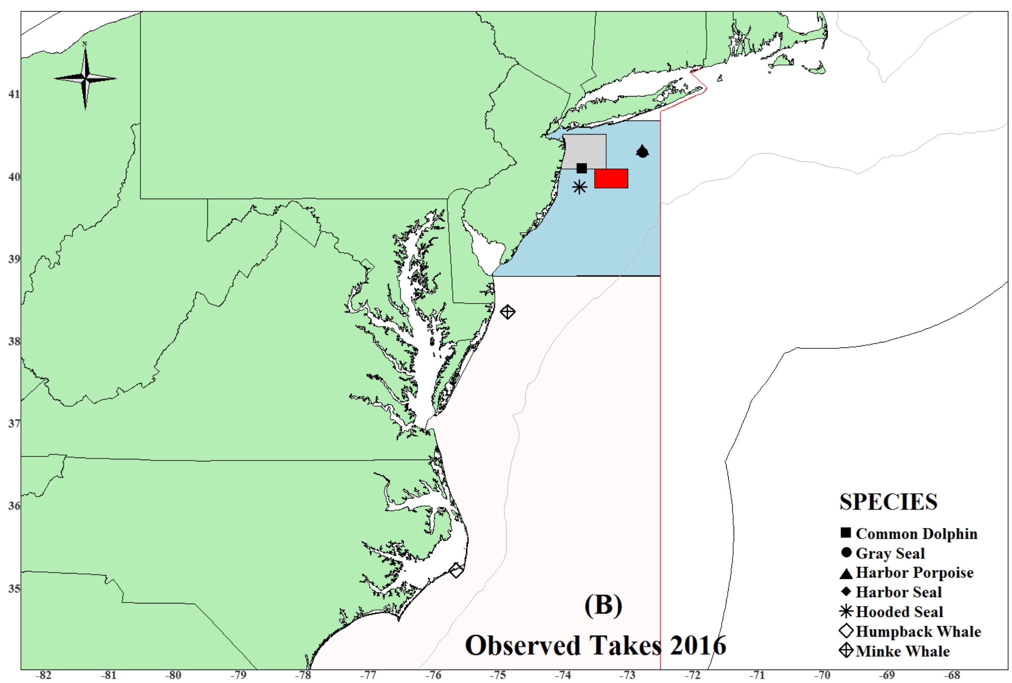
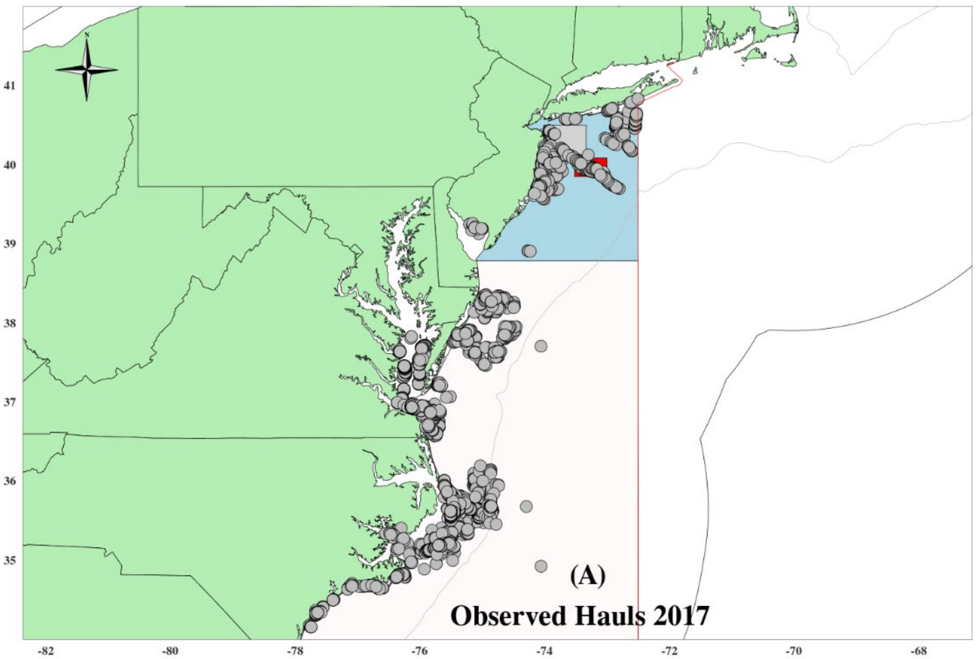


Figure 9. 2017 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor Porpoise Take Reduction Plan Management Areas:

Southern Mid-Atlantic Waters | New Jersey Mudhole | Mudhole South | Waters off New Jersey

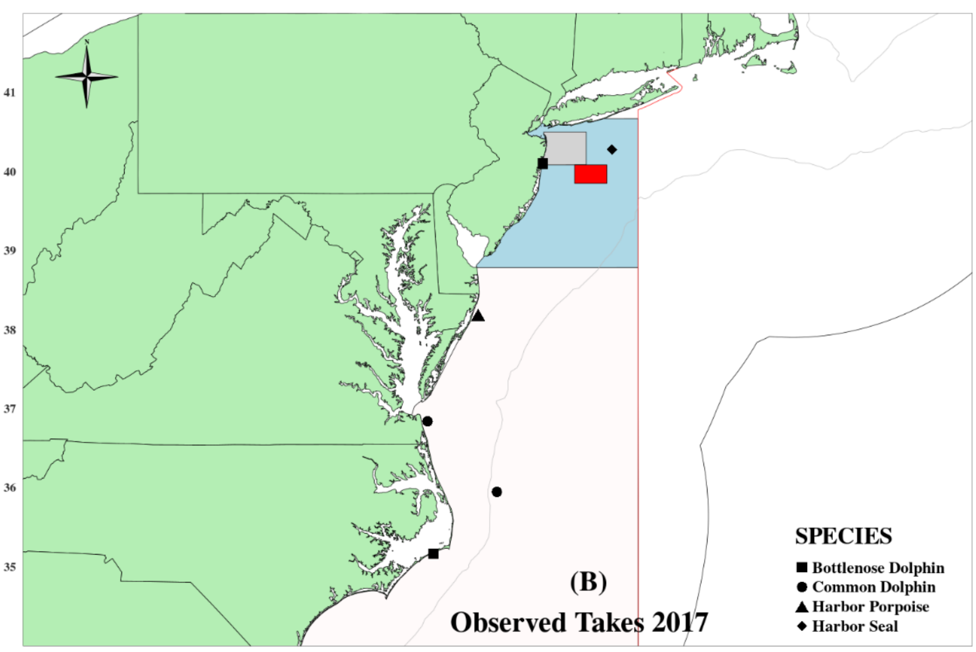
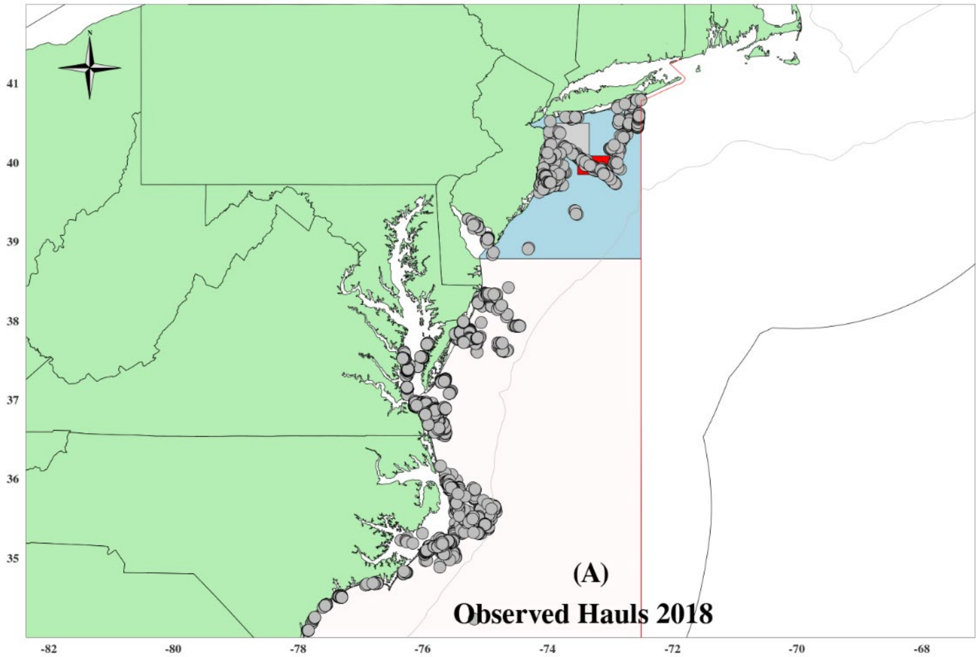


Figure 10. 2018 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor Porpoise Take Reduction Plan Management Areas:
Southern Mid-Atlantic Waters | New Jersey Mudhole | Mudhole South | Waters off New Jersey

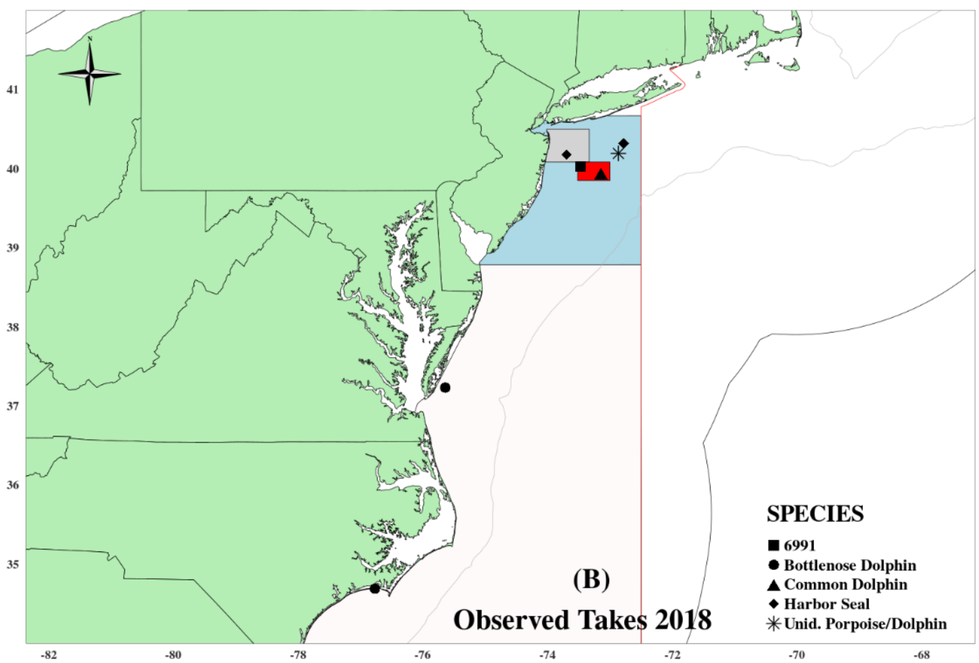


Figure 11. 2014 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

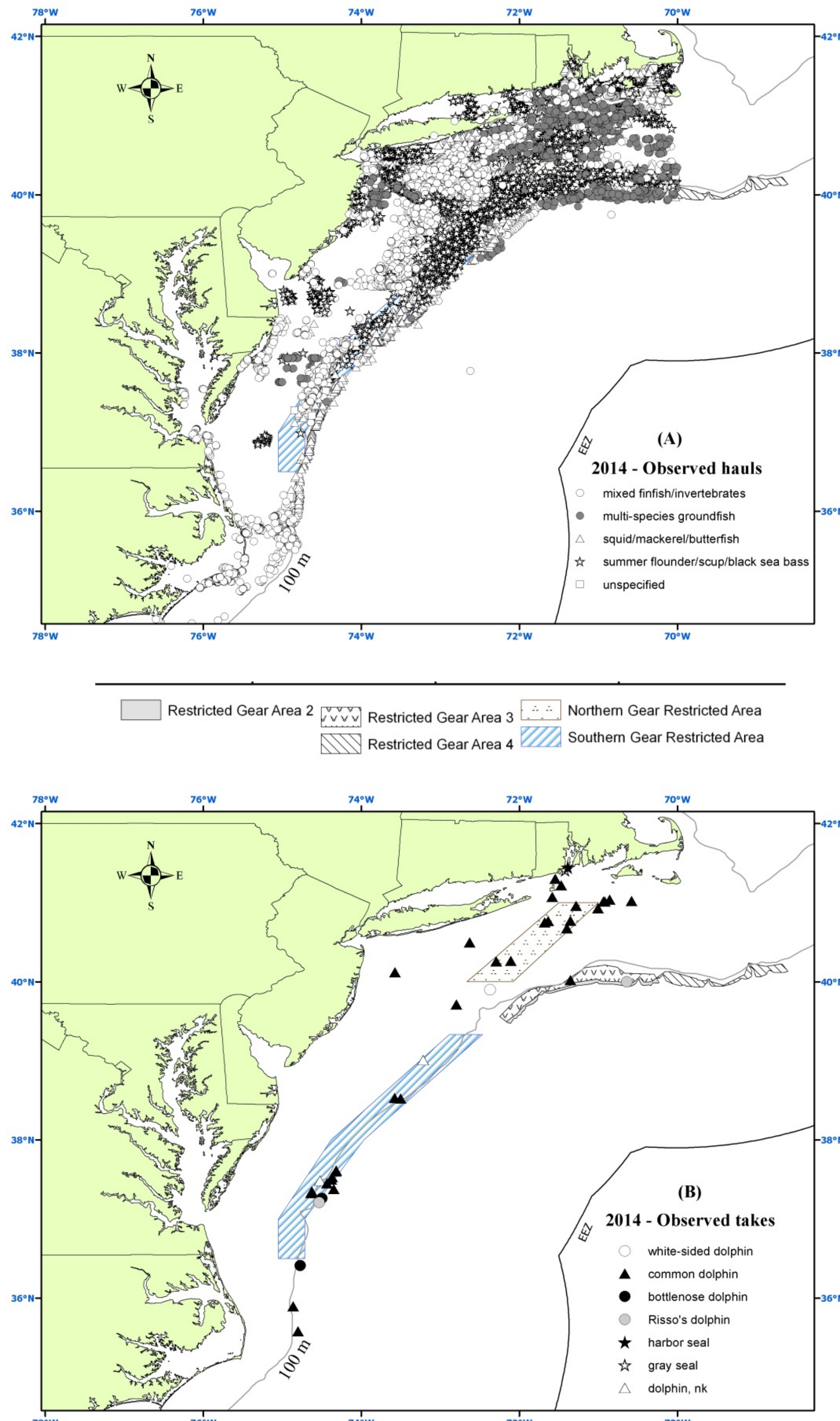


Figure 12. 2015 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

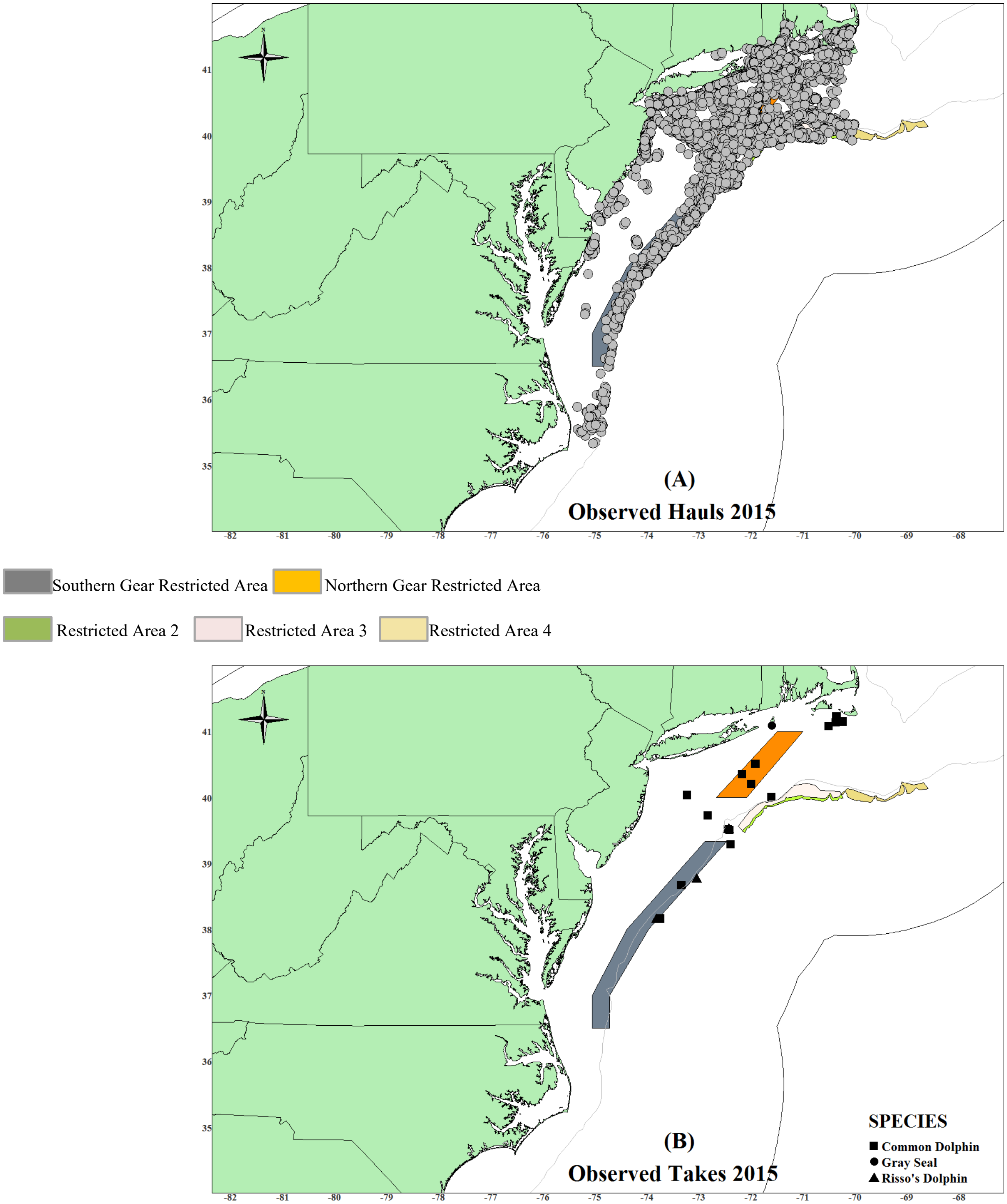
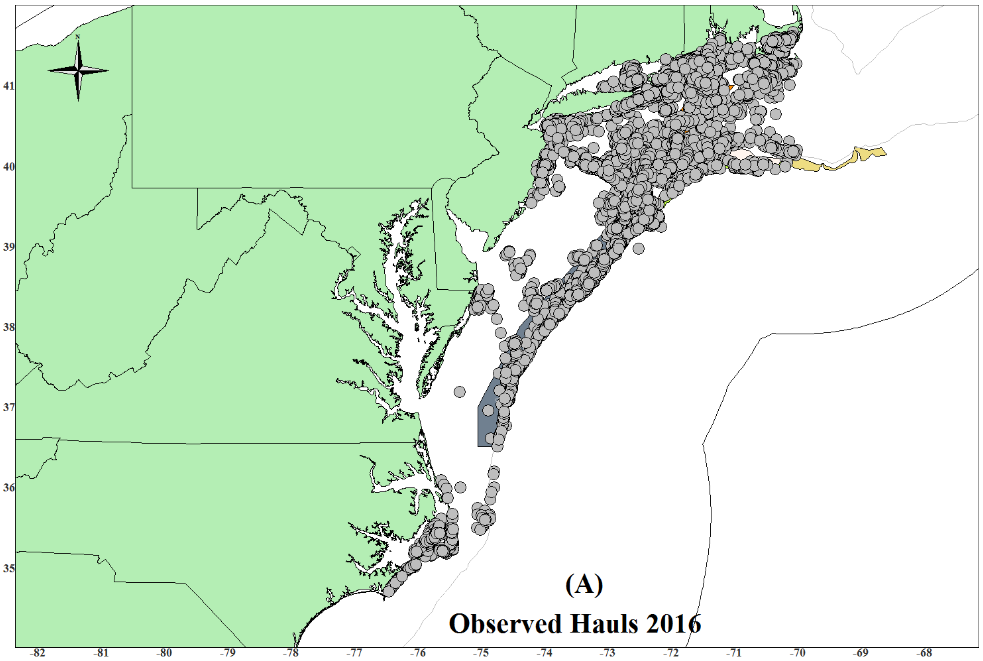


Figure 13. 2016 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).



Southern Gear Restricted Area Northern Gear Restricted Area

Restricted Area 2 Restricted Area 3 Restricted Area 4

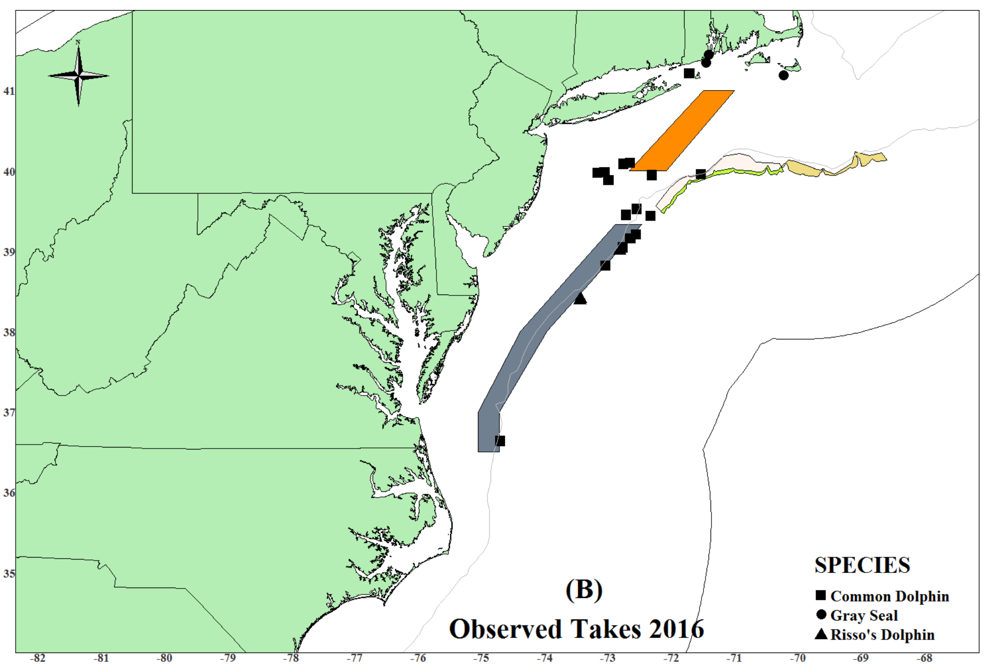


Figure 14. 2017 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

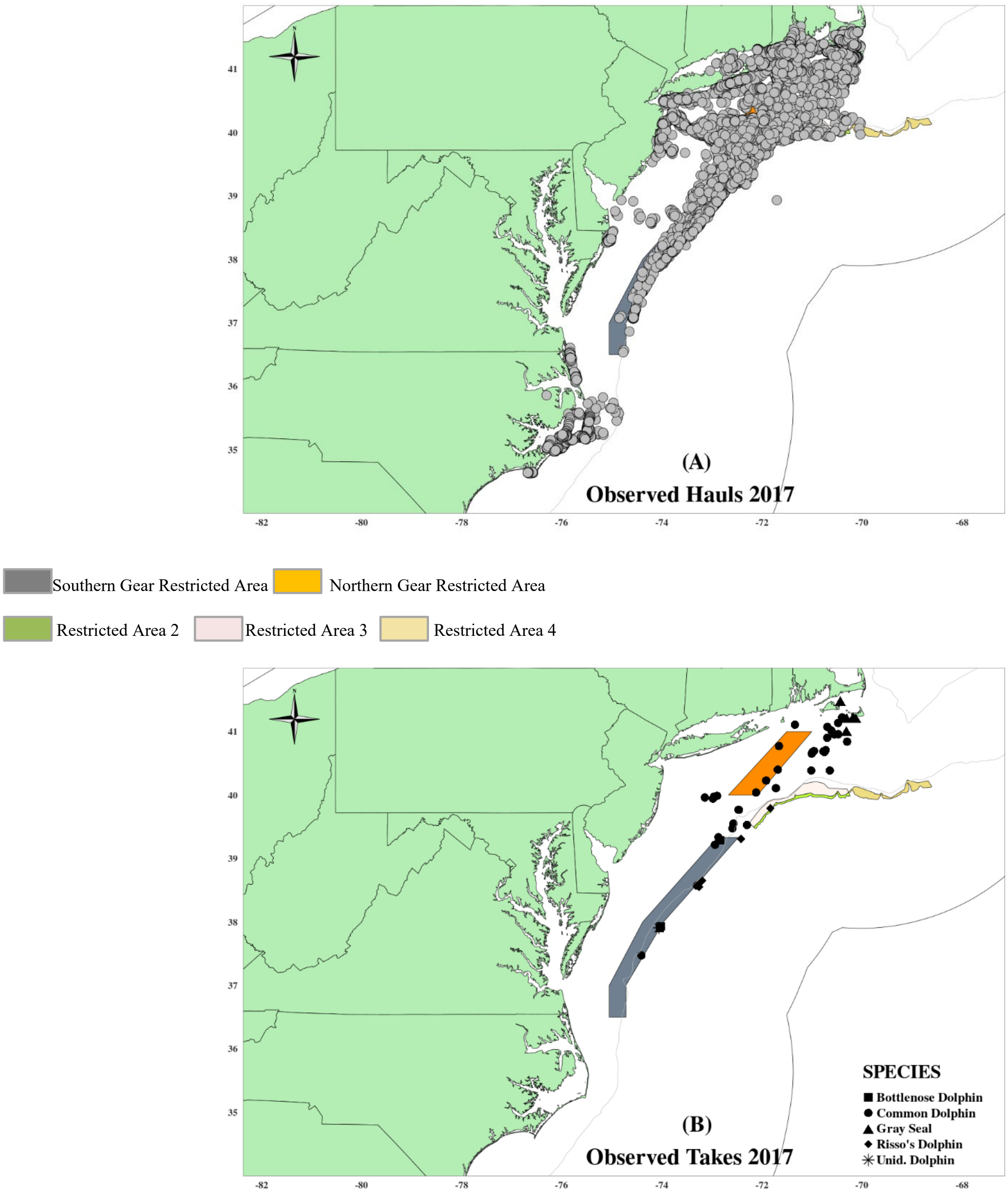


Figure 15. 2018 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

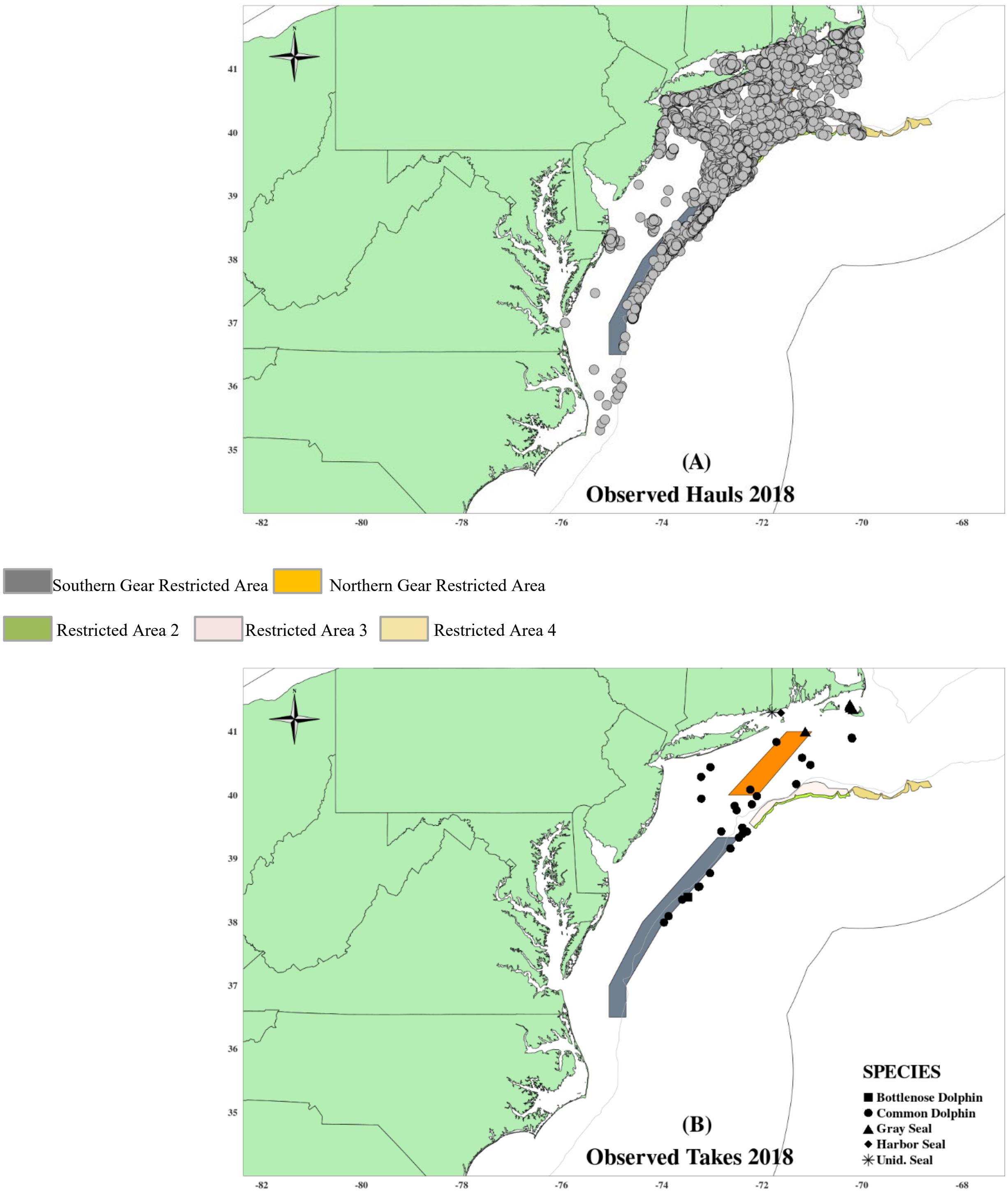


Figure 16. 2014 Northeast bottom trawl observed tows (A) and observed takes (B).

