NOAA Technical Memorandum NMFS-NE-264

US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2019

US DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, Massachusetts
July 2020

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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 3 years for stocks determined to be non-strategic. Included in this report as appendices are: 1) a summary of serious injury/mortality estimates of marine mammals in observed U.S. fisheries (Appendix I), 2) a summary of NMFS records of large whale human-caused serious injury and mortality (Appendix II), 3) detailed fisheries information (Appendix III), 4) 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 a list of reports not updated in the current year (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 2018 publication. Most of the changes incorporate new information into sections on population size and/or mortality estimates. A total of 35 of the Atlantic and Gulf of Mexico stock assessment reports were revised for 2019, and two new reports were written. NMFS is in the process of writing separate stock assessment reports for each of the 31 individual stocks contained in the Northern Gulf of Mexico Bay, Sound, and Estuary common bottlenose dolphin report. For the draft 2019 SARs, two new individual reports were completed separating out St. Andrew Bay and West Bay. The revised SARs include 5 strategic and 32 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 May 2019 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 to calculate it. 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, the NMFS held a workshop to review proposed additions and revisions to the guidelines for preparing SARs (Wade and Angliss 1997). Guidelines developed at the workshop were followed in preparing the 1996 through 2015 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 2020 date-stamp at the top right corner at the beginning of each report. Stocks not updated in 2019 are listed in Appendix VI.

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, 211 pp.

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, 93 pp.

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

Total Annual S.I. (serious injury) and Mortality and Annual Fisheries S.I. and Mortality are mean annual figures for the period 2013-2017. The "SAR revised" column indicates 2019 stock assessment reports that have been revised relative to the 2018 reports (Y=yes, N=no). If abundance, mortality, PBR or status have been revised, they are indicated with the letters "a", "m", "p" and "status" respectively. For those species not updated in this edition, the year of last revision is indicated. Unk = unknown and undet=undetermined (PBR for species with outdated abundance estimates is considered "undetermined").

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
North Atlantic right whale	Western North Atlantic	NEC	428	0	418	0.04[a]	0.1	0.8	6.85[a]	5.55[a]	Y	Y (a, m, p)
Humpback whale	Gulf of Maine	NEC	1,396	0	1,380	0.065	0.5	22	12.15[b]	7.75[b]	N	Y (a, m, p, status)
Fin whale	Western North Atlantic	NEC	7,418	0.25	6,029	0.04	0.1	12	2.35[c]	1.55[c]	Y	Y (a, m, p)
Sei whale	Nova Scotia	NEC	6,292	1.015	3,098	0.04	0.1	6.2	1.0[d]	0.2 [d]	Y	Y (a, m, p)
Minke whale	Canadian east coast	NEC	24,202	0.30	18,902	0.04	0.5	189	8.20[e]	7.0 [e]	N	Y (a, m, p)
Blue whale	Western North Atlantic	NEC	unk	unk	402	0.04	0.1	0.8	0	0	Y	Y (a, m, p)
Sperm whale	North Atlantic	NEC	4,349	0.28	3,451	0.04	0.1	3.9	0	0	Y	Y (a, m, p)
Dwarf sperm whale	Western North Atlantic	SEC	7,750[h]	0.38	5,689 [h]	0.04	0.4	46	0	0	N	Y (a, m, p)
Pygmy sperm whale	Western North Atlantic	SEC	7,750 [h]	0.38	5,689 [h]	0.04	0.4	46	0	0	N	Y (a, m, p)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Killer whale	Western North Atlantic	NEC	unk	unk	unk	0.04	0.5	unk	0	0	N	N (2014)
Pygmy killer whale	Western North Atlantic	SEC	unk	unk	unk	0.04	0.5	unk	0	0	N	Y (m)
False killer whale	Western North Atlantic	SEC	1,791	0.56	1,154	0.04	0.5	12	0	0	N	Y (a, m, p)
Northern bottlenose whale	Western North Atlantic	NEC	unk	unk	unk	0.04	0.5	unk	0	0	N	N (2014)
Cuvier's beaked whale	Western North Atlantic	NEC	5,744 [g]	0.36	4,282 [g]	0.04	0.5	43	0.2	0	N	Y (a, m, p)
Blainville's beaked whale	Western North Atlantic	NEC	10,107 [g]	0.27	8,085 [g]	0.04	0.5	81	0.2	0	N	Y (a, m, p)
Gervais beaked whale	Western North Atlantic	NEC	10,107 [g]	0.27	8,085[g]	0.04	0.5	81	0	0	N	Y (a, m, p)
Sowerby's beaked whale	Western North Atlantic	NEC	10,107 [g]	0.27	8,085 [g]	0.04	0.5	81	0	0	N	Y (a, m, p)
True's beaked whale	Western North Atlantic	NEC	10,107 [g]	0.27	8,085 [g]	0.04	0.5	81	0.2	0.2	N	Y (a, m, p)
Melon-headed whale	Western North Atlantic	SEC	unk	unk	unk	0.04	0.5	unk	0	0	N	Y (m)
Risso's dolphin	Western North Atlantic	NEC	35,493	0.19	30,289	0.04	0.5	303	54.3	53.9 (0.24)	N	Y (a, m, p)
Pilot whale, long-finned	Western North Atlantic	NEC	39,215	0.30	30,627	0.04	0.5	306	21	21 (0.22)	N	Y (a, m, p)
Pilot whale, short-finned	Western North Atlantic	SEC	28,924	0.24	23,637	0.04	0.5	236	160	160 (0.12)	N	Y (m)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Atlantic white-sided dolphin	Western North Atlantic	NEC	93,233	0.71	54,443	0.04	0.5	544	26	26 (0.20)	N	Y (a, m, p)
White-beaked dolphin	Western North Atlantic	NEC	536,016	0.31	415,344	0.04	0.5	4,153	0	0	N	Y (a, m, p)
Common dolphin	Western North Atlantic	NEC	172,825	0.21	145,216	0.04	0.5	1,452	419	419 (0.10)	N	Y (a, m, p)
Atlantic spotted dolphin	Western North Atlantic	SEC	39,921	0.27	32,032	0.04	0.5	320	0	0	N	Y (a, m, p)
Pantropical spotted dolphin	Western North Atlantic	SEC	6,593	0.52	4,367	0.04	0.5	44	0	0	N	Y (a, m, p)
Striped dolphin	Western North Atlantic	NEC	67,036	0.29	52,939	0.04	0.5	529	0	0	N	Y (a, m, p)
Fraser's dolphin	Western North Atlantic	SEC	unk	unk	unk	0.04	0.5	unk	0	0	N	Y (m)
Rough-toothed dolphin	Western North Atlantic	SEC	136	1.0	67	0.04	0.5	0.7	0	0	N	N (2018)
Clymene dolphin	Western North Atlantic	SEC	4,237	1.03	2,071	0.04	0.5	21	0	0	N	Y (a, m, p)
Spinner dolphin	Western North Atlantic	SEC	4,102	0.99	2,045	0.04	0.5	20	0	0	N	Y (a, m, p)
Common bottlenose dolphin	Western North Atlantic, offshore	SEC	62,851 [f]	0.23	51,914	0.04	0.5	519	28	28 (0.34)	N	Y (a, m, p)
Common bottlenose dolphin	Western North Atlantic, northern migratory coastal	SEC	6,639	0.41	4,759	0.04	0.5	48	6.1-13.2 [k]	6.1-13.2 [k]	Y	N (2017)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Common bottlenose dolphin	Western North Atlantic, southern migratory coastal	SEC	3,751	.060	2,353	0.04	0.5	23	0-14.3 [k]	0-14.3 [k]	Y	N (2017)
Common bottlenose dolphin	Western North Atlantic, S. Carolina/Georgia coastal	SEC	6,027	0.34	4,569	0.04	0.5	46	1.4-1.6 [k]	1.0-1.2 [k]	Y	N (2017)
Common bottlenose dolphin	Western North Atlantic, northern Florida coastal	SEC	877	0.49	595	0.04	0.5	6.0	0.6 [k]	0 [k]	Y	N (2017)
Common bottlenose dolphin	Western North Atlantic, central Florida coastal	SEC	1,218	0.35	913	0.04	0.5	9.1	0.4 [k]	0.4 [k]	Y	N (2017)
Common bottlenose dolphin	Northern North Carolina Estuarine System	SEC	823	0.06	782	0.04	0.5	7.8	0.8-18.2 [k]	0.2-17.6 [k]	Y	N (2017)
Common bottlenose dolphin	Southern North Carolina Estuarine System	SEC	unk	unk	unk	0.04	0.5	undet	0.4-0.6 [k]	0.4-0.6 [k]	Y	N (2017)
Common bottlenose dolphin	Northern South Carolina Estuarine System	SEC	unk	unk	unk	0.04	0.5	unk	0.2 [k]	0.2 [k]	Y	N (2015)
Common bottlenose dolphin	Charleston Estuarine System	SEC	unk	unk	unk	0.04	0.5	undet	unk[k]	unk[k]	Y	N (2015)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Common bottlenose dolphin	Northern Georgia/ Southern South Carolina Estuarine System	SEC	unk	unk	unk	0.04	0.5	unk	1.4 [k]	1.4 [k]	Y	N (2015)
Common bottlenose dolphin	Central Georgia Estuarine System	SEC	192	0.04	185	0.04	0.5	1.9	unk[k]	unk[k]	Y	N (2015)
Common bottlenose dolphin	Southern Georgia Estuarine System	SEC	194	0.05	185	0.04	0.5	1.9	unk[k]	unk[k]	Y	N (2015)
Common bottlenose dolphin	Jacksonville Estuarine System	SEC	unk	unk	unk	0.04	0.5	unk	1.2 [k]	1.2 [k]	Y	N (2015)
Common bottlenose dolphin	Indian River Lagoon Estuarine System	SEC	unk	unk	unk	0.04	0.5	unk	4.4 [k]	4.4 [k]	Y	N (2015)
Common bottlenose dolphin	Biscayne Bay	SEC	unk	unk	unk	0.04	0.5	unk	unk [k]	unk [k]	Y	N (2013)
Common bottlenose dolphin	Florida Bay	SEC	unk	unk	unk	0.04	0.5	undet	unk [k]	unk [k]	N	N (2013)
Harbor porpoise	Gulf of Maine/Bay of Fundy	NEC	95,543	0.31	74,034	0.046	0.5	851	217	217 (0.15)	N	Y (a, m, p)
Harbor seal	Western North Atlantic	NEC	75,834	0.15	66,884	0.12	0.5	2,006	350	338 (0.12)	N	Y (m)
Gray seal	Western North Atlantic	NEC	27,131	0.19	23,158	0.12	1.0	1,389	5,410	940 (0.09)	N	Y (m)
Harp seal	Western North Atlantic	NEC	unk	unk	unk	0.12	1.0	unk	232,422	65 (0.21)	N	Y (m)
Hooded seal	Western North Atlantic	NEC	unk	unk	unk	0.12	0.75	unk	1,680	0.6(1.12)	N	N (2018)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Sperm whale	Gulf of Mexico	SEC	763	0.38	560	0.04	0.1	1.1	0	0	Y	N (2015)
Bryde's whale	Gulf of Mexico	SEC	33	1.07	16	0.04	0.1	0.03	0.8	0	Y	N (2017)
Cuvier's beaked whale	Gulf of Mexico	SEC	74	1.04	36	0.04	0.5	0.4	0	0	N	N (2012)
Blainville's beaked whale	Gulf of Mexico	SEC	149[g]	0.91	77	0.04	0.5	0.8	0	0	N	N (2012)
Gervais' beaked whale	Gulf of Mexico	SEC	149[g]	0.91	77	0.04	0.5	0.8	0	0	N	N (2012)
Common bottlenose dolphin	Gulf of Mexico, Continental shelf	SEC	51,192	0.10	46,926	0.04	0.5	469	0.8 [k]	0.6 [k]	N	N (2015)
Common bottlenose dolphin	Gulf of Mexico, eastern coastal	SEC	12,388	0.13	11,110	0.04	0.5	111	1.6 [k]	1.6 [k]	N	N (2015)
Common bottlenose dolphin	Gulf of Mexico, northern coastal	SEC	7,185	0.21	6,044	0.04	0.5	60	0.4 [k, l]	0.4 [k]	N	N (2015)
Common bottlenose dolphin	Gulf of Mexico, western coastal	SEC	20,161	0.17	17,491	0.04	0.5	175	0.6 [k]	0.6 [k]	N	N (2015)
Common bottlenose dolphin	Gulf of Mexico, Oceanic	SEC	5,806	0.39	4,230	0.04	0.5	42	6.5	6.5 (0.65)	N	N (2014)
Common bottlenose dolphin	Laguna Madre[j]	SEC	80	1.57	unk	0.04	0.5	undet	0.4 [k]	0.2 [k]	Y	N (2018)
Common bottlenose dolphin	Neuces Bay/Corpus Christi Bay[j]	SEC	58	0.61	unk	0.04	0.5	undet	0 [k]	0 [k]	Y	N (2018)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Common bottlenose dolphin	Copano Bay/Aransas Bay/San Antonio Bay/Redfish Bay/Espiritu Santo Bay[j]	SEC	55	0.82	unk	0.04	0.5	undet	0.2 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	Matagorda Bay/Tres Palacios Bay/Lavaca Bay[j]	SEC	61	0.45	unk	0.04	0.5	undet	0.4 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	West Bay	SEC	48	0.03	46	0.04	0.5	0.5	0.2 [k]	0.2 [k]	N	Y (a, m, p)
Common bottlenose dolphin	Galveston Bay/East Bay/Trinity Bay[j]	SEC	152	0.43	unk	0.04	0.5	undet	0.4 [k]	0.4 [k]	Y	N (2018)
Common bottlenose dolphin	Sabine Lake[j]	SEC	0	-	-	0.04	0.4	undet	0.2 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	Calcasieu Lake[j]	SEC	0	-	-	0.04	0.4	undet	0.2 [k]	0.2 [k]	Y	N (2018)
Common bottlenose dolphin	Vermilion Bay/West Cote Blanche Bay/Atchafalaya Bay[j]	SEC	0	-	-	0.04	0.4	undet	0 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	Terrebonne Bay/Timbalier Bay	SEC	3870	0.15	3426	0.04	0.4	27	0.2 [k]	0 [k]	N	N (2018)
Common bottlenose dolphin	Barataria Bay	SEC	2,306	0.09	2,138	0.04	0.4	17	160 [k]	0.8 [k]	Y	N (2017)
Common bottlenose dolphin	Mississippi River Delta[j]	SEC	332	0.93	170	0.04	0.4	1.4	32.7[k, m]	0 [k]	Y	N (2018)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Common bottlenose dolphin	Mississippi Sound, Lake Borgne, Bay Boudreau	SEC	3,046	0.06	2,896	0.04	0.4	23	310 [k]	1.0 [k]	Y	N (2017)
Common bottlenose dolphin	Mobile Bay/Bonsecour Bay[j]	SEC	122	0.34	unk	0.04	0.4	undet	36.6 [k, m]	0.8 [k]	Y	N (2018)
Common bottlenose dolphin	Perdido Bay[j]	SEC	0	-	-	0.04	0.4	undet	0.6 [k]	0.2 [k]	Y	N (2018)
Common bottlenose dolphin	Pensacola Bay/East Bay[j]	SEC	33	0.80	unk	0.04	0.4	undet	0.2 [k]	0.2 [k]	Y	N (2018)
Common bottlenose dolphin	Choctawhatchee Bay	SEC	179	0.04	unk	0.04	0.5	undet	0.4 [k]	0.4 [k]	Y	N (2015)
Common bottlenose dolphin	St. Andrew Bay	SEC	199	0.09	185	0.04	0.4	1.5	0.2 [k]	0.2 [k]	N	Y (a, m, p)
Common bottlenose dolphin	St. Joseph Bay	SEC	142	0.17	123	0.04	0.4	1.0	unk [k]	unk [k]	N	Y (a, m, p)
Common bottlenose dolphin	St. Vincent Sound/Apalachicola Bay/St. George Sound[j]	SEC	439	0.14	unk	0.04	0.4	undet	0 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	Apalachee Bay[j]	SEC	491	0.39	unk	0.04	0.4	undet	0 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	Waccasassa Bay/Withlacoochee Bay/Crystal Bay[j]	SEC	unk	-	unk	0.04	0.4	undet	0 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	St. Joseph Sound/Clearwater Harbor[j]	SEC	unk	-	unk	0.04	0.4	undet	0.4 [k]	0.4 [k]	Y	N (2018)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Common bottlenose dolphin	Tampa Bay[j]	SEC	unk	-	unk	0.04	0.4	undet	0.6 [k]	0.6 [k]	Y	N (2018)
Common bottlenose dolphin	Sarasota Bay/Little Sarasota Bay[j]	SEC	158	0.27	126	0.04	0.4	1.0	0.6 [k]	0.6 [k]	N	N (2018)
Common bottlenose dolphin	Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay[j]	SEC	826	0.09	unk	0.04	0.4	undet	1.6 [k]	1.0 [k]	Y	N (2018)
Common bottlenose dolphin	Caloosahatchee River[j]	SEC	0	-	-	0.04	0.4	undet	0.4 [k]	0.4 [k]	Y	N (2018)
Common bottlenose dolphin	Estero Bay[j]	SEC	unk	-	unk	0.04	0.4	undet	0.2 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	Chokoloskee Bay/Ten Thousand Islands/Gullivan Bay[j]	SEC	unk	-	unk	0.04	0.4	undet	0 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	Whitewater Bay[j]	SEC	unk	-	unk	0.04	0.4	undet	0 [k]	0 [k]	Y	N (2018)
Common bottlenose dolphin	Florida Keys (Bahia Honda to Key West)[j]	SEC	unk	-	unk	0.04	0.4	undet	0 [k]	0 [k]	Y	N (2018)
Atlantic spotted dolphin	Gulf of Mexico	SEC	unk	unk	unk	0.04	0.5	undet	42	42 (0.45)	N	N (2015)
Pantropical spotted dolphin	Gulf of Mexico	SEC	50,880	0.27	40,699	0.04	0.5	407	4.4	4.4	N	N (2015)
Striped dolphin	Gulf of Mexico	SEC	1,849	0.77	1,041	0.04	0.5	10	0	0	N	N (2012)
Spinner dolphin	Gulf of Mexico	SEC	11,441	0.83	6,221	0.04	0.5	62	0	0	N	N (2012)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Rough-toothed dolphin	Gulf of Mexico	SEC	624	0.99	311	0.04	0.4	2.5	0.8	0.8 (1.0)	N	N (2016)
Clymene dolphin	Gulf of Mexico	SEC	129	1.00	64	0.04	0.5	0.6	0	0	N	N (2012)
Fraser's dolphin	Gulf of Mexico	SEC	unk	unk	unk	0.04	0.5	undet	0	0	N	N (2012)
Killer whale	Gulf of Mexico	SEC	28	1.02	14	0.04	0.5	0.1	0	0	N	N (2012)
False killer whale	Gulf of Mexico	SEC	unk	unk	unk	0.04	0.5	undet	0	0	N	N (2012)
Pygmy killer whale	Gulf of Mexico	SEC	152	1.02	75	0.04	0.5	0.8	0	0	N	N (2012)
Dwarf sperm whale	Gulf of Mexico	SEC	186[h]	1.04	90	0.04	0.5	0.9	0	0	N	N (2012)
Pygmy sperm whale	Gulf of Mexico	SEC	186[h]	1.04	90	0.04	0.5	0.9	0.3	0.3 (1.0)	N	N (2012)
Melon-headed whale	Gulf of Mexico	SEC	2,235	0.75	1,274	0.04	0.5	13	0	0	N	N (2012)
Risso's dolphin	Gulf of Mexico	SEC	2,442	0.57	1,563	0.04	0.5	16	7.9	7.9 (0.85)	N	N (2015)
Pilot whale, short-finned	Gulf of Mexico	SEC	2,415[i]	0.66	1456	0.04	0.5	15	0.5	0.5 (1.0)	N	N (2015)
Sperm Whale	Puerto Rico and U.S. Virgin Islands	SEC	unk	unk	unk	0.04	0.1	unk	unk	unk	Y	N (2010)
Common bottlenose dolphin	Puerto Rico and U.S. Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)
Cuvier's beaked whale	Puerto Rico and U.S. Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)