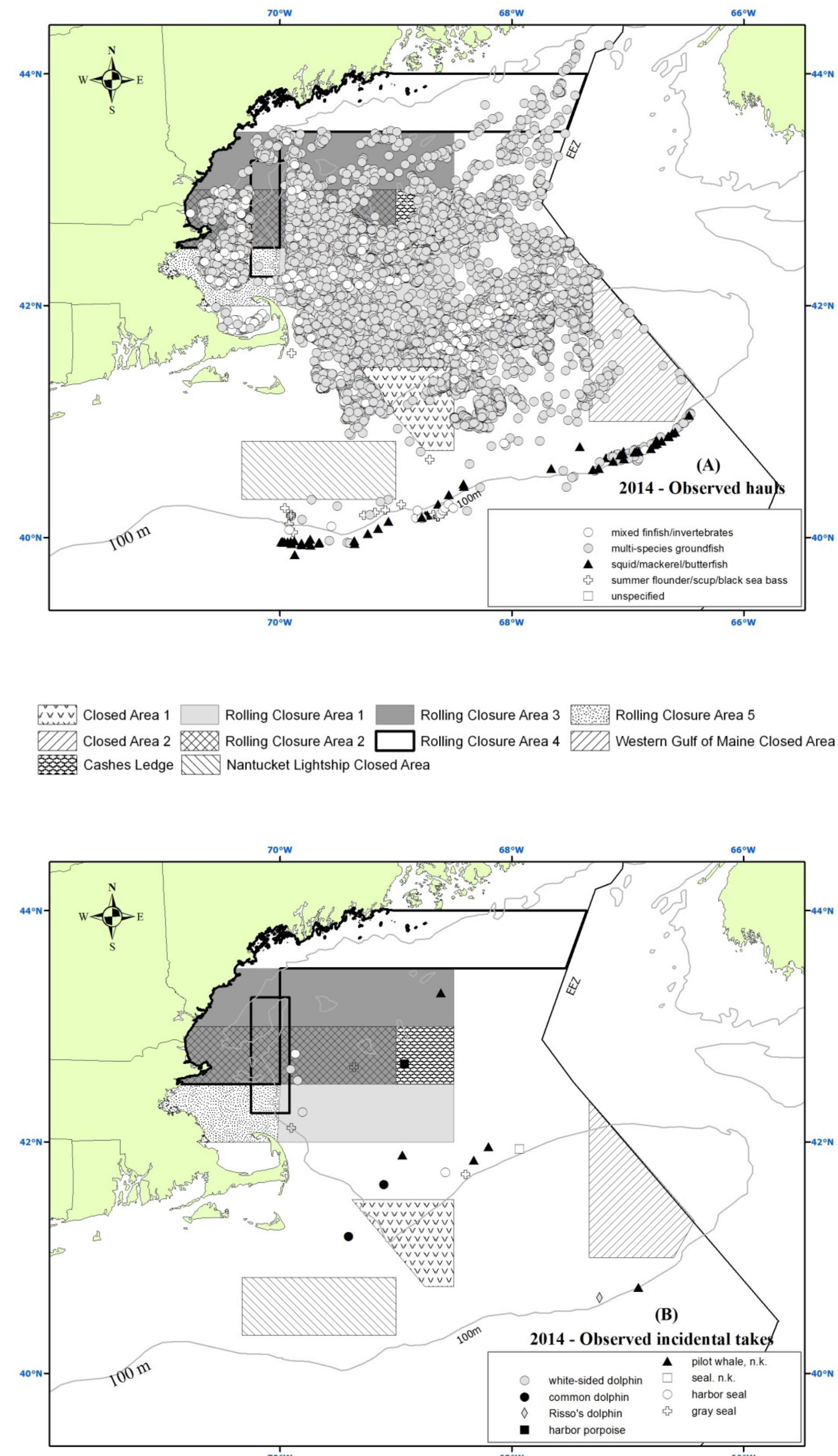
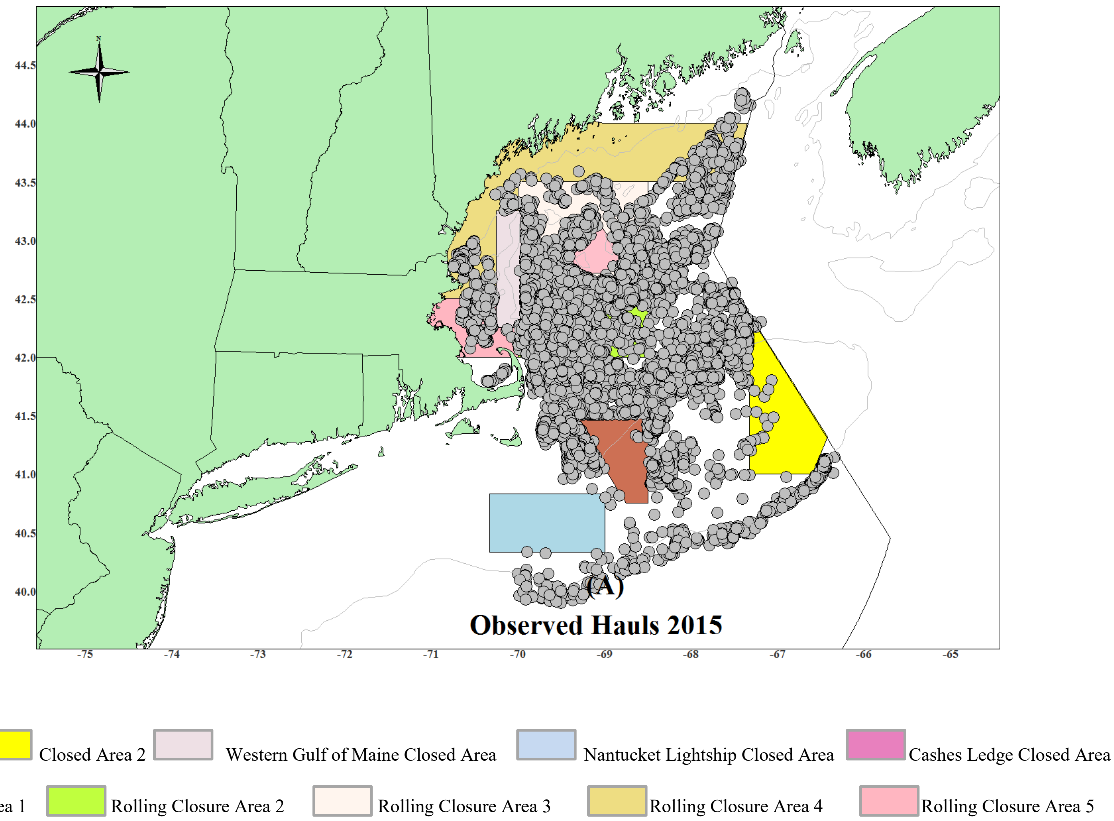


Figure 17. 2015 Northeast bottom trawl observed tows (A) and observed takes (B).



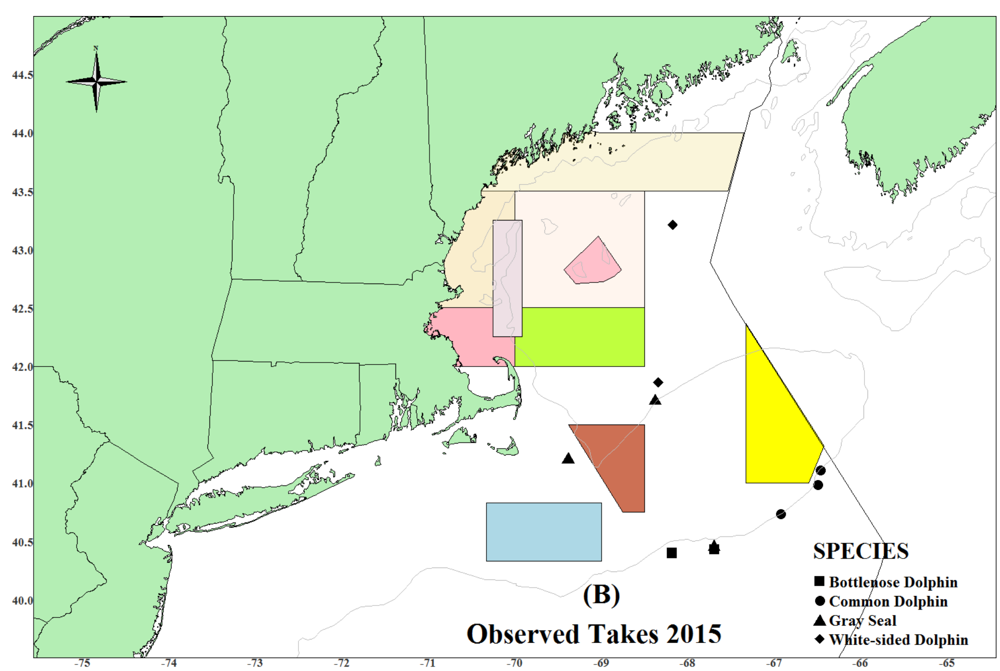
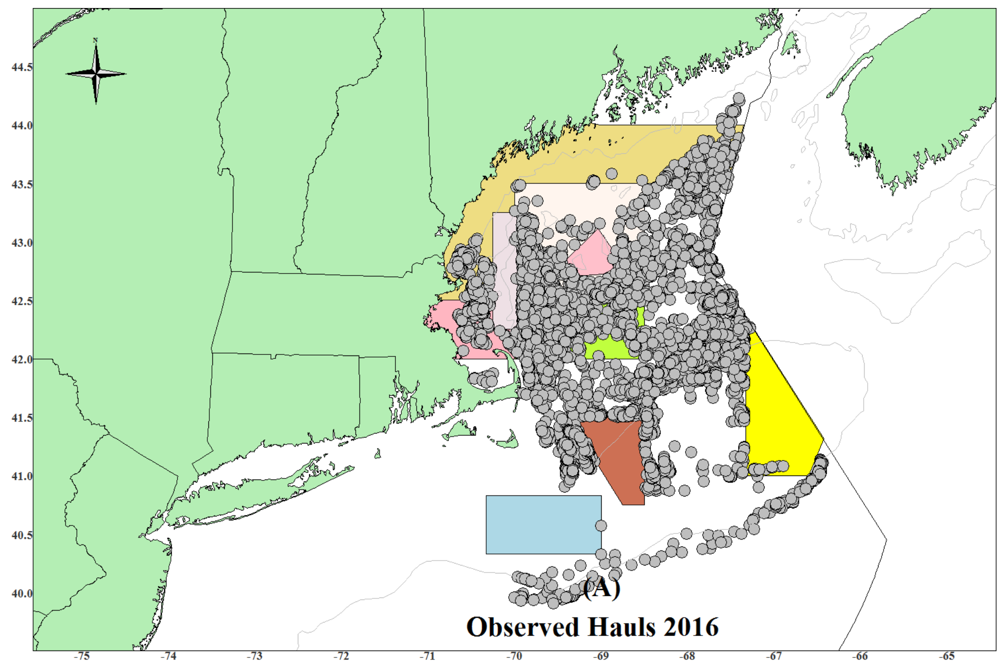
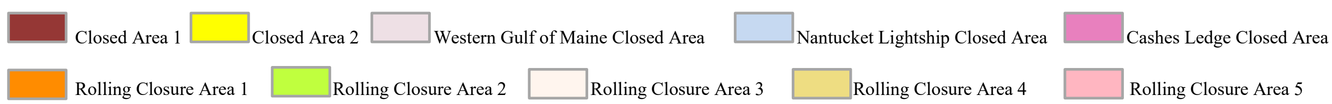


Figure 18. 2016 Northeast bottom trawl observed tows (A) and observed takes (B).





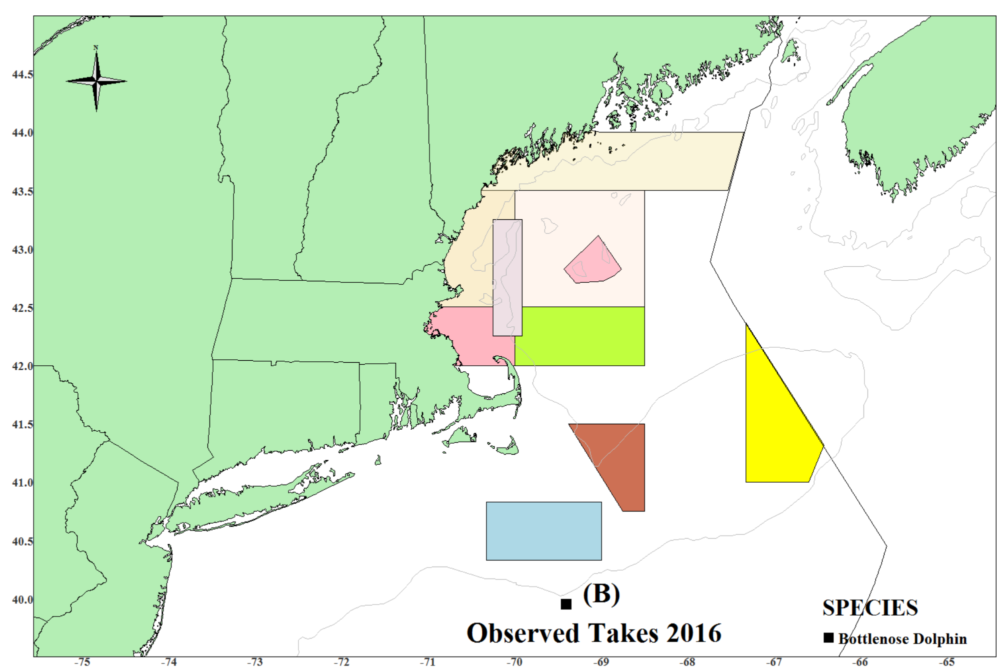


Figure 19. 2017 Northeast bottom trawl observed tows (A) and observed takes (B).

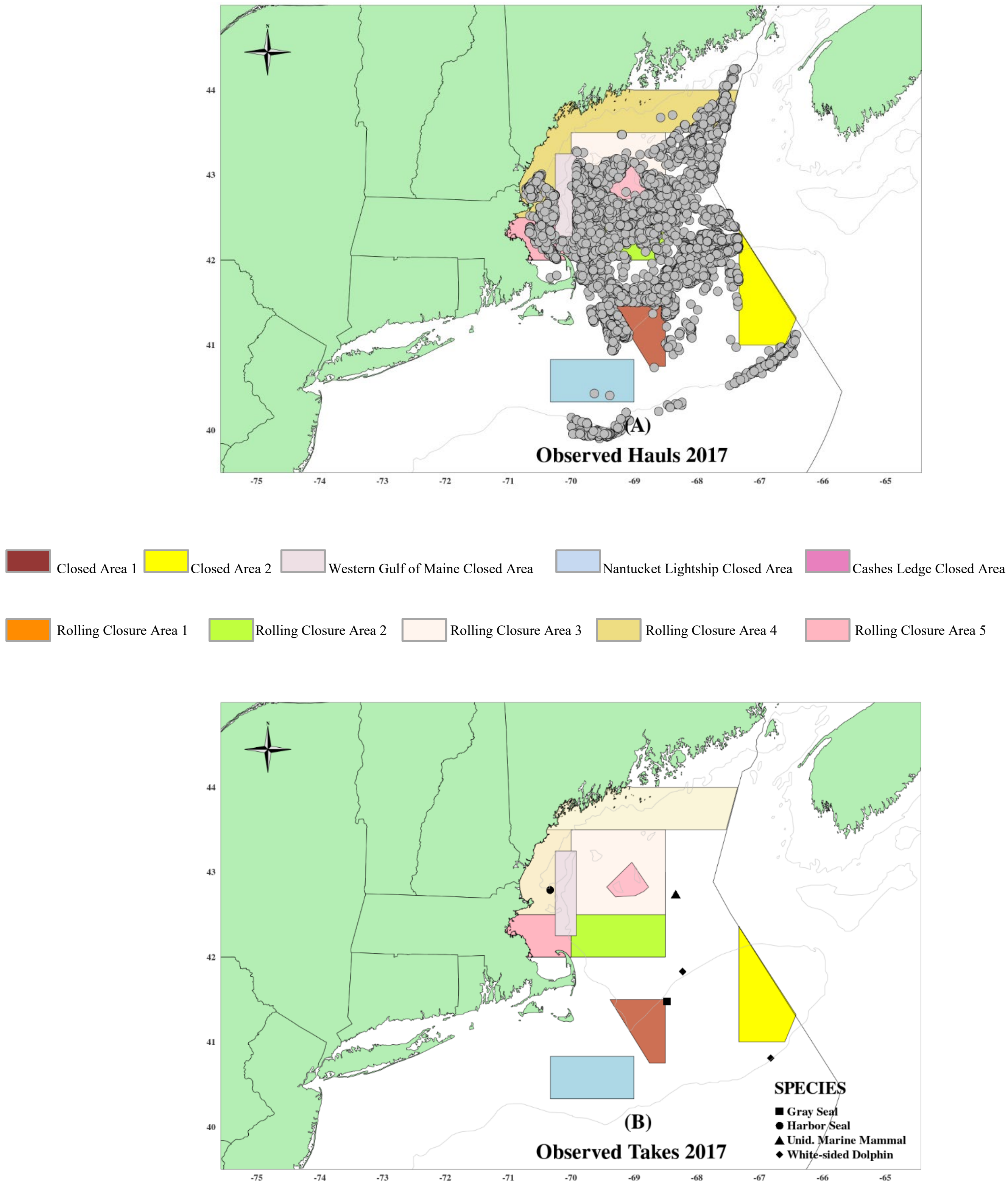


Figure 20. 2018 Northeast bottom trawl observed tows (A) and observed takes (B).

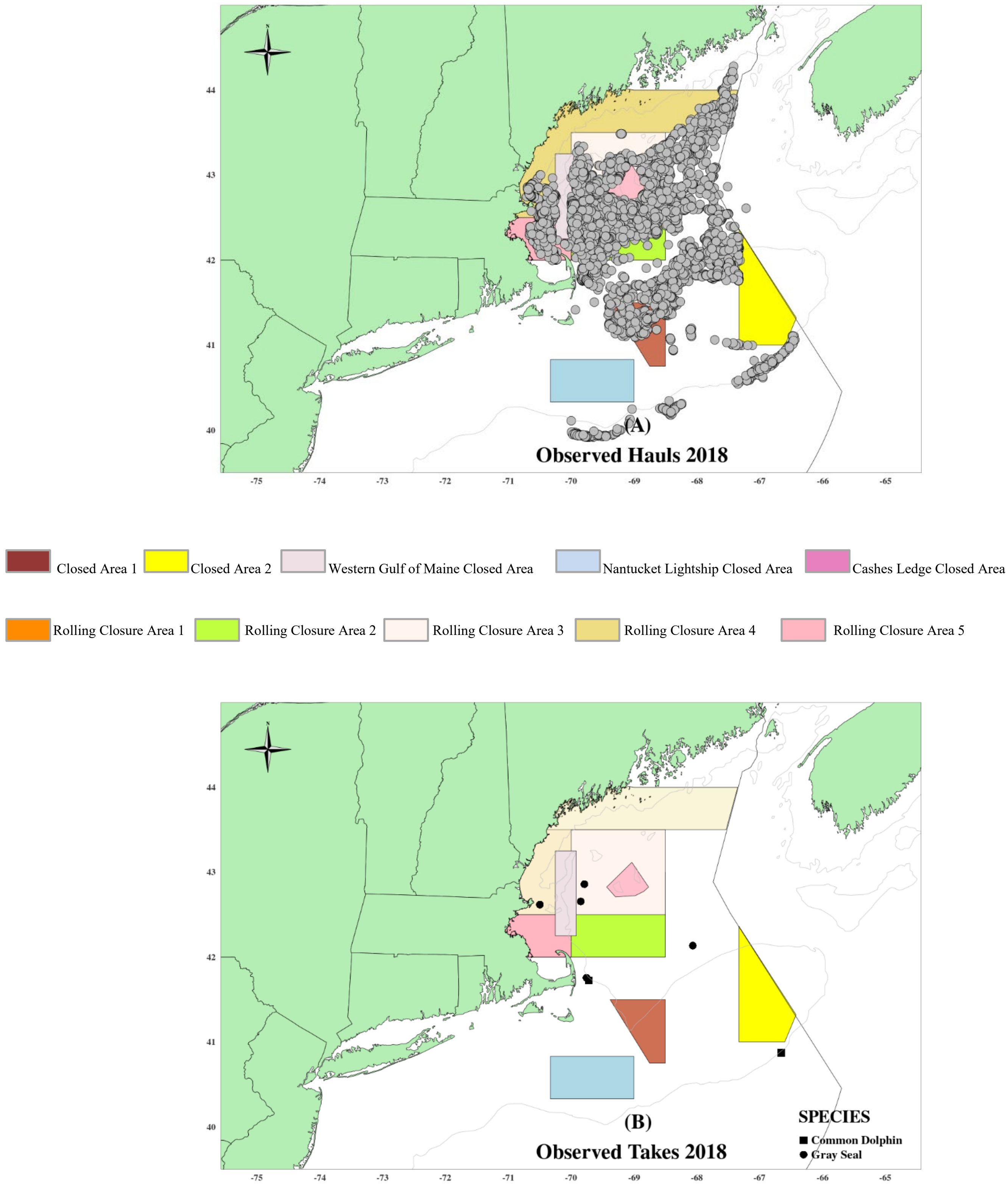
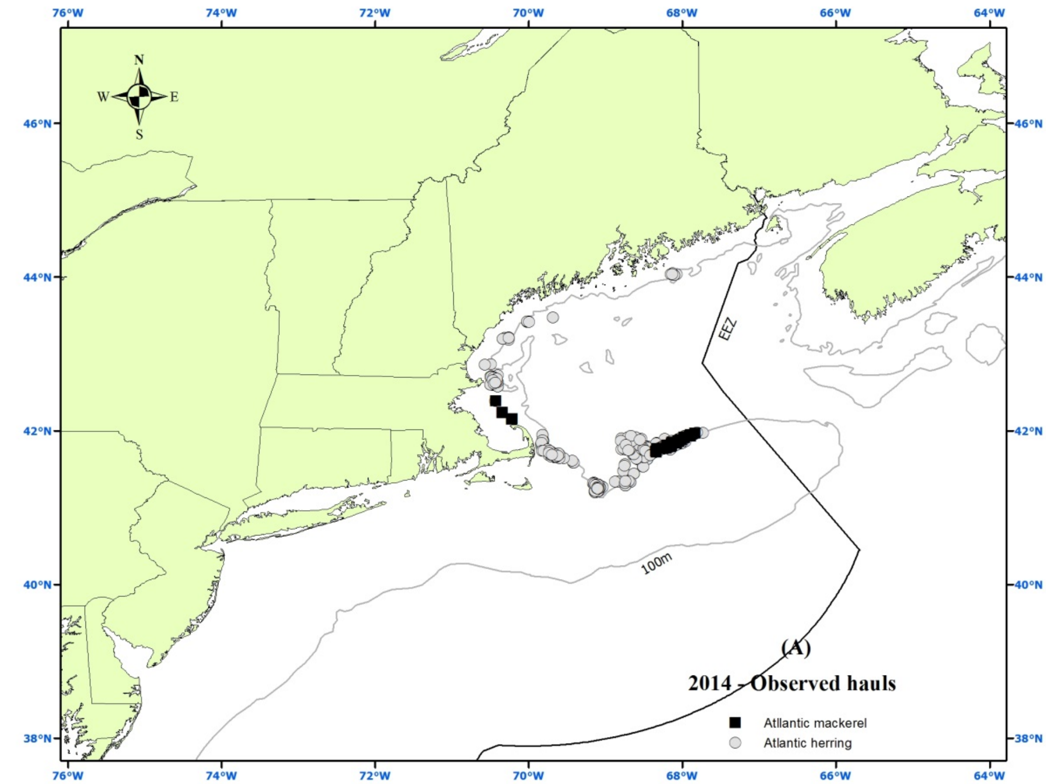


Figure 21. 2014 Northeast mid-water trawl observed tows (A) and observed takes (B).



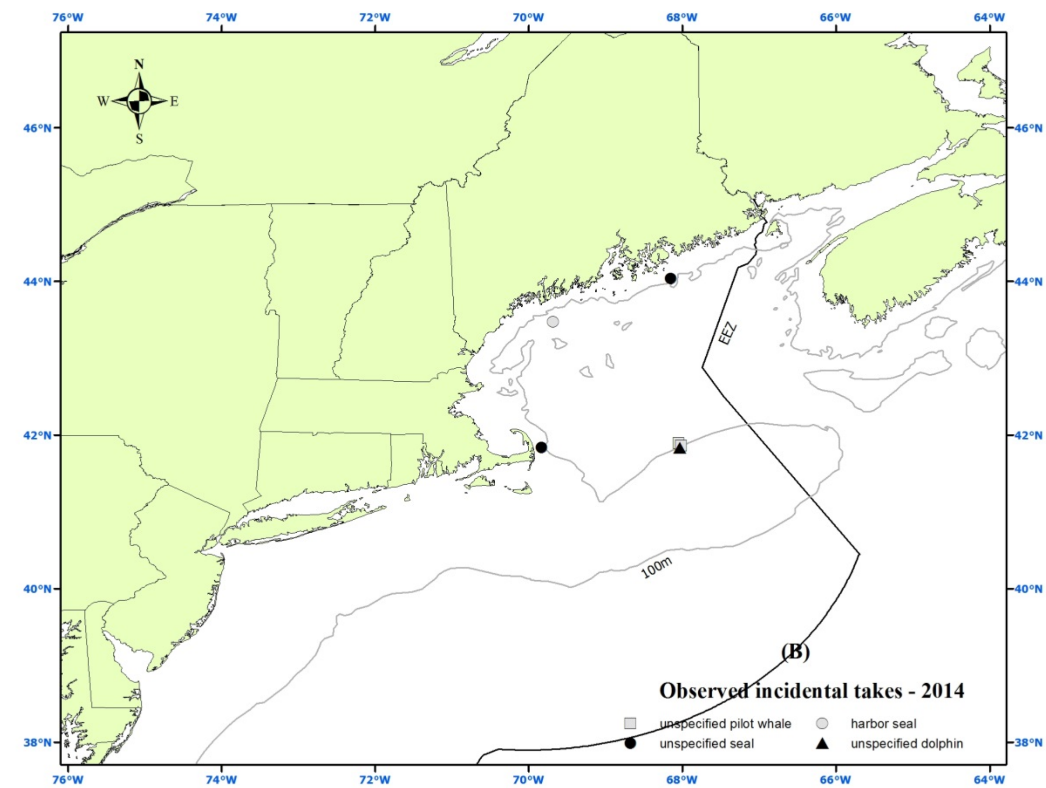


Figure 22. 2015 Northeast mid-water trawl observed tows (A) and observed takes (B).

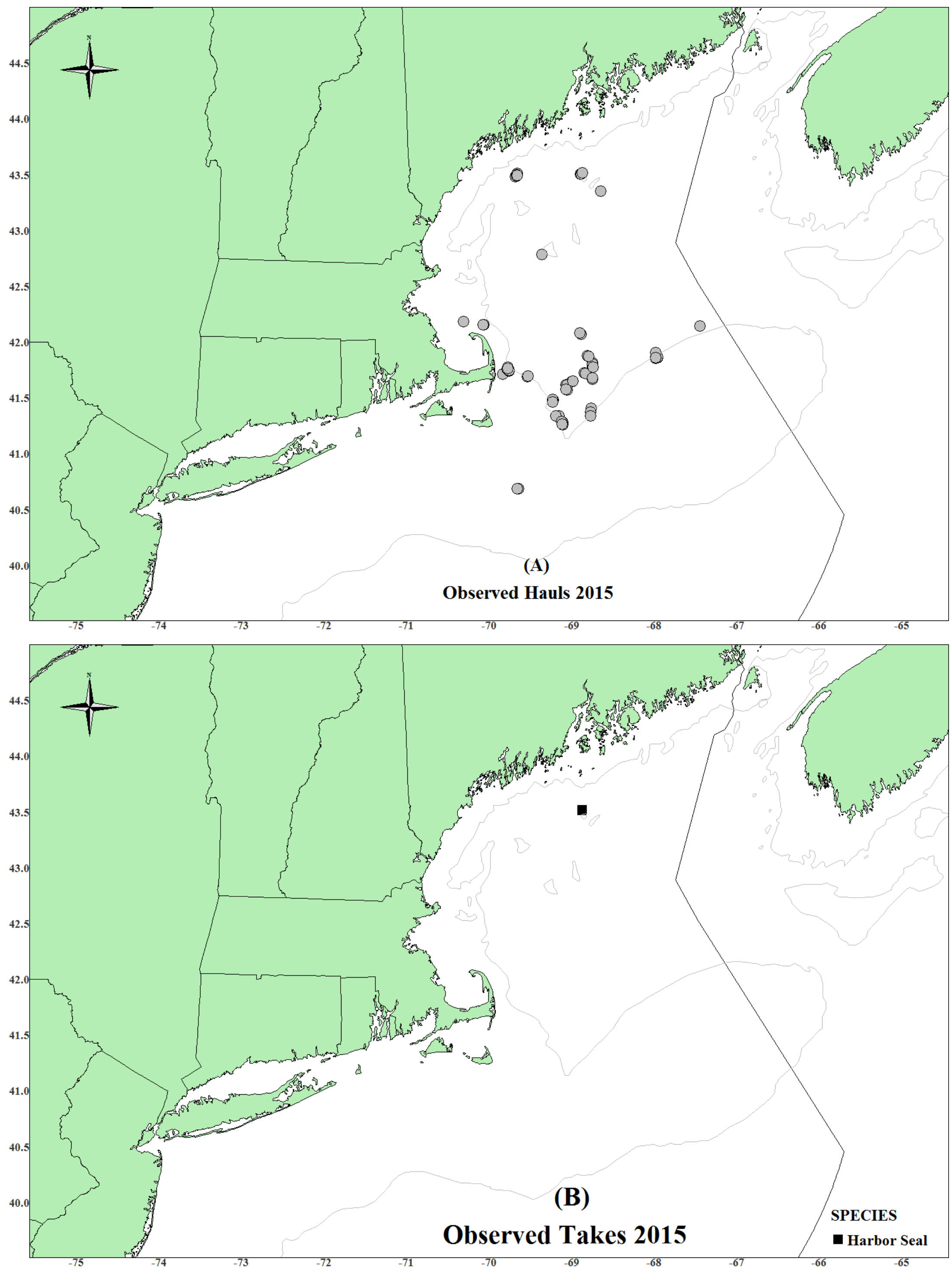


Figure 23. 2016 Northeast mid-water trawl observed tows (A) and observed takes (B).

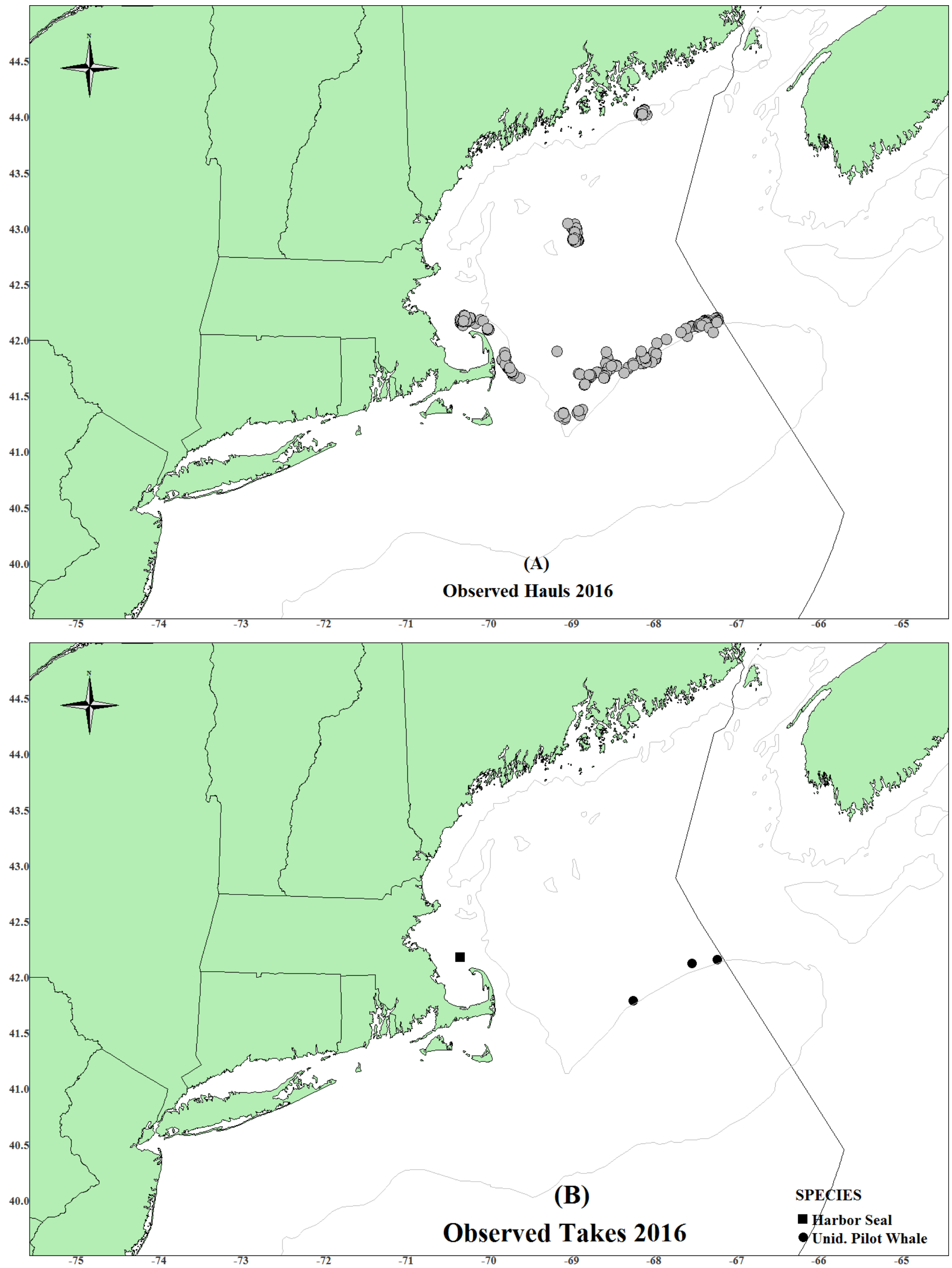
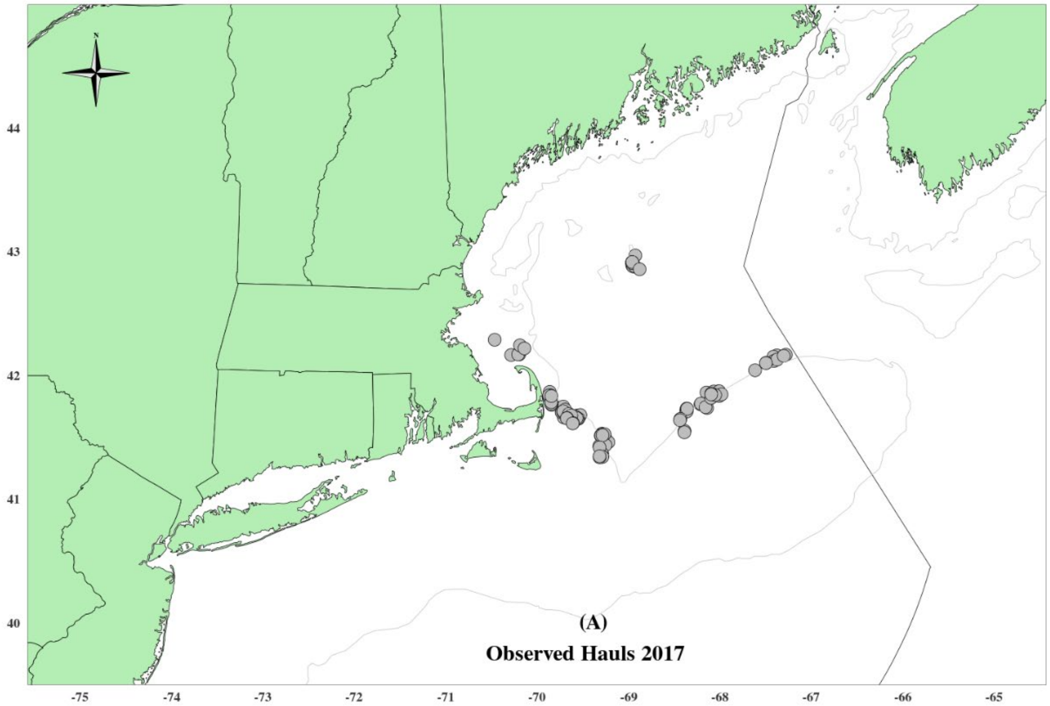


Figure 24. 2017 Northeast mid-water trawl observed tows (A) and observed takes (B).



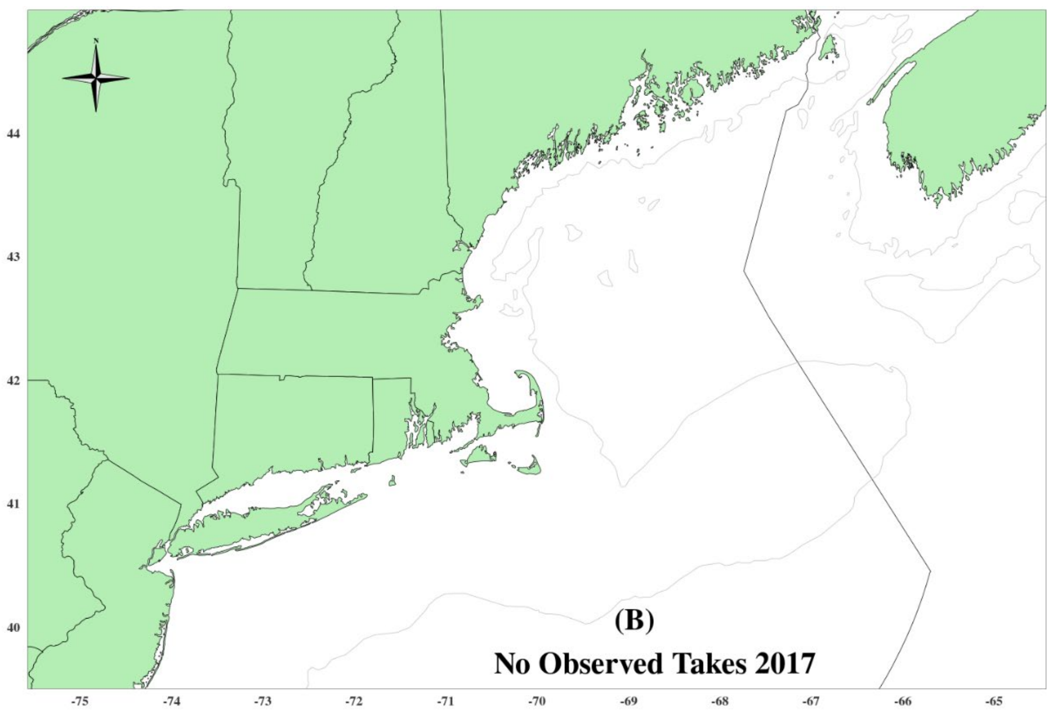


Figure 25. 2018 Northeast mid-water trawl observed tows (A) and observed takes (B).

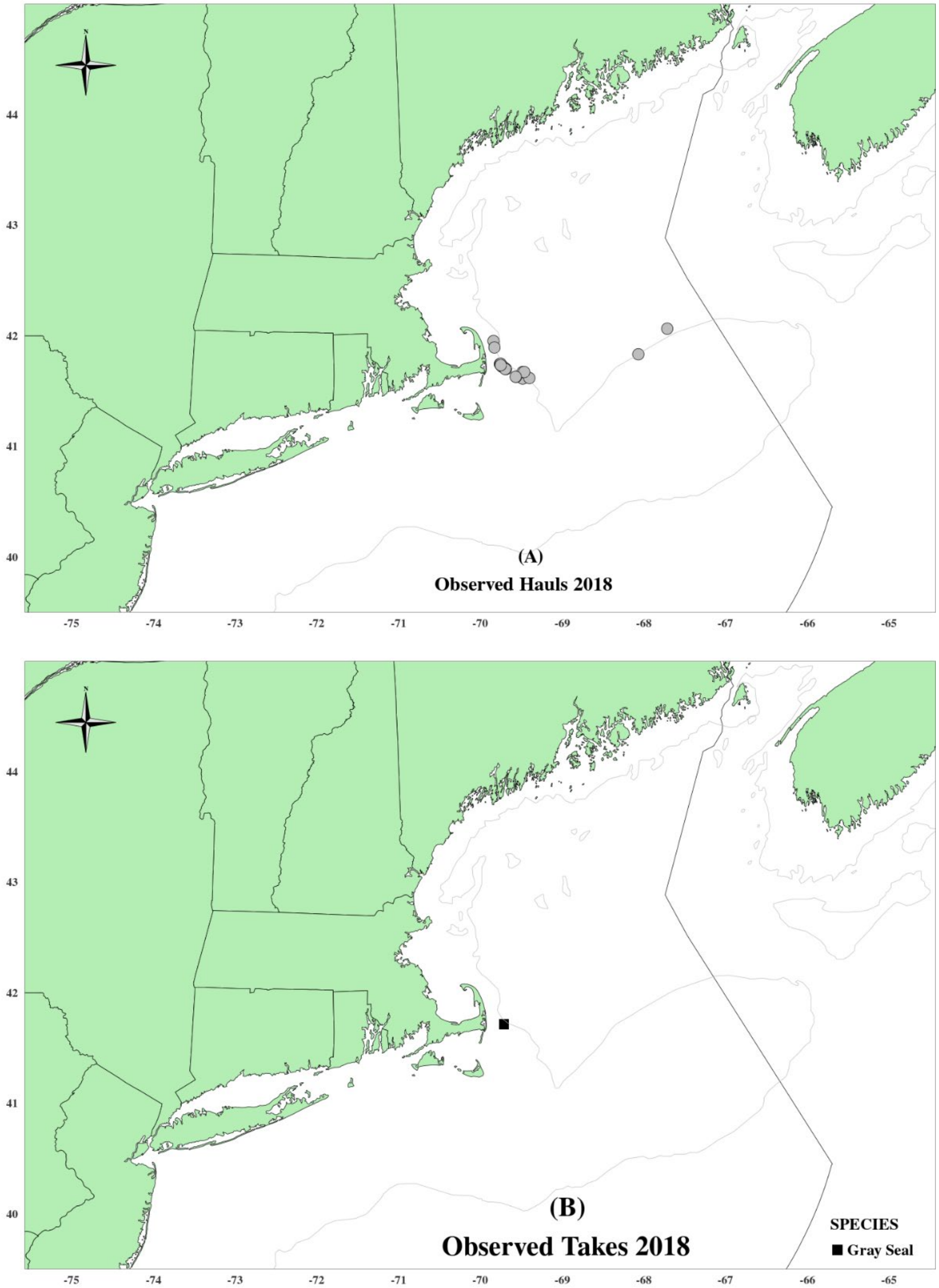
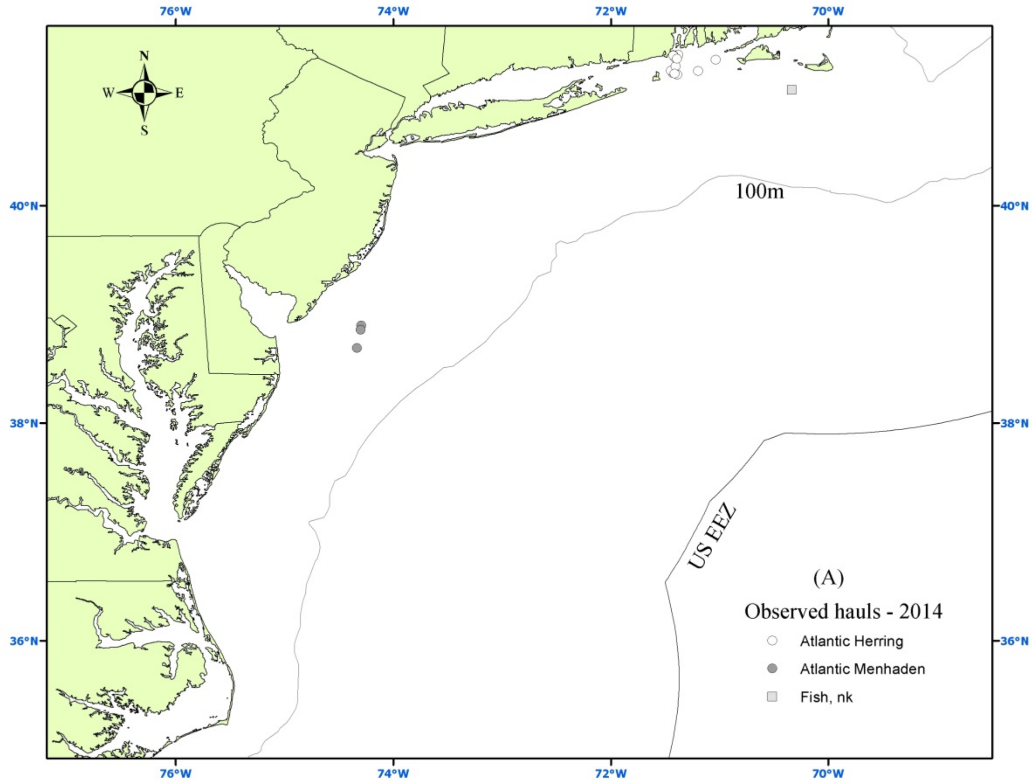


Figure 26. 2014 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



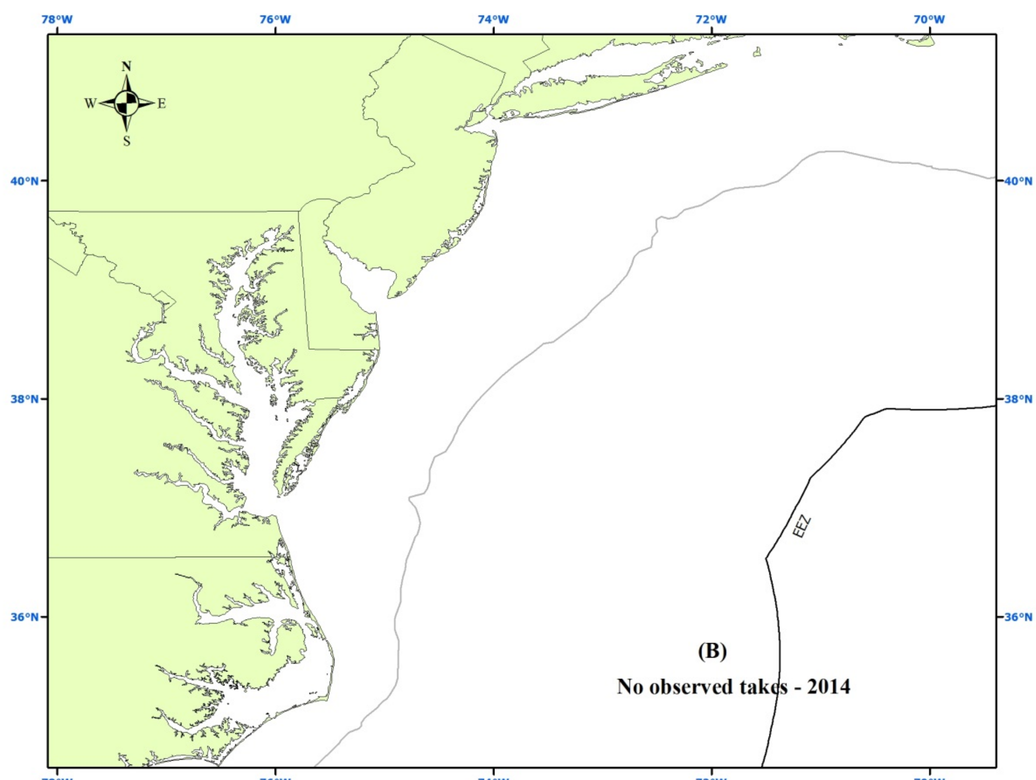
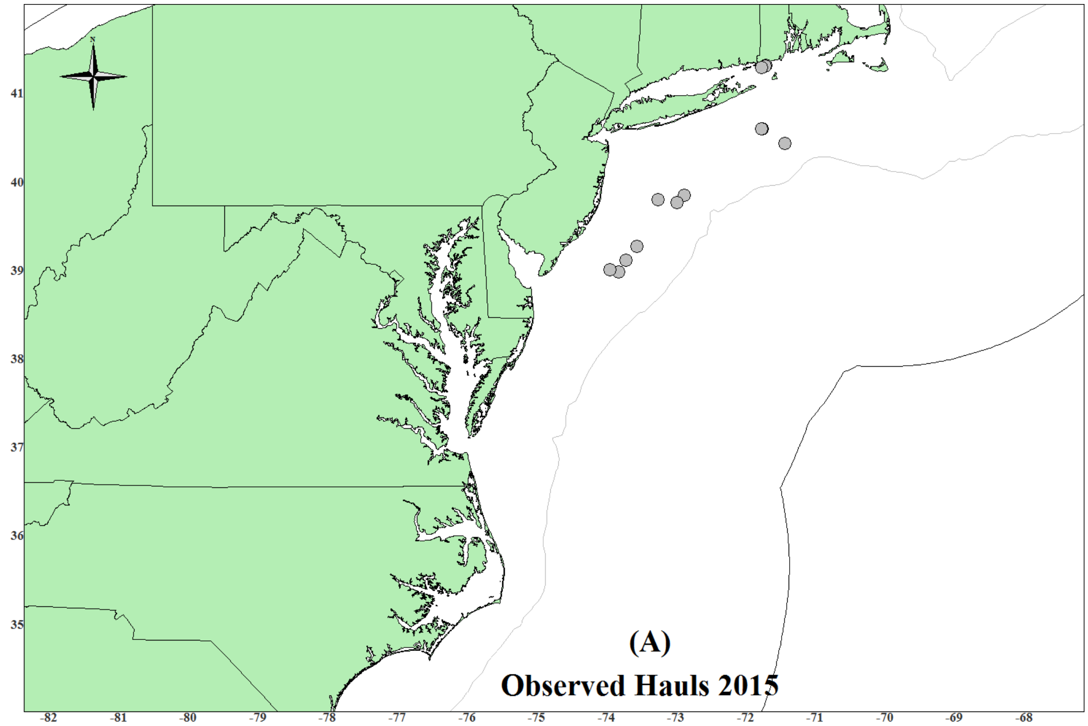


Figure 27. 2015 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



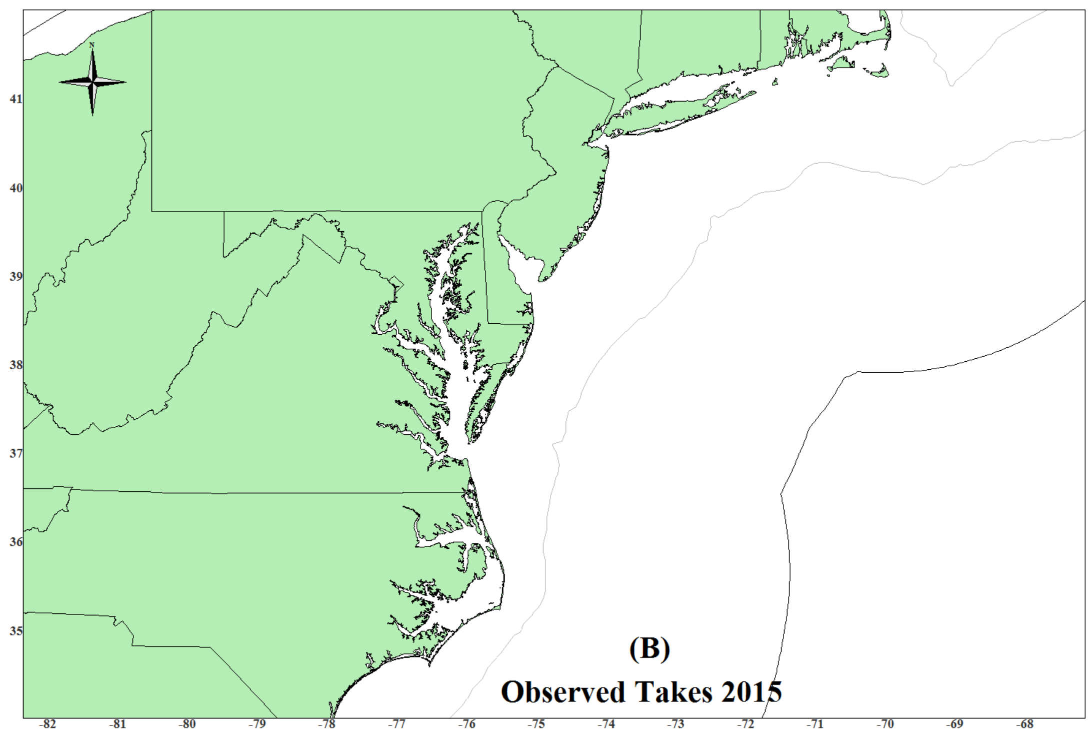
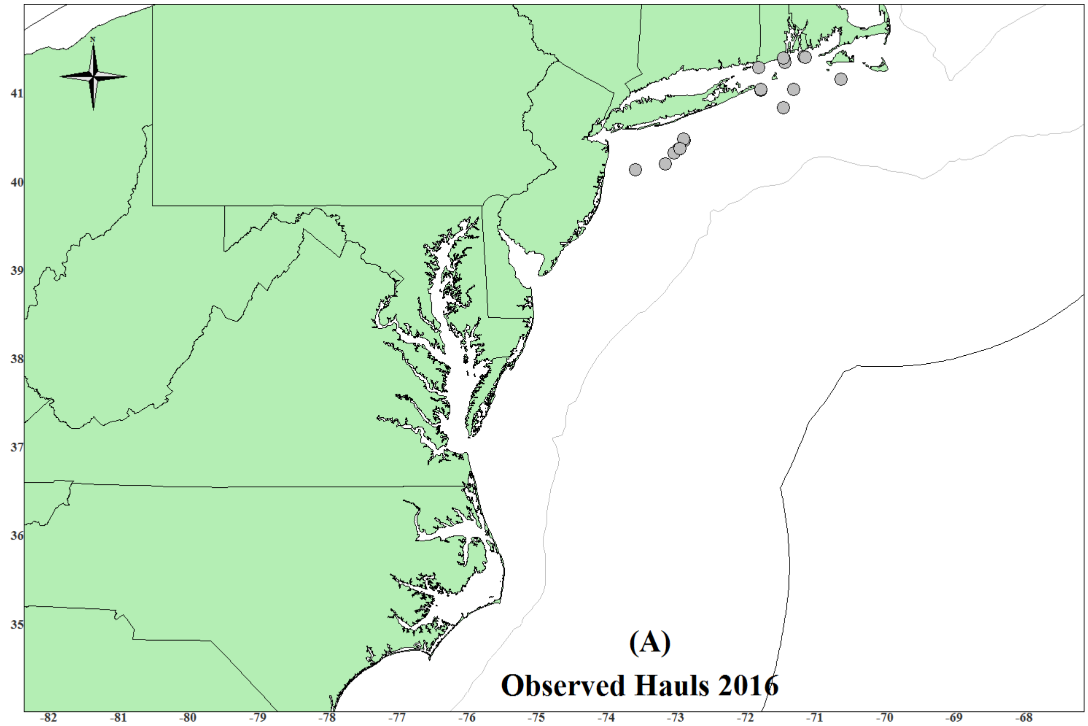


Figure 28. 2016 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



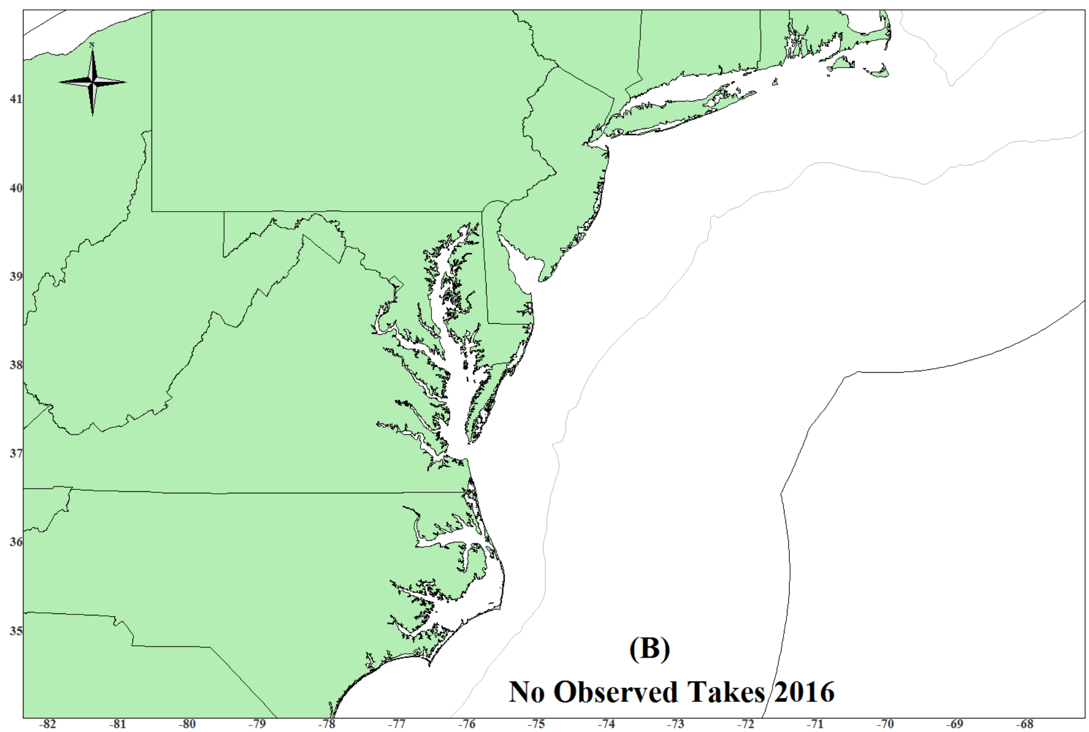
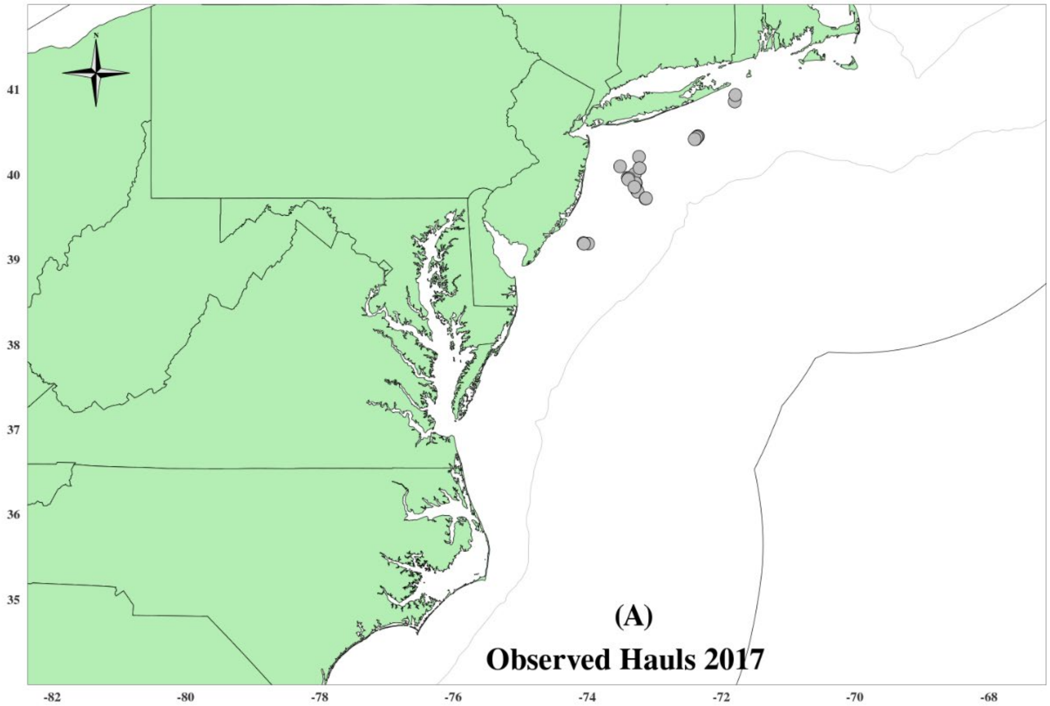


Figure 29. 2017 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



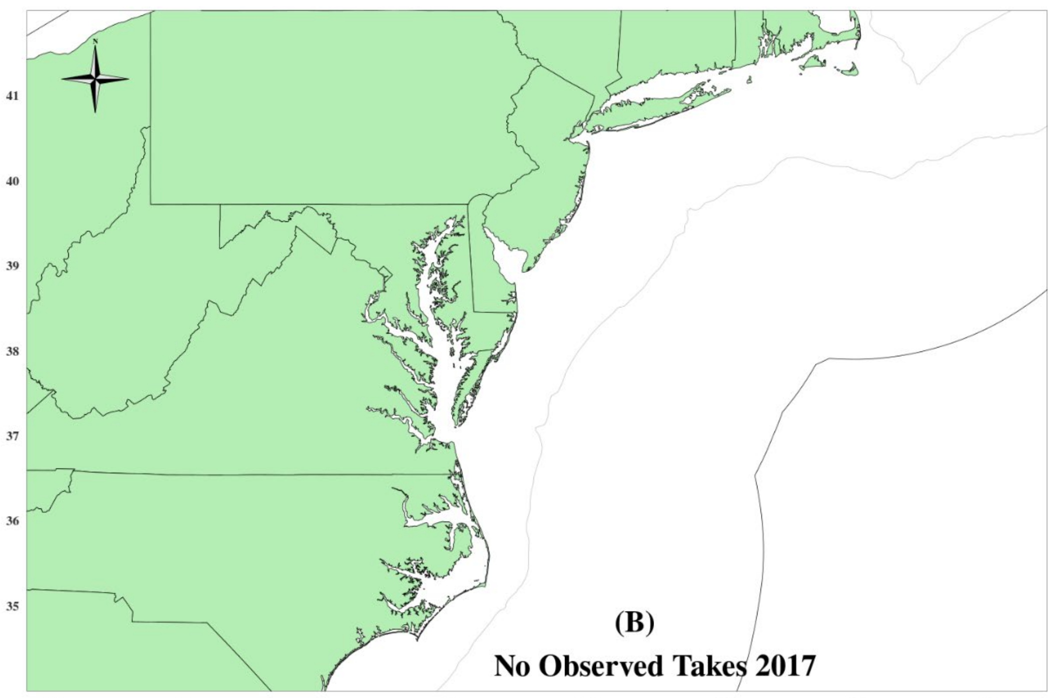
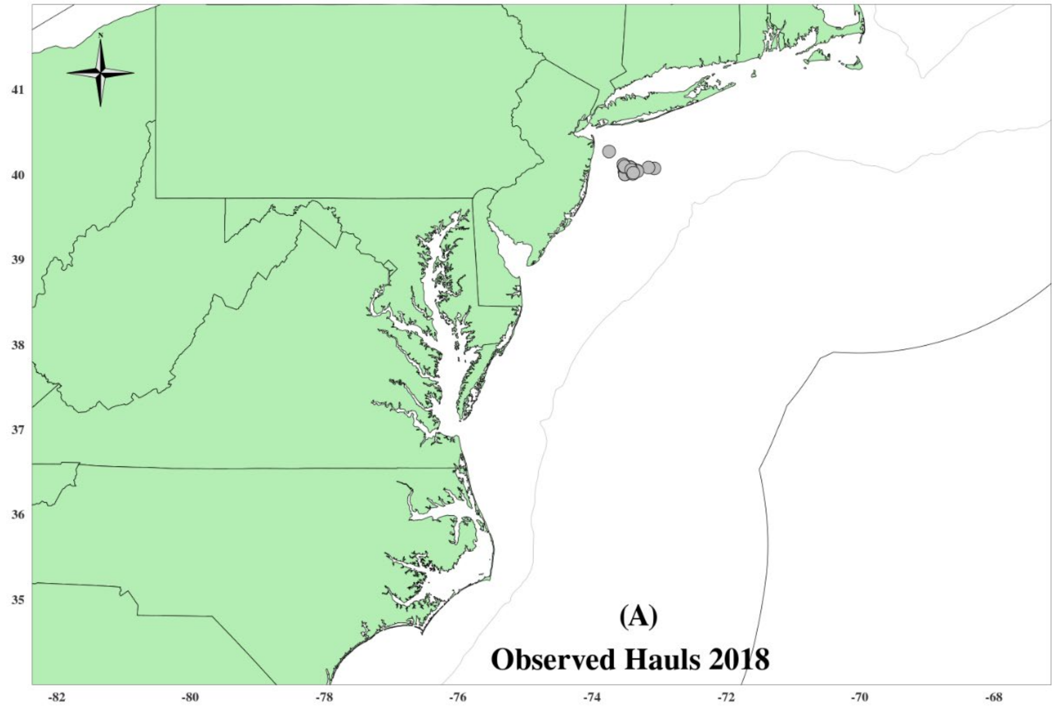