Species	Stock Area	NMFS Ctr.	Nbest	Nbest CV	Nmin	Rmax	Fr	PBR	Total Annual S.I and Mort.	Annual Fish. S.I. and Mort. (cv)	Strategic Status	SAR Revised
Pilot whale, short-finned	Puerto Rico and U.S. Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)
Spinner dolphin	Puerto Rico and U.S. Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)
Atlantic spotted dolphin	Puerto Rico and U.S. Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)

a. The R given for right whales is the default Rmax of 0.04. The total estimated human-caused mortality and serious injury to right whales is estimated at 6.85 per year. This is derived from two components: 1) non-observed fishery entanglement records at 5.55 per year, and 2) ship strike records at 1.3 per year.
b. The total estimated human-caused mortality and serious injury to the Gulf of Maine humpback whale stock is estimated as 12.15 per year. This average is derived from two components: 1) incidental fishery interaction records 7.75; 2) records of vessel collisions, 4.4.
c. The total estimated human-caused mortality and serious injury to the Western North Atlantic fin whale stock is estimated as 2.35 per year . This average is derived from two components: 1) incidental fishery interaction records 1.55; 2) records of vessel collisions, 0.8.
d. The total estimated human-caused mortality and serious injury to the Nova Scotia sei whale stock is estimated as 1.0 per year. This average is derived from two components: 1) incidental fishery interaction records 0.2; 2) records of vessel collisions, 0.8.
e. The total estimated human-caused mortality and serious injury to the Canadian East Coast minke whale stock is estimated as 8.0 per year. This average is derived from four components: 1) 6.6 minke whales per year (unknown CV) from U.S. and Canadian fisheries using strandings and entanglement data; 2) 1.0 per year from vessel strikes; and 3) 0.2 from U.S. observed fisheries, and 4) 0.2 non-fishery entanglement takes.
f. Estimates may include sightings of the coastal form.
g. This estimate includes Gervais' beaked whales and Blainville's beaked whales for the Gulf of Mexico stocks, and all undifferentiated beaked whales in the Atlantic.
h. This estimate includes both the dwarf and pygmy sperm whales.
i. This estimate includes all *Globicephala* sp., though it is presumed that only short-finned pilot whales are present in the Gulf of Mexico.
j. Details for these 24 stocks are included in the collective report: Common bottlenose dolphin (*Tursiops truncatus truncatus*), Northern Gulf of Mexico Bay, Sound, and Estuary Stocks. However, each stock has been given its own row in this table.
k The total annual human-caused mortality and serious injury for these stocks of common bottlenose dolphins is unknown because these stocks may interact with unobserved fisheries. Also, for Gulf of Mexico BSE stocks, mortality estimates for the shrimp trawl fishery are calculated at the state level and have not been included within mortality estimates for individual BSE stocks. Therefore, minimum counts of human-caused mortality and serious injury for these stocks are presented.
l. This minimum count does not include projected mortality estimates for 2012–2016 due to the DWH oil spill.
m. This minimum count includes projected mortality estimates for 2012–2016 due to the DWH oil spill.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

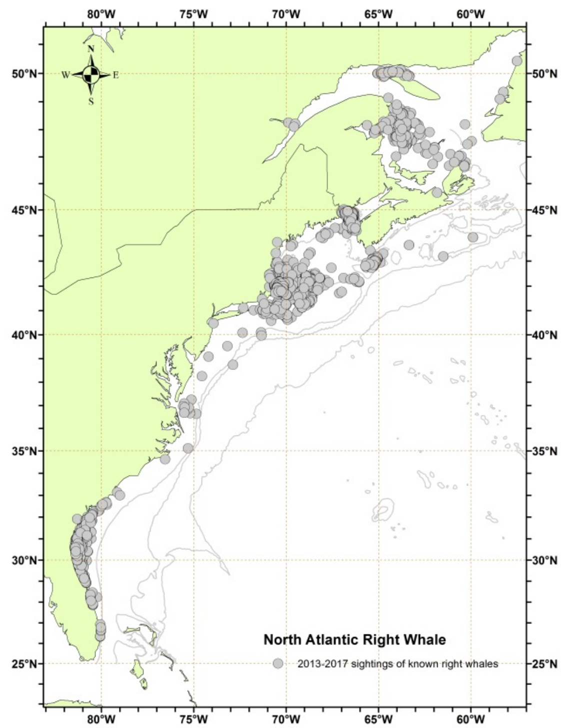

Figure 1. Distribution of sightings of known North Atlantic right whales, 2013-2017. Isobaths are the 100-m, 1000-m and 4000-m depth contours.

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. 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. However, Knowlton *et al.* (1992) reported several long-distance movements as far north as Newfoundland, the Labrador Basin, and southeast of Greenland. In addition, 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), and in the Azores (Silva *et al.* 2012). The September 1999 Norwegian sighting represents one of only two published sightings in the 20th century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described. A few published records from the Gulf of Mexico (Moore and Clark 1963; Schmidly *et al.* 1972; Ward-Geiger *et al*. 2011) likely represent occasional wanderings of individuals beyond the sole known calving and wintering ground in the waters of the southeastern U. S. The location of much of the population is unknown during the winter. Davis *et al.* (2017) recently pooled together detections from a large number of passive acoustic devices and documented broad-scale use of much more of the U.S. eastern seaboard than previously believed. Further, there has been an apparent shift in habitat use patterns (Davis *et al*. 2017). Surveys flown in an area from 31 to 160 km from the shoreline off northeastern Florida and southeastern Georgia since 1996 report the majority of right whale sightings occur within 90 km of the shoreline. One sighting occurred ~140 km offshore (NMFS unpub. data) and an offshore survey in March 2010 observed the birth of a right whale in waters 75 km off Jacksonville, Florida (Foley *et al.* 2011). Although habitat models predict that right whales are not likely to occur farther than 90 km from the shoreline (Gowan and Ortega-Ortiz 2015), the frequency with which right whales occur in offshore waters in the southeastern U.S. remains unclear.

Visual and acoustic surveys have demonstrated the existence of seven areas where western North Atlantic right whales aggregate seasonally: the coastal waters of the southeastern U.S.; the Great South Channel; Jordan Basin; Georges Basin along the northeastern edge of Georges Bank; Cape Cod and Massachusetts Bays; the Bay of Fundy; and the Roseway Basin on the Scotian Shelf (Brown *et al*. 2001; Cole *et al.* 2013). 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). 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). All of this work further demonstrates the highly mobile nature of right whales. Movements within and between habitats are extensive, and the area off the mid-Atlantic states is 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 rather 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 newborn was detected in Cape Cod Bay in 2012 (Center for Coastal Studies, Provincetown, MA USA, unpub. data).