Figure 30. 2018 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



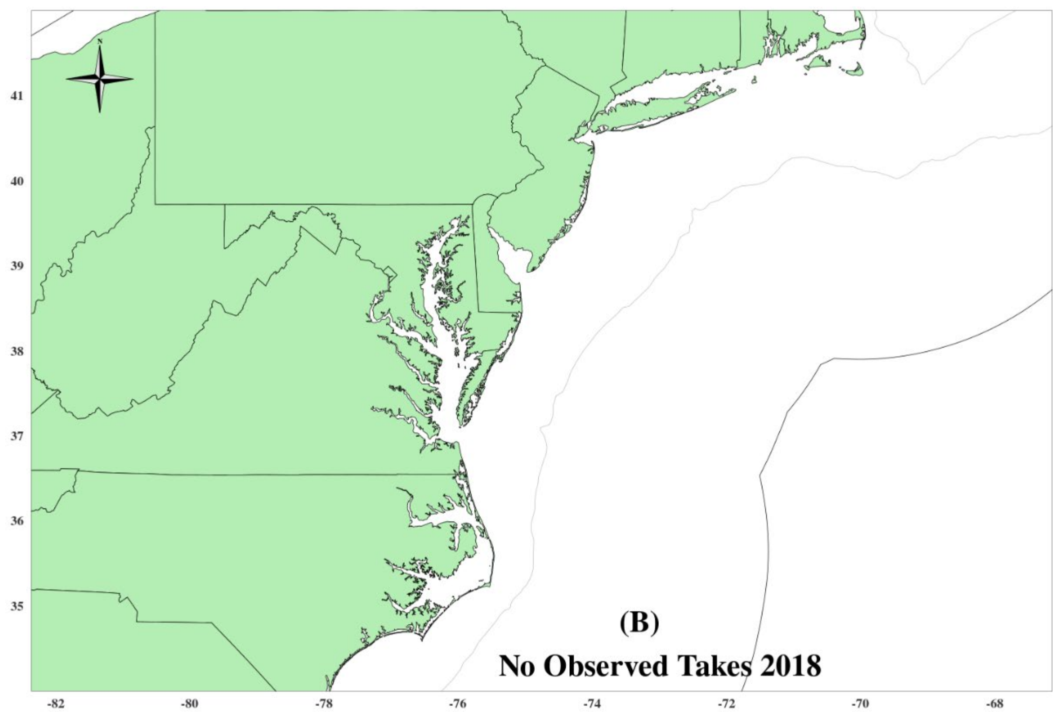
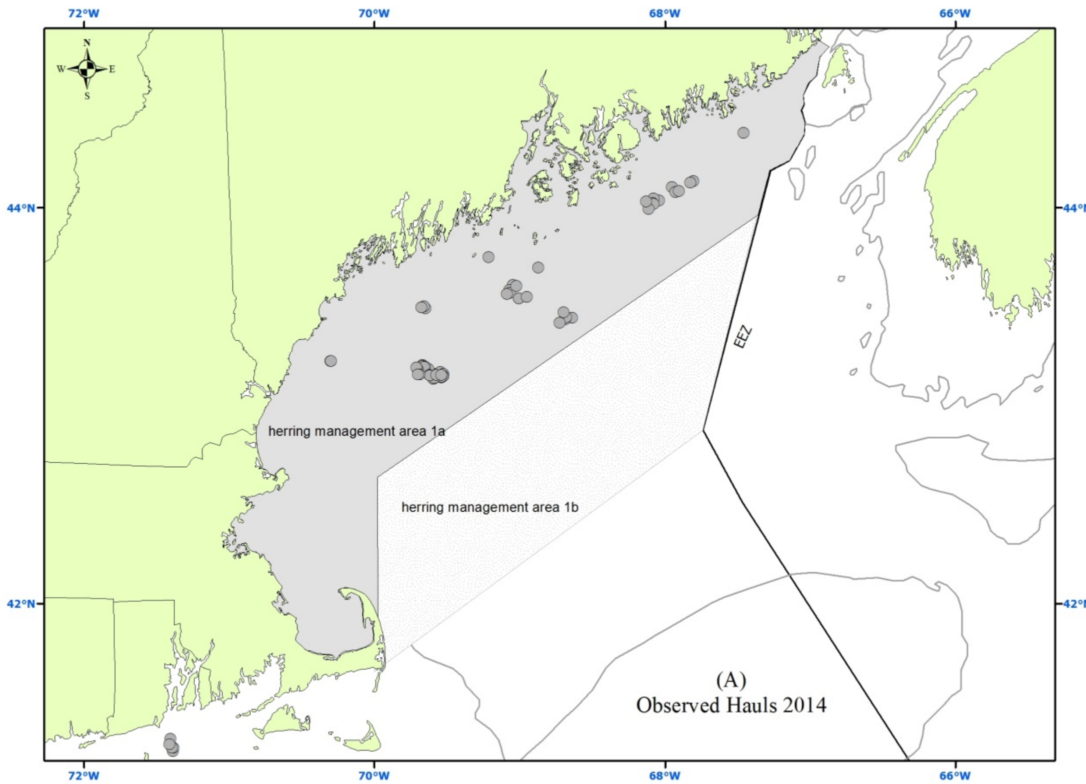


Figure 31. 2014 Herring purse seine observed hauls (A) and observed takes (B).



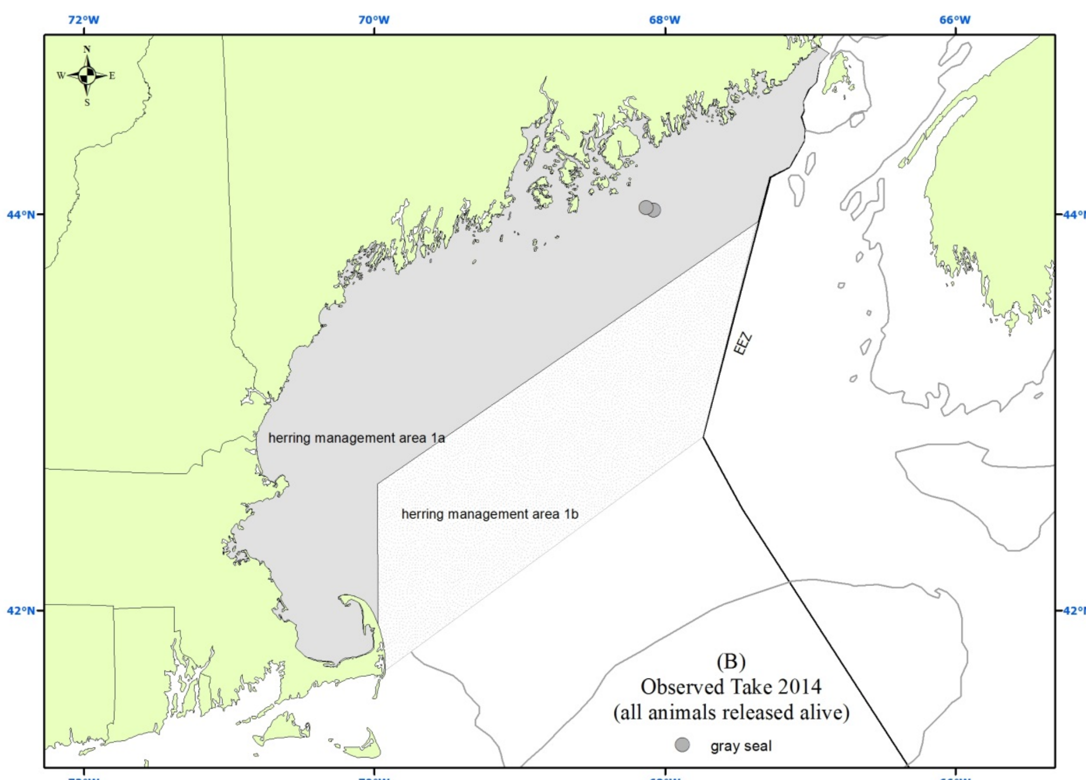


Figure 32. 2015 Herring purse seine observed hauls (A) and observed takes (B).

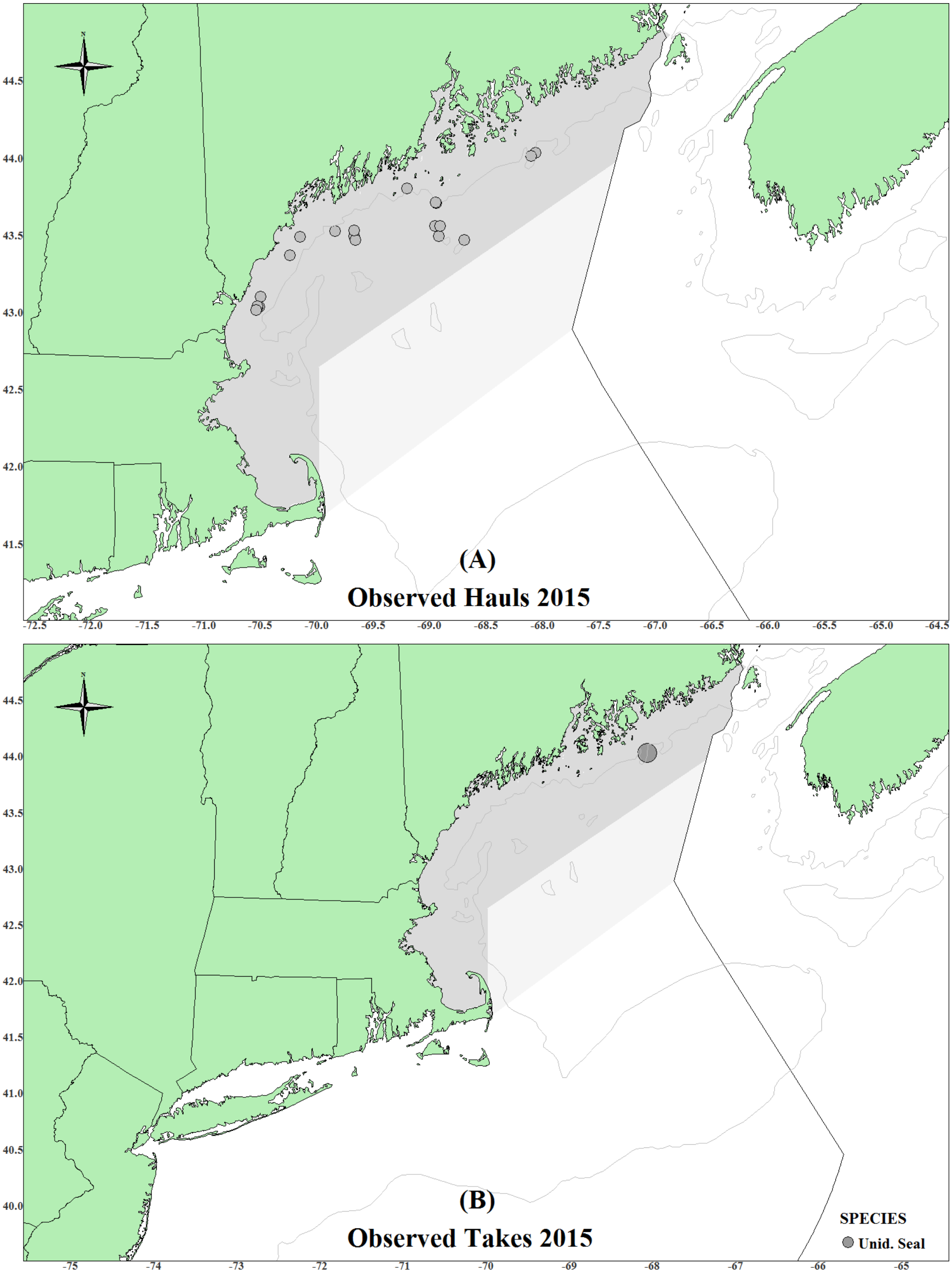
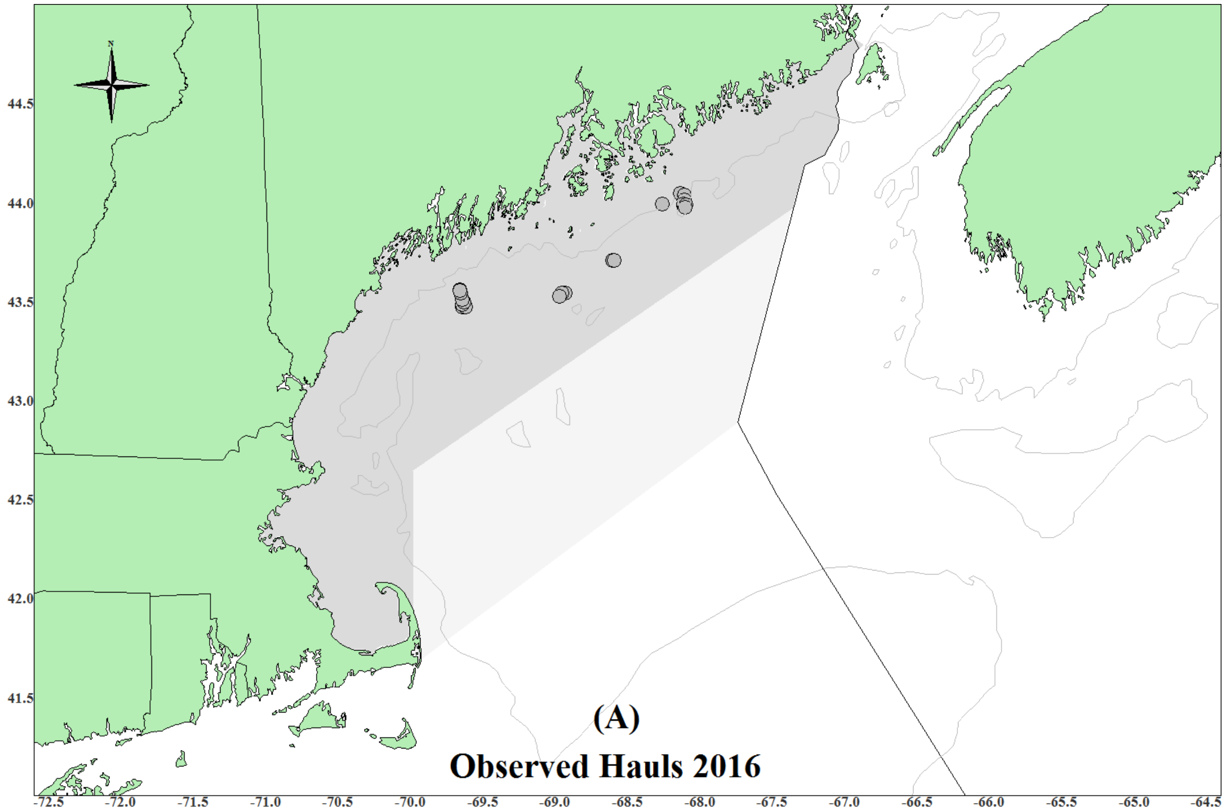


Figure 33. 2016 Herring purse seine observed hauls (A) and observed takes (B).



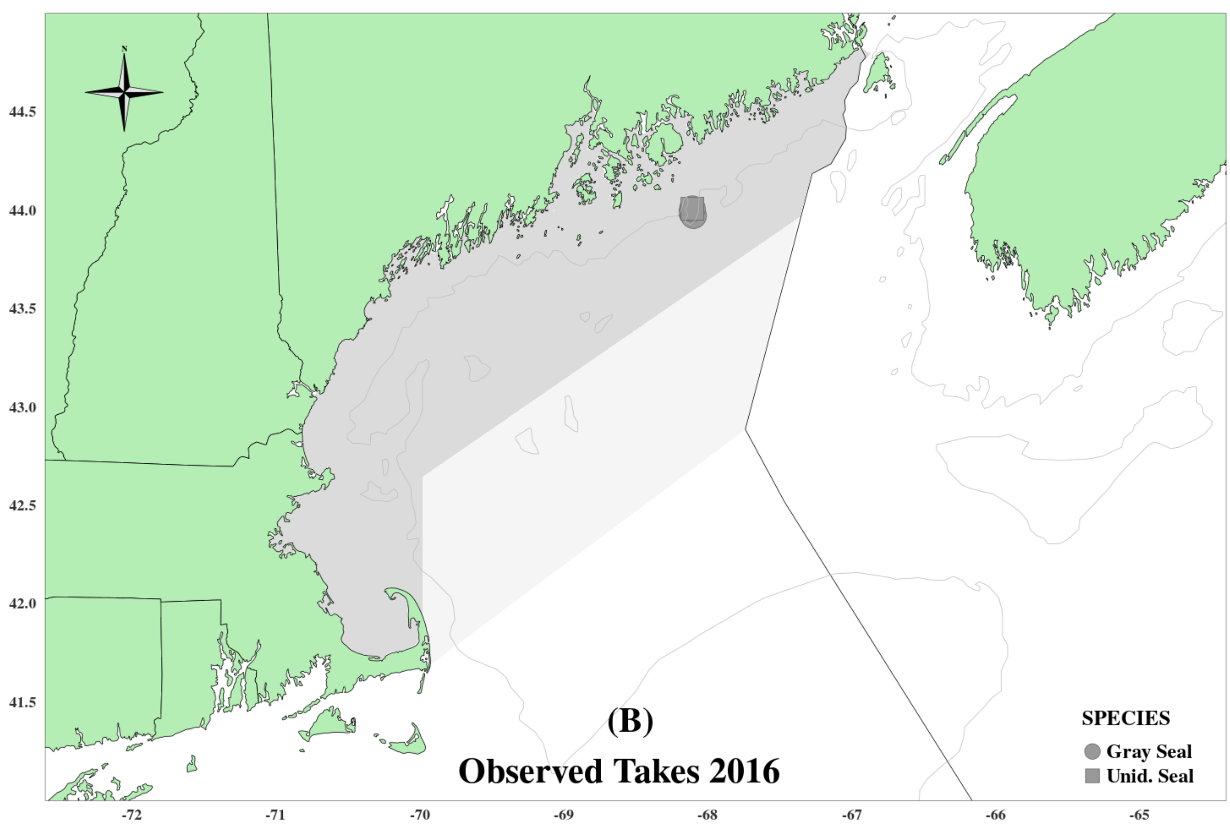


Figure 34. 2017 Herring purse seine observed hauls (A) and observed takes (B).

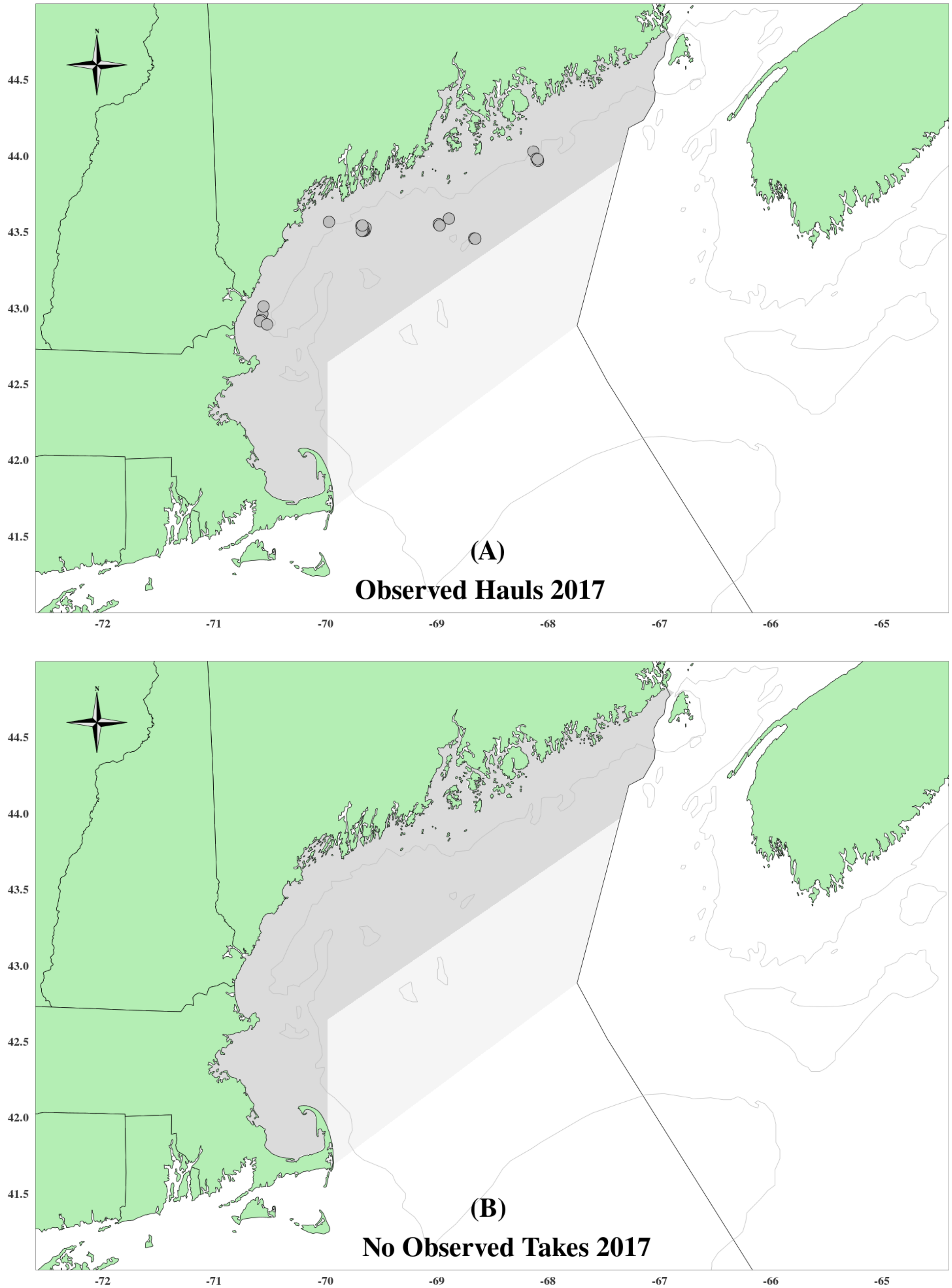


Figure 35. 2018 Herring purse seine observed hauls (A) and observed takes (B).

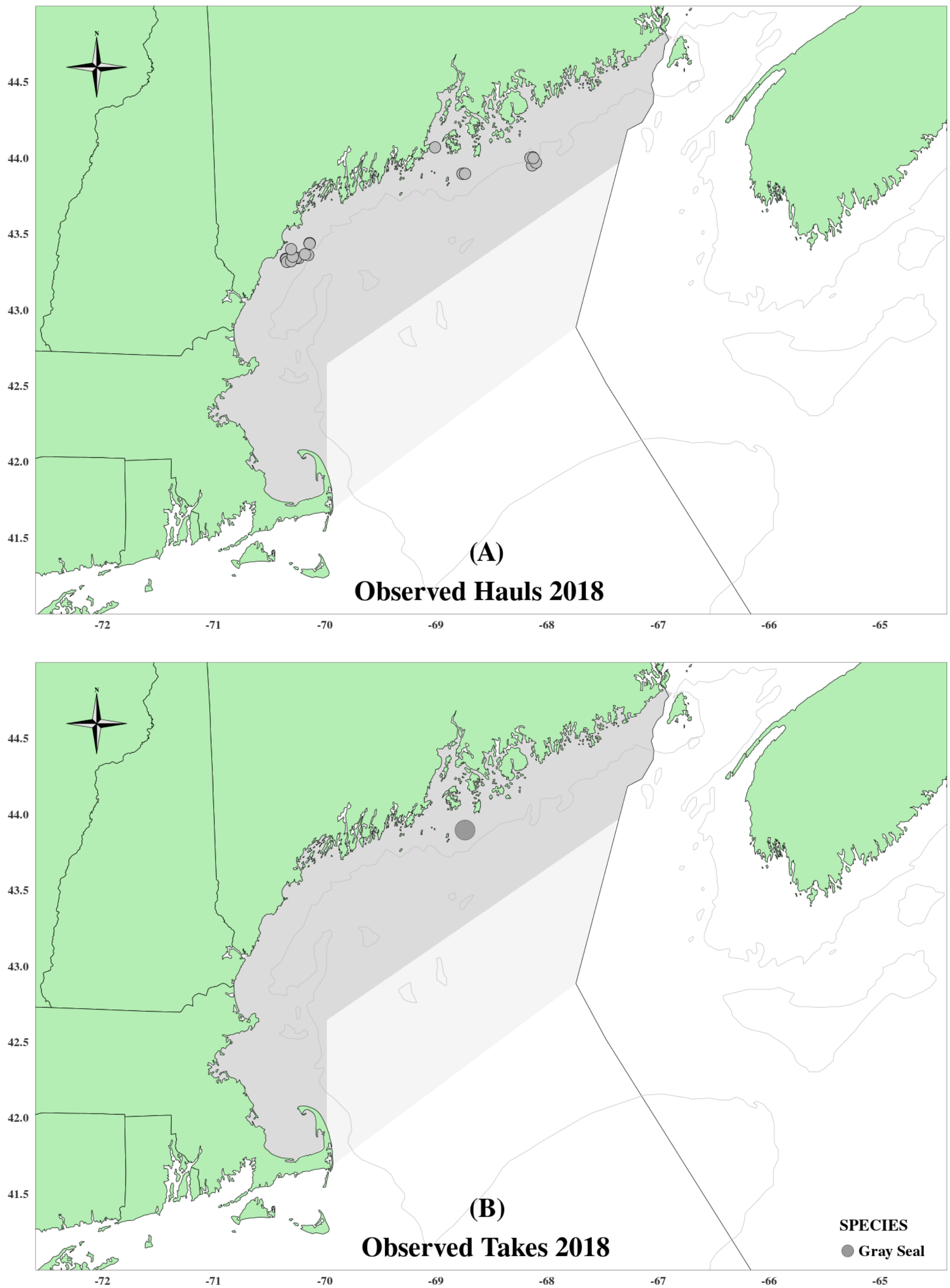


Figure 36. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2014. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

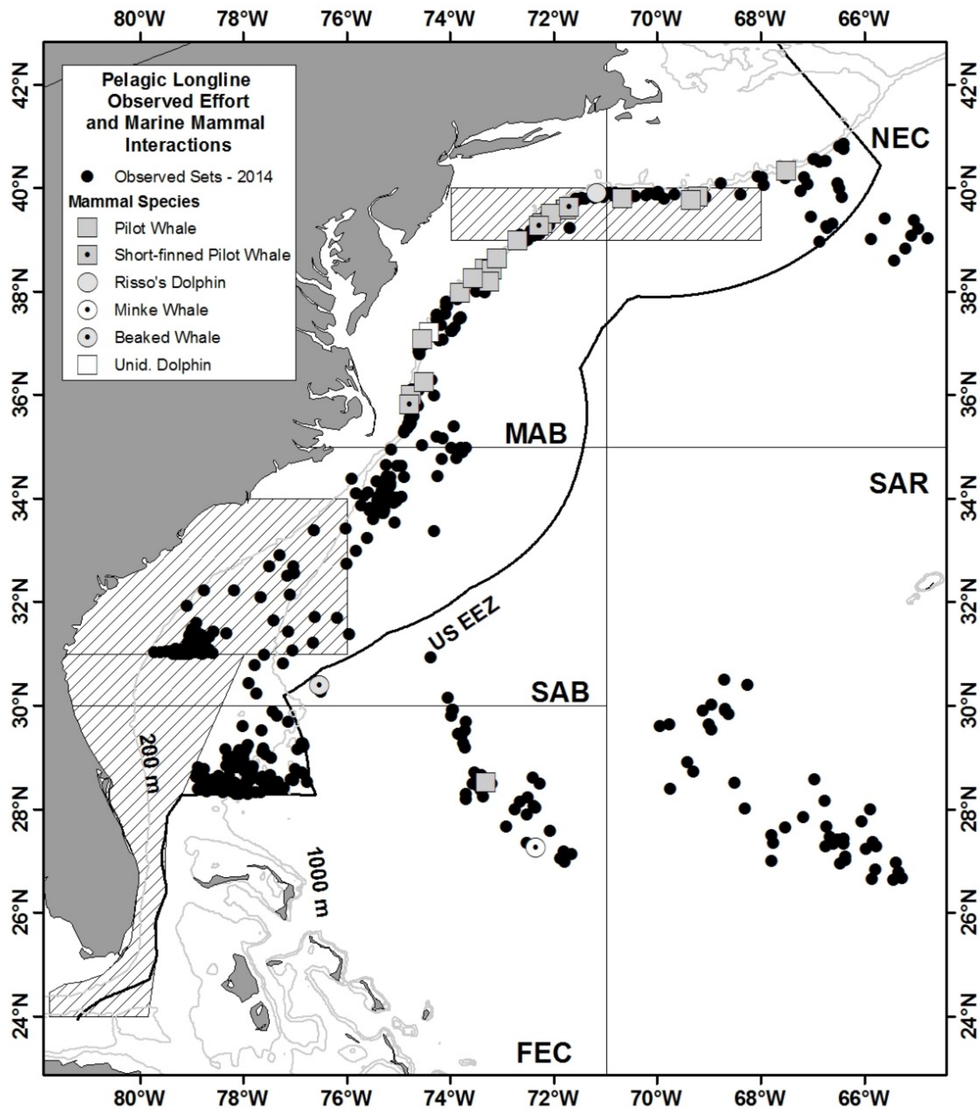


Figure 37. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2015. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

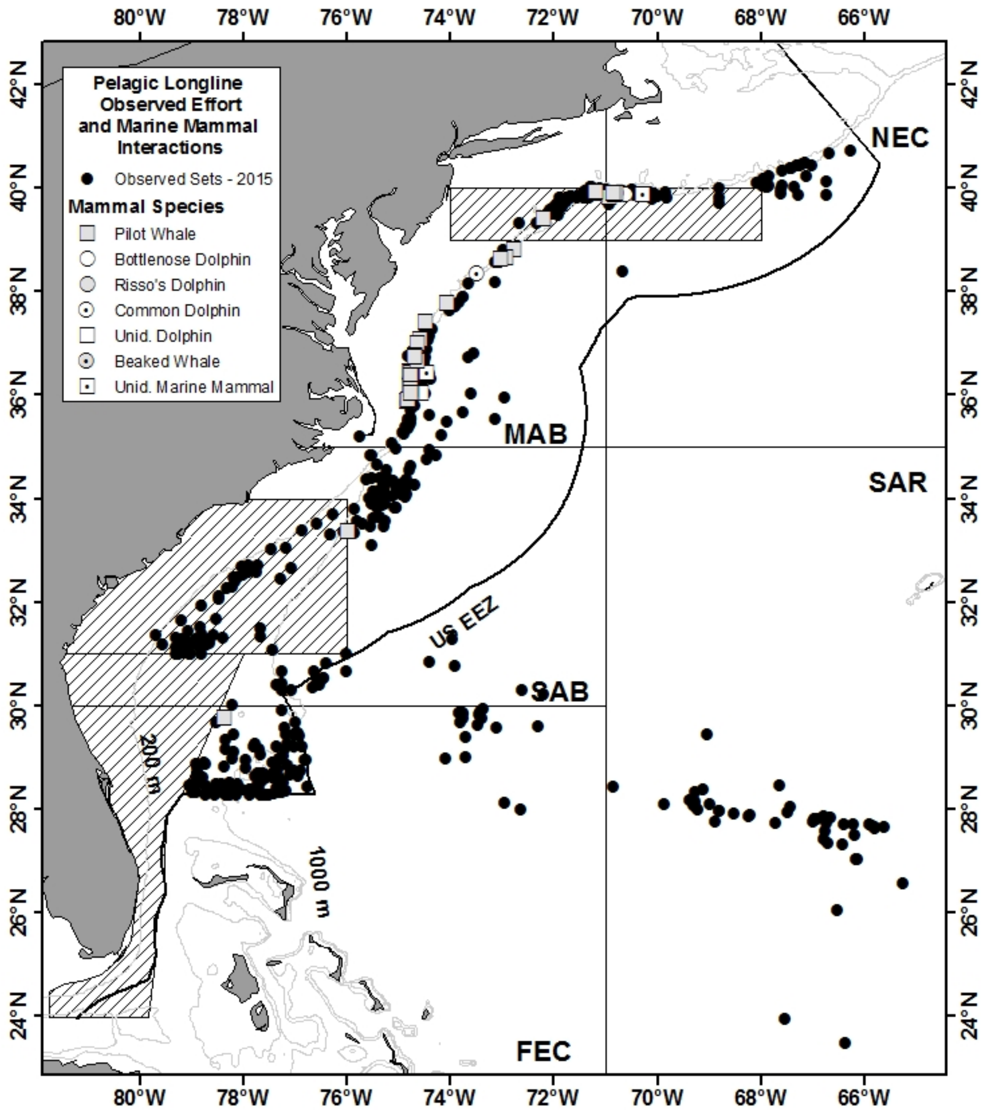


Figure 38. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2016. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

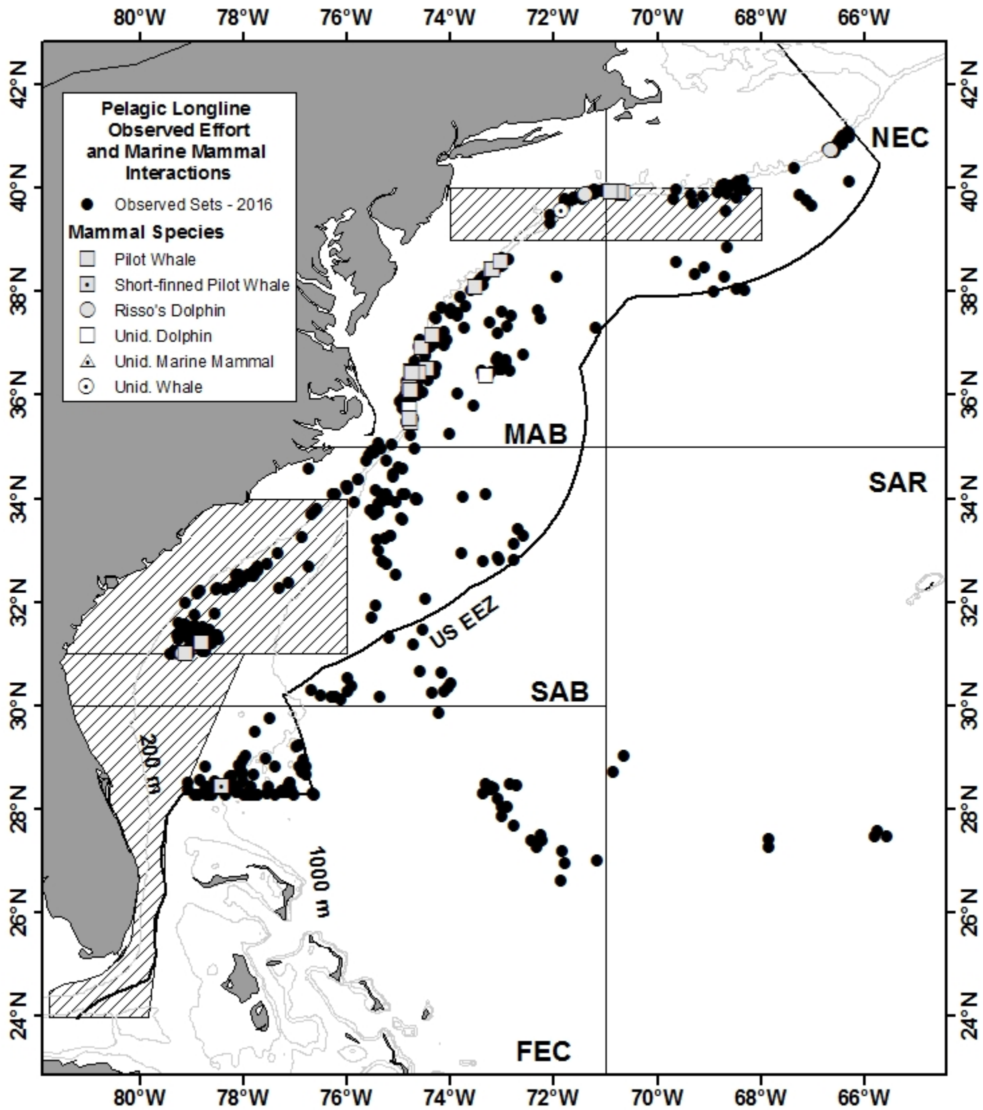


Figure 39. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2017. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

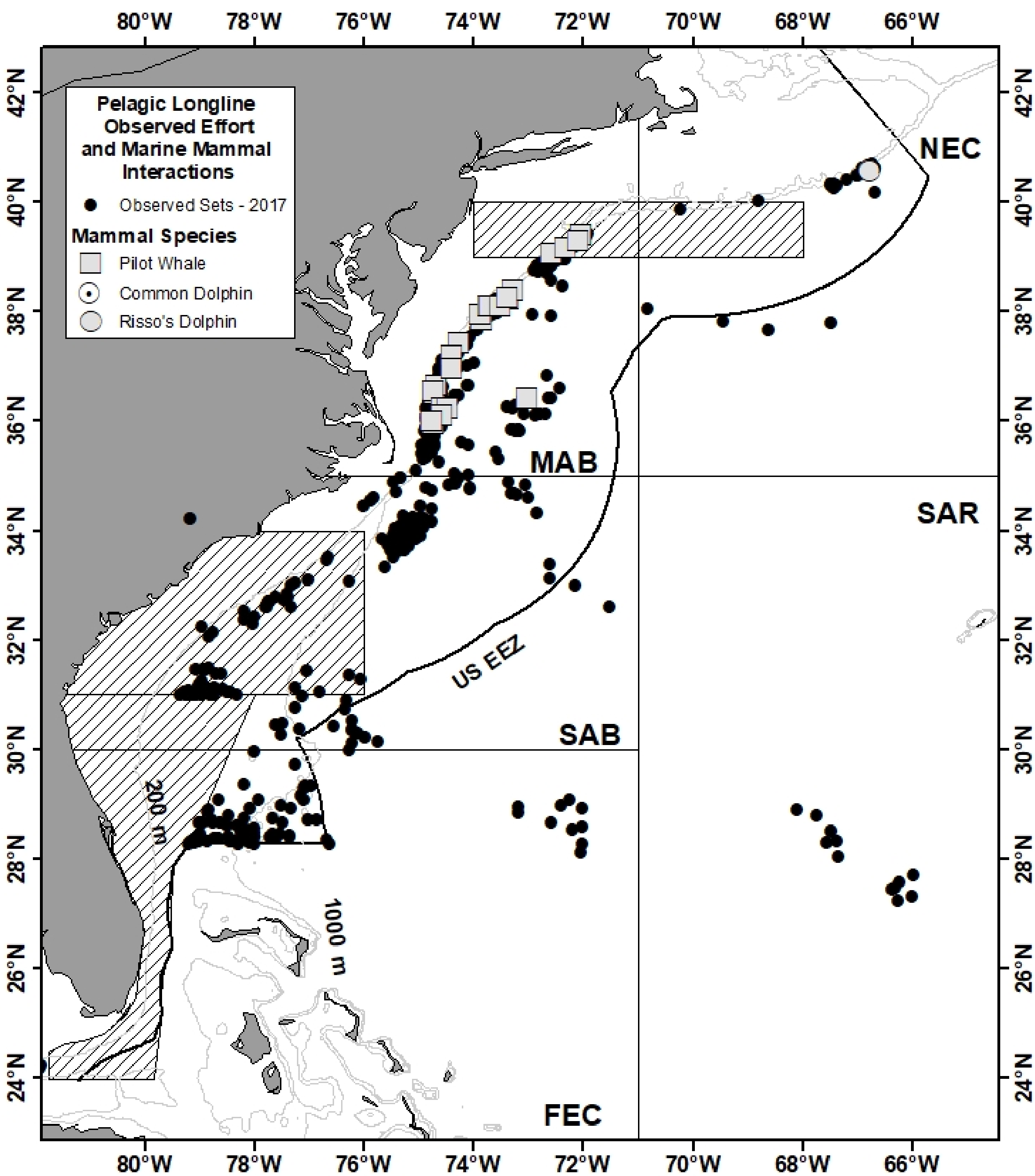


Figure 40. Observed sets and marine mammal interactions in the Pelagic Longline Fishery along the U.S. Atlantic coast during 2018. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

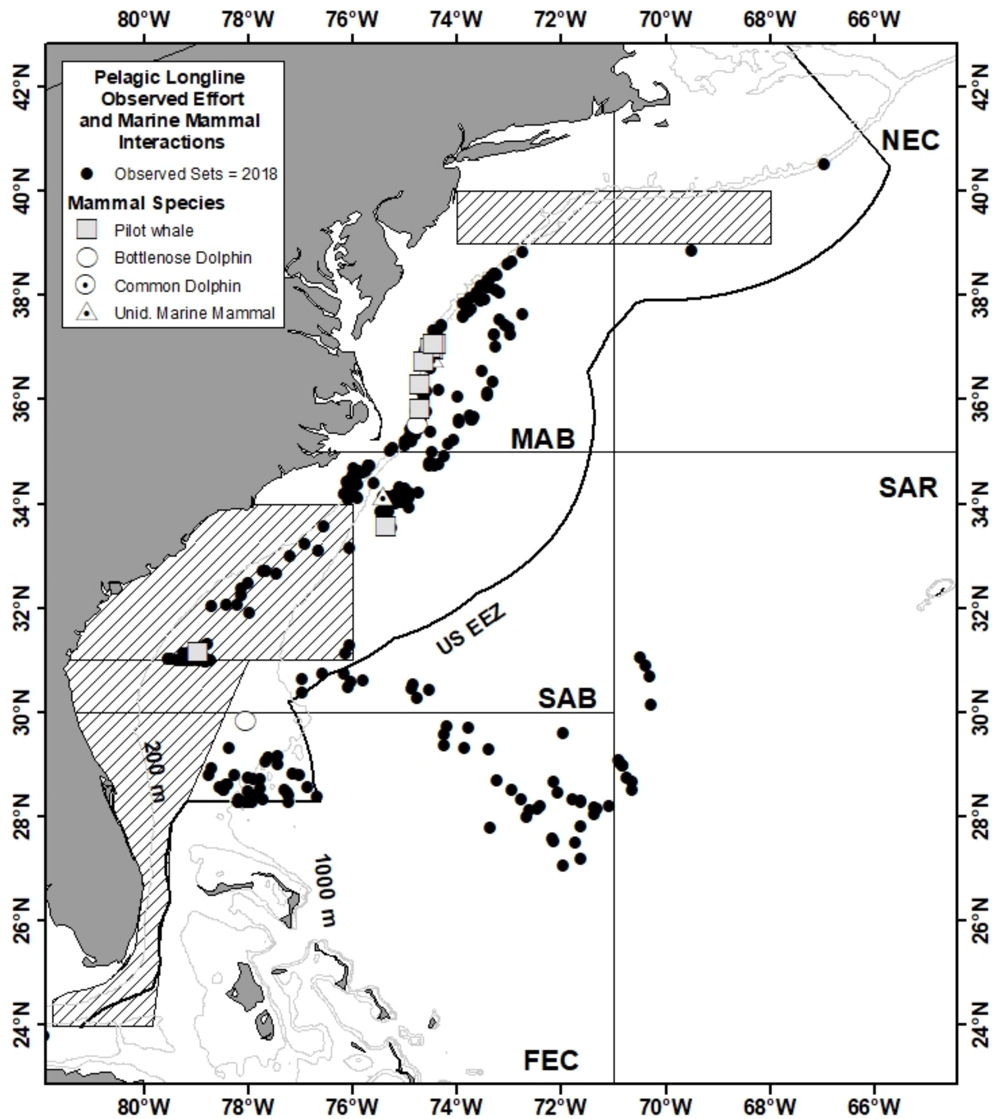


Figure 41. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2014. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.

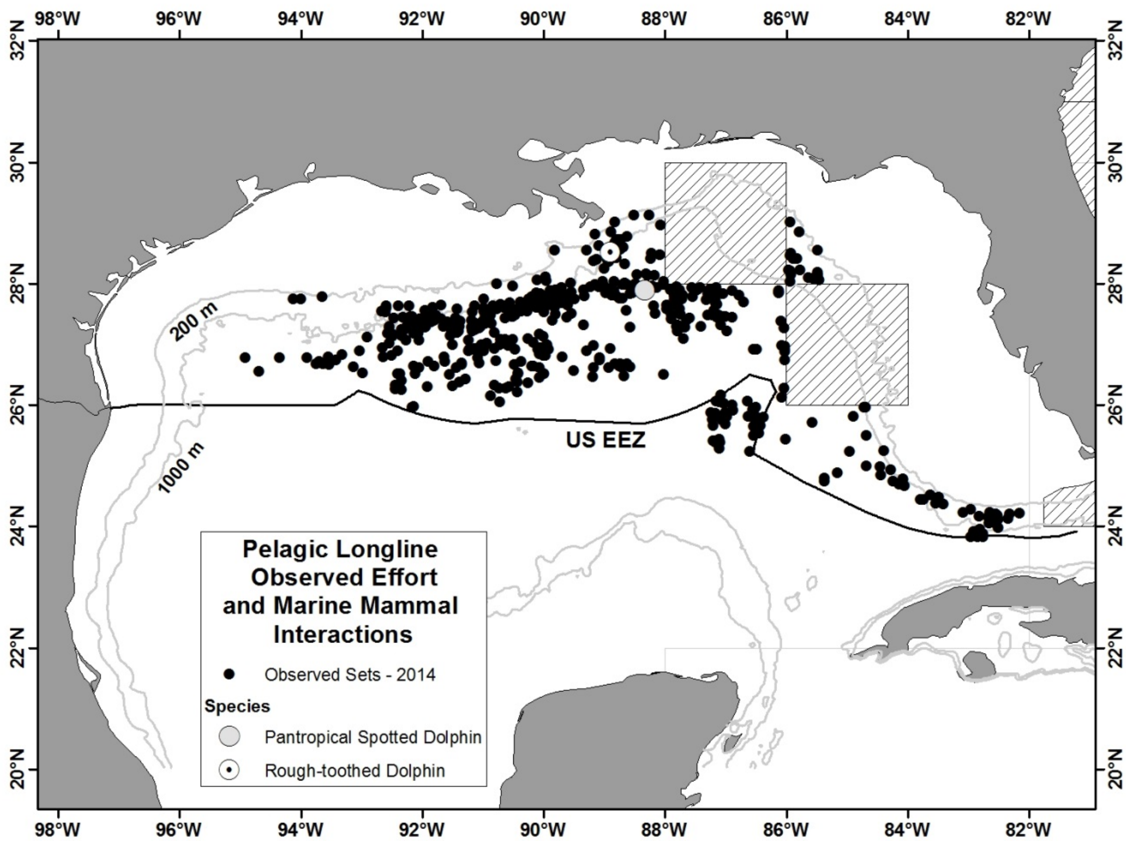


Figure 42. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2015. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.

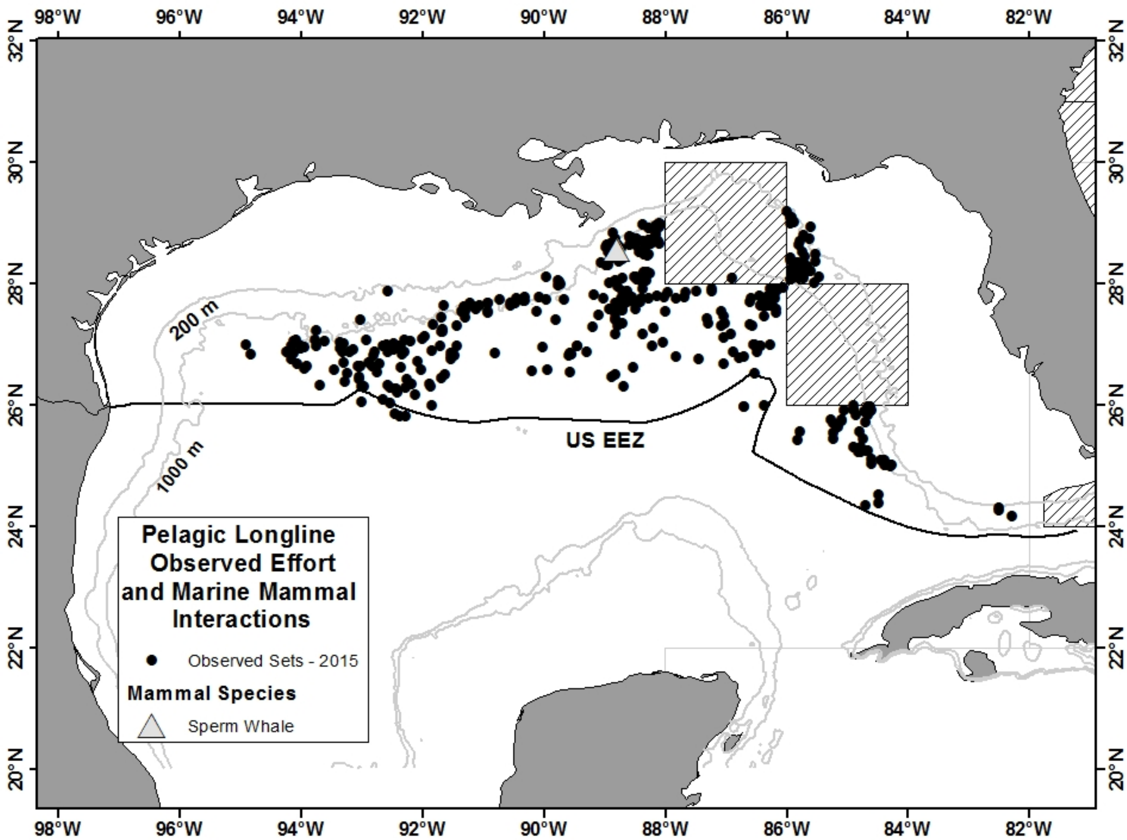


Figure 43. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2016. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.

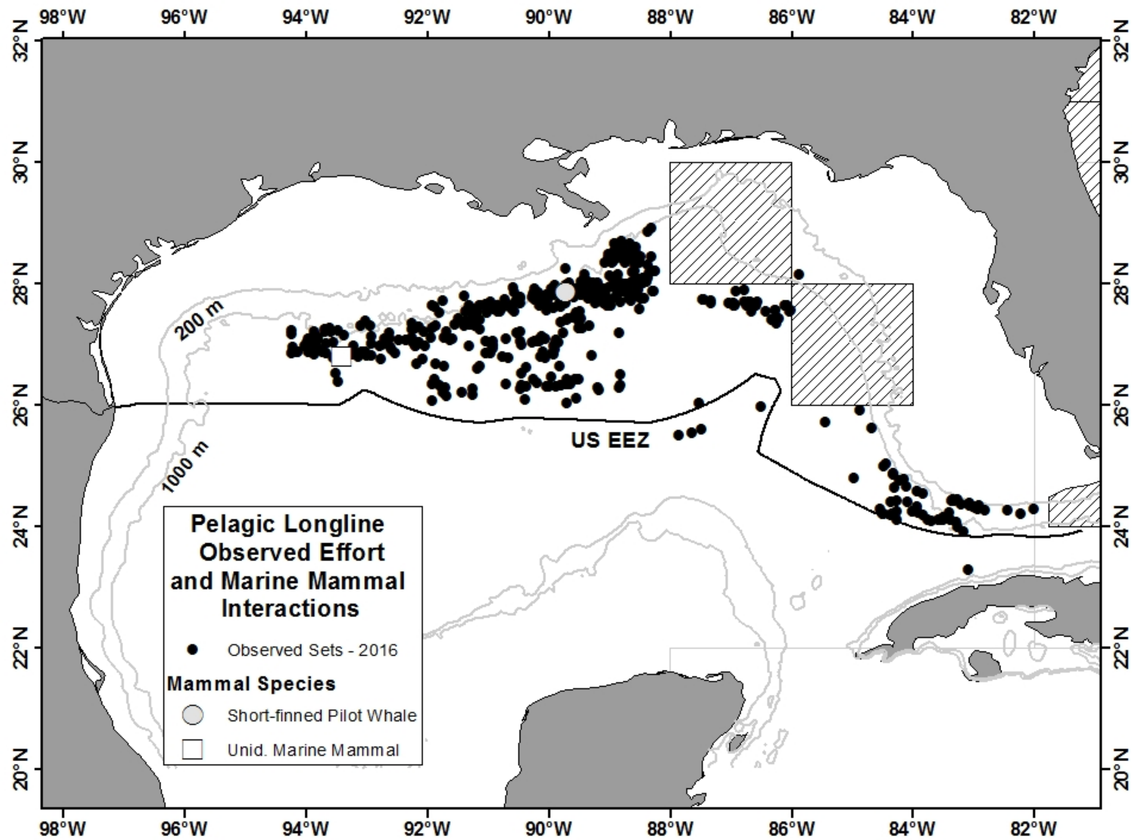


Figure 44. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2017. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.

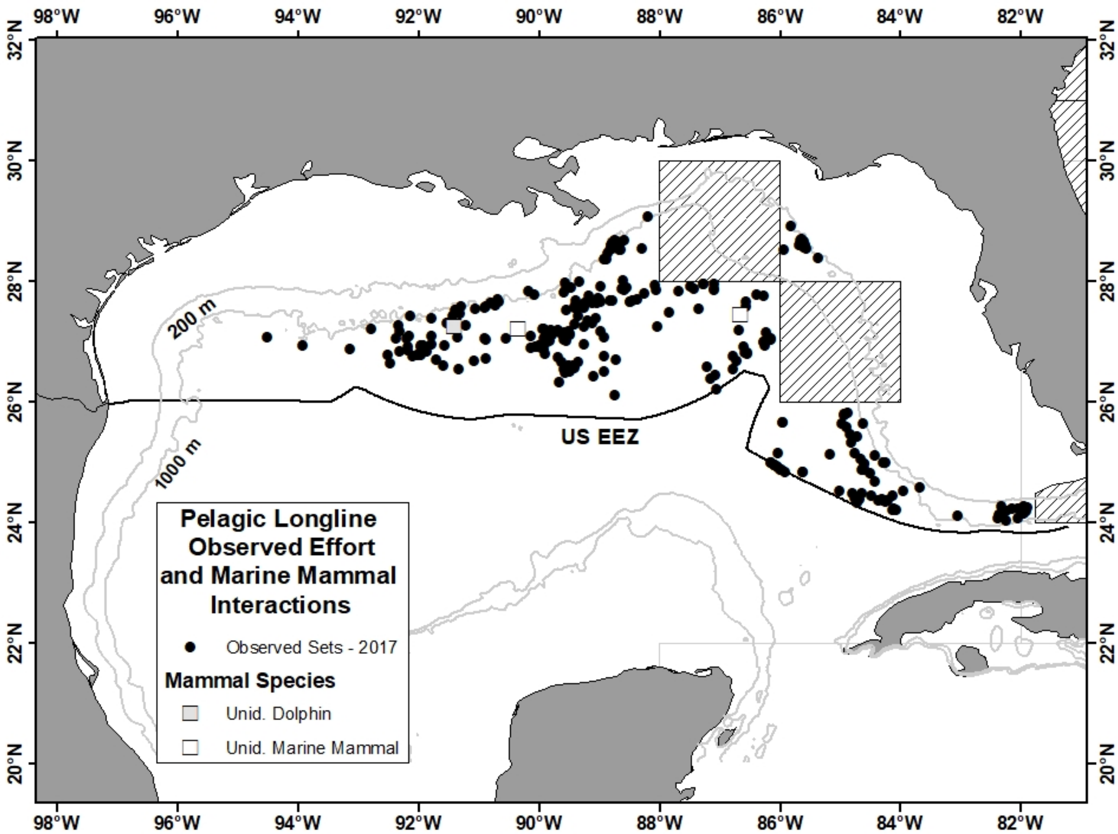
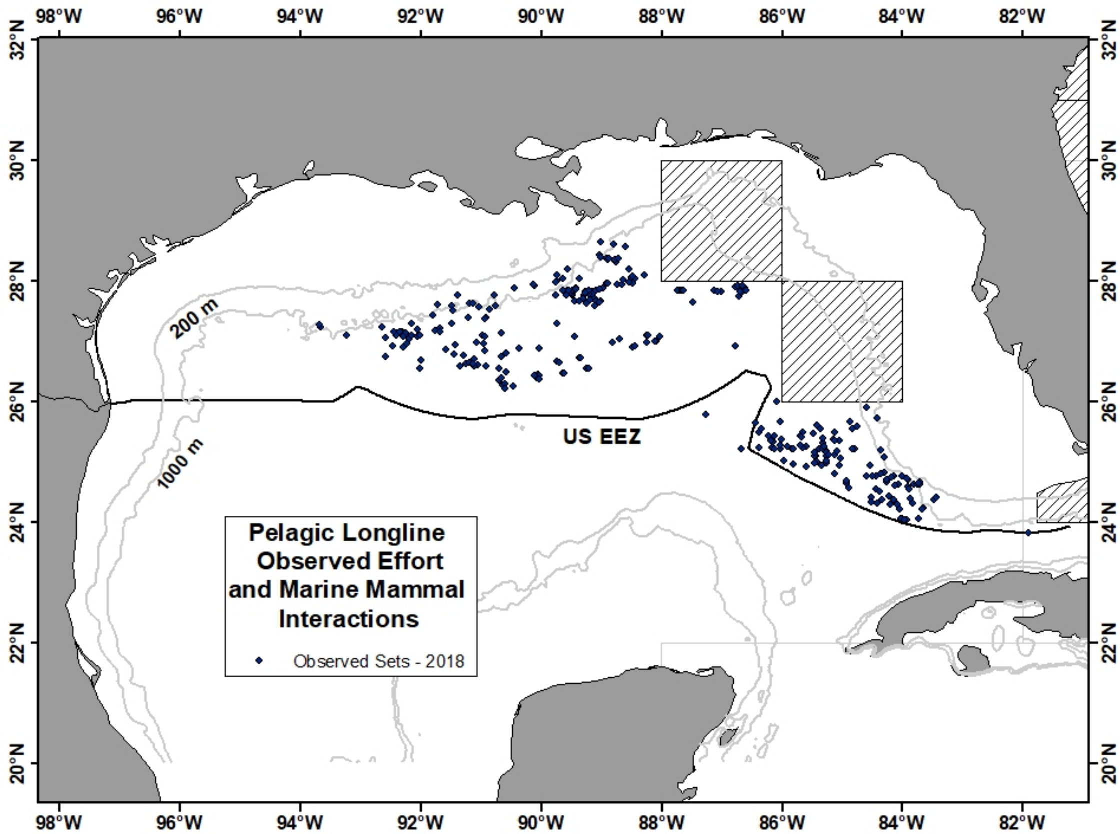


Figure 45. Observed sets in the Pelagic Longline Fishery in the Gulf of Mexico during 2018. Closed areas in the DeSoto Canyon instituted in 2001 are shown as hatched areas.



Appendix IV: Table A. Surveys.