Right whale calls have been detected by autonomous passive acoustic sensors deployed between 2005 and 2010 at three sites (Massachusetts Bay, Stellwagen Bank, and Jeffreys Ledge) in the southern Gulf of Maine (Morano *et al*. 2012, Mussoline *et al*. 2012). Comparisons between detections from passive acoustic recorders and observations from aerial surveys in Cape Cod Bay between 2001 and 2005 demonstrated that aerial surveys found whales on approximately two-thirds of the days during which acoustic monitoring detected whales (Clark *et al*. 2010). These data suggest that the current understanding of the distribution and movements of right whales in the Gulf of Maine and surrounding waters is incomplete. Additionally, the aforementioned apparent shift in habitat use patterns since 2010, highlighted by Davis *et al.* (2017), includes increased use of Cape Cod Bay (Mayo *et al.* 2018) and decreased use of the Great South Channel.

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 (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. Analysis of the sightings data has shown that the utilization of these areas has a strong seasonal component (Pace and Merrick 2008). Although right whales are 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). In 2016, the Northeastern U.S. Foraging Area Critical Habitat was expanded to include nearly all U.S. waters of the Gulf of Maine (81 FR 4837, 26 February 2016).

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 have detected fewer individuals in the Great South Channel and the Bay of Fundy. In addition, late winter use of a region south of Martha's Vineyard and Nantucket Islands was recently described (Leiter *et al.* 2017). A large increase in aerial surveys of the Gulf of St. Lawrence documented at least 34, 105, and 131 unique individuals using the region, respectively, during the summers of 2015, 2017, and 2018 (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 has been completed for >75% of all North Atlantic right whales identified through 2006. This work has improved our understanding of genetic variability, the number of reproductively active individuals, reproductive fitness, parentage, and relatedness of individuals (Frasier *et al.* 2007, 2009). One emerging result 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.* 2007). 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). However, a recent study compared photo-identification and pedigree genetic data for animals known or presumed to be alive during 1980-2016 and 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 2018. Using a hierarchical, state-space Bayesian open population model of these histories produced a median abundance value. The best abundance estimate available for the North Atlantic right whale stock is 428 individuals (95% credible intervals 406-447). 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 is used above to characterize that uncertainty (as opposed to a CV that may appear in other stock assessment reports).

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.* 2007). 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 428. The minimum population estimate as of January 2017 is 418 and stands as Nmin. The 17 known mortalities from 2017 are not accounted for in this estimate.

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 imilar 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 99.99% chance that abundance declined from 2011 to 2017 when the final estimate was 428 individuals. As noted above, there seems to have been a considerable change in right whale habitat use patterns in areas where most of the population has been observed in previous years 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, the chance of seeing an individual that is alive has decreased. However, the methods in Pace *et al.* (2017) account for changes in capture probability.

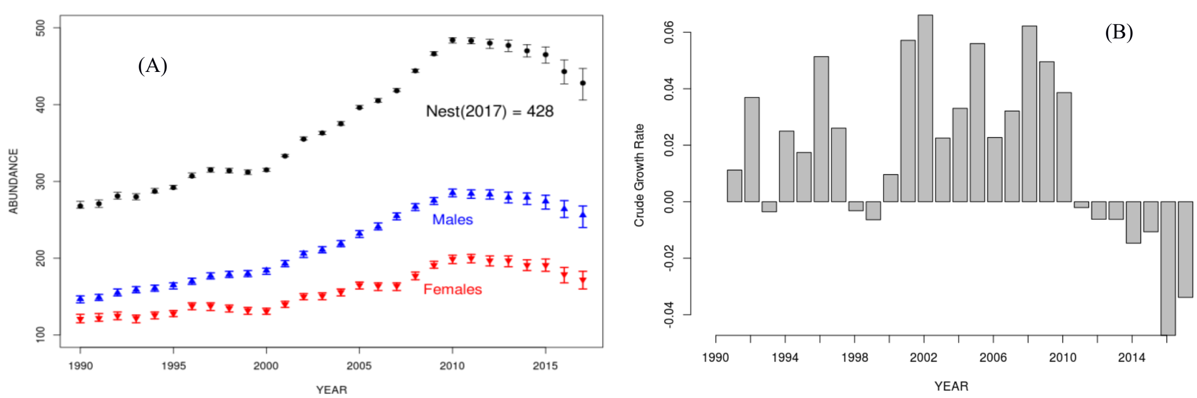

Figure 2. (A) Abundance estimates for North Atlantic right whales. Estimates are the median values of a posterior distribution from modeled capture histories. Also shown are sex-specific abundance estimates. Cataloged whales may include some but not all calves produced each year. (B) Crude annual growth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

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). 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, and 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%). Their 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, Kenny (2018) 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 be 588 by 2016.

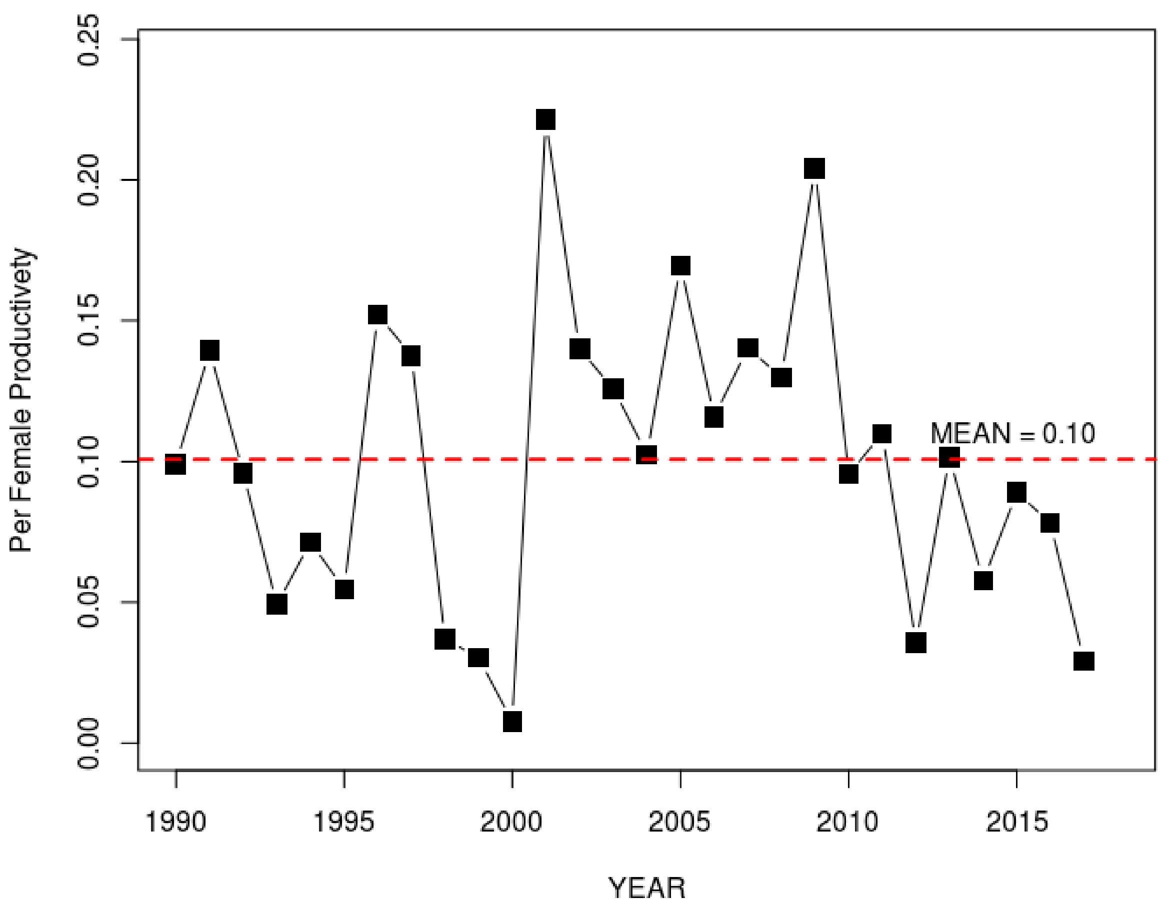

Figure 3. Productivity in the North Atlantic right whale population as characterized by calves detected/(estimated number of females).

North Atlantic right whales have thinner blubber than southern right whales off South Africa (Miller *et al.* 2011). Blubber thickness of male North Atlantic right whales (males were selected to avoid the effects of pregnancy and lactation) varied with *Calanus* abundance in the Gulf of Maine (Miller *et al.* 2011). Sightings of North Atlantic right whales correlated with satellite-derived sea-surface chlorophyll concentration (as a proxy for productivity), and calving rates correlated with chlorophyll concentration prior to gestation (Hlista *et al.* 2009). On a regional scale, observations of North Atlantic right whales correlate well with copepod concentrations (Pendleton *et al*. 2009). 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).

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). 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 418. The maximum productivity rate is 0.04, the default value for cetaceans. PBR for the Western Atlantic stock of the North Atlantic right whale is 0.8.

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2013 through 2017, the minimum rate of annual human-caused mortality and serious injury to right whales averaged 6.85 per year. This is derived from two components: 1) incidental fishery entanglement records at 5.55 per year, and 2) vessel strike records at 1.3 per year. Early analyses of the effectiveness of the ship 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 ship 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 are applied to previously reported vessel strikes between 2000 and 2017 (Henry *et al.* 2020), there is no apparent trend (Fig 4). However, the odds of an entanglement event are now increasing by 6.3% per year.Although PBR analyses in this SAR reflect data collected through 2016, There were 17 right whale mortalities in 2017 (Daoust *et al.* 2017). 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. Therefore, the decline in the right whale population will continue for at least an additional 2 years.

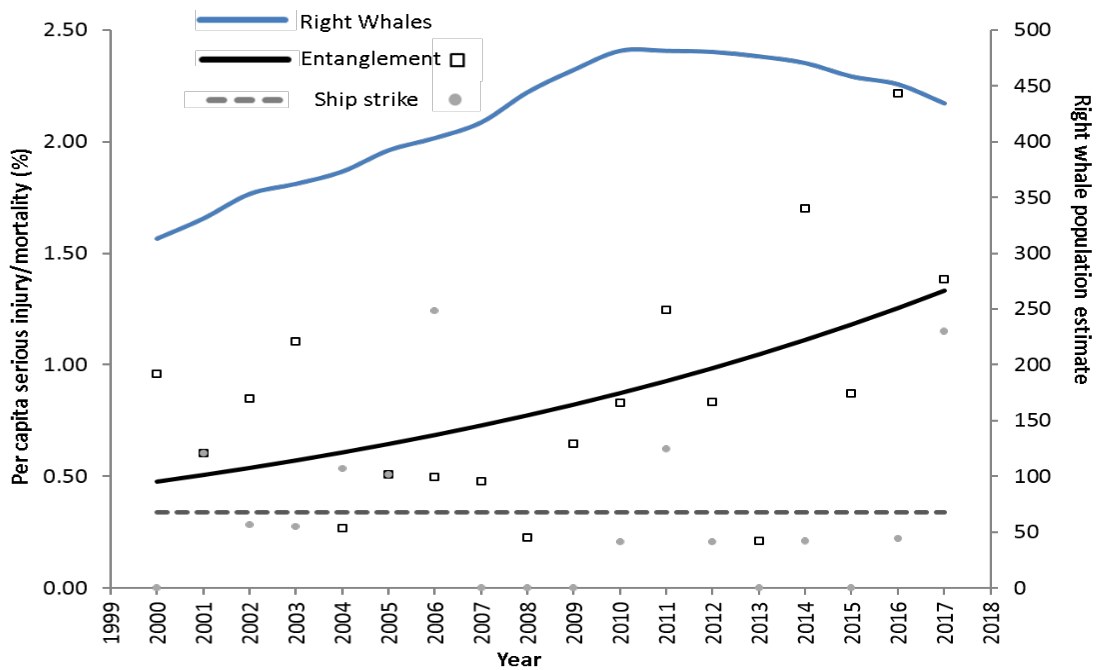

Figure 4. North Atlantic right whale serious injury/mortality rates from known sources 2000-2017. The right whale population trend is overlaid and referenced to right y-axis

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. It is important to stress that serious injury determinations are made based upon the best available information; these determinations may change with the availability of new information (Henry *et al.* 2020). For the purposes of this report, discussion is limited to those records considered confirmed human-caused mortalities or serious injuries. Annual rates calculated from detected mortalities should be considered a low-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. A key uncertainty is the fraction of the actual human-caused mortality represented by the detected serious injuries and mortalities. Research on small cetaceans has shown the actual number of deaths can be several times higher than that observed (Wells and Allen 2015; Williams *et al.* 2011). For North Atlantic right whales, estimates of the total mortality exceed or equal the number of detected serious injuries and mortalities (Figure 5) and currently 72% of mortalities since 2000 are estimated to have been observed. Because annual population estimates are now available (Pace *et al.* 2017), it is possible to estimate total annual mortality (and the number of undetected mortalities) by applying the basic population dynamic formula (Williams *et al.* 2002):

$N_{t+1} = N_t + B_t - D_t$

Where N_t is the number of animals in a population in year *t*, N_{t+1} is the number of animals in the population in year *t+1*, B_t is the number of births in the population in year *t*, and D_t is the number of deaths in the population in year *t*.

Solving for D_t yields: $D_t = N_t + B_t - N_{t+1}$ which can then be used to estimate undetected mortality as: D_t- observed deaths= undetected deaths.