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
1	1982	year-round	Plane	211,585	Cape Hatteras, NC to Nova Scotia, (continental shelf & shelf edge waters)	CETAP	Line transect analyses of distance data	N	CETAP 1982
2	1990	Aug	Ship (Chapman)	2,067	Cape Hatteras, NC to Southern New England (north wall of Gulf Stream)	NEC	One team data analyzed by DISTANCE	N	NMFS 1990
3	1991	Jul–Aug	Ship (Abel-J)	1,962	Gulf of Maine, lower Bay of Fundy, southern Scotian Shelf	NEC	Two independent team data analyzed with modified direct duplicate method	Y	Palka 1995
4	1991	Aug	Boat (Sneak Attack)	640	Inshore bays of Maine	NEC	One team data analyzed by DISTANCE	Y	Palka 1995
5	1991	Aug–Sep	Plane 1 (AT-11)	9,663	Cape Hatteras, NC to Nova Scotia (continental shelf & shelf edge waters)	NEC/SEC	One team data analyzed by DISTANCE	N	NMFS 1991
6	1991	Aug–Sep	Plane 2 (Twin Otter)		Cape Hatteras, NC to Nova Scotia (continental shelf & shelf edge waters)	NEC/SEC	One team data analyzed by DISTANCE	N	NMFS 1991
7	1991	Jun–Jul	Ship (Chapman)	4,032	Cape Hatteras to Georges Bank, (between 200 & 2,000m isobaths)	NEC	One team data analyzed by DISTANCE	N	Waring *et al.* 1992; Waring 1998
8	1992	Jul–Sep	Ship (Abel-J)	3,710	N. Gulf of Maine & lower Bay of Fundy	NEC	Two independent team data analyzed with modified direct duplicate method	Y	Smith *et al.* 1993
9	1993	Jun–Jul	Ship (Delaware II)	1,874	S. edge of Georges Bank, across the Northeast Channel, to the SE edge of the Scotian Shelf	NEC	One team data analyzed by DISTANCE		NMFS 1993
10	1994	Aug–Sep	Ship (Relentless)	534	Georges Bank (shelf edge & slope waters)	NEC	One team data analyzed by DISTANCE	N	NMFS 1994
11	1995	Aug–Sep	Plane (Skymaster)	8,427	Gulf of St. Lawrence	DFO	One team data analyzed using Quenouille's Jackknife Bias Reduction Method that modeled the left truncated sighting curve	N	Kingsley and Reeves 1998

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
12	1995	Jul–Sep	2 Ships (Abel-J & Pelican) & Plane (Twin Otter)	32,600	Virginia to the mouth of the Gulf of St. Lawrence	NEC	Ship: Two independent team data analyzed with modified direct duplicate method. Plane: One team data analyzed by DISTANCE.	Y/N	Palka 1996
13	1996	Jul–Aug	Plane	3,993	Northern Gulf of St. Lawrence	DFO	Quenouille's Jackknife Bias Reduction Method on line-transect methods that modeled the left truncated sighting curve	N	Kingsley and Reeves 1998
14	1998	Jul–Aug	Ship	4,163	South of Maryland	SEC	One team data analyzed by DISTANCE	N	Mullin and Fulling 2003
15	1998	Aug–Sep	Plane		Gulf of St. Lawrence	DFO			Kingsley and Reeves 1998
16	1998	Jul–Sep	Ship (Abel-J) & Plane (Twin Otter)	15,900	North of Maryland	NEC	Ship: Two independent team data analyzed with the modified direct duplicate or Palka & Hammond analysis methods, depending on the presence of responsive movement. Plane: One team data analyzed by DISTANCE.	Y	
17	1999	Jul–Aug	Ship (Abel-J) & Plane (Twin Otter)	6,123	South of Cape Cod to mouth of Gulf of St. Lawrence	NEC	Ship: Two independent team data analyzed with modified direct duplicate or Palka & Hammond analysis methods, depending on the presence of responsive movement. Plane: Circle-back data pooled with aerial data collected in 1999, 2002, 2004, 2006, 2007, and 2008 to calculate pooled g(0)'s and year-species specific abundance estimates for all years except 2008.	Y	
18	2002	Jul–Aug	Plane (Twin Otter)	7,465	Georges Bank to Maine	NEC	Same as for plane in survey 17	Y	Palka 2006
19	2002	Feb–Apr	Ship (Gunter)	4,592	SE US continental shelf - Delaware to Florida	SEC	One team data analyzed by DISTANCE	N	
20	2002	Jun–Jul	Plane	6,734	Florida to New Jersey	SEC	Two independent team data analyzed with modified direct duplicate method	Y	
21	2004	Jun–Aug	Ship (Gunter)	5,659	Florida to Maryland	SEC	Two independent team data analyzed with modified direct duplicate method	Y	Garrison *et al.* 2010
22	2004	Jun–Aug	Ship (Endeavor) & plane (Twin Otter)	10,761	Maryland to Bay of Fundy	NEC	Same methods used in survey 17	Y	Palka 2006
23	2006	Aug	Plane (Twin Otter)	10,676	Georges Bank to Bay of Fundy	NEC	Same as for plane in survey 17	Y	Palka 2005

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
24	2007	Aug	Ship (Bigelow) & Plane (Twin Otter)	8,195	Georges Bank to Bay of Fundy	NEC	Ship: Tracker data analyzed by DISTANCE. Plane: Same as for plane in survey 17	Y	Palka 2005
25	2007	Jul–Aug	Plane	46,804	Nova Scotia to Newfoundland	DFO	Uncorrected counts	N	Lawson and Gosselin 2009
26	2008	Aug	Plane (Twin Otter)	6,267	New York to Maine	NEC	Same as for plane in survey 17	Y	Palka 2005
27	2001	May–Jun	Plane		Maine Coast	NEC, UM	Corrected counts	N	Gilbert *et al.* 2005
28	1999	Mar	Plane		Cape Cod	NEC	Uncorrected counts	N	Barlas 1999
29	1983–1986	1983 (Fall), 1984 (Winter, Spring, Summer), 1985 (Summer, Fall), 1986 (Winter)	Plane (Beechcraft D-18S, modified with a bubblenose)	103,490	Northern Gulf of Mexico bays & sounds (coastal waters from shoreline to 18m isobath, & OCS waters from 18m isobath to 9.3km past the 18m isobath)	SEC	One team data analyzed with line-transect theory	N	Scott *et al.* 1989
30	1991–1994	Apr–Jun	Ship (Oregon II)	22,041	Northern Gulf of Mexico (from 200m to U.S. EEZ)	SEC	One team data analyzed by DISTANCE	N	Hansen *et al.* 1995
31	1992–1993	Sep–Oct	Plane (Twin Otter)		Northern Gulf of Mexico bays & sounds (coastal waters from shoreline to 18m isobath, & OCS waters from 18m isobath to 9.3km past the 18m isobath)	GOMEX92, GOMEX93	One team data analyzed by DISTANCE	N	Blaylock and Hoggard 1994
33	1996–1997, 1999–2001	Apr–Jun	Ship (Oregon II & Gunter)	12,162	Northern Gulf of Mexico (from 200m to U.S. EEZ)	SEC	One team data analyzed by DISTANCE	N	Mullin and Fulling 2004
34	1998–2001	End of Aug–Early Oct	Ship (Gunter & Oregon II)	2,196	Northern Gulf of Mexico (OCS waters from 20–200 m)	SEC	One team data analyzed by DISTANCE	N	Fulling *et al.* 2003
36	2004	12Jan–13 Jan	Helicopter		Sable Island	DFO	Pup count	na	Bowen *et al.* 2007
37	2004		Plane		Gulf of St Lawrence & Nova Scotia Eastern Shore	DFO	Pup count	na	Hammill 2005
38	2009	10Jun–13Aug	Ship	4,600	Northern Gulf of Mexico (from 200m to U.S. EEZ)	SEC	One team data analyzed by DISTANCE		

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
39	2007	17Jul–08Aug	Plane		Northern Gulf of Mexico (from shore to 200m, majority of effort 0–20m)	SEC	One team data analyzed by DISTANCE		
40	2011	04Jun–01Aug	Ship (Bigelow)	3,107	Virginia to Massachusetts (waters that were deeper than the 100m depth contour out to beyond the US EEZ)	NEC	Two-independent teams, both using big-eyes. Analyzed using DISTANCE, the independent observer option assuming point independence	Y	Palka 2012
41	2011	07Aug–26Aug	Plane (Twin Otter)	5,313	Massachusetts to New Brunswick, Canada (waters north of New Jersey & shallower than the 100m depth contour, through the US & Canadian Gulf of Maine & up to & including the lower Bay of Fundy)	NEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	Palka 2012
42	2011	19Jun–01Aug	Ship (Gunter)	4,445	Florida to Virginia	SEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	Garrison 2016
43	2012	May–Jun	Plane		Maine Coast	NEC	Corrected counts	N	Waring *et al.* 2015
44	1992	Jan–Feb	Ship (Oregon II)	3,464	Cape Canaveral to Cape Hatteras, US EEZ	SEC		N	NMFS 1992
45	2010	24Jul–14Aug	Plane	7,944	Southeastern Florida to Cape May, New Jersey	SEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.		
46	2011	06Jul–29Jul	Plane	8,665	Southeastern Florida to Cape May, New Jersey	SEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence.		Garrison 2016
47	2016	27Jun–25Aug	Ship & Plane	5,354	Central Virginia to the lower Bay of Fundy	NEC	Two-independent teams. Analyzed using DISTANCE, the independent observer option assuming point independence.		Palka 2020

Survey Number	Year(s)	Time of Year	Platform	Track Line Length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference(s)
48	2016	30Jun–19Aug	Ship & Plane	4,399	Central Florida to Virginia	SEC	Two-independent teams. Analyzed using DISTANCE, the independent observer option assuming point independence.		Garrison 2020
49	2016	Aug & Sep	Plane	50,160	Gulf of St. Lawrence, Bay of Fundy, Scotian Shelf, Newfoundland, Labrador	DFO	NAISS		Lawson and Gosselin 2018
50	2017, 2018	02Jul–25Aug 2017, 11Aug–06Oct 2018	Ship (Gunter)	13,775	Northern Gulf of Mexico (waters from 200m to U.S. EEZ)	SEC	Two-independent teams. Analyzed using DISTANCE, the independent observer option assuming point independence.	Y	Garrison *et al.* 2020

Appendix IV: Table B. Abundance Estimates.

"Survey Number" refers to surveys described in Table A. "Best" estimate for each species is in bold font.

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Humpback Whale	Gulf of Maine	1992	501			Minimum population size estimated from photo-ID data
		1993	652	0.29		YONAH sampling (Clapham *et al.* 2003)
		1997	497			Minimum population size estimated from photo-ID data
		1999	902	0.45	17	
		2002	521	0.67	18	Palka 2006
		2004	359	0.75	22	Palka 2006
		2006	847	0.55	23	Palka 2005
		2008	823			Mark-recapture estimate (Robbins 2010)
		2011	335	0.42	40+41	Palka 2012
		2015	896			Minimum population size estimated from photo-ID data
		2016	2,368			
		2016	**1,396**	**na**		State-space mark-recapture (Pace 2017)
Fin Whale	Western North Atlantic	1995	2,200	0.24	12	Palka 1996
		1999	2,814	0.21	18	Palka 2006
		2002	2,933	0.49	18	Palka 2006
		2004	1,925	0.55	22	Palka 2006
		2006	2,269	0.37	23	Palka 2005
		2007	3,522	0.27	25	Lawson and Gosselin 2009
		2011	1,595	0.33	40+41	Palka 2012
		2011	23	0.87	42	
		2011	1,618	0.33	40+41+42	Estimate summed from north and south surveys
		2016	3,006	0.40	47+48	Garrison 2020; Palka 2020
		2016	2,235	0.41	49	Bay of Fundy/Scotian Shelf (Lawson and Gosselin 2018)
		2016	2,177	0.47	49	Newfoundland/Labrador (Lawson and Gosselin 2018)
		2016	**7,418**	**0.25**	**47+48+49**	
Sei Whale	Nova Scotia Stock	1977	1,393–2,248			Based on tag-recapture data (Mitchell and Chapman 1977)
		1977	870			Based on census data (Mitchell and Chapman 1977)
		1982	280		1	CETAP 1982
		2002	71	1.01	18	Palka 2006
		2004	386	0.85	22	Palka 2006
		2006	207	0.62	23	Palka 2005
		2011	357	0.52	40+41	Palka 2012
		2010–2013	**6,292**	**1.02**		Springtime average abundance estimate generated from spatially- and temporally-explicit density models derived from visual two-team abundance survey data collected between 2010 and 2013 (Palka *et al.* 2017)
		1999–2013	627	0.14		Spring habitat-based density estimates (Roberts *et al.* 2016)
		1995–2013	717	0.30		Summer habitat-based density estimates (Roberts *et al.* 2016)
		2016	28	0.55	47	Palka 2016
		1982	320	0.23	1	CETAP 1982
		1992	2,650	0.31	3+8	
		1993	330	0.66	9	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Minke Whale	Canadian East Coast	1995	2,790	0.32	12	Palka 1996
		1995	1,020	0.27	11	
		1996	620	0.52	13	
		1999	2,998	0.19	17	
		2002	756	0.9	18	Palka 2006
		2004	600	0.61	22	Palka 2006
		2006	3,312	0.74	23	
		2007	20,741	0.3	25	Lawson and Gosselin 2009
		2011	2,591	0.81	40+41	Palka 2012
		2016	5,036	0.68	47	Palka 2020
		2016	6,158	0.40	49	Bay of Fundy/Scotian Shelf (Lawson and Gosselin 2018)
		2016	13,008	0.46	49	Newfoundland/Labrador (Lawson and Gosselin 2018)
		2016	**24,202**	**0.30**	**47+49**	
Sperm Whale	North Atlantic	1982	219	0.36	1	CETAP 1982
		1990	338	0.31	2	
		1991	736	0.33	7	Waring *et al.*1992, Warring 1998
		1991	705	0.66	6	
		1991	337	0.5	5	
		1993	116	0.4	9	
		1994	623	0.52	10	
		1995	2,698	0.67	12	Palka 1996
		1998	2,848	0.49	16	
		1998	1,181	0.51	14	Mullin and Fulling 2003
		2004	2,607	0.57	22	Palka 2006
		2004	2,197	0.47	21	Garrison *et al.* 2010
		2004	4,804	0.38	21+22	Estimate summed from north and south surveys
		2011	1,593	0.36	40+41	Palka 2012
		2011	695	0.39	42	
		2011	2,288	0.28	40+41+42	Estimate summed from north and south surveys
		2016	3,321	0.35	47	Palka 2020
		2016	1,028	0.35	48	Garrison 2020
		2016	**4,349**	**0.28**	**47+48**	Estimate summed from north and south surveys
***Kogia* spp.**	Western North Atlantic	1998	115	0.61	16	
		1998	580	0.57	14	Mullin and Fulling 2003
		2004	358	0.44	22	Palka 2006
		2004	37	0.75	21	Garrison *et al.* 2010
		2004	395	0.4	21+22	Estimate summed from north and south surveys
		2011	1,783	0.62	40+41	Palka 2012
		2011	2,002	0.69	42	
		2011	3,785	0.47	40+41+42	Estimate summed from north and south surveys
		2016	4,548	0.49	47	Palka 2020
		2016	3,202	0.59	48	Garrison 2020
		2016	**7,750**	**0.38**	**47+48**	Estimate summed from north and south surveys
		1982	120	0.71	1	CETAP 1982
		1990	442	0.51	2	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Beaked Whales	Western North Atlantic	1991	262	0.99	7	Waring *et al.*1992, Warring 1998
		1991	370	0.65	6	
		1991	612	0.73	5	
		1993	330	0.66	9	
		1994	99	0.64	10	
		1995	1,519	0.69	12	Palka 1996
		1998	2,600	0.4	16	
		1998	541	0.55	14	Mullin and Fulling 2003
		2004	2,839	0.78	22	Palka 2006
		2004	674	0.36	21	Garrison *et al.* 2010
		2004	3,513	0.63	21+22	Estimate summed from north and south surveys
		2006	922	1.47	23	
		2011	5,500	0.67	40+41	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*; Palka 2012)
		2011	1,592	0.67	42	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*)
		2011	**7,092**	**0.54**	**40+41+42**	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*); Estimate summed from north and south surveys
		2016	6,760	0.37	47	Palka 2020
		2016	3,347	0.29	48	Garrison 2020
		2016	**10,107**	**0.27**	**47+48**	Estimate summed from north and south surveys
Cuvier's Beaked Whale	Western North Atlantic	2011	4,962	0.37	40+41	Palka 2012
		2011	1,570	0.65	42	
		2011	6,532	0.32	40+41+42	Estimate summed from north and south surveys
		2016	3,897	0.47	47	Palka 2020
		2016	1,847	0.49	48	Garrison 2020
		2016	**5,744**	**0.36**	**47+48**	Estimate summed from north and south surveys
Risso's Dolphin	Western North Atlantic	1982	4,980	0.34	1	CETAP 1982
		1991	11,017	0.58	7	Waring *et al.*1992, Warring 1998
		1991	6,496	0.74	5	
		1991	16,818	0.52	6	
		1993	212	0.62	9	
		1995	5,587	1.16	12	Palka 1996
		1998	18,631	0.35	17	
		1998	9,533	0.5	15	
		1998	28,164	0.29	15+17	Estimate summed from north and south surveys
		2002	69,311	0.76	18	Palka 2006
		2004	15,053	0.78	21	Garrison *et al.* 2010
		2004	5,426	0.54	22	Palka 2006

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		2004	20,479	0.59	21+22	Estimate summed from north and south surveys
		2006	14,408	0.38	23	
		2011	15,197	0.55	40+41	Palka 2012
		2011	3,053	0.44	42	
		2011	18,250	0.46	40+41+42	Estimate summed from north and south surveys
		2016	7,245	0.44	48	Garrison 2020
		2016	22,175	0.23	47	Palka 2020
		2016	6,073	0.45	49	Lawson and Gosselin 2018
		2016	**35,493**	**0.19**	**47+48+49**	
Pilot Whale	Western North Atlantic	1951	50,000			Derived from catch data from 1951–1961 drive fishery (Mitchell 1974)
		1975	43,000–96,000			Derived from population models (Mercer 1975)
		1982	11,120	0.29	1	CETAP 1982
		1991	3,636	0.36	7	Waring *et al.*1992, Warring 1998
		1991	3,368	0.28	5	
		1991	5,377	0.53	6	
		1993	668	0.55	9	
		1995	8,176	0.65	12	Palka 1996
		1995	9,776	0.55	12+16	Sum of US (#12) and Canadian (#16) surveys
		1998	1,600	0.65	16	
		1998	9,800	0.34	17	
		1998	5,109	0.41	15	
		2002	5,408	0.56	18	Palka 2006
		2004	15,728	0.34	22	Palka 2006
		2004	15,411	0.43	21	Garrison *et al.* 2010
		2004	31,139	0.27	21+22	Estimate summed from north and south surveys
		2006	26,535	0.35	23	Estimate summed from north and south surveys
		2007	16,058	0.79	25	Long-finned pilot whales (Lawson and Gosselin 2009)
		2011	5,636	0.63	40+41	Long-finned pilot whales
		2011	11,865	0.57	40+41	Unidentified pilot whales
		2011	4,569	0.57	40+41	Short-finned pilot whales
		2011	16,946	0.43	42	Short-finned pilot whales
		2011	21,515	0.37	40+41+42	Best estimate for short-finned pilot whales alone; Estimate summed from north and south surveys
		2016	3,810	0.42	47	Short-finned pilot whales (Garrison and Palka 2018)
		2016	25,114	0.27	48	Short-finned pilot whales (Garrison and Palka 2018)
		2016	**28,924**	**0.24**	**47+48**	Best estimate for short-finned pilot whales alone; Estimate summed from north and south surveys
		2016	10,997	0.51	47	Long-finned pilot whales (Garrison 2020; Palka 2020)
		2016	28,218	0.36	48	Long-finned pilot whales (Garrison 2020; Palka 2020)
		2016	**39,215**	**0.30**	**47+48**	Best estimate for long-finned pilot whales alone; Estimate summed from north and south surveys
		1982	28,600	0.21	1	
		1992	20,400	0.63	2+7	
		1993	729	0.47	9	
		1995	27,200	0.43	12	Palka 1996

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Atlantic White-sided Dolphin	Western North Atlantic	1995	11,750	0.47	11	
		1996	560	0.89	13	
		1999	51,640	0.38	17	
		2002	109,141	0.3	18	Palka 2006
		2004	2,330	0.8	22	Palka 2006
		2006	17,594	0.3	23	
		2006	63,368	0.27	(18+23)/2	Average of #18 and #23
		2007	5,796	0.43	25	Lawson and Gosselin 2009
		2011	48,819	0.61	40+41	Palka 2012
		2016	31,912	0.61	47	Palka 2020
		2016	61,321	1.04	49	Canadian part of Gulf of Maine and all of Gulf of St. Lawrence population (Lawson and Gosselin 2018)
		2016	**93,233**	**0.71**	**47+49**	
White-beaked Dolphin	Western North Atlantic	1982	573	0.69	1	CETAP 1982
			5,500			Alling and Whitehead 1987
		1982	3,486	0.22		Alling and Whitehead 1987
		2006	2,003	0.94	23	
		2007	11,842		25	
		2008			26	
		2016	**536,016**	**0.31**	**49**	Lawson and Gosselin 2018
Common Dolphin	Western North Atlantic	1982	29,610	0.39	1	
		1991	22,215	0.4	7	Waring *et al.*1992; Warring 1998
		1993	1,645	0.47	9	
		1995	6,741	0.69	12	Palka 1996
		1998	30,768	0.32	17	
		1998	0		15	
		2002	6,460	0.74	18	
		2004	90,547	0.24	22	Palka 2006
		2004	30,196	0.54	21	Garrison *et al.* 2010
		2004	120,743	0.23	21+22	Estimate summed from north and south surveys
		2006	84,000	0.36	24	
		2007	173,486	0.55	25	Lawson and Gosselin 2009
		2011	67,191	0.29	40+41	Palka 2012
		2011	2,993	0.87	42	
		2011	70,184	0.28	40+41+42	Estimate summed from north and south surveys
		2016	80,227	0.31	47	Palka 2020
		2016	900	0.57	48	Garrison 2020
		2016	48,574	0.48	49	Newfoundland/Labrador (Lawson and Gosselin 2018)
		2016	43,124	0.28	49	Bay of Fundy/Scotian Shelf (Lawson and Gosselin 2018)
		2016	**172,825**	**0.21**	**47+48+49**	Estimate summed from north, south and Canadian surveys
		1982	6,107	0.27	1	CETAP 1982
		1995	4,772	1.27	12	Palka 1996
		1998	32,043	1.39	16	
		1998	14,438	0.63	14	Mullin and Fulling 2003
		2004	3,578	0.48	22	Palka 2006
		2004	47,400	0.45	21	Garrison *et al.* 2010

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Atlantic Spotted Dolphin	Western North Atlantic	2004	50,978	0.42	21+22	Estimate summed from north and south surveys
		2011	26,798	0.66	40+41	Palka 2012
		2011	17,917	0.42	42	
		2011	44,715	0.43	40+41+42	Estimate summed from north and south surveys
		2016	8,247	0.24	47	Palka 2020
		2016	31,674	0.33	48	Garrison 2020
		2016	**39,921**	**0.27**	**47+48**	Estimate summed from north and south surveys
Pantropical Spotted Dolphin	Western North Atlantic	1982	6,107	0.27	1	CETAP 1982
		1995	4,772	1.27	12	Palka 1996
		1998	343	1.03	16	
		1998	12,747	0.56	14	Mullin and Fulling 2003
		2004	0		22	Palka 2006
		2004	4,439	0.49	21	Garrison *et al.* 2010
		2004	4,439	0.49	21+22	Estimate summed from north and south surveys
		2011	0	0	40+41	Palka 2012
		2011	3,333	0.91	42	
		2011	3,333	0.91	40+41+42	Estimate summed from north and south surveys
		2016	0	-	47	Palka 2020
		2016	6,593	0.52	48	Garrison 2020
		2016	**6,593**	**0.52**	**47+48**	Estimate summed from north and south surveys
Striped Dolphin	Western North Atlantic	1982	36,780	0.27	1	
		1995	31,669	0.73	12	Palka 1996
		1998	39,720	0.45	16	
		1998	10,225	0.91	14	Mullin and Fulling 2003
		2004	52,055	0.57	22	
		2004	42,407	0.53	21	Garrison *et al.* 2010
		2004	94,462	0.4	21+22	Estimate summed from north and south surveys
		2011	46,882	0.33	40+41	Palka 2012
		2011	7,925	0.66	42	
		2011	54,807	0.3	40+41+42	Estimate summed from north and south surveys
		2016	42,783	0.25	47	Palka 2020
		2016	24,163	0.66	48	Garrison 2020
		2016	**67,036**	**0.29**	**47+48**	Estimate summed from north and south surveys
Rough-toothed Dolphin	Western North Atlantic	2011	0	0	40+41	Palka 2012
		2011	271	1	42	
		2011	**271**	**1**	**40+41+42**	Estimate summed from north and south surveys
Bottlenose Dolphin	Western North Atlantic: Offshore	1998	16,689	0.32	16	
		1998	13,085	0.4	14	Mullin and Fulling 2003
		2002	26,849	0.19	20	
		2002	5,100	0.41	18	Palka 2006
		2004	9,786	0.56	22	Palka 2006
		2004	44,953	0.26	21	Garrison *et al.* 2010
		2006	2,989	1.11	23	
		2011	26,766	0.52	40+41	Palka 2012
		2011	50,766	0.55	42	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		2011	**77,532**	**0.4**	**40+41+42**	Estimate summed from north and south surveys
		2016	17,958	0.33	47	Palka 2020
		2016	44,893	0.29	48	Garrison 2020
		2016	**62,851**	**0.23**	**47+48**	Estimate summed from north and south surveys
Harbor Porpoise	Gulf of Maine, Bay of Fundy	1991	37,500	0.29	3	Palka 1995
		1992	67,500	0.23	8	Smith *et al.* 1993
		1995	74,000	0.2	12	Palka 1996
		1995	12,100	0.26	11	
		1996	21,700	0.38	14	Mullin and Fulling 2003
		1999	89,700	0.22	17	Survey discovered portions of the range not previously surveyed (Palka 2006)
		2002	64,047	0.48	21	Palka 2006
		2004	51,520	0.65	23	Palka 2006
		2006	89,054	0.47	24	
		2007	4,862	0.31	25	Lawson and Gosselin 2009
		2011	79,883	0.32	40+41	Palka 2012
		2016	75,079	0.38	47	Palka 2020
		2016	20,464	0.39	48	Garrison 2020
		2016	**95,543**	**0.31**	**47+48**	Estimate summed from north and south surveys
Harbor Seal	Western North Atlantic	2001	99,340	0.097	27	Gilbert *et al.* 2005
		2012	**75,834**	**0.15**	**43**	Waring *et al.* 2015
Gray Seal	Western North Atlantic	1999	5,611		28	Barlas 1999
		2001	1,731		27	Gilbert *et al.* 2005
		2004	52,500	0.15	37	Gulf of St Lawrence and Nova Scotia Eastern Shore
		2004	208,720–223,220	0.08–0.14	36	Sable Island
		2012	331,000	95%CI= 263,000-458,000		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2013)
		2014	505,000	95%CI= 329,000–682,000		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2014)
		2016	424,300	95%CI= 263,600–578,300		Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island (DFO 2017)
		2016	**27,131**	**95%CI= 18,768–39,221**		Derived from total population size to pup ratios in Canada applied to U.S. pup counts
Bryde's Whale	Northern Gulf of Mexico	1991–1994	35	1.1	30	Hansen *et al.* 1995
		1996–2001	40	0.61	33	Mullin and Fulling 2004
		2003–2004	15	1.98	35	
		2009	33	1.07	38	
		2017–2018	**51**	**0.50**	**50**	Garrison *et al.* 2020
		1991–1994	530	0.31	30	Hansen *et al.* 1995
		1996–2001	1,349	0.23	33	Mullin and Fulling 2004

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Sperm Whale	Northern Gulf of Mexico	2003–2004	1,665	0.2	35	
		2009	763	0.38	38	
		2017–2018	**1,307**	**0.33**	**50**	Garrison *et al.* 2020
***Kogia* spp.**	Northern Gulf of Mexico	1991–1994	547	0.28	30	Hansen *et al.* 1995
		1996–2001	742	0.29	33	Mullin and Fulling 2004
		2003–2004	453	0.35	35	
		2009	186	1.04	38	
		2017–2018	**336**	**0.35**	**50**	Garrison *et al.* 2020
Cuvier's Beaked Whale	Northern Gulf of Mexico	1991–1994	30	0.5	30	Hansen *et al.* 1995
		1996–2001	95	0.47	33	Mullin and Fulling 2004
		2003–2004	65	0.67	35	
		2009	74	1.04	38	
		2017–2018	**18**	**0.75**	**50**	Garrison *et al.* 2020
***Mesoplodon* spp.**	Northern Gulf of Mexico	1996–2001	106	0.41	33	Mullin and Fulling 2004
		2003–2004	57	1.4	35	
		2009	149	0.91	38	
		2017–2018	**98**	**0.46**	**50**	Garrison *et al.* 2020
Killer Whale	Northern Gulf of Mexico	1991–1994	277	0.42	30	Hansen *et al.* 1995
		1996–2001	133	0.49	33	Mullin and Fulling 2004
		2003–2004	49	0.77	35	
		2009	28	1.02	38	
		2017–2018	**267**	**0.75**	**50**	Garrison *et al.* 2020
False Killer Whale	Northern Gulf of Mexico	1991–1994	381	0.62	30	Hansen *et al.* 1995
		1996–2001	1,038	0.71	33	Mullin and Fulling 2004
		2003–2004	777	0.56	35	
		2017–2018	**494**	**0.79**	**50**	Garrison *et al.* 2020
Short-finned Pilot Whale	Northern Gulf of Mexico	1991–1994	353	0.89	30	Hansen *et al.* 1995
		1996–2001	2,388	0.48	33	Mullin and Fulling 2004
		2003–2004	716	0.34	35	
		2009	2,415	0.66	38	
		2017–2018	**1,321**	**0.43**	**50**	Garrison *et al.* 2020
Melon-headed Whale	Northern Gulf of Mexico	1991–1994	3,965	0.39	30	Hansen *et al.* 1995
		1996–2001	3,451	0.55	33	
		2003–2004	2,283	0.76	35	
		2009	2,235	0.75	38	
		2017–2018	**1,749**	**0.68**	**50**	Garrison *et al.* 2020

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Pygmy Killer Whale	Northern Gulf of Mexico	1991–1994	518	0.81	30	Hansen *et al.* 1995
		1996–2001	408	0.6	33	Mullin and Fulling 2004
		2003–2004	323	0.6	35	
		2009	152	1.02	38	
		2017–2018	**613**	**1.15**	**50**	Garrison *et al.* 2020
Risso's Dolphin	Northern Gulf of Mexico	1991–1994	2,749	0.27	30	Hansen *et al.* 1995
		1996–2001	2,169	0.32	33	Mullin and Fulling 2004
		2003–2004	1,589	0.27	35	
		2009	2,442	0.57	38	
		2017–2018	**1,974**	**0.46**	**50**	Garrison *et al.* 2020
Pantropical Spotted Dolphin	Northern Gulf of Mexico	1991–1994	31,320	0.2	30	Hansen *et al.* 1995
		1996–2001	91,321	0.16	33	Mullin and Fulling 2004
		2003–2004	34,067	0.18	35	
		2009	50,880	0.27	38	
		2017–2018	**37,195**	**0.24**	**50**	Garrison *et al.* 2020
Striped Dolphin	Northern Gulf of Mexico	1991–1994	4,858	0.44	30	Hansen *et al.* 1995
		1996–2001	6,505	0.43	33	Mullin and Fulling 2004
		2003–2004	3,325	0.48	35	
		2009	1,849	0.77	38	
		2017–2018	**1,817**	**0.56**	**50**	Garrison *et al.* 2020
Spinner Dolphin	Northern Gulf of Mexico	1991–1994	6,316	0.43	30	Hansen *et al.* 1995
		1996–2001	11,971	0.71	33	Mullin and Fulling 2004
		2003–2004	1,989	0.48	35	
		2009	11,441	0.83	38	
		2017–2018	**2,991**	**0.54**	**50**	Garrison *et al.* 2020
Clymene Dolphin	Northern Gulf of Mexico	1991–1994	5,571	0.37	30	Hansen *et al.* 1995
		1996–2001	17,355	0.65	33	Mullin and Fulling 2004
		2003–2004	6,575	0.36	35	
		2009	129	1	38	
		2017–2018	**513**	**1.03**	**50**	Garrison *et al.* 2020
Atlantic Spotted Dolphin	Northern Gulf of Mexico	Oceanic (1991–1994)	3,213	0.44	30	Hansen *et al.* 1995
		Oceanic (1996–2001)	175	0.84	33	Mullin and Fulling 2004
		OCS (1998–2001)	37,611	0.28	34	Abundance estimate is from 2000-2001 surveys only (from Fulling *et al.* 2003). Current best population size estimate is unknown because data from the continental shelf portion of this species' range are more than 8 years old.
		Oceanic (2003–2004)	0	-	35	
		2009	2968	0.67	38	
Fraser's	Northern Gulf of Mexico	1991–1994	127	0.9	30	Hansen *et al.* 1995
		1996–2001	726	0.7	33	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Dolphin		2003–2004	0	-	35	
		2009	0	-	38	
		2017–2018	**213**	**1.03**	**50**	Garrison *et al.* 2020
Rough-toothed Dolphin	Northern Gulf of Mexico	Oceanic (1991–1994)	852	0.31	30	
		Oceanic (1996–2001)	985	0.44	33	Mullin and Fulling 2004
		OCS (1998–2001)	1,145	0.83	34	Abundance estimate is from 2000-2001 surveys only (from Fulling *et al.* 2003). Current best population size estimate is unknown because data from the continental shelf portion of this species' range are more than 8 years old.
		Oceanic (2003–2004)	1,508	0.39	35	
		2009	624	0.99	38	
Bottlenose Dolphin	Northern Gulf of Mexico: Oceanic	1996–2001	2,239	0.41	33	Mullin and Fulling 2004
		2003–2004	3,708	0.42	35	
		2009	5,806	0.39	38	
		2017–2018	**213**	**1.03**	**50**	Garrison *et al.* 2020
Bottlenose Dolphin	Northern Gulf of Mexico: Continental Shelf	1998–2001	17,777	0.32	34	Abundance estimate is from 2000-2001 surveys only (from Fulling *et al.* 2003). Current best population size estimate is unknown because data from the continental shelf are more than 8 years old.
Bottlenose Dolphin	Northern Gulf of Mexico: Coastal (3 stocks)	Eastern (1994)	9,912	0.12	32	
		Eastern (2007)	**7,702**	**0.19**	**39**	
		Northern (1993)	4,191	0.21	31	Current best population size estimate for this stock is unknown because data are more than 8 years old (Blaylock and Hoggard 1994)
		Northern (2007)	**2,473**	**0.25**	**39**	
		Western (1992)	3,499	0.21	31	Current best population size estimate for this stock is unknown because data are more than 8 years old (Blaylock and Hoggard 1994)
		Choctawhatchee Bay (2007)	179	0.04		Conn *et al.* 2011
		St. Joseph Bay (2011)	142	0.17		Balmer *et al.* 2018
		St. Vincent Sound, Apalachicola Bay, St. George Sound (2008)	439	0.14		Tyson *et al.* 2011
		Sarasota Bay, Little Sarasota Bay (2015)	158	0.27		Tyson and Wells 2016
		Mississippi River Delta (2011–2012)	332	0.93		

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Bottlenose Dolphin	Northern Gulf of Mexico: Bay, Sound and Estuarine (31 stocks)	Mississippi Sound, Lake Borgne, Bay Boudreau	901	0.63		
		Mississippi Sound, Lake Borgne, Bay Boudreau	**3,046**	**0.06**		**Mullin *et al.* 2017**
		Barataria Bay	**2,306**	**0.09**		**McDonald *et al.* 2017**
		Pine Island Sound, Charlotte Harbor, Gasparilla Sound, Lemon Bay (2006)	826	0.09		Bassos-Hull *et al.* 2013
		Laguna Madre	80	1.57		
		Neuces Bay, Corpus Christi Bay	58	0.61		
		Copano Bay, Aransas Bay, San Antonio Bay, Redfish Bay, Espiritu Santo Bay	55	0.82		
		Matagorda Bay, Tres Palacios Bay, Lavaca Bay[j]	61	0.45		
		West Bay	48	0.03		Litz *et al.* 2019
		Galveston Bay, East Bay, Trinity Bay[j]	152	0.43		
		Terrebonne Bay, Timbalier Bay	3,870	0.15		***Litz*** *et al.* ***2018***
		Mobile Bay, Bonsecour Bay	122	0.34		
		Pensacola Bay, East Bay	33	0.80		
		St. Andrew Bay	199	0.09		***Balmer*** *et al.* ***2019***
		Apalachee Bay	491	0.39		
		Remaining 11 stocks	unknown	undetermined	31	Blaylock and Hoggard 1994; Current best population size estimate for each of these 11 stocks is unknown because data are more than 8 years old

REFERENCES CITED

Alling, A.K. and H.P. Whitehead 1987. A preliminary study of the status of white-beaked dolphins, *Lagenorhynchus albirostris*, and other small cetaceans off the coast of Labrador. Can. Field-Nat. 101(1): 131–135.