The total mortality estimated described above is based on the assumption that all animals that exit from the population in the model are actual deaths and that all entries into the population are births. If immigration were occurring, new mature animals would be documented and captured in the estimate of B_t. There is a lack of any evidence for permanent emigration from the population. Temporary emigration (*e.g.* the animal is not observed in the survey area for multiple years) only adds to individual capture heterogeneity, which is accommodated by the model given the longevity of the data sets. Importantly, these assumptions are not novel to the total mortality estimate, but a core part of the published Pace *et al.* (2017) population estimate. A method to assign cause to these undetected mortalities is currently under development; as such these additional mortalities are not counted towards PBR at this time. Another uncertainty is assigning many of the detected entanglements to country of origin. Gear recovered is often not adequately marked and whales have been known to carry gear for long periods of time and over great distances before being detected.

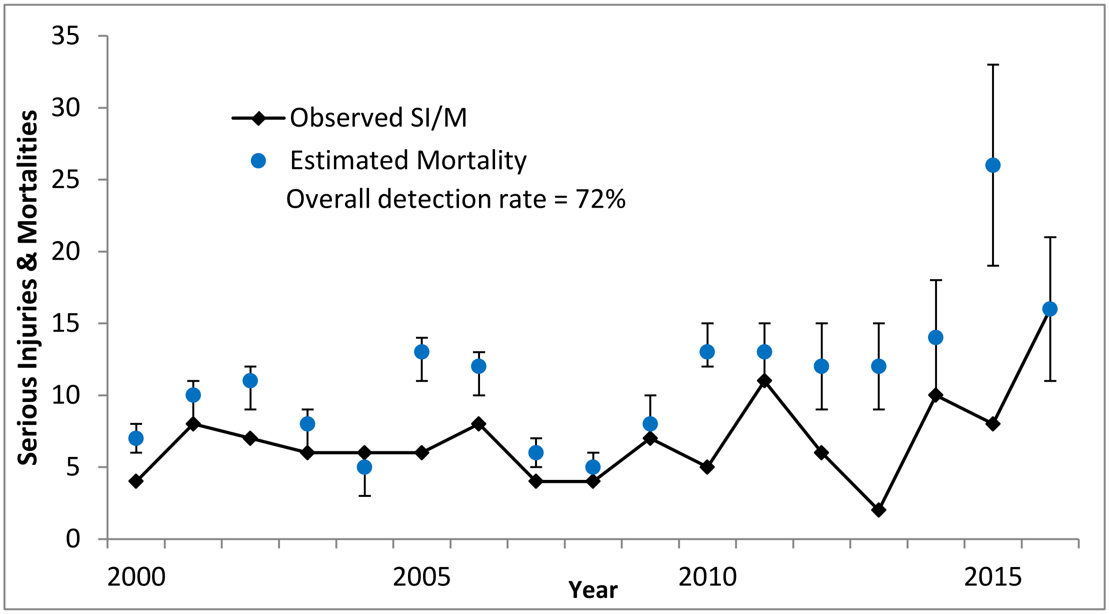

Figure 5. Time series of observed annual total serious injuries and mortalities (SI/M; black line) versus estimated total mortalities (blue points with associated error bars).

Background

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 assigned cause is based on the best judgment of the available data; additional information may result in revisions. When reviewing Table 1 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) in entanglements, several types of gear may be involved.

Further, 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 retarding 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 was attributable to human impacts (calves accounted for six deaths from ship strikes and two from entanglements). One should be cautious in applying these percentages as more than minimum rates as they only represent carcasses, and exclude serious injury which is highly skewed towards entanglement. A recent analysis of human-caused serious injury and mortality during 2000–2017 (Figure 4) 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).

Finally, entanglement or minor vessel collisions may not kill an animal directly, but may weaken or otherwise affect it so that it is more likely to become vulnerable to further injury. Serious injury determinations for large whales commonly include animals carrying gear when these entanglements are constricting or appear to interfere with foraging (Henry *et al*. 2020).

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 (Table 1). From 2013 through 2017, 28 of those examined records of mortality or serious injury (including records from both U.S. and Canadian waters, prorated to 27.75 using serious injury guidelines) involved entanglement or fishery interactions. For this time frame, the average reported mortality and serious injury to right whales due to fishery entanglement was 5.55 whales per year. 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 of disentanglement teams averted a likely serious-injury determination. Seven serious injuries were prevented by intervention during 2013–2017 (Henry *et al.* 2020). Sometimes, even with disentanglement, an animal may die of injuries sustained from fishing gear. A female yearling right whale, #3107, was first sighted with gear wrapping its caudal peduncle on 6 July 2002 near Briar Island, Nova Scotia. Although the gear was removed on 1 September by the New England Aquarium disentanglement team, and the animal seen alive during an aerial survey on 1 October, its carcass washed ashore at Nantucket on 12 October 2002 with deep entanglement injuries on the caudal peduncle. Additionally, but infrequently, a whale listed as seriously injured becomes gear-free without a disentanglement effort and is seen later in reasonable health. Such was the case for whale #1980, listed as a serious injury in 2008 but seen gear-free and apparently healthy in 2011.

Incidents of entanglements in waters of Atlantic Canada and the U.S. east coast were summarized by Read (1994) and Johnson *et al.* (2005). Despite the long history of known fishing interactions, the only bycatch of a right whale observed by the Northeast Fisheries Observer Program was in the pelagic drift gillnet fishery in 1993. No mortalities or serious injuries have been documented by fisheries observers in any of the other fisheries monitored by NMFS.

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. About a quarter of the individuals identified in each year (26%) were entangled in that year. 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. More recently, 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 over time that efforts made since 1997 to reduce right whale entanglement have not worked. Working from 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. Vessel strikes and entanglements were the two leading causes of death for known mortalities of right whales for which a cause of death could be determined. Across all 8 species of large whales, there was no detectable change in causes of anthropogenic mortality over time (van der Hoop *et al*. 2012). Pace *et al.* (2015) analyzed entanglement rates and serious injuries due to entanglement during 1999–2009 and 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. Assessment efforts are underway but rely on 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 2013 through 2017 have been summarized in Table 1. For this time frame, the average reported mortality and serious injury to right whales due to vessel strikes was 1.3 whales per year.

An Unusual Mortality Event was established for North Atlantic right whales in June 2017 due to elevated stranding along the Atlantic coast, especially in the Gulf of St. Lawrence region of Canada (https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2018-north-atlantic-right-whale-unusual-mortality-event).

Table 1. Confirmed human-caused mortality and serious injury records of right whales: 2013–2017[a]

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
07/12/2013	Prorated Injury	3123	off Virginia Beach, VA	EN	.75	XU	NR	Constricting gear cutting into mouthline; Partially disentangled; final configuration unknown. No resights post Jul/2013
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	.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.
1/29/2016	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.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
5/19/2016	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.
5/03/2016	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.
7/26/2016	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.
8/1/2016	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.
8/13/2016	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.
8/16/2016	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.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
8/28/2016	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.
8/31/2016	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.
9/23/2016	Mortality	3694	off Seguin Island, MA	EN	1	XC	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.

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
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.
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.

Assigned Cause	**Five-year mean (US/CN/XU/XC)**
Vessel strike	01.3 (0.50/ 0.80/ 0.00/ 0.00)
Entanglement	5.55 (0.20/ 1.20/ 2.45/ 1.70)

a. For more details on events please see Henry *et al.* 2020.
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.

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 (see 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.

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HUMPBACK WHALE (*Megaptera novaeangliae*): Gulf of Maine Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

In the western North Atlantic, humpback whales feed during spring, summer and fall over a geographic range encompassing the eastern coast of the United States (including the Gulf of Maine), the Gulf of St. Lawrence, Newfoundland/Labrador, and western Greenland (Katona and Beard 1990). Other North Atlantic feeding grounds occur off Iceland and in the Norwegian Sea, including off northern Norway, Bear Island, Jan Mayen, and Franz Josef Land (Christensen *et al.* 1992; Palsbøll *et al.* 1997). These six regions represent relatively discrete subpopulations, fidelity to which is determined matrilineally (Clapham and Mayo 1987), which is supported by studies of the mitochondrial genome (Palsbøll *et al.* 1995; Palsbøll *et al.* 2001) and individual animal movements (Stevick *et al*. 2006). During the 2002 Comprehensive Assessment of North Atlantic humpback whales, the International Whaling Commission acknowledged the evidence for treating the Gulf of Maine as a separate management unit (IWC 2002).

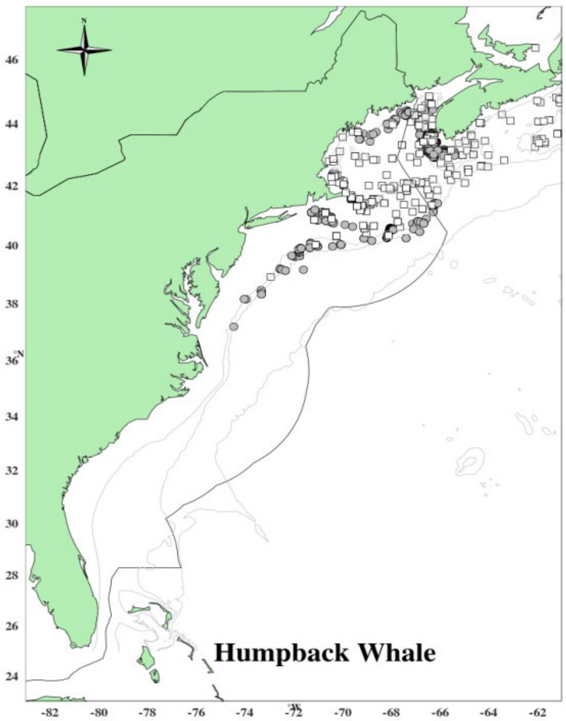

Figure 1. Distribution of humpback whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares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

During the summers of 1998 and 1999, the Northeast Fisheries Science Center conducted surveys for humpback whales on the Scotian Shelf to establish the occurrence and population identity of the animals found in this region, which lies between the well-studied populations of the Gulf of Maine and Newfoundland. Photographs from both surveys were compared to both the overall North Atlantic Humpback Whale Catalog and a large regional catalog from the Gulf of Maine (maintained by the College of the Atlantic and the Center for Coastal Studies, respectively); this work is summarized in Clapham *et al.* (2003). The match rate between the Scotian Shelf and the Gulf of Maine was 27% (14 of 52 Scotian Shelf individuals from both years). Comparable rates of exchange were obtained from the southern (28%, n=10 of 36 whales) and northern (27%, *n*=4 of 15 whales) ends of the Scotian Shelf (one whale was observed in both areas). In contrast, all of the 36 humpback whales identified by the same NMFS surveys elsewhere in the Gulf of Maine (including Georges Bank, southwestern Nova Scotia, and the Bay of Fundy) had been previously observed in the Gulf of Maine region. The sighting histories of the 14 Scotian Shelf whales matched to the Gulf of Maine suggested that many of them were transient through the latter area. There were no matches between the Scotian Shelf and any other North Atlantic feeding ground, except the Gulf of Maine; however, instructive comparisons are compromised by the often low sampling effort in other regions in recent years. Overall, it appears that the northern range of many members of the Gulf of Maine stock does not extend onto the Scotian Shelf. Some uncertainty in the stock definition for the Gulf of Maine stock of humpback whales is where along the Scotian shelf stock boundaries are drawn in a relatively contiguous range. However, exact placement of the boundary should have little effect on conservation status because the whales along the southern Scotian shelf represent a relatively small fraction of either the Gulf of Maine or Labrador stocks.

During winter, whales from most North Atlantic feeding areas (including the Gulf of Maine) mate and calve in the West Indies, where spatial and genetic mixing among feeding groups occurs (Katona and Beard 1990; Clapham *et al.* 1993; Palsbøll *et al.* 1997; Stevick *et al.* 1998; Kennedy *et al.* 2013). Some whales using eastern North Atlantic feeding areas migrate to the Cape Verde Islands (Reiner *et al.* 1996; Wenzel *et al.* 2009; Stevick *et al.* 2016), and some individuals have been recorded in both the Cape Verde Islands and the southeast Caribbean (Stevick e*t al*. 2016). In the West Indies, the majority of whales are found in the waters of the Dominican Republic, notably on Silver Bank and Navidad Bank, and in Samana Bay (Balcomb and Nichols 1982; Whitehead and Moore 1982; Mattila *et al.* 1989, 1994). Humpback whales also are found at much lower densities throughout the remainder of the Antillean arc (Winn *et al.* 1975; Levenson and Leapley 1978; Price 1985; Mattila and Clapham 1989). Although recognition of 2 breeding areas for North Atlantic humpbacks is the prevailing model, our knowledge of breeding season distribution is far from complete (see Smith and Pike 2009; Stevick *et al.* 2016).

Not all whales from this stock migrate to the West Indies every winter, because significant numbers of animals are found in mid- and high-latitude regions at this time (Clapham *et al.* 1993; Swingle *et al.* 1993) and some individuals have been sighted repeatedly within the same winter season (Clapham *et al.* 1993; Robbins 2007). Acoustic recordings made within the Massachusetts Bay area detected some level of humpback song and non-song sounds in almost all months, with two prominent periods, March through May and September through December (Clark and Clapham 2004; Vu *et al*. 2012; Murray *et al*. 2013). This pattern of acoustic occurrence, especially for song, confirms the presence of male humpback whales in the area (a mid-latitude feeding ground) during periods that bracket male occurrence in the Caribbean region, where singing is highest during winter months. A complementary pattern of humpback singer occurrence was observed during the January–May period in deep-ocean regions north and west of the Caribbean and to the east of Bermuda during April (Clark and Gagnon 2002). These acoustic observations from both coastal and deep-ocean regions support the conclusion that at least male humpbacks are seasonally distributed throughout broad regions of the western North Atlantic. In addition, photographic records from Newfoundland have shown a number of adult humpbacks remain there year-round, particularly on the island's north coast.

Within the U.S. Atlantic EEZ, humpback whales have been sighted well away from the Gulf of Maine. Sightings of humpback whales in the vicinity of the Chesapeake and Delaware Bays occurred in 1992 (Swingle *et al.* 1993). Wiley *et al.* (1995) reported that 38 humpback whale strandings occurred during 1985–1992 in the U.S. mid-Atlantic and southeastern states. Humpback whale strandings increased, particularly along the Virginia and North Carolina coasts, and most stranded animals were sexually immature; in addition, the small size of many of these whales strongly suggested that they had only recently separated from their mothers. Wiley *et al.* (1995) concluded that these mid-Atlantic areas were becoming an increasingly important habitat for juvenile humpback whales. For the period 2013–2017, there are records of 95 humpback whale strandings between Maine and Florida in the Marine Mammal Health and Stranding Response database (accessed 23 October 2018). There have also been a number of wintertime humpback sightings in coastal waters of the southeastern U.S. (Zoodsma *et al.* 2016; Surrey-Marsden *et al.* 2018) Other sightings of note include 46 sightings of humpbacks in the New York-New Jersey Harbor Estuary documented between 2011 and 2016 (Brown *et al*. 2017). Multiple humpbacks were observed feeding off Long Island during July of 2016 (https://www.greateratlantic.fisheries.noaa.gov/mediacenter/2016/july/26_humpback_whales_visit_new_york.html, accessed 28 April 2017) and there were sightings during November–December 2016 near New York City (https://www.greateratlantic.fisheries.noaa.gov/mediacenter/2016/december/09_humans_and_humpbacks_of_new_york_2.html, accessed 28 April 2017). Additional sightings occurring during summer (about July – August) along the shelf break east of New Jersey and New York during the NEFSC abundance surveys have been increasing since about 2004 (2016 survey described below, Palka 2020). Whether the increased numbers of sightings represent a distributional change, or are simply due to an increase in sighting effort and/or whale abundance, is unknown.

Stock identity of individuals observed off southeastern and mid-Atlantic states was investigated using fluke photographs of living and dead whales observed in the region (Barco *et al.* 2002). In this study, photographs of 40 whales (alive or dead) were of sufficient quality to be compared to catalogs from the Gulf of Maine (i.e., the closest feeding ground) and other areas in the North Atlantic. Of 21 live whales, 9 (43%) matched to the Gulf of Maine, 4 (19%) to Newfoundland, and 1 (4.8%) to the Gulf of St Lawrence. Of 19 dead humpbacks, 6 (31.6%) were known Gulf of Maine whales. Although the population composition of the mid-Atlantic is apparently dominated by Gulf of Maine whales, lack of photographic effort in Newfoundland makes it likely that the observed match rates under-represent the true presence of Canadian whales in the region. Barco *et al.* (2002) suggested that the mid-Atlantic region primarily represents a supplemental winter feeding ground used by humpbacks. With populations recovering, additional surveys that include photo identification and genetic sampling should be conducted to determine which stocks are currently using the mid-Atlantic region.

In New England waters, feeding is the principal activity of humpback whales, and their distribution in this region has been largely correlated to abundance of prey species, although behavior and bathymetry are factors influencing foraging strategy (Payne *et al.* 1986, 1990). Humpback whales are frequently piscivorous when in New England waters, feeding on herring (*Clupea harengus*), sand lance (*Ammodytes* spp.), and other small fishes. In the northern Gulf of Maine, euphausiids are also frequently taken (Paquet *et al.* 1997). Humpback whales were densest over the sandy shoals in the southwestern Gulf of Maine favored by the sand lance during much of the late 1970s and early 1980s, and humpback distribution appeared to have shifted to this area (Payne *et al.* 1986). An apparent reversal began in the mid-1980s, and herring and mackerel increased as sand lance again decreased (Fogarty *et al.* 1991). Humpback whale abundance in the northern Gulf of Maine increased markedly during 1992–1993, along with a major influx of herring (P. Stevick, pers. comm.). Humpback whales were few in nearshore Massachusetts waters in the 1992–1993 summer seasons. They were more abundant in the offshore waters of Cultivator Shoal, the Northeast Peak of Georges Bank, and Jeffreys Ledge; these latter areas are traditional locations of herring occurrence. In 1996 and 1997, sand lance and therefore humpback whales were once again abundant in the Stellwagen Bank area. However, unlike previous cycles, when an increase in sand lance corresponded to a decrease in herring, herring remained relatively abundant in the northern Gulf of Maine, and humpbacks correspondingly continued to occupy this portion of the habitat, where they also fed on euphausiids (Weinrich *et al.* 1997). Recent observations of humpbacks foraging along the shelf break off New York and New Jersey may be indicative of changing forage distribution**.**

POPULATION SIZE

The best abundance estimate available for the Gulf of Maine humpback whale stock is 1,396 (95% credible intervals 1363−1429). This is based on a 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 data through October 2016. The median abundance value was produced using a hierarchical, state-space Bayesian open population model of these histories.

Gulf of Maine stock - Earlier estimates

Please see Appendix IV for earlier estimates. As recommended in the 2016 guidelines for preparing stock assessment reports (NMFS 2016), estimates older than eight years are deemed unreliable and should not be used for PBR determinations.

Gulf of Maine Stock - Recent surveys and abundance estimates

Humpback whales are uniquely identifiable based primarily on coloration patterns of the ventral side of the fluke and identification can be augmented by other features such as dorsal fin shape, scars and genetic data (Smith *et al.* 1999). A recent count of the minimum number alive (MNA) for 2015 was produced by counting the number of unique individuals seen in 2015 in the Gulf of Maine stock area as well as seen both before and after 2015 (data provided by J. Robbins, Center for Coastal Studies, Provincetown, MA, USA). The humpback MNA for 2015 was 896 and includes not only cataloged whales but some calves born in 2015 but not yet identifiable. By comparison, an abundance of 335 (CV=0.42) humpback whales was estimated from a line-transect survey conducted during June–August 2011 by ship and plane (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines over waters north of New Jersey and shallower than the 100-m depth contour through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a two-simultaneous-team data collection procedure, which allows estimation of abundance corrected for perception bias (Laake and Borchers 2004). Estimation of abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009). This estimate did not include the portion of the Scotian Shelf that is known to be part of the range used by Gulf of Maine humpback whales. This estimate should not be compared to previous estimates that were derived using a different methodology. The now-outdated estimate of 823 humpbacks in the Gulf of Maine and Bay of Fundy in 2008 was based on a minimum number alive calculation. While that type of estimate is generally more accurate than one derived from line-transect survey, the 2016 GAMMS guidelines (NMFS 2016) notes the decline of confidence in the reliability of abundance estimates older than eight years. For this report, two new independent estimates are available from different methods- one based upon ship and aerial line transect surveys, and a second from applying mark and recapture methods to photo identification records from the J. Robbins studies (Robbins and Pace 2018).

An abundance estimate of 2,368 (CV=0.48) humpback whales was generated from a shipboard and aerial survey conducted during 27 June–28 September 2016 (Figure 2, Palka 2020) in a region covering 425,192 km^2. The aerial portion covered 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 covered 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) using standard mark-recapture distance sampling with covariates to assist in defining the detection function. The estimates were also corrected for availability bias which was estimated from dive patterns recorded from tagged humpbacks. The abundance resulting from the aerial survey in the U.S. portion of the Gulf of Maine was 1,372 (CV=0.70), where the availability bias correction factor was 1.541 (CV=0.185); thus, the uncorrected abundance was 890 (CV=0.68). The abundance resulting from the shipboard survey on the shelf break was 996 (CV=0.59), where the availability bias correction factor was 1.0.

Abundance estimates of 8,439 (CV=0.49) for the Newfoundland/Labrador portion and 1,854 (CV=0.40) for the Bay of Fundy/Scotian Shelf/Gulf of St. Lawrence portion were generated from the Canadian Northwest Atlantic International Sightings Survey (NAISS) survey conducted in August–September 2016. 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).

According to Clapham *et al.* (2003), and as has been done in previous Stock Assessment Reports, the abundance of the Gulf of Maine humpback whale when derived from visual sighting survey data would consist of those from the U.S. waters (2,368 (CV=0.48)) plus 2/3 of the humpback whales in the Canadian Bay of Fundy and Scotian shelf up to about Halifax, Nova Scotia. The Canadian portion of the Gulf of Maine stock has not been estimated, though the number of sightings of Gulf of Maine humpbacks from the Canadian 2016 NAISS survey are approximately 0.6 of the Bay of Fundy/Scotian Shelf/Gulf of St. Lawrence portion of the Canadian 2016 NAISS estimate reported above. Based on this, one might estimate 742 ($\approx$ 1854 * 2/3 * 0.6) for the Canadian portion of the Gulf of Maine humpback population by line transect methodology as a rough number to add to the estimate from U.S. waters.

As an alternative approach to estimating whale abundance, we analyzed the photo-identification database (Robbins and Pace 2018) and applied mark and recapture methods using a state-space model of the sighting histories of individual whales following the methodology described in Pace et al. 2017.

Sightings histories of Gulf of Maine humpback whales were constructed from the photo-ID recapture database as it existed in April 2019. The data were provided by an annual spatially arranged survey dedicated to gathering photo-id data on Gulf of Maine humpbacks. The estimation process was greatly enhanced by using photographic captures from sources other than the primary survey including additional research efforts by the principal survey team but outside of the dedicated survey effort, other cetacean research groups, and cooperating whale watch vessels. These later data were used to inform the known state matrix. All sightings from the primary survey were bounded by the hatched area noted on Figure 2 for capture-mark-recapture (CMR) sampling strata. A hierarchical, state-space Bayesian open population model of these histories produced a median abundance value of 1,396 individuals (95% credible intervals 1363−1429). 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 data structure. Because the statistically-based uncertainty is asymmetric about N, the credible interval is used above to characterize that uncertainty (as opposed to a CV that may appear in other stock assessment reports).

The CMR estimate of 1,396 stands as the best available estimate for this stock assessment report for several reasons. First it is in agreement with updated line transect survey estimate of 2,368, which has a large CV that ranges from 5,781 to 970. Second, the CMR estimate provides a tighter confidence interval and therefore is more precise. Third, the long term nature of the CMR database enabled the calculation of historical annual population estimates backwards in time through 2000, thus allowing trend analysis. Furthermore, humpback whales meet the key criteria for applying mark and recapture methodology as an animal with an established stock and home-range that is also uniquely identifiable. There is some spatial difference in sampling strata between the CMR and Line transect survey, which result in the CMR estimate better representing the population abundance. The line transect estimate is most accurate when fully sampling the defined seasonal range of the stock. The current estimate includes many sightings from the continental shelf areas east of New York and New Jersey. While, this region is typically included for the Gulf of Maine stock particularly when assigning cases of anthropogenic mortality, further research is required to confirm that surveys south of the Gulf of Maine might detect animals from other stocks using U.S. waters during good forage conditions. At the same time the aerial portion of the line transect estimate did not go into the Canadian waters in the Bay of Fundy region, nor east of the Hague line. The CMR estimate can be generated from a sampling a subregion of a species range, if that region is used by the entire population, as was the case here where sampling of the GOM humpback stock was conducted throughout the Gulf of Maine. CMR estimates are potentially subject to error if there is permanent immigration/emigration into or out of the population. However there is little evidence for this, given a lack of photo ID reports for GOM animals observed permanently relocating to other stock regions which would be indicative of emigration. Similarly there is little evidence for immigration, given the lack of regular 'new entrants' into the population as adults. Temporary emigration (*e.g.* the animal is not observed in the survey area for multiple years) only adds to individual capture heterogeneity, which is accommodated by the model given the longevity of the data sets. Given the efficiency of the method, NMFS should consider investment to ensure the continuation of this data record.

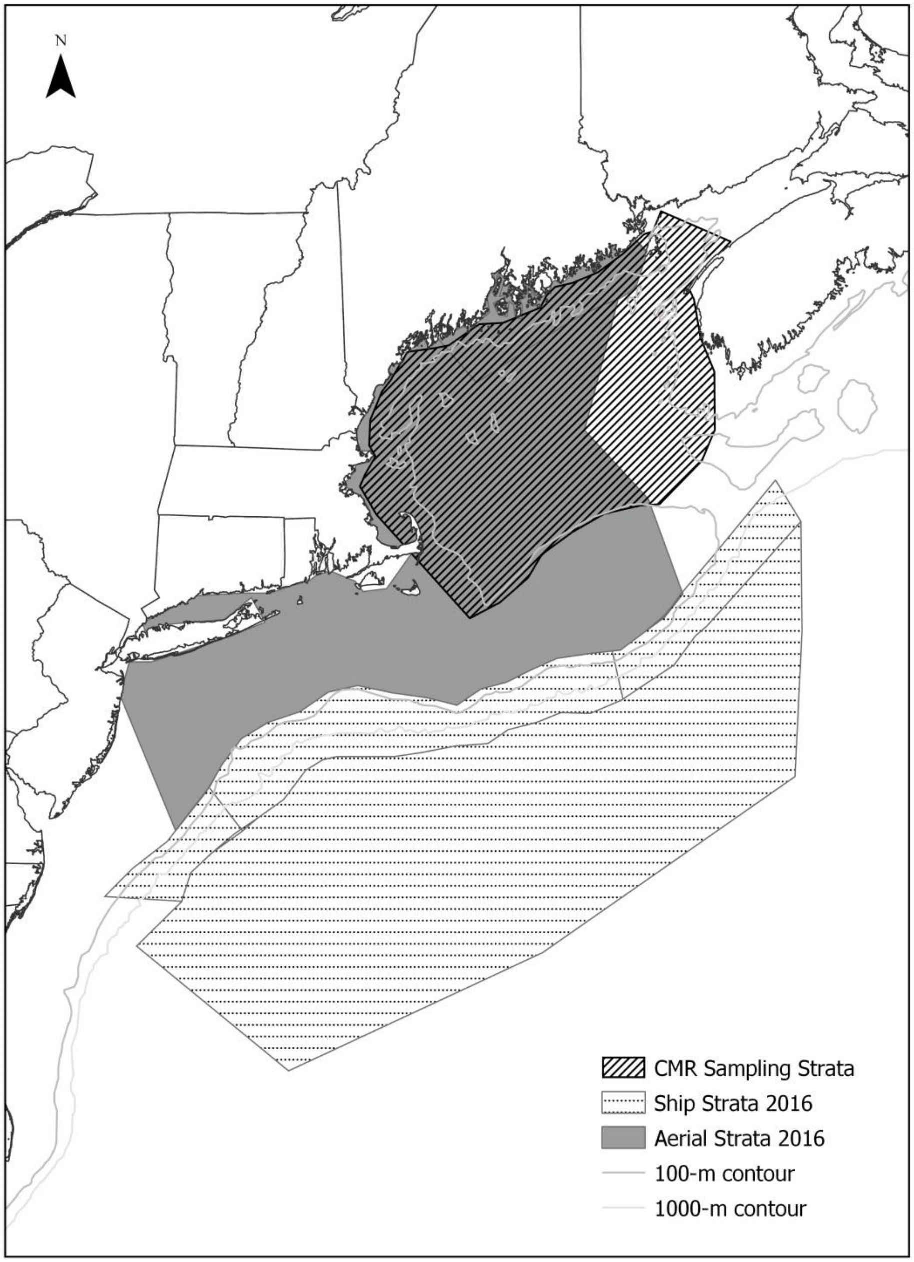

Figure 2. Map of sampling strata for two reported humpback whale population estimation methods. Capture-Mark-Recapture (CMR) strata identifies long term photo ID survey area in Gulf of Maine waters (note, western edge of CMR strata extends to coastline). Line transect strata of 2016 aerial (US waters) and ship-based surveys are also defined.

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 CMR 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 minimum population estimate is 1,380 using the CMR method.

Table 1. Summary of abundance estimates for Gulf of Maine humpback whales with month, year, and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

Month/Year	Type	N_{best}	CV
Jun–Aug 2011	Virginia to lower Bay of Fundy	335	0.42
Jun–Oct 2015	Gulf of Maine and Bay of Fundy	896	0
Jun-Sep 2016	Central Virginia to lower Bay of Fundy	2,368	0.48
Mid-summer 2016	State-space mark-recapture estimates	1,396	n/a

Current Population Trend

As detailed below, previous analyses concluded that the Gulf of Maine humpback whale stock is characterized by a positive trend in abundance. This was consistent with an estimated average trend of 3.1% (SE=0.005) in the North Atlantic population overall for the period 1979–1993 (Stevick *et al.* 2003), although there are no feeding-area-specific estimates. An analysis of demographic parameters for the Gulf of Maine (Clapham *et al.* 2003) suggested a lower rate of increase than the 6.5% reported by Barlow and Clapham (1997). Examination of the abundance estimates for the years 2000‒2016 (Figure 3) suggests that abundance increased at about 2.8% per annum (Robbins and Pace 2018).

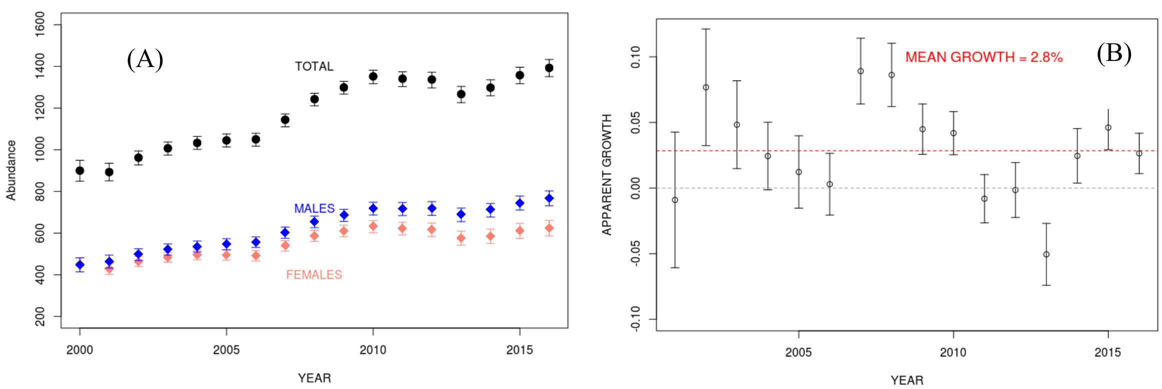

Figure 3. (A) Abundance estimates for Gulf of Maine humpback whales using the methodology described in Pace *et al.* 2017. Estimates are the median values of a posterior distribution from modeled capture histories. Also shown are sex-specific abundance estimates. Cataloged whales may include some but not all calves produced each year. (B) Crude annual growth rates from the abundance values.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Zerbini *et al.* (2010) reviewed various estimates of maximum productivity rates for humpback whale populations, and, based on simulation studies, they proposed that 11.8% be considered as the maximum rate at which the species could grow. Barlow and Clapham (1997), applying an interbirth interval model to photographic mark-recapture data, estimated the population growth rate of the Gulf of Maine humpback whale stock at 6.5% (CV=0.012). Maximum net productivity is unknown for this population, although a theoretical maximum for any humpback population can be calculated using known values for biological parameters (Brandão *et al.* 2000; Clapham *et al.* 2001). For the Gulf of Maine stock, data supplied by Barlow and Clapham (1997) and Clapham *et al.* (1995) give values of 0.96 for survival rate, 6 years as mean age at first parturition, 0.5 as the proportion of females, and 0.42 for annual pregnancy rate. From this, a maximum population growth rate of 0.072 is obtained according to the method described by Brandão *et al.* (2000). This suggests that the observed rate of 6.5% (Barlow and Clapham 1997) is close to the maximum for this stock.

Clapham *et al.* (2003) updated the Barlow and Clapham (1997) analysis using data from the period 1992 to 2000. The population growth estimate was either 0% (for a calf survival rate of 0.51) or 4.0% (for a calf survival rate of 0.875). Although uncertainty was not strictly characterized by Clapham *et al.* (2003), their work might reflect a decline in population growth rates from the earlier study period. More recent work by Robbins (2007) places apparent survival of calves at 0.664 (95% CI: 0.517-0.784), a value between those used by Barlow and Clapham (1997) and in addition found productivity to be highly variable and well less than maximum.

Despite the uncertainty accompanying the more recent estimates of observed population growth rate for the Gulf of Maine stock, the maximum net productivity rate was assumed to be 6.5% calculated by Barlow and Clapham (1997) because it represents an observation greater than the default of 0.04 for cetaceans (Barlow *et al.* 1995) but is conservative in that it is well below the results of Zerbini *et al.* (2010).

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 Gulf of Maine stock is 1,380 whales. The maximum productivity rate is 0.065 (based on Barlow and Clapham 1997). In the 2015 and prior SARs, the recovery factor was 0.10 because this stock was listed as an endangered species under the Endangered Species Act (ESA). The 2016 revision to the ESA listing of humpback whales concluded that the West Indies Distinct Population Segment (of which the Gulf of Maine stock is a part) did not warrant listing (81 FR 62259, September 8, 2016). Consequently, in the 2016 SAR the recovery factor was revised to 0.5, the default value for stocks of unknown status relative to OSP (Wade and Angliss 1997). PBR for the Gulf of Maine humpback whale stock is 22.

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2013 through 2017, the minimum annual rate of detected human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 12.15 animals per year. This value includes incidental fishery interaction records, 7.75; and records of vessel collisions, 4.4 (Table 2; Henry *et al.* 2020). In addition to the total 60.75 (38.75 entanglement, 22 vessel) anthropogenic mortalities and serious injuries for this time period, 11 carcasses examined found no detected human interaction. In contrast to stock assessment reports before 2007, these averages include humpback mortalities and serious injuries that occurred in the southeastern and mid-Atlantic states that could not be confirmed as involving members of the Gulf of Maine stock. In past reports, only events involving whales confirmed to be members of the Gulf of Maine stock were counted against the PBR. Starting in the 2007 report, we assumed whales were from the Gulf of Maine unless they were identified as members of another stock. At the time of this writing, no whale was identified as a member of another stock. These determinations may change with the availability of new information. Canadian records from the southern side of Nova Scotia were incorporated into the mortality and serious injury rates, to reflect the effective range of this stock as described above. For the purposes of this report, the discussion is primarily limited to those records considered to be confirmed human-caused mortalities or serious injuries.

It is important to stress that serious injury determinations are made based upon the best available information; these determinations may change with the availability of new information (Henry *et al.* 2020). For the purposes of this report, takes against PBR are limited to those records considered confirmed human-caused mortalities or serious injuries. Annual rates calculated from detected mortalities should be considered a low-biased accounting of human-caused mortality; they represent a definitive lower bound. Detections of mortality and serious injury are haphazard, incomplete, and not the result of a designed sampling scheme. A key uncertainty is the fraction of the actual human-caused mortality and serious injury represented by the detected mortalities and serious injuries. Research on small cetaceans has shown the actual number of deaths can be several times higher than that observed (Wells and Allen 2015; Williams *et al.* 2011). Because annual population estimates are now available (Pace *et al*. 2017), it is possible to estimate total annual mortality (and the number of undetected.

$N_{t+1} = N_t + B_t - D_t$

Where N_t is the number of animals in a population in year *t*, N_{t+1} is the number of animals in the population in year *t+1*, B_t is the number of births in the population in year *t*, and D_t is the number of deaths in the population in year *t*.

Solving for D_t yields: $D_t = N_t + B_t - N_{t+1}$ which can then be used to estimate undetected mortality as: D_t- observed deaths= undetected deaths

The total mortality estimated described above is based on the assumption that all animals that exit from the population in the model are actual deaths and that all entries into the population are births. If immigration were occurring, new mature animals would be documented and captured in the estimate of B_t. The total mortality estimate assumes all departures from the population are deaths, given the lack of any evidence for emigration from the population. Temporary emigration (*e.g.* the animal is not observed in the survey area for multiple years) only adds to individual capture heterogeneity, which is accommodated by the model given the longevity of the data sets. Importantly, these assumptions are not novel to the total mortality estimate, but a core part of the published Pace *et al*. (2017) method. A method to assign cause to these undetected mortalities is currently under development; as such these additional mortalities are not counted towards PBR at this time. Regardless, these estimates exceed or equal the number of detected serious injuries and mortalities (Figure 4) and currently roughly 20% of mortalities since 2000 are estimated to have been observed. For all the mortality observed in humpbacks, the current minimum fraction of anthropogenic mortality is 0.85. If this proportion were assigned to all unseen mortalities, the estimated annual anthropogenic mortality for this time period would be 53 and exceed PBR. While NMFS will be working to publish methodology for apportioning unseen mortality, it is worth noting that anthropogenic mortality in humpbacks would still exceed PBR if only 0.37 of unseen mortality were attributed to anthropogenic causes and it is very likely that is has exceeded PBR for the past several years (Figure 4).

There is mounting evidence that humpback whales have been over PBR for some time, and likely will be formally determined to be so in a future report. This is further supported by the NMFS declaration of Unusual Mortality Event No. 63.7 which includes cases from 2016 to the time of this writing in 2019 (https://www.fisheries.noaa.gov/national/marine-life-distress/2016-2019-humpback-whale-unusual-mortality-event-along-atlantic-coast).The literature and review of records described here suggest that there are significant human impacts beyond those recorded in the data assessed for serious injury and mortality. For example, a study of entanglement-related scarring on the caudal peduncle of 134 individual humpback whales in the Gulf of Maine suggested that between 48% and 65% had experienced entanglements (Robbins and Mattila 2001) and that 12-16% encounter gear annually (Robbins 2012).

To better assess human impacts (both vessel collision and commercial fishery mortality and serious injury) there needs to be greater emphasis on the timely recovery of carcasses and complete necropsies. Decomposed and/or unexamined animals (e.g., carcasses reported but not retrieved or no necropsy performed) represent 'lost data', some of which may relate to human impacts.