Balmer, B.C., T. McDonald, F. Hornsby, J. Adams, J. Allen, A. Barleycorn, P. Clarke, C. Cush, A. Honaker, K. McHugh, T. Speakman, R. Wells and L. Schwacke. 2018. Long-term trends in a northern Gulf of Mexico common bottlenose dolphin (*Tursiops truncatus*) population in the wake of the *Deepwater Horizon* oil spill. J. Cetacean Res. Manage. 18:1–9.

Balmer, B., S. Watwood, B. Quigley, T. Speakman, K. Barry, K. Mullin, P. Rosel, C. Sinclair, E. Zolman and L. Schwacke. 2019. Common bottlenose dolphin (*Tursiops truncatus*) abundance and distribution patterns in St. Andrew Bay, Florida, USA. Aquat. Conserv.: Mar. Freshw. Ecosyst. 2019:1–13.

Barlas, M.E. 1999. The distribution and abundance of harbor seals (*Phoca vitulina concolor*) and gray seals (*Halichoerus grypus*) in southern New England, winter 1998-summer 1999. M.A. thesis. Graduate School of Arts and Sciences Boston University, Boston, MA. 52 pp.

Bassos-Hull, K., R.M. Perrtree, C.C. Shepard, S. Schilling, A.A. Barleycorn, J.B. Allen, B.C. Balmer, W.E. Pine and R.S. Wells. 2013. Long-term site fidelity and seasonal abundance estimates of common bottlenose dolphins (*Tursiops truncatus*) along the southwest coast of Florida and responses to natural perturbations. J. Cetacean Res. Manage. 13(1): 19–30.

Blaylock, R.A. and W. Hoggard 1994. Preliminary estimates of bottlenose dolphin abundance in southern U.S. Atlantic and Gulf of Mexico continental shelf waters. NOAA Tech. Memo. NMFS-SEFSC-356. 10 pp.

Bowen, W.D., J.I. McMillan and W. Blanchard 2007. Reduced population growth of gray seals at Sable Island: Evidence from pup production and age of primiparity. Mar. Mamm. Sci. 23: 48–64.

CETAP 1982. A characterization of marine mammals and turtles in the mid- and north Atlantic areas of the U.S. outer continental shelf, final report. Washington, DC, Bureau of Land Management. #AA551-CT8-48 538 pp.

Clapham, P.J., J. Barlow, T. Cole, D. Mattila, R. Pace, D. Palka, J. Robbins and R. Seton 2003. Stock definition, abundance and demographic parameters of humpback whales from the Gulf of Maine. J. Cetacean Res. Manage. 5(13–22).

Conn, P.B., A.M. Gorgone, A.R. Jugovich, B.L. Byrd and L.J. Hansen. 2011. Accounting for transients when estimating abundance of bottlenose dolphins in Choctawhatchee Bay, Florida. J. Wildl. Manage. 75(3): 569–579.

DFO 2013. Stock assessment of Northwest Atlantic grey seals (*Halichoerus grypus*). DFO Can. Sci. Advis. Sec. Sci. Advis. Rep.2014/008. Available at http://www.dfo-mpo.gc.ca/csas.

Fulling, G.L., K.D. Mullin and C.W. Hubard 2003. Abundance and distribution of cetaceans in outer continental shelf waters of theU.S. Gulf of Mexico. Fish. Bull. 101: 923–932.

Garrison, L.P., A. Martinez and K. Maze-Foley. 2010. Habitat and abundance of cetaceans in Atlantic Ocean continental slope waters off the eastern USA. J. Cetacean Res. Manage. 11(3): 267–277.

Garrison, L.P., S.L. Swartz, A. Martinez, C. Burks and J. Stamates 2003. A marine mammal assessment survey of the Southeast US continental shelf, February-April 2002. NOAA Tech. Memo. NMFS-SEFSC-492.

Garrison, L.P. 2016. Abundance of marine mammals in waters of the U.S. East Coast during summer 2011. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution #PRBD-2016-08, 21 pp.

Garrison, L.P. and D. Palka. 2018. Abundance of short-finned pilot whales along the U.S. east coast from summer 2016 vessel surveys. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2018-07, 18 pp.

Garrison, L.P. 2020. Abundance of cetaceans along the southeast U.S. east coast from a summer 2016 vessel survey. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRD Contribution # PRD-2020-04, 17 pp.

Garrison, L.P., J. Ortega-Ortiz and G. Rappucci. 2020. Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution # PRBD-2020-07. 55 pp. Available from: https://repository.library.noaa.gov/view/noaa/26505

Gilbert, J.R., G.T. Waring, K.M. Wynne and N. Guldager 2005. Changes in abundance and distribution of harbor seals in Maine, 1981-2001. Mar. Mamm. Sci. 21: 519-535.
Hammill, M.O. 2005. Abundance of Northwest Atlantic grey seals in the Gulf of St. Lawrence and along the Nova Scotia Eastern Shore. Canadian Department of Fisheries and Oceans. Ottawa, Ontario. DFO Research Document 2005/036. 11 pp. http://www.dfo-mpo.gc.ca/csas/Csas/Publications/ResDocs-DocRech/2005/2005_036_e.
Hansen, L.J., K.D. Mullin and C.L. Roden 1995. Estimates of cetacean abundance in the northern Gulf of Mexico from vessel surveys. Southeast Fisheries Science Center, Miami, FL. Contribution No. MIA-94/95-25. 9 pp.
Kingsley, M.C.S. and R.R. Reeves 1998. Aerial surveys of cetaceans in the Gulf of St. Lawrence in 1995 and 1996. Can. J. Zool. 76: 1529–1550.
Lawson, J.W. and J.-F. Gosselin 2009. Distribution and preliminary abundance estimates for cetaceans seen during Canada's Marine Megafauna Survey - A component of the 2007 TNASS. Can. Sci. Advisory Sec. Res. Doc. 2009/031. 33 pp.
Lawson, J.W. and J.-F. Gosselin 2011. Fully-corrected cetacean abundance estimates from the Canadian TNASS survey.Working Paper 10. National Marine Mammal Peer Review Meeting. Ottawa, Can. 28 pp.
Lawson J, and J-F. Gosselin 2018. Estimates of cetacean abundance from the 2016 NAISS aerial surveys of eastern Canadian waters, with a comparison to estimates from the 2007 TNASS NAAMCO SC/25/AE/09. 40 pp.
Litz, J., C. Sinclair and L. Garrison. 2018. Terrebonne-Timbalier Bay, Louisiana, common bottlenose dolphin abundance estimate: 2016. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2018-03, 10 pp. Available from: https://grunt.sefsc.noaa.gov/P_QryLDS/download/PRB94_PRBD_2018-03-C.pdf?id=LDS
Litz, J.A., E.L. Ronje, H.R. Whitehead and L.P. Garrison. 2019. Updated abundance estimate for common bottlense dolphins (*Tursiops truncatus*) inhabiting West Bay, Texas. Aquatic Conserv.: Mar. Freshw. Ecosyst. 2019:1–12.
McDonald, T.L., F.E. Hornsby, T.R. Speakman, E.S. Zolman, K.D. Mullin, C. Sinclair, P.E. Rosel, L. Thomas and L.H. Schwacke. 2017. Survival, density, and abundance of common bottlenose dolphins in Barataria Bay following the *Deepwater Horizon* oil spill. Endanger. Spec. Res. 33:193–209.
Mercer, M.C. 1975. Modified Leslie-DeLury population models of the long-finned pilot whale (*Globicephala melaena*) and annual production of the short-finned squid (*Illex illecebrosus*) based upon their interactions at Newfoundland. J. Fish. Res. Board Can 32(7): 1145-1154.
Miller, C. 2003. Abundance trends and environmental habitat usage patterns of bottlenose dolphins (*Tursiops truncatus*) in lower Barataria and Caminada Bays, Louisiana. Ph.D. thesis. Louisiana State University, Baton Rouge. 125 pp.
Mitchell, E. 1974. Present status of northwest Atlantic fin and other whale stocks. Pages 108–169 in: W. E. Schevill, (ed.) The whale problem: A status report. Harvard University Press, Cambridge, MA.
Mitchell, E. and D.G. Chapman 1977. Preliminary assessment of stocks of northwest Atlantic sei whales (*Balaenoptera borealis*). Rep. Int. Whal. Comm. (Special Issue) 1: 117–120.
Mullin, K.D. 2007. Abundance of cetaceans in the oceanic Gulf of Mexico based on 2003-2004 ship surveys. Available from: NMFS, Southeast Fisheries Science Center, P.O. Drawer 1207, Pascagoula, MS 39568, 26 pp.
Mullin, K.D., T. McDonald, R.S. Wells, B.C. Balmer, T. Speakman, C. Sinclair, E.S. Zolman, F. Hornsby, S.M. McBride, K.A. Wilkinson and L.H. Schwacke. 2017. Density, abundance, survival, and ranging patters of common bottlenose dolphins (*Tursiops truncatus*) in Mississippi Sound following the *Deepwater Horizon* oil spill. PLoS ONE 12(10):e0186265.
Mullin, K.D. and G.L. Fulling 2003. Abundance of cetaceans in the southern U.S. North Atlantic Ocean during summer 1998. Fish. Bull. 101: 603–613.
Mullin, K.D. and G.L. Fulling 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico. Mar. Mamm. Sci. 20(4): 787–807.
NMFS 1990. Cruise results, NOAA Ship CHAPMAN, Cruise No. 90-05. Marine Mammal Sighting Survey. National Marine Fisheries Service. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026. 5 pp.
NMFS 1991. Northeast cetacean aerial survey and inter-platform study. National Marine Fisheries Service. NOAA-NMFS-SEFSC and NEFSC. 4 pp.

NMFS 1993. Cruise results, NOAA ship DELAWARE II, Cruise No. DEL 93-06, Marine Mammal Survey. National Marine Fisheries Service. NOAA NMFS NEFSC, Woods Hole Laboratory, Woods Hole, MA, 5 pp.

NMFS 1994. Cruise results, NOAA ship RELENTLESS, Cruise No. RS 9402, Marine Mammal Survey/Warm Core Ring Study. National Marine Fisheries Service. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-102. 8 pp.

Palka, D.L. 1995. Abundance estimate of Gulf of Maine harbor porpoise. Rep. Int. Whal. Comm. (Special Issue) 16: 27-50.

Palka, D.L. 1996. Update on abundance of Gulf of Maine/Bay of Fundy harbor porpoises. Northeast Fish. Sci. Cent. Ref. Doc. 96-04. 35 pp. https://repository.library.noaa.gov/view/noaa/4008.

Palka, D.L. 2006. Summer abundance estimates of cetaceans in US North Atlantic Navy Operating Areas. Northeast Fish. Sci. Cent. Ref. Doc. 06-03. 41 pp. https://repository.library.noaa.gov/view/noaa/5258.

Palka, D.L. 2012. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2011 line transect survey. Northeast Fish. Sci. Cent. Ref. Doc. 12-29. 37 pp. https://repository.library.noaa.gov/view/noaa/4312.

Palka, D.L., S. Chavez-Rosales, E. Josephson, D. Cholewiak, H.L. Haas, L. Garrison, M. Jones, D. Sigourney, G. Waring, M. Jech, E. Broughton, M. Soldevilla, G. Davis, A. DeAngelis, C.R. Sasso, M.V.Winton, R.J. Smolowitz, G. Fay, E. LaBrecque, J.B. Leiness, Dettloff, M. Warden, K. Murray, and C. Orphanides. 2017. Atlantic Marine Assessment Program for Protected Species: 2010–2014. OCS Study BOEM 2017-071. US Dept. of the Interior, Bureau of Ocean Energy Management, Atlantic OCS Region, Washington, DC. 211 pp. https://espis.boem.gov/final%20reports/5638.pdf

Palka, D. 2020. Cetacean abundance estimates in US northwestern Atlantic Ocean waters from summer 2016 line transect surveys conducted by the Northeast Fisheries Science Center. Northeast Fish. Sci. Cent. Ref. Doc. 20-05.

Roberts, J.J., B.D. Best, L. Mannocci, E. Fujioka, P.N. Halpin, D.L. Palka, L.P. Garrison, K.D. Mullin, T.V.N. Cole, C.B. Khan, W. A. McLellan, D.A. Pabst and G.G. Lockhart. 2016. Habitat-based cetacean density models for the US Atlantic and Gulf of Mexico. Sci. Rep. 6:22615.

Scott, G.P., D.M. Burn, L.J. Hansen and R.E. Owen 1989. Estimates of bottlenose dolphin abundance in the Gulf of Mexico from regional aerial surveys. CRD 88/89-07. Available from: NMFS, Southeast Fisheries Science Center, 75 Virginia Beach Dr., Miami, FL 33149.

Smith, T., D. Palka and K. Bisack 1993. Biological significance of by-catch of harbor porpoise in the Gulf of Maine Demersal Gillnet Fishery. Northeast Fish. Sci. Cent. Ref. Doc. 93-23. 15 pp.

Spradlin T.R. and D.A. Pabst 2008. Seasonal abundance and distribution patterns of common bottlenose dolphins (*Tursiops truncatus*) near St. Joseph Bay, Florida, USA. J. Cetacean Res. Manage. 10(2): 157-167.Tyson, R.B. and R.S. Wells. 2016. Sarasota Bay/Little Sarasota Bay bottlenose dolphin abundance estimates: 2015. Prepared for National Marine Fisheries Service, northern Gulf of Mexico bay, sound, and estuary bottlenose dolphin stock blocks B20 and B35, combined. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, FL 33140. PRBD Contribution # PRBD-2016-02, 21 pp. Available from: https://www.sefsc.noaa.gov/P_QryLDS/download/PRB84_PRBD-2016-02.pdf?id=LDS

Tyson, R.B., S.M. Nowacek and D.P. Nowacek. 2011. Community structure and abundance of bottlenose dolphins Tursiops truncates in coastal waters of the northeast Gulf of Mexico. Mar. Ecol. Prog. Ser. 438: 253–265.

Waring, G.T. 1998. Results of the summer 1991 R/V Chapman marine mammal sighting survey. Northeast Fish. Sci. Cent. Ref. Doc. 98-09. 21 pp.

Waring, G.T., C.P. Fairfield, C.M. Ruhsam and M. Sano 1992. Cetaceans associated with Gulf Stream Features off the Northeastern USA Shelf. ICES [Int. Counc. Explor. Sea] C.M. 1992/N:12.

Waring, G.T., R.A. DiGiovanni Jr, E. Josephson, S. Wood, and J.R. Gilbert. 2015. 2012 population estimate for the harbor seal (*Phoca vitulina concolor*) in New England waters. NOAA Tech. Memo. NMFS NE-235. 15 pp.

Wells, R.S. 2009. Learning from nature: Bottlenose dolphin care and husbandry. Zoo Biol. 28: 1–17.

Appendix V: Fishery Bycatch Summaries

Part A: By Fishery

Northeast Sink Gillnet

	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin		Long-finned Pilot Whale		Harbor Seal		Gray Seal		Harp Seal	
Year	**M/SI**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	2900	0.32	0	0	0	0	0	0	0	0	0	0	602	0.68	0	0	0	0
1991	2000	0.35	0	0	49	0.46	0	0	0	0	0	0	231	0.22	0	0	0	0
1992	1200	0.21	0	0	154	0.35	0	0	0	0	0	0	373	0.23	0	0	0	0
1993	1400	0.18	0	0	205	0.31	0	0	0	0	0	0	698	0.19	0	0	0	0
1994	2100	0.18	0	0	240	0.51	0	0	0	0	0	0	1330	0.25	19	0.95	861	0.58
1995	1400	0.27	0	0	80	1.16	0	0	0	0	0	0	1179	0.21	117	0.42	694	0.27
1996	1200	0.25	0	0	114	0.61	63	1.39	0	0	0	0	911	0.27	49	0.49	89	0.55
1997	782	0.22	0	0	140	0.61	0	0	0	0	0	0	598	0.26	131	0.5	269	0.5
1998	332	0.46	0	0	34	0.92	0	0	0	0	0	0	332	0.33	61	0.98	78	0.48
1999	270	0.28	0	0	69	0.7	146	0.97	0	0	0	0	1446	0.34	155	0.51	81	0.78
2000	507	0.37	132	1.16	26	1	0	0	15	1.06	0	0	917	0.43	193	0.55	24	1.57
2001	53	0.97	0	0	26	1	0	0	0	0	0	0	1471	0.38	117	0.59	26	1.04
2002	444	0.37	0	0	30	0.74	0	0	0	0	0	0	787	0.32	0	0	0	0
2003	592	0.33	0	0	31	0.93	0	0	0	0	0	0	542	0.28	242	0.47	0	0
2004	654	0.36	1[a]	na	7	0.98	0	0	0	0	0	0	792	0.34	504	0.34	303	0.3
2005	630	0.23	0	0	59	0.49	5	0.8	15	0.93	0	0	719	0.2	574	0.44	35	0.68
2006	514	0.31	0	0	41	0.71	20	1.05	0	0	0	0	87	0.58	248	0.47	65	0.66
2007	395	0.37	0	0	0	0	11	0.94	0	0	0	0	92	0.49	886	0.24	119	0.35
2008	666	0.48	0	0	81	0.57	34	0.77	0	0	0	0	242	0.41	618	0.23	238	0.38
2009	591	0.23	0	0	0	0	43	0.77	0	0	0	0	513	0.28	1063	0.26	415	0.27
2010	387	0.27	0	0	66	0.9	42	0.81	0	0	3	0.8	540	0.25	1155	0.28	253	0.61
2011	273	0.2	0	0	18	0.43	64	0.71	0	0	0	0	343	0.19	1491	0.22	14	0.46
2012	277.3	0.59	0	0	9	0.92	95	0.4	6	0.87	0	0	252	0.26	542	0.19	0	0
2013	399	0.33	27	5	4	1.03	104	0.47	23	0.97	0	0	147	0.3	1127	0.2	22	0.75
2014	128	0.27	0	0	10	0.66	111	0.46	0	0	0	0	390	0.39	917	0.14	17	0.53
2015	177	0.28	0	0	0	0	55	0.54	0	0	0	0	474	0.17	1021	0.25	119	0.34
2016	125	0.34	0	0	0	0	80	0.38	0	0	0	0	245	0.29	498	0.33	85	0.5
2017	136	0.28	8	0.92	0	0	133	0.28	0	0	0	0	298	0.18	930	0.16	44	0.37
2018	92	0.52	0	0	0	0	93	0.45	0	0	0	0	188	0.36	1113	0.32	14	0.8

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/northeast-sink-gillnet-fishery-mmpa-list-fisheries.
[a] Unextrapolated mortalities
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Mid-Atlantic Sink Gillnet

	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin		Pilot Whale, Unidentified		Harbor Seal		Gray Seal		Harp Seal	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1994	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1995	103	0.57	56	1.66	0	0	7.4	0.69	0	0	0	0	0	0	0	0	0	0
1996	311	0.31	64	0.83	0	0	43	0.79	0	0	0	0	0	0	0	0	0	0
1997	572	0.35	0	0	45	0.82	0	0	0	0	0	0	0	0	0	0	0	0
1998	446	0.36	63	0.94	0	0	0	0	0	0	7	0	11	0.77	0	0	17	1.02
1999	53	0.49	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2000	21	0.76	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2001	26	0.95	na	na	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2002	unk	na	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2003	76	1.13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2004	137	0.91	0	0	0	0	0	0	0	0	0	0	15	0.86	69	0.92	0	0
2005	470	0.51	1[a]	na	0	0	0	0	0	0	0	0	63	0.67	0	0	0	0
2006	511	0.32	0	0	0	0	0	0	0	0	0	0	26	0.98	0	0	0	0
2007	58	1.03	0	0	0	0	0	0	34	0.73	0	0	0	0	0	0	38	0.9
2008	350	0.75	0	0	0	0	0	0	0	0	0	0	88	0.74	0	0	176	0.74
2009	201	0.55	0	0	0	0	0	0	0	0	0	0	47	0.68	0	0	0	0
2010	259	0.88	0	0	0	0	30	0.48	0	0	0	0	89	0.39	267	0.75	0	0
2011	123	0.41	0	0	0	0	29	0.53	0	0	0	0	21	0.67	19	0.60	0	0
2012	63.41	0.83	0	0	0	0	15	0.93	0	0	0	0	0	0	14	0.98	0	0
2013	19	1.06	26	0.95	0	0	62	0.67	0	0	0	0	0	0	0	0	0	0
2014	22	1.03	0	0	0	0	17	0.86	0	0	0	0	19	1.06	22	1.09	0	0
2015	60	1.16			0	0	30	0.55	0	0	0	0	48	0.52	15	1.04	0	0
2016	23	0.64			0	0	7	0.97	0	0	0	0	18	0.95	7	0.93	0	0
2017	9	0.95	0	0	0	0	22	0.71	0	0	0	0	3	0.62	0	0	0	0
2018	0	0	0	0	0	0	8	0.91	0	0	0	0	26	0.52	0	0	0	0

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-gillnet-fishery-mmpa-list-fisheries. For bottlenose dolphin stocks not listed in this table (Northern Migratory Coastal Stock, Southern Migratory Coastal Stock, Northern NC Estuarine Stock, Southern NC Estuarine Stock), see Lyssikatos & Garrison 2018 and Lyssikatos 2021.

[a] Unextrapolated mortalities

na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

New England/North Atlantic Bottom Trawl

	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Harbor Seal		Gray Seal		Harp Seal		Minke Whale	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1990	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1991	0	0	91	0.97	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1992	0	0	0	0	110	0.97	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1993	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1994	0	0	0	0	182	0.71	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1995	0	0	0	0	0	0	142	0.77	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1996	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	93	1.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	137	0.34	27	0.29	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2001	0	0	0	0	161	0.34	30	0.3	0	0	21	0.27	0	0	0	0	0	0	49	1.1	0	0
2002	0	0	0	0	70	0.32	26	0.29	0	0	22	0.26	0	0	0	0	0	0	0	0	0	0
2003	*	*	0	0	216	0.27	26	0.29	0	0	20	0.26	0	0	0	0	0	0	0	0	0	0
2004	0	0	0	0	200	0.30	26	0.29	0	0	15	0.29	0	0	0	0	0	0	0	0	0	0
2005	7.2	0.48	0	0	213	0.28	32	0.28	0	0	15	0.30	0	0	0	0	unk	unk	unk	unk	0	0
2006	6.5	0.49	0	0	40	0.50	25	0.28	0	0	14	0.28	0	0	0	0	0	0	0	0	0	0
2007	5.6	0.46	48	0.95	29	0.66	24	0.28	3	0.52	0	0	0	0	0	0	unk	unk	0	0	0	0
2008	5.6	0.97	19	0.88	13	0.57	6	0.99	2	0.56	0	0	21	0.51	0	0	16	0.52	0	0	7.8	0.69
2009	0	0	18	0.92	171	0.28	24	0.60	3	0.53	0	0	13	0.70	0	0	22	0.46	5	1.02	0	0
2010	0	0	4	0.53	37	0.32	114	0.32	2	0.55	0	0	30	0.43	0	0	30	0.34	0	0	0	0
2011	5.9	0.71	10	0.84	141	0.24	72	0.37	3	0.55	0	0	55	0.18	9	0.58	58	0.25	3	1.02	0	0
2012	0	0	0	0	27	0.47	40	0.54	0	0	0	0	33	0.32	3	1	37	0.49	0	0	0	0
2013	7	0.98	0	0	33	0.31	17	0.54	0	0	0	0	16	0.42	4	0.89	20	0.37	0	0	0	0
2014	5.5	0.86	0	0	16	0.5	17	0.53	4.2	0.91	0	0	32	0.44	11	0.63	19	0.45	0	0	0	0
2015	3.7	0.49	19	0.65	15	0.52	22	0.45	0	0	0	0	0	0	0	0	23	0.46	0	0	0	0
2016	0	0	33.5	0.89	28	0.46	16	0.46	17	0.88	0	0	29	0.58	0	0	0	0	0	0	0	0
2017	0	0	0	0	15	0.64	0	0	0	0	0	0	0	0	8.3	0	16	0.24	0	0	0	0
2018	0	0	0	0	0	0	28	0.54	0	0	0	0	0	0	0	0	32	0.42	0	0	0	0

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/northeast-bottom-trawl-fishery-mmpa-list-fisheries
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Mid-Atlantic Bottom Trawl

	Bottlenose Dolphin, Atlantic Offshore Stock		White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Harbor Seal		Gray Seal	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1997	0	0	161	1.58	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	0	0	228	1.03	0	0	0	0
2000	0	0	27	0.17	0	0	0	0	0	0	0	0	0	0
2001	0	0	27	0.19	103	0.27	0	0	39	0.3	0	0	0	0
2002	0	0	25	0.17	87	0.27	0	0	38	0.36	0	0	0	0
2003	0	0	31	0.25	99	0.28	0	0	31	0.31	0	0	0	0
2004	0	0	26	0.2	159	0.3	0	0	35	0.33	0	0	0	0
2005	0	0	38	0.29	141	0.29	0	0	31	0.31	0	0	0	0
2006	0	0	3	0.53	131	0.28	0	0	37	0.34	0	0	0	0
2007	11	0.42	2	1.03	66	0.27	33	0.34	0	0	0	0	0	0
2008	16	0.36	0	0	23	1	39	0.69	0	0	0	0	0	0
2009	21	0.45	0	0	167	0.46	23	0.5	0	0	24	0.92	38	0.7
2010	20	0.34	0	0	21	0.96	54	0.74	0	0	11	1.1	0	0
2011	34	0.31	0	0	271	0.25	62	0.56	0	0	0	0	25	0.57
2012	16	1.00	0	0	323	0.26	8	1	0	0	23	1	30	1.1
2013	0	0	0	0	269	0.29	42	0.71	0	0	11	0.96	29	0.67
2014	25	0.66	9.7	0.94	329	0.29	21	0.93	0	0	10	0.95	7	0.96
2015	0	0	0	0	250	0.32	40	0.63	0	0	7.4	1.0	0	0
2016	7.3	0.93	0	0	177	0.33	39	0.56	0	0	0	0	26	0.57
2017	22.1	0.66	0	0	380	0.23	31	0.51	0	0	0	0	26	0.40
2018	6.3	0.91	0	0	205	0.21	0	0	0	0	5.6	0.94	56	0.58

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-bottom-trawl-fishery-mmpa-list-fisheries
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Northeast Mid-Water Trawl

	White-sided Dolphin		Common Dolphin		Pilot Whale, Unidentified		Long-finned Pilot Whale		Harbor Seal		Gray Seal	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1999	0	0	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	4.6	0.74	0	0	0	0	0	0
2001	unk	na	0	0	11	0.74	0	0	0	0	0	0
2002	unk	na	0	0	8.9	0.74	0	0	0	0	0	0
2003	22	0.97	0	0	14	0.56	0	0	0	0	0	0
2004	0	0	0	0	5.8	0.58	0	0	0	0	0	0
2005	9.4	1.03	0	0	1.1	0.68	0	0	0	0	0	0
2006	0	0	0	0	0	0	0	0	0	0	0	0
2007	0	0	0	0	0	0	0	0	0	0	0	0
2008	0	0	0	0	0	0	16	0.61	0	0	0	0
2009	0	0	0	0	0	0	0	0	1.3	0.81	0	0
2010	0	0	1[a]	na	0	0	0	0	2[a]	na	0	0
2011	0	0	0	0	0	0	1	0	0	0	0	0
2012	0	0	1[a]	na	0	0	1	0	1[a]	na	1[a]	na
2013	0	0	0	0	0	0	3	0	0	0	1[a]	na
2014	0	0	0	0	0	0	4	na	1[a]	na	0	0
2015	0	0	0	0	0	0	0	na	2[a]	na	0	0
2016	0	0	0	0	0	0	3	na	1[a]	na	0	0
2017	0	0	0	0	0	0	0	na	0	na	0	0
2018											1[a]	na

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/northeast-mid-water-trawl-fishery-mmpa-list-fisheries

[a] Unextrapolated mortalities

na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Mid-Atlantic Mid-Water Trawl

	White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Harbor Seal		Gray Seal	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1999	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	0	0	0	0	0	0
2001	unk	na	0	0	0	0	0	0	0	0
2002	unk	na	0	0	0	0	0	0	0	0
2003	0	0	0	0	0	0	0	0	0	0
2004	22	0.99	0	0	0	0	0	0	0	0
2005	58	1.02	0	0	0	0	0	0	0	0
2006	29	0.74	0	0	0	0	0	0	0	0
2007	12	0.98	3.2	0.7	0	0	0	0	0	0
2008	15	0.73	0	0	1[a]	na	0	0	0	0
2009	4	0.92	0	0	0	0	0	0	0	0
2010	0	0	0	0	0	0	1[a]	na	1[a]	na
2011	0	0	0	0	0	0	0	0	0	0
2012	0	0	0	0	0	0	0	0	0	0
2013	0	0	0	0	0	0	0	0	0	0
2014	0	0	0	0	0	0	0	0	0	0
2015	0	0	0	0	0	0	0	0	0	0
2016	0	0	0	0	0	0	0	0	0	0
2017	0	0	0	0	0	0	0	0	0	0
2018										

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-mid-water-trawl-includes-pair-trawl-fishery-mmpa
[a] Unextrapolated mortalities
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Pelagic Longline

	Pantropical Spotted Dolphin, GMex		Bottlenose Dolphin, Atlantic Offshore Stock		Common Dolphin		Risso's Dolphin, Atlantic		Risso's Dolphin, Gmex		Pilot Whale, Unidentified & Long-finned, Atlantic		Short-finned Pilot Whale, Atlantic		Beaked Whale, Unidentified	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1992	0	0	0	0	0	0	0	0	0	0	22	0.23	0	0	0	0
1993	0	0	0	0	0	0	13	0.19	0	0	0	0	0	0	0	0
1994	0	0	0	0	0	0	7	1	0	0	137	0.44	0	0	0	0
1995	0	0	0	0	0	0	103	0.68	0	0	345	0.51	0	0	0	0
1996	0	0	0	0	0	0	99	1	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	57	1	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	22	1	0	0	381	0.79	0	0	0	0
2000	0	0	0	0	0	0	64	1	0	0	133	0.88	0	0	0	0
2001	0	0	0	0	0	0	69	0.57	0	0	79	0.48	0	0	0	0
2002	0	0	0	0	0	0	28	0.86	0	0	54	0.46	0	0	0	0
2003	0	0	0	0	0	0	40	0.63	0	0	21	0.77	0	0	5.3	1
2004	0	0	0	0	0	0	28	0.72	0	0	74	0.42	0	0	0	0
2005	0	0	0	0	0	0	3	1	0	0	212	0.21	0	0	0	0
2006	0	0	0	0	0	0	0	0	0	0	185	0.47	0	0	0	0
2007	0	0	0	0	0	0	9	0.65	0	0	57	0.65	0	0	0	0
2008	0	0	0	0	0	0	16.8	0.73	8.3	0.63	0	0	80	0.42	0	0
2009	16	0.69	8.8	1	8.5	1	11.8	0.711	0	0	0	0	17	0.7	0	0
2010	0	0	0	0	0	0	0	0	0	0	0	0	127	0.78	0	0
2011	0	0	0	0	0	0	12	0.70	1.5	1	0	0	305	0.29	0	0
2012	0	0	62	0.68	0	0	15	1	30	1	0	0	170.1	0.33	0	0
2013	2.1	1	0	0	0	0	1.9	1	15	1	0	0	124	0.32	0	0
2014	0	0	0	0	0	0	7.7	1	0	0	9.6	0.43	233	0.24	0	0
2015	0	0	0	0	9.05	1	8.4	0.71	0	0	2.2	0.49	200	0.24	0	0
2016	0	0	0	0	0	0	16	0.57	0	0	1.1	0.6	111	0.31	0	0
2017	0	0	0	0	4.92	1	0.2	1	0	0	3.3	0.98	133	0.29	0	0
2018	0	0	17.3	0.73	1.44	1	0.2	0.94	0	0	0.4	0.93	102	0.39	0	0

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/atlantic-ocean-caribbean-gulf-mexico-large-pelagics-longline; na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Pelagic Drift Gillnet

	White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Bottlenose Dolphin, Atlantic Offshore Stock		Beaked Whale, Unidentified		Sowerby's Beaked Whales		Harbor Porpoise	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1989	4.4	0.71	0	0	87	0.52	0	0	0	0	72	0.18	60	0.21	0	0	0.7	7
1990	6.8	0.71	0	0	144	0.46	0	0	0	0	115	0.18	76	0.26	0	0	1.7	2.65
1991	0.9	0.71	223	0.12	21	0.55	30	0.26	0	0	26	0.15	13	0.21	0	0	0.7	1
1992	0.8	0.71	227	0.09	31	0.27	33	0.16	0	0	28	0.1	9.7	0.24	0	0	0.4	1
1993	2.7	0.17	238	0.08	14	0.42	31	0.19	0	0	22	0.13	12	0.16	0	0	1.5	0.34
1994	0	0.71	163	0.02	1.5	0.16	20	0.06	0	0	14	0.04	0	0	3	0.09	0	0
1995	0	0	83	0	6	0	9.1	0	0	0	5	0	3	0	6	0	0	0
1996	0	0	0	0	0	0	0	0	0	0	0	0	2	0.25	9	0.12	0	0
1997	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	9	0	0	0	0	0	3	0	7	0	2	0	0	0
1999	0	0	0	0	0	0	20	0	0	0	0	0	0	0	0	0	0	0

Note: This table only includes observed bycatch.