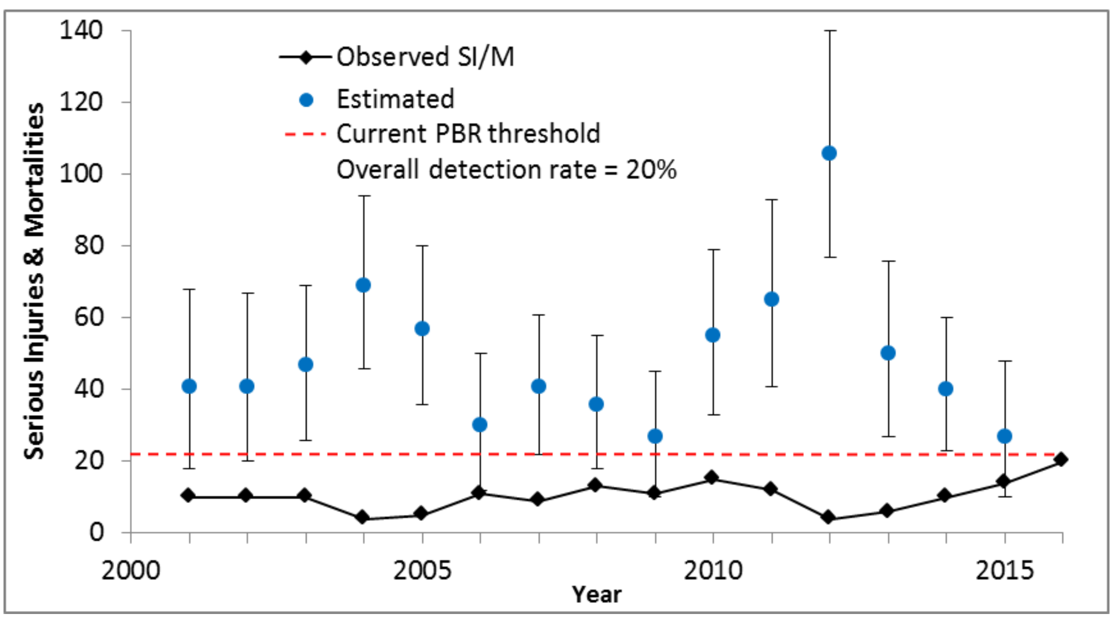

Figure 4. Time series of observed annual total serious injuries and mortalities (SI/M; bottom black line) observed versus total annual estimated mortalities (blue circles with associated error bars).Dashed line indicates current PBR threshold of 22.

Background

As with right whales, human impacts (vessel collisions and entanglements) may be slowing recovery of the humpback whale population. Van der Hoop *et al.* (2013) reviewed 1,762 mortalities and serious injuries recorded for 8 species of large whales in the Northwest Atlantic for the 40 years 1970–2009. Of 473 records of humpback whales, cause of death could be attributed for 203. Of the 203, 116 (57%) mortalities were caused by entanglements in fishing gear, and 31 (15%) were attributable to vessel strikes.

Inferences made from scar prevalence and multistate models of GOM humpback whales report that (1) younger animals are more likely to become entangled than adults, (2less than 10% of humpback entanglements are ever reported, and (3) 3% of the population may be dying annually as the result of entanglements (Robbins 2009, 2010, 2011, 2012). Humpback whale entanglements also occur in relatively high numbers in Canadian waters. Reports of interactions with fixed fishing gear set for groundfish around Newfoundland averaged 365 annually from 1979 to 1987 (range 174–813). An average of 50 humpback whale entanglements (range 26–66) was reported annually between 1979 and 1988, and 12 of 66 humpback whales entangled in 1988 died (Lien *et al.* 1988). A total of 965 humpbacks was reported entangled in fishing gear in Newfoundland and Labrador from 1979 to 2008 (Benjamins *et al.* 2012). Volgenau *et al.* (1995) reported that gillnets were the primary cause of entanglements and entanglement mortalities (20%) of humpbacks in the Gulf of Maine between 1975 and 1990. More recently, Johnson *et al*. (2005) found that 40% of humpback entanglements were in trap/pot gear and 50% were in gillnets, but sample sizes were small and much uncertainty still exists about the frequency of certain gear types involved in entanglement. A recent review (Cassoff *et al*. 2011) describes in detail the types of injuries that baleen whales, including humpbacks, suffer as a result of entanglement in fishing gear.

More than 2 decades ago, Wiley *et al.* (1995) reported that serious injuries attributable to ship strikes were more common and probably more serious than those from entanglements, but this claim is not supported by more recent analysis. Non-lethal interactions with gear and vessels are common (see Robbins 2010, 2011, 2012; Hill *et al* 2017), but recent analysis suggests entanglement serious injuries and mortalities are more common than ship strikes (van der Hoop *et al.* 2013). Furthermore, in the NMFS records for 2013 through 2017, there are only 23 reports of serious injuries and mortalities as a result of collision with a vessel and 56 records of injuries (prorated or serious) and mortalities attributed to entanglement. Similarly, a recent analysis of the past 20 years of mortalities in North Atlantic right whales, which have considerable overlap in distribution, shows a steady increase in the rate of entanglement (Hayes *et al.* 2019- this SAR report). Because it has never been shown that serious injuries and mortalities related to ships or to fisheries interactions are equally detectable, it is unclear as to which human source of mortality is more prevalent. A major aspect of vessel collision that will be cryptic as a serious injury is blunt trauma; when lethal it is usually undetectable from an external exam (Moore *et al.* 2013). No whale involved in the recorded vessel collisions had been identified as a member of a stock other than the Gulf of Maine stock at the time of drafting this report.

Fishery-Related Serious Injuries and Mortalities

A description of fisheries is provided in Appendix III. See Appendix V for more information on historical takes.

Confirmed human-caused mortalities and serious injuries from the last five years reported to the NMFS Greater Atlantic and Southeast regional offices and to Atlantic Canadian Maritime stranding networks (Henry *et al*. 2020) are listed in Table 2. When there was no evidence to the contrary, events were assumed to involve members of the Gulf of Maine stock. While these records are not statistically quantifiable in the same way as observer fishery records, they provide some indication of the minimum frequency of entanglements. Specifically to this stock, if the calculations of Robbins (2011, 2012) are reasonable then the 3% mortality due to entanglement that she calculates equates to a minimum average rate of 25.

Although disentanglement is often unsuccessful or not possible for many cases, there are several documented cases of entanglements for which the intervention of disentanglement teams averted a likely serious-injury determination. Twenty-four serious injuries were prevented by intervention during 2013–2017 (Henry *et al.* 2020).

Table 2. Confirmed human-caused mortality and serious injury records of humpback whales (Megaptera novaeangliae) where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2013–2017 [a]

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
3-Apr-13	Mortality	-	off Ft Story, VA	VS	1	US	-	Fractured orbitals & ribs w/ associated bruising
13-Sep-13	Mortality	-	York River, VA	VS	1	US	-	6 lacerations penetrate into muscle w/ associated hemorrhaging
16-Sep-13	Prorated Injury	-	off Chatham, MA	EN	0.75	XU	NR	Partial disentanglement; original & final configurations unknown
28-Sep-13	Mortality	-	off Saltaire, NY	EN	1	XU	GN	Embedded line in mouth w/ associated hemorrhaging & necrosis; evidence of constriction at pectorals, peduncle & fluke w/ associated hemorrhaging; emaciated. Previously reported as GU.
1-Oct-13	Mortality	-	Buzzards Bay, MA	EN	1	US	NP	Evidence of underwater entrapment & subsequent drowning.
4-Oct-13	Serious Injury	-	off Chatham, MA	EN	1	XU	NR	Full configuration unknown, but evidence of health decline: emaciation & pale skin
02-Jun-14	Prorated Injury	-	15 mi E of Monomoy Island, MA	EN	0.75	XU	NR	Free-swimming with buoy and highflier trailing 100ft aft of flukes. Attachment point(s) unknown. Unable to confirm if resighted on 21Jun2014.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
21-Jun-14	Prorated Injury	-	5 mi E of Gloucester, MA	EN	0.75	XU	NR	Free-swimming trailing a buoy and possibly another buoy/highflier aft. Attachment point(s) unknown. Unable to confirm if this is a resight of 02Jun2014.
18-Jul-14	Serious Injury	-	Provincetown Harbor, MA	EN	1	XU	NR	Free-swimming, trailing short amount of line from left side of mouth. No other gear noted, but evidence of previously more complicated, constricting entanglement. Current configuration deemed non-life threatening. Unsuccessful disentanglement attempt. In poor condition - emaciated with some cyamids. No resights
03-Sep-14	Prorated Injury		off Long Island Beach, NJ	EN	.75	XU	NE	Full/final config. unknown. Seen with new vessel strike lacerations on 14Aug2014. No resights. Previously reported as gear unknown and being gear free (SI value=0) but gear status determined to be unconfirmed.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
11-Sep-14	Mortality	Spinnaker	10 nm SE of Frenchboro, ME	EN	1	XU	GN	Free-swimming with gillnet gear. Found anchored on 12Sep2014. Gillnet panel lodged in mouth and tightly wrapping forward part of body. Panel entangled in pots with 20+ wraps of pot lines around flukes and peduncle. Mostly disentangled--left with short section of gillnet in mouth expecting to shed. Animal entangled again (14May2015 - anchored and disentangled). Carcass found 11Jun2015. Necropsy revealed gillnet from 2014 entanglement embedded deep into the maxilla and through the vomer. Bone had started to grow around the line. Gillnet is unknown origin. Pot/trap is US gear.
20-Sep-14	Prorated Injury	NYC0010	off Rockaway Beach, Long Island, NY	EN	.75	US	GN	Free-swimming with netting and rope with floats wrapping flukes. Entanglement noticed during photo processing. Full configuration unknown. No resights. Previously reported as gear unknown.
01-Oct-14	Prorated Injury	-	15 mi E of Metompkin Inlet, VA	EN	.75	XU	NR	Free-swimming whale with line &/or netting on left fluke blade. Gear appeared heavy. Full configuration unknown. No resights.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
15-Dec-14	Prorated Injury	-	8.5 nm S of Grand Manan, NB	EN	.75	CN	PT	Fisherman found animal entangled in trawl. Grappled line, animal dove. Upon surfacing, appeared free of gear, but unable to confirm gear free. Original and final configuration unknown. Previously reported as XC.
25-Dec-14	Mortality	Triomphe	Little Cranberry Island, ME	EN	1	XU	NP	Fresh carcass with evidence of extensive constricting entanglement. No necropsy, but robust body condition and histopathology results of samples support EN as COD.
01-Feb-15	Serious Injury	-	off Beaufort, NC	EN	1	XU	NE	Constricting wrap at fluke insertion with line and monofilament netting trailing from flukes. Partial disentanglement by fisherman. Left with embedded gear and at least 40 ft of trailing line and netting. Unknown if there are additional attachment points. No resights. Gear previously reported as NR.
03-Feb-15	Mortality	-	Corolla, NC	EN	1	US	NP	Fresh carcass with injuries consistent with constricting gear. No gear present. Full stomach indicating fed recently. COD likely peracute under water entrapment.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
13-Apr-15	Mortality	-	off Fire Island, NY	VS	1	US	-	Extensive bruising and hemorrhaging at left gape and pectoral, throat, and right and left lateral thorax.
18-Apr-15	Mortality	-	Smith Point, NY	VS	1	US	-	Multifocal hemorrhage and edema in right lateral abdomen.
29-Jun-15	Mortality	-	Fire Island, NY	VS	1	US	-	Extensive fracturing of cranial bones with associated bruising. Additional extensive bruising along dorsal and right lateral body.
09-Jul-15	Prorated Injury	-	off Sandy Hook, NJ	EN	0.75	XU	NR	High flier trailing 30 ft aft of flukes. Attachment point(s) and configuration unknown. No resights.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
02-Aug-15	Serious Injury	-	off Race Point, Provincetown, MA	EN	1	XU	GN	Free-swimming with two sets of gear through its mouth: Primary gear=a closed bridle of gillnet joining mid-belly and trailing just past flukes and restricting movement; Secondary gear=an open bridle with one end leading to a buoy and the other to a pot. Disentangled from both sets of gear. Left with very short amount of gillnet through mouth that is expected to shed. Emaciated. No resights. Gillnet is primary cause of injury and of unknown origin. Pot/trap is US gear.
02-Aug-15	Prorated Injury	-	off Chatham, MA	EN	0.75	XU	NR	Calf with line around tail leading to buoys 4 ft aft of flukes. Full configuration unknown. No resights post 22Aug2015.
07-Sep-15	Prorated Injury	-	off Race Point, Provincetown, MA	EN	0.75	XU	MF	Monofilament line trailing from flukes. Attachment point(s) and configuration unknown. No resights.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
24-Sep-15	Prorated Injury	-	off Hampton, NH	EN	0.75	US	Anchor system	Became entangled in anchor line of fishing vessel during the night. Believed to be towing the entire system--45 lb anchor, 20 ft of chain, 350 ft of anchor line, 150 ft of float line, polyball and acorn buoy--in an unknown configuration. No resights.
25-Sep-15	Serious Injury	-	off Menemsha Harbor, MA	EN	1	XU	NR	Evidence of constricting body wrap, unable to confirm if gear embedded. Trailing 10 ft of line from flukes, full configuration unknown. Animal emaciated with heavy cyamids. No resights.
17-Oct-15	Mortality	-	Lloyd Neck Harbor, NY	VS	1	US	-	Extensive bruising and edema around right cranial and pectoral.
04-Dec-15	Prorated Injury	-	off Brier Island, NS	EN	0.75	CN	PT	Likely anchored in gear. Partially disentangled by fishermen. Left free-swimming with a body wrap aft of blowholes and 2 balloon floats close to body. Final configuration unknown. No resights.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
15-Dec-15	Prorated Injury	-	off North East Harbour, NS	EN	0.75	CN	PT	Likely anchored in gear. Partially disentangled by fishermen. Left free-swimming with buoy and lines around front of whale and lines on the peduncle. Attachment point(s) and final configuration unknown. No resights.
07-Jan-16	Prorated Injury	--	off Greenwich, CT	EN	0.75	US	PT	Anchored in gear with line through mouth and around tail. Partially disentangled - all gear removed from mouth and some from tail. Post intervention whale was using pectorals to swim and tail was down, but unable to confirm if any gear remained and in what configuration. No resights.
09-Jan-16	Serious Injury	MAHWC-254	off Fort Story, VA	VS	1	US	-	Deep laceration across back - penetrating into muscle and impacting ability to dive. No resights.
03-Mar-16	Serious Injury	MAHWC-251	off Virginia Beach, VA	VS	1	US	-	Deep laceration on left fluke blade, near insertion. Fluke blade necrotic. No resights.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
24-Apr-16	Prorated Injury	-	off Race Point, Provincetown, MA	EN	0.75	XU	NR	Free-swimming with 2 buoys - submerged orange at 5 ft and white bullet at 10 ft - trailing behind flukes. Line appears to wrap flukes. Subsequent sighting only reported white buoy, but only one surfacing and no photos. Attachment point(s) and configuration unknown. No resights.
25-Apr-16	Mortality	-	Marshfield, MA	VS	1	US	-	Bruising deep to muscle and fascia by right pectoral and mandible at the base of the skull. Limited necropsy but depth and area of bruising consistent with blunt trauma from vessel strike.
25-Apr-16	Mortality	-	Napreague Bay, NY	VS	1	XU	-	Extensive bruising to ventral thoracic region along with fractured ribs.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
18-May-16	Serious Injury	Foggy	off Gloucester, MA	EN	1	XU	GU	Anchored with lines through mouth and 2 embedded body wraps with large float alonside by right body. Entangling gear fouled in 2 other sets of gear. Animal in emaciated. Partial disentanglement - left with an open bridle of 2 lines through the mouth. Subsequent sightings show lines had relooped into a closed bridle and health continued to decline. No resights post July 2016.
21-May-16	Prorated Injury	-	off Mantoloking, NJ	EN	0.75	XU	GN	Full configuration unknown, but minimally wrapped in gear from head to dorsal. Unknown amount of gear removed by public. Unable to confirm if gear free. No resights.
15-Jun-16	Mortality	-	off Fenwick Island, DE	VS	1	US	-	Large area of hemorrhaging around neck and head. Organs displaced forward in body cavity. Full stomach.
24-Jun-16	Mortality	-	off Shinnecock Inlet, NY	VS	1	US	-	Extensive bruising to connective tissue and muscles of the left side, back, and right peduncle.
26-Jun-16	Mortality	Snowplow	off Rockport, MA	VS	1	US	-	Limited necropsy, but significant evidence of blunt trauma to left head and pectoral consistent with vessel strike.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
05-Jul-16	Serious Injury	-	off Chatham, MA	EN	1	XU	GU	Free-swimming with embedded wraps at base of flukes and buoy trailing 50 ft. Partially disentangled. Peduncle wraps loosened and expect to shed. Pronosis poor - flukes compromised and deteriorating. Animal swimming with flippers. No resights.
02-Sep-16	Prorated Injury	-	off Gloucester, MA	EN	0.75	XU	NR	Free-swimming and trailing red buoy. Attachment point(s) and configuration unknown. No resights.
10-Sep-16	Mortality	-	Martha's Vineyard, MA	EN	1	XU	NP	No gear present, but evidence of constricting entanglement with associated reactive tissue at fluke insertions. State of decomposition at time of exam precluded COD determination, but injuries and thin blubber layer are consistent with chronic entanglement.
16-Oct-16	Mortality	GOM-1626	off Ipswich, MA	EN	1	US	PT	No necropsy, but extensive entanglement. Line through mouth with constricting wraps on both flippers, body, and peduncle. Entanglement as COD most parsimonious. Confirmed as same individual released from weir on 27Sep2016.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
13-Nov-16	Prorated Injury	NYC0052	off Belmar, NJ	EN	0.75	XU	MF	Free-swimming with monofilament over peduncle and trailing from flukes. Attachment point(s) and configuration unknown. No resights.
14-Nov-16	Prorated Injury	-	off Stone Harbor, NJ	EN	0.75	XUS	PT	Free-swimming with line wrapping left flipper and flukes and trailing. Full configuration unclear. No resights. Previously reported as XC, gear not recovered.
04-Dec-16	Prorated Injury	-	off Quogue, NY	EN	0.75	XU	NR	Free-swimming with high flier near flukes. Attachment point(s) and configuration unknown. No resights.
16-Dec-16	Mortality	HDRVA078	off Dam Neck, VA	EN	1	US	NP	No gear present, but evidence of extensive constricting entanglement. Fresh carcass with digestive system full of fish. COD dry drowning due to entanglement.
19-Dec-16	Prorated Injury	-	off Tiverton, NS	EN	0.75	Sure!XC	NR	Free-swimming with line around tail and buoy trailing. Full configuration unknown. No resights.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
02-Feb-17	Mortality	-	Chesapeake Bay, VA	VS	1	US	-	Four lacerations that penetrated body cavity. Robust condition with full stomach. COD exsanguination and asphyxia from sharp trauma consistent with vessel strike.
05-Feb-17	Mortality	-	Chesapeake Bay, VA	VS	1	US	-	Extensive skeletal fracturing with associated hemorrhaging consistent with blunt trauma from vessel strike.
11-Feb-17	Mortality	-	Fort Story, VA	VS	1	US	-	Three lacerations that penetrated body cavity. Robust condition with full stomach. COD exsanguination from sharp trauma consistent with vessel strike.
14-Feb-17	Serious Injury	-	Virginia Beach, VA	VS	1	US	-	Two new, deep lacerations fore and aft of dorsal fin. No resights.
03-Apr-17	Mortality	-	Rockaway, NY	VS	1	US	-	Extensive hemorrhage and edema along back and side consistent with blunt trauma from vessel strike.
04-May-17	Mortality	-	Rehobeth Beach, DE	VS	1	US	-	Disarticulated left jaw and cervical vertebrae with associated hemorrhaging. Limited necropsy but injuries consistent with blunt trauma from vessel strike.
15-Jun-17	Mortality	-	Jamestown, RI	VS	1	US	-	Muscle contusions and associated cranial fractures consistent with blunt trauma from vessel strike.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
18-Jun-17	Mortality	GOM-1625	Chatham, MA	EN	1	XU	NP	No gear present, but evidence of constricting entanglement with associated hemorrhaging at insertion of pectorals and fluke. Poor health condition.
15-Jul-17	Prorated Injury	2016 Calf of Thumper	off Race Point, Provincetown, MA	EN	.75	US	NR	Free-swimming with hook and monofilament trailing from right fluke blade. Attachment point(s) and full configuration unknown. No resights.
01-Aug-17	Mortality	2017 Calf of Cajun	off Gloucester, MA	EN	1	US	GN	Dependent calf with gillnet exiting right side of mouth. Evidence of unwitnessed extensive ent. Carcass found on 24Feb2018, not recovered for necropsy. Prox. COD = ent., Ultimate COD = unk.
19-Aug-17	Prorated Injury	-	off Long Island, NY	EN	.75	XU	NR	Free-swimming with buoy trailing aft of flukes. Attachment point(s) and configuration unknown. No resights post 11Sep2017.
18-Sep-17	Prorated Injury	-	off Jonesport, ME	EN	.75	CN	PT	Anchored in gear. Fisher responded later, animal not relocated and gear missing section of pots and line. Final configuration unknown. No resights.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
01-Oct-17	Mortality	-	off Narragansett, RI	VS	1	XU	-	Hemorrhaging along dorsal and left side consistent with blunt trauma from vessel strike.
10-Oct-17	Prorated Injury	-	off Gloucester, MA	EN	.75	US	PT	Anchored in gear. Partially disentangled. Unable to confirm gear free. Final configuration unknown. No resights.
14-Oct-17	Prorated Injury	-	off Race Point, Provincetown, MA	EN	.75	XU	NR	Free-swimming with buoy along right flank. Attachment point(s) and full configuration unknown. No resights.
21-Oct-17	Prorated Injury	GOM-1747	off Long Island, NY	EN	.75	XU	NR	Free-swimming with buoy in tow. Attachment point(s) and full configuration unknown. No resights.
12-Nov-17	Prorated Injury	-	off Atlantic Beach, NY	EN	.75	US	MF	Free-swimming with monofilament trailing from right fluke. Attachment point(s) and full configuration unknown. No resights.
30-Nov-17	Prorated Injury	-	off Grand Manan, NB	EN	.75	CN	PT	Anchored at tail area, partially disentangled. Unable to confirm gear free or that all gear recovered. Final configuration unknown. No resights.
26-Dec-17	Mortality	-	East Atlantic Beach, NY	VS	1	US	-	Extensive bruising and edema on both sides of body consistent with blunt trauma from vessel strike.

Assigned Cause	Five-year mean (US/CN/XU/XC)
Vessel strike	4.4 (4.0/ 0.00/ 0.40/ 0.00)
Entanglement	7.75 (2.05/ 0.75/ 4.8/ 0.15

a. For more details on events please see Henry *et al.* 2020.
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, NE=netting, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

Other Mortality

Between November 1987 and January 1988, at least 14 humpback whales died after consuming Atlantic mackerel containing a dinoflagellate saxitoxin (Geraci *et al.* 1989). The whales subsequently stranded or were recovered in the vicinity of Cape Cod Bay and Nantucket Sound, and it is highly likely that other unrecorded mortalities occurred during this event. During the first six months of 1990, seven dead juvenile (7.6 to 9.1 m long) humpback whales stranded between North Carolina and New Jersey. The significance of these strandings is unknown.

Between July and September 2003, an Unusual Mortality Event (UME) that included 16 humpback whales was invoked in offshore waters of coastal New England and the Gulf of Maine. Biotoxin analyses of samples taken from some of these whales found saxitoxin at very low/questionable levels and domoic acid at low levels, but neither were adequately documented and therefore no definitive conclusions could be drawn. Seven humpback whales were considered part of a large whale UME in New England in 2005. Twenty-one dead humpback whales found between 10 July and 31 December 2006 triggered a humpback whale UME declaration. Additionally, in January 2016 a humpback whale UME was declared for the U.S. Atlantic coast due to elevated numbers of mortalities (a total of 88 strandings in 2016–2018; https://www.fisheries.noaa.gov/national/marine-life-distress/2016-2018-humpback-whale-unusual-mortality-event-along-atlantic-coast). This most recent UME is ongoing.

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 (Head *et al*. 2010; Grieve *et al.* 2017; Nye *et al*. 2009; Pinsky *et al*. 2013; Hare *et al*. 2016). 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.

STATUS OF STOCK

NMFS conducted a global status review of humpback whales (Bettridge *et al*. 2015) and recently revised the ESA listing of the species (81 FR 62259, September 8, 2016). The Distinct Population Segments (DPSs) that occur in waters under U.S. jurisdiction, as established in the Final Rule, do not necessarily equate to the existing MMPA stocks. NMFS is evaluating the stock structure of humpback whales under the MMPA, but no changes to current stock structure are proposed at this time. As noted within the humpback whale ESA-listing Final Rule, in the case of a species or stock that achieved its depleted status solely on the basis of its ESA status, such as the humpback whale, the species or stock would cease to qualify as depleted under the terms of the definition set forth in MMPA Section 3(1) if the species or stock is no longer listed as threatened or endangered. The final rule indicated that until the stock delineations are reviewed in light of the DPS designations, NMFS would consider stocks that do not fully or partially coincide with a listed DPS as not depleted for management purposes. Therefore, the Gulf of Maine stock is considered not depleted because it does not coincide with any ESA-listed DPS. The detected level of U.S. fishery-caused mortality and serious injury derived from the available records, (average of 12.5 for 2013–2017) does not exceed the calculated PBR of 22 and, therefore, this is not a strategic stock if the recovery factor is set at 0.5. Because the observed mortality is estimated to be only 20% of all mortality (Figure 4), total annual mortality may be 60-70 animals in this stock. If anthropogenic causes are responsible for as little as 31% of potential total mortality, this stock could be over PBR. While detected mortalities yield an estimated minimum fraction of anthropogenic mortality as 0.85, additional research is being done before apportioning mortality to anthropogenic versus natural causes for undetected mortalities. Therefore, the accounting of human caused mortality is biased low and the uncertainties associated with this assessment may have produced an incorrect determination of strategic status.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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).

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).

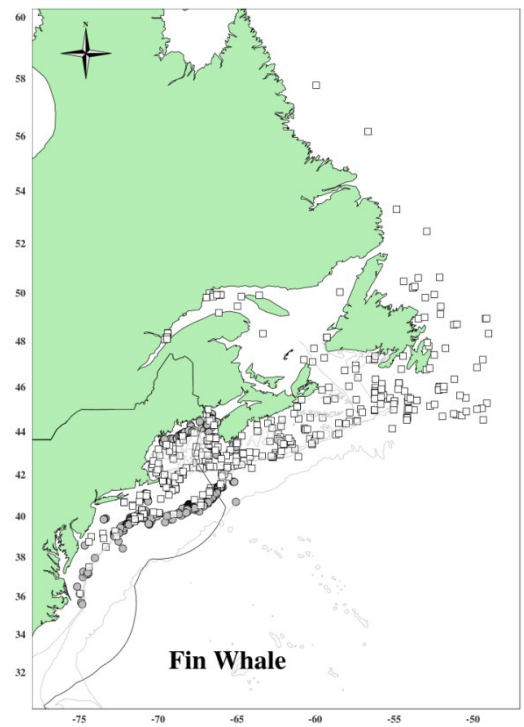

Figure 1.Distribution of fin whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 100-m and 4000-m depth contours. Circle symbols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

Fin whales are common in waters of the U. S. Atlantic Exclusive Economic