Pelagic Pair Trawl

	White-sided Dolphin		Common Dolphin		Risso's Dolphin, Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Bottlenose Dolphin, Atlantic Offshore	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1989	0	0	0	0	0	0	0	0	0	0	0	0
1990	0	0	0	0	0	0	0	0	0	0	0	0
1991	0	0	0	0	0.6	1	0	0	0	0	13	0.52
1992	0	0	0	0	4.3	0.76	0	0	0	0	73	0.49
1993	0	0	0	0	3.2	1	0	0	0	0	85	0.41
1994	0	0	0	0	0	0	2	0.49	0	0	4	0.4
1995	0	0	0	0	3.7	0.45	22	0.33	0	0	17	0.26
1996	0	0	0	0	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	0	0	0	0
1999	0	0	0	0	0	0	0	0	0	0	0	0

Note: This table only includes observed bycatch.
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Gulf of Mexico Shrimp Otter Trawl

	Atlantic Spotted Dolphin		Bottlenose Dolphin, Continental Shelf Stock		Bottlenose Dolphin, Western Coastal Stock		Bottlenose Dolphin, Northern Coastal Stock		Bottlenose Dolphin, Eastern Coastal Stock		Bottlenose Dolphin, TX BSE Stocks		Bottlenose Dolphin, LA BSE Stocks		Bottlenose Dolphin, AL/MS BSE Stocks		Bottlenose Dolphin, FL BSE Stocks	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1997	128	0.44	172	0.42	217	0.84	13	0.80	18	0.99	0	-	29	1.00	37	0.82	3	0.99
1998	146	0.44	180	0.43	148	0.80	20	0.95	23	0.99	0	-	31	0.99	37	0.83	2	0.99
1999	120	0.44	159	0.42	289	0.91	31	0.72	11	0.99	0	-	38	0.89	52	0.85	3	0.99
2000	105	0.44	156	0.43	242	0.86	15	0.72	15	0.99	0	-	21	0.86	47	0.77	8	0.99
2001	115	0.45	169	0.42	291	0.85	15	0.79	11	0.99	0	-	28	0.99	55	0.74	6	0.99
2002	128	0.44	166	0.42	223	0.80	29	0.84	12	0.99	0	-	118	0.98	69	0.84	6	0.99
2003	75	0.45	122	0.43	133	0.79	15	0.71	5	0.99	0	-	72	1.00	52	0.82	5	0.99
2004	84	0.46	132	0.43	111	0.80	14	0.88	5	0.99	0	-	77	0.90	26	0.90	2	0.99
2005	55	0.49	94	0.43	66	0.84	11	0.64	1	0.99	0	-	57	0.96	15	0.72	3	0.99
2006	49	0.44	77	0.43	105	0.89	16	0.67	6	0.99	0	-	55	0.97	17	0.64	3	0.99
2007	43	0.45	60	0.43	81	0.85	20	0.67	3	0.99	0	-	47	0.90	26	0.77	1	0.99
2008	37	0.53	46	0.44	56	0.80	22	0.77	1	0.99	0	-	61	1.00	28	0.76	1	0.99
2009	49	0.50	56	0.43	77	0.89	35	0.67	3	0.99	0	-	116	1.02	45	0.73	6	0.99
2010	44	0.42	57	0.40	57	0.83	17	0.64	3	0.99	0	-	113	1.09	58	0.64	6	0.99
2011	35	0.48	63	0.44	67	0.91	13	0.65	1	0.99	0	-	104	0.98	47	0.64	3	0.99
2012	28	0.44	49	0.37	48	0.79	12	0.68	0.6	1.01	0	-	31	0.76	12	0.80	0.2	1.01
2013	27	0.43	57	0.38	23	0.74	6.0	0.83	0.7	1.01	0	-	19	0.74	14	0.95	1.1	1.01
2014	23	0.43	58	0.40	57	0.84	8.3	0.74	1.1	0.98	0	-	40	0.94	2.8	0.66	1.2	0.98

Note: This table only includes observed bycatch. For a complete list of marine mammal species interactions with this fishery, please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/southeastern-us-atlantic-gulf-mexico-shrimp-trawl-fishery-mmpa.
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Appendix V: Fishery Bycatch Summaries

Part B: By Species

Harbor Porpoise

	Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Pelagic Drift Gillnet	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	0	0	2900	0.32	1.7	2.65
1991	na	na	0	0	2000	0.35	0.7	1
1992	na	na	0	0	1200	0.21	0.4	1
1993	na	na	0	0	1400	0.18	1.5	0.34
1994	na	na	0	0	2100	0.18		
1995	103	0.57	0	0	1400	0.27		
1996	311	0.31	0	0	1200	0.25		
1997	572	0.35	0	0	782	0.22		
1998	446	0.36	0	0	332	0.46		
1999	53	0.49	0	0	270	0.28		
2000	21	0.76	0	0	507	0.37		
2001	26	0.95	0	0	53	0.97		
2002	unk	na	0	0	444	0.37		
2003	76	1.13	*	*	592	0.33		
2004	137	0.91	0	0	654	0.36		
2005	470	0.51	7.2	0.48	630	0.23		
2006	511	0.32	6.5	0.49	514	0.31		
2007	58	1.03	5.6	0.46	395	0.37		
2008	350	0.75	5.6	0.97	666	0.48		
2009	201	0.55	0	0	591	0.23		
2010	259	0.88	0	0	387	0.27		
2011	123	0.41	5.9	0.71	273	0.2		
2012	63.41	0.83	0	0	277.3	0.59		
2013	19	1.06	7	0.98	399	0.33		
2014	22	1.03	5.5	0.86	128	0.27		
2015	60	1.16	3.7	0.49	177	0.28		
2016	23	0.64	0	0	125	0.34		
2017	9	0.95	0	0	136	0.28		
2018	0	0	0	0	92	0.52		

Note: This table only includes observed bycatch.

na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Common Bottlenose Dolphin, Atlantic Offshore Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Pelagic Drift Gillnet		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1991	na	na	na	na	91	0.97	0	0	26	0.15	0	0
1992	na	na	na	na	0	0	0	0	28	0.1	0	0
1993	na	na	na	na	0	0	0	0	22	0.13	0	0
1994	na	na	na	na	0	0	0	0	14	0.04	0	0
1995	na	na	56	1.66	0	0	0	0	5	0	0	0
1996	na	na	64	0.83	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	0	0			0	0
1998	0	0	63	0.94	0	0	0	0			0	0
1999	0	0	0	0	0	0	0	0			0	0
2000	0	0	0	0	0	0	132	1.16			0	0
2001	0	0	na	na	0	0	0	0			0	0
2002	0	0	0	0	0	0	0	0			0	0
2003	0	0	0	0	0	0	0	0			0	0
2004	0	0	0	0	0	0	1[a]	na			0	0
2005	0	0	1[a]	na	0	0	0	0			0	0
2006	0	0	0	0	0	0	0	0			0	0
2007	11	0.42	0	0	48	0.95	0	0			0	0
2008	16	0.36	0	0	19	0.88	0	0			0	0
2009	21	0.45	0	0	18	0.92	0	0			8.8	1
2010	20	0.34	0	0	4	0.53	0	0			0	0
2011	34	0.31	0	0	10	0.84	0	0			0	0
2012	16	1	0	0	0	0	0	0			61.8	0.68
2013	0	0	0	0	0	0	26	0.95			0	0
2014	25	0.66	0	0	0	0	0	0			0	0
2015	0	0	0	0	18.6	0.65	0	0			0	0
2016	7.3	0.93	0	0	33.5	0.89	0	0			0	0
2017	22.1	0.66	0	0	0	0	8	0.92			0	0
2018	6.3	0.91	0	0	0	0	0	0			17.3	0.73

Note: This table only includes observed bycatch.

[a] Unextrapolated mortalities

na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

White-sided Dolphin

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Drift Gillnet	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	na	na	na	na	0	0	0	0	na	na		
1991	na	na	na	na	na	na	0	0	49	0.46	na	na	0	0
1992	na	na	na	na	na	na	110	0.97	154	0.35	na	na	110	0.97
1993	na	na	na	na	na	na	0	0	205	0.31	na	na	0	0
1994	na	na	0	0	na	na	182	0.71	240	0.51	na	na	182	0.71
1995	na	na	0	0	na	na	0	0	80	1.16	na	na	0	0
1996	na	na	0	0	na	na	0	0	114	0.61	na	na		
1997	161	1.58	45	0.82	na	na	0	0	140	0.61	na	na		
1998	0	0	0	0	na	na	0	0	34	0.92	na	na		
1999	0	0	0	0	0	0	0	0	69	0.7	0	0		
2000	27	0.17	0	0	0	0	137	0.34	26	1	0	0		
2001	27	0.19	0	0	unk	na	161	0.34	26	1	unk	na		
2002	25	0.17	0	0	unk	na	70	0.32	30	0.74	unk	na		
2003	31	0.25	0	0	0	0	216	0.27	31	0.93	22	0.97		
2004	26	0.2	0	0	22	0.99	200	0.3	7	0.98	0	0		
2005	38	0.29	0	0	58	1.02	213	0.28	59	0.49	9.4	1.03		
2006	3	0.53	0	0	29	0.74	40	0.5	41	0.71	0	0		
2007	2	1.03	0	0	12	0.98	29	0.66	0	0	0	0		
2008	0	0	0	0	15	0.73	13	0.57	81	0.57	0	0		
2009	0	0	0	0	4	0.92	171	0.28	0	0	0	0		
2010	0	0	0	0	0	0	37	0.32	66	0.9	0	0		
2011	0	0	0	0	0	0	141	0.24	18	0.43	0	0		
2012	0	0	0	0	0	0	27	0.47	9	0.92	0	0		
2013	0	0	0	0	0	0	33	0.31	4	1.03	0	0		
2014	9.7	0.94	0	0	0	0	16	0.50	10	0.66	0	0		
2015	0	0	0	0	0	0	15	0.52	0	0	0	0		
2016	0	0	0	0	0	0	28	0.46	0	0	0	0		
2017	0	0	0	0	0	0	15	0.64	0	0	0	0		
2018	0	0	0	0	0	0	0	0	0	0	0	0		

Note: This table only includes observed bycatch; na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Risso's Dolphin, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1996	0	0	0	0	0	0	0	0	99	1
1997	0	0	0	0	0	0	0	0	0	0
1998	0	0	0	0	0	0	0	0	57	1
1999	0	0	0	0	0	0	0	0	22	1
2000	0	0	0	0	0	0	15	1.06	64	1
2001	0	0	0	0	0	0	0	0	69	0.57
2002	0	0	0	0	0	0	0	0	28	0.86
2003	0	0	0	0	0	0	0	0	40	0.63
2004	0	0	0	0	0	0	0	0	28	0.72
2005	0	0	0	0	0	0	15	0.93	3	1
2006	0	0	0	0	0	0	0	0	0	0
2007	33	0.34	34	0.73	3	0.52	0	0	9	0.65
2008	39	0.69	0	0	2	0.56	0	0	16.8	0.732
2009	23	0.5	0	0	3	0.53	0	0	11.8	0.711
2010	54	0.74	0	0	2	0.55	0	0	0	0
2011	62	0.56	0	0	3	0.55	0	0	11.8	0.699
2012	8	1	0	0	0	0	6	0.87	15.1	1
2013	42	0.71	0	0	0	0	23	0.97	1.9	1
2014	21	0.93	0	0	4.2	0.91	0	0	7.7	1.0
2015	40	0.63	0	0	0	0	0	0	8.4	0.71
2016	39	0.56	0	0	17	0.88	0	0	16.1	0.57
2017	31	0.51	0	0	0	0	0	0	0.2	1

Note: This table only includes observed bycatch.
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Long-finned Pilot Whale, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Midwater Trawl		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
2008	0	0	0	0	21	0.51	0	0	16	0.61	na	na
2009	0	0	0	0	13	0.7	0	0	0	0	na	na
2010	0	0	0	0	30	0.43	3	0.82	0	0	na	na
2011	0	0	0	0	55	0.18	0	0	1	0	na	na
2012	0	0	0	0	33	0.32	0	0	1	0	na	na
2013	0	0	0	0	16	0.42	0	0	3	0	na	na
2014	0	0	0	0	32	0.44	0	0	4	na	9.6	0.43
2015	0	0	0	0	0	0	0	0	0	na	2.2	0.49
2016	0	0	0	0	29	0.58	0	0	3	na	1.1	0.6
2017	0	0	0	0	0	0	0	0	0	na	3.3	0.98
2018	0	0	0	0	0	0	0	0	0	0	0.4	0.93

Note: This table only includes observed bycatch.
na = not applicable; unk = observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd = to be determined

Short-finned Pilot Whale, Western North Atlantic Stock

	Pelagic Longline	
Year	**M/SI (est)**	**CV**
2008	80	0.42
2009	17	0.7
2010	127	0.78
2011	305	0.29
2012	170	0.33
2013	124	0.32
2014	233	0.24
2015	200	0.24
2016	111	0.31
2017	133	0.29
2018	102	0.39

Note: This table only includes observed bycatch.
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Common Dolphin, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl		Pelagic Drift Gillnet		Pelagic Longline	
Year	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**	**M/SI (est)**	**CV**
1990	na	na	na	na	0	0	0	0	na	na			na	na
1991	na	na	na	na	0	0	0	0	na	na	223	0.12	na	na
1992	na	na	na	na	0	0	0	0	na	na	227	0.09	0	0
1993	na	na	na	na	0	0	0	0	na	na	238	0.08	0	0
1994	na	na	0	0	0	0	0	0	na	na	163	0.02	0	0
1995	na	na	7.4	0.69	142	0.77	0	0	na	na	83	0	0	0
1996	na	na	43	0.79	0	0	63	1.39	na	na			0	0
1997	0	0	0	0	93	1.06	0	0	na	na			0	0
1998	0	0	0	0	0	0	0	0	na	na			0	0
1999	0	0	0	0	0	0	146	0.97	0	0			0	0
2000	0	0	0	0	27	0.29	0	0	0	0			0	0
2001	103	0.27	0	0	30	0.3	0	0	0	0			0	0
2002	87	0.27	0	0	26	0.29	0	0	0	0			0	0
2003	99	0.28	0	0	26	0.29	0	0	0	0			0	0
2004	159	0.3	0	0	26	0.29	0	0	0	0			0	0
2005	141	0.29	0	0	32	0.28	5	0.8	0	0			0	0
2006	131	0.28	0	0	25	0.28	20	1.05	0	0			0	0
2007	66	0.27	0	0	24	0.28	11	0.94	0	0			0	0
2008	23	1	0	0	6	0.99	34	0.77	0	0			0	0
2009	167	0.46	0	0	24	0.6	43	0.77	0	0			8.8	1
2010	21	0.96	30	0.48	114	0.32	42	0.81	1[a]	na			0	0
2011	271	0.25	29	0.53	72	0.37	64	0.71	0	0			0	0
2012	323	0.26	15	0.93	40	0.54	95	0.4	1[a]	0			61.8	.68
2013	269	0.29	62	0.67	17	0.54	104	0.46	0	0			0	0
2014	17	0.53	17	0.86	17	0.53	111	0.47	0	0			0	0
2015	250	0.32	30	0.55	22	0.45	55	0.54	0	0			9.1	1.0
2016	177	0.33	7	0.97	16	0.46	80	0.38	0	0			0	0
2017	380	0.23	22	0.71	0	0	133	0.28	0	0			4.92	1
2018	205	0.54	98	0.91	28	0.54	93	0.45	0	0			1.44	1

Note: This table only includes observed bycatch; na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

[a] Unextrapolated mortalities

Harbor Seal

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl	
Year	M/SI (est)	CV	M/SI (est)	CV	M/SI (est)	CV	M/SI (est)	CV	M/SI (est)	CV	M/SI (est)	CV	M/SI (est)	CV
1990	na	na	na	na	na	na	na	na	0	0	602	0.68	na	na
1991	na	na	na	na	na	na	na	na	0	0	231	0.22	na	na
1992	na	na	na	na	na	na	na	na	0	0	373	0.23	na	na
1993	na	na	na	na	na	na	na	na	0	0	698	0.19	na	na
1994	na	na	na	na	na	na	na	na	0	0	1330	0.25	na	na
1995	na	na	na	na	0	0	na	na	0	0	1179	0.21	na	na
1996	na	na	na	na	0	0	na	na	0	0	911	0.27	na	na
1997	na	na	0	0	0	0	na	na	0	0	598	0.26	na	na
1998	na	na	0	0	11	0.77	na	na	0	0	332	0.33	na	na
1999	na	na	0	0	0	0	na	na	0	0	1446	0.34	0	0
2000	na	na	0	0	0	0	0	0	0	0	917	0.43	0	0
2001	na	na	0	0	0	0	0	0	0	0	1471	0.38	0	0
2002	na	na	0	0	0	0	0	0	0	0	787	0.32	0	0
2003	0	0	0	0	0	0	0	0	0	0	542	0.28	0	0
2004	0	0	0	0	15	0.86	0	0	0	0	792	0.34	0	0
2005	0	0	0	0	63	0.67	0	0	0	0	719	0.2	0	0
2006	na	na	0	0	26	0.98	0	0	0	0	87	0.58	0	0
2007	0	0	0	0	0	0	0	0	0	0	92	0.49	0	0
2008	0	0	0	0	88	0.74	0	0	0	0	242	0.41	0	0
2009	0	0	24	0.92	47	0.68	0	0	0	0	513	0.28	1.3	0.81
2010	0	0	11	1.1	89	0.39	1[a]	0	0	0	540	0.25	2	0
2011	1[a]	0	0	0	21	0.67	0	0	9	0.58	343	0.19	0	0
2012	0	0	23	1	0	0	0	0	3	1	252	0.26	1	0
2013	0	0	11	0.96	0	0	0	0	4	0.89	147	0.3	0	0
2014	0	0	10	0.95	19	1.06	0	0	11	0.63	390	0.39	na	ma
2015	0	0	7.4	1.0	48	0.52	0	0	0	0	474	0.17	2[a]	na
2016	0	0	0	0	18	0.95	0	0	0	0	245	0.29	1[a]	na
2017	0	0	0	0	3	0.62	0	0	0	0	298	0.18	0	0
2018	0	0	6	0.94	26	0.52	0	0	0	0	188	0.36	0	0

Note: This table only includes observed bycatch; na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

[a] Unextrapolated mortalities

Gray Seal

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gillnet		Northeast Midwater Trawl	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1994	na	na	na	na	0	0	0	0	0	0	19	0.95	0	0
1995	na	na	na	na	0	0	0	0	0	0	117	0.42	0	0
1996	na	na	na	na	0	0	0	0	0	0	49	0.49	0	0
1997	na	na	0	0	0	0	0	0	0	0	131	0.5	0	0
1998	na	na	0	0	0	0	0	0	0	0	61	0.98	0	0
1999	na	na	0	0	0	0	0	0	0	0	155	0.51	0	0
2000	na	na	0	0	0	0	0	0	0	0	193	0.55	0	0
2001	na	na	0	0	0	0	0	0	0	0	117	0.59	0	0
2002	na	na	0	0	0	0	0	0	0	0	0	0	0	0
2003	0	0	0	0	0	0	0	0	0	0	242	0.47	0	0
2004	0	0	0	0	69	0.92	0	0	0	0	504	0.34	0	0
2005	0	0	0	0	0	0	0	0	unk	unk	574	0.44	0	0
2006	na	na	0	0	0	0	0	0	0	0	248	0.47	0	0
2007	0	0	0	0	0	0	0	0	unk	unk	886	0.24	0	0
2008	0	0	0	0	0	0	0	0	16	0.52	618	0.23	0	0
2009	0	0	38	0.7	0	0	0	0	22	0.46	1063	0.26	0	0
2010	0	0	0	0	267	0.75	1[a]	0	30	0.34	1155	0.28	0	0
2011	0	0	25	0.57	19	0.6	0	0	58	0.25	1491	0.22	0	0
2012	0	0	30	1.1	14	0.98	0	0	37	0.49	542	0.19	1[a]	na
2013	0	0	29	0.67	0	0	0	0	20	0.37	1127	0.2	1[a]	na
2014	0	0	7	0.96	22	1.09	0	0	19	0.45	917	0.14	0	0
2015	0	0	0	0	15	1.04	0	0	23	0.46	1021	0.25	0	0
2016	0	0	26	0.57	7	0.93	0	0	0	0	498	0.33	0	0
2017	0	0	26	0.40	0	0	0	0	16	0.24	930	0.16	0	0
2018	0	0	56	0.58	0	0	0	0	32	0.42	1113	0.32	1[a]	na

Note: This table only includes observed bycatch.
[a] Unextrapolated mortalities
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Harp Seal

	Mid-Atlantic Gillnet		Northeast Bottom Trawl		NE Sink Gillnet	
Year	**M/SI**	**CV**	**M/SI**	**CV**	**M/SI**	**CV**
1994	0	0	0	0	861	0.58
1995	0	0	0	0	694	0.27
1996	0	0	0	0	89	0.55
1997	0	0	0	0	269	0.5
1998	17	1.02	0	0	78	0.48
1999	0	0	0	0	81	0.78
2000	0	0	0	0	24	1.57
2001	0	0	49	1.1	26	1.04
2002	0	0	0	0	0	0
2003	0	0	*	*	0	0
2004	0	0	0	0	303	0.3
2005	0	0	0	0	35	0.68
2006	0	0	0	0	65	0.66
2007	38	0.9	0	0	119	0.35
2008	176	0.74	0	0	238	0.38
2009	0	0	5	1.02	415	0.27
2010	0	0	0	0	253	0.61
2011	0	0	3	1.02	14	0.46
2012	0	0	0	0	0	0
2013	0	0	0	0	22	0.75
2014	0	0	0	0	57	0.42
2015	0	0	0	0	119	0.34
2016	0	0	0	0	85	0.50
2017	0	0	0	0	44	0.37
2018	0	0	0	0	14	0.80

Note: This table only includes observed bycatch.
na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Appendix VI: Table C. Estimates of Human-caused Mortality Resulting from the *Deepwater Horizon* Oil Spill

Estimates of human-caused mortality are a result of a population model developed to estimate the injury and time to recovery for stocks affected by the *Deepwater Horizon* (DWH) oil spill, taking into account long-term impacts resulting from mortality, reproductive failure, reduced survival rates, and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015).

	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Beaked Whales[a]	15.96	13.49	11.42	9.68	8.21	6.28	4.81	3.68	2.79	2.09	1.52	1.05	0.65	0.31	0
Common Bottlenose Dolphin, Oceanic Stock	96.55	81.93	69.71	59.39	50.63	38.86	29.86	22.88	17.40	13.03	9.48	6.54	4.06	1.91	0
Bryde's Whale	1.44	1.22	1.03	0.88	0.74	0.57	0.44	0.33	0.25	0.19	0.14	0.09	0.06	0.03	0
Clymene Dolphin	26.23	22.12	18.71	15.86	13.45	10.28	7.86	6.00	4.55	3.40	2.46	1.70	1.05	0.49	0
False Killer Whale	6.67	5.64	4.78	4.05	3.44	2.63	2.01	1.54	1.17	0.87	0.63	0.44	0.27	0.13	0
***Kogia* spp.**	111.92	91.48	75.08	61.80	50.98	37.92	28.27	21.04	15.56	11.33	8.03	5.40	3.27	1.50	0
Melon-headed Whale	29.33	24.83	21.04	17.84	15.13	11.56	8.85	6.76	5.13	3.83	2.78	1.92	1.19	0.56	0
Pantropical Spotted Dolphin	748.73	631.49	534.21	452.68	384.00	293.38	224.47	171.38	129.89	96.96	70.37	48.47	30.04	14.12	0
Pygmy Killer Whale	4.94	4.19	3.56	3.03	2.57	1.97	1.51	1.16	0.88	0.66	0.48	0.33	0.21	0.10	0
Risso's Dolphin	16.18	13.73	11.68	9.95	8.48	6.51	5.00	3.83	2.92	2.18	1.59	1.10	0.68	0.32	0
Rough-toothed Dolphin	113.72	96.50	82.11	69.96	59.64	45.78	35.18	26.96	20.50	15.35	11.17	7.72	4.79	2.26	0
Shelf Dolphins[b]	912.14	774.01	658.54	561.05	478.31	367.12	282.07	216.17	164.39	123.07	89.55	61.82	38.38	18.07	0
Short-finned Pilot Whale	10.79	9.13	7.73	6.56	5.56	4.25	3.25	2.49	1.88	1.41	1.02	0.71	0.44	0.21	0
Sperm Whale	29.82	25.12	21.20	17.90	15.14	11.53	8.79	6.70	5.07	3.78	2.74	1.89	1.17	0.55	0
Spinner Dolphin	352.31	297.15	251.37	213.01	180.70	138.05	105.63	80.65	61.13	45.63	33.12	22.82	14.14	6.65	0
Striped Dolphin	39.30	33.15	28.04	23.76	20.16	15.40	11.78	9.00	6.82	5.09	3.69	2.54	1.58	0.74	0

a. Beaked whales include Blainville's beaked whales, Gervais' beaked whales, and Cuvier's beaked whales

b. Shelf dolphins include common bottlenose dolphins and Atlantic spotted dolphins

DWH MMIQT [*Deepwater Horizon* Marine Mammal Injury Quantification Team]. 2015. Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the *Deepwater Horizon* Oil Spill, MM_TR.01_Schwacke_Quantification.of.Injury.to.GOM.Cetaceans. Southeast Fisheries Science Center, Protected Resources and Biodiversity Division, 75 Virginia Beach Dr., Miami, Florida 33140. PRBD Contribution #: PRBD-2020-02.

Procedures for Issuing Manuscripts in the Northeast Fisheries Science Center Reference Document (CRD) and the Technical Memorandum (TM) Series

The mission of NOAA's National Marine Fisheries Service (NMFS) is "stewardship of the nation's ocean resources and their habitat." As the research arm of the NMFS's Greater Atlantic Region, the Northeast Fisheries Science Center (NEFSC) supports the NMFS mission by "conducting ecosystem-based research and assessments of living marine resources, with a focus on tsshe Northeast Shelf, to promote the recovery and long-term sustainability of these resources and to generate social and economic opportunities and benefits from their use." Results of NEFSC research are largely reported in primary scientific media (*e.g.*, anonymously peer reviewed scientific journals). However, to assist itself in providing data, information, and advice to its constituents, the NEFSC occasionally releases its results in its own series.

NOAA Technical Memorandum NMFS-NE -- This series is issued irregularly. The series typically includes: data reports of long-term field or lab studies of important species or habitats; synthesis reports for important species or habitats; annual reports of overall assessment or monitoring programs; manuals describing program-wide surveying or experimental techniques; literature surveys of important species or habitat topics; proceedings and collected papers of scientific meetings; and indexed and/or annotated bibliographies. All issues receive internal scientific review and most issues receive technical and copy editing.

Northeast Fisheries Science Center Reference Document -- This series is issued irregularly. The series typically includes: data reports on field and lab studies; progress reports on experiments, monitoring, and assessments; background papers for, collected abstracts of, and/or summary reports of scientific meetings; and simple bibliographies. Issues receive internal scientific review and most issues receive copy editing.

CLEARANCE

All manuscripts submitted for issuance as CRDs must have cleared the NEFSC's manuscript/abstract/webpage review process. If your manuscript includes material from another work which has been copyrighted, then you will need to work with the NEFSC's Editorial Office to arrange for permission to use that material by securing release signatures on the "NEFSC Use-of-Copyrighted-Work Permission Form."

For more information, NEFSC authors should see the NEFSC's online publication policy manual, "Manuscript/abstract/webpage preparation, review, and dissemination: NEFSC author's guide to policy, process, and procedure."

STYLE

The CRD series is obligated to conform with the style contained in the current edition of the United States Government Printing Office Style Manual. That style manual is silent on many aspects of scientific manuscripts. The CRD series relies more on the CSE Style Manual. Manuscripts should be prepared to conform with these style manuals.

The CRD series uses the Integrated Taxonomic Information System, American Fisheries Society's guides, and the Society for Marine Mammalogy's guide for verifying scientific species'names.

For in-text citation, use the name-date system. A special effort should be made to ensure that all necessary bibliographic information is included in the list of references cited. Personal communications must include date, full name, and full mailing address of the contact.

PREPARATION

Once your document has cleared the review process, the Editorial Office will contact you with publication needs – for example, revised text (if necessary) and separate digital figures and tables if they are embedded in the document. Materials may be submitted to the Editorial Office as email attachments or intranet downloads. Text files should be in Microsoft Word, tables may be in Word or Excel, and graphics files may be in a variety of formats (JPG, GIF, Excel, PowerPoint, etc.).

PRODUCTION AND DISTRIBUTION

The Editorial Office will perform a copyedit of the document and may request further revisions. The Editorial Office will develop the inside and outside front covers, the inside and outside back covers, and the title and bibliographic control pages of the document.

Once the CRD is ready, the Editorial Office will contact you to review it and submit corrections or changes before the document is posted online. A number of organizations and individuals in the Northeast Region will be notified by e-mail of the availability of the document online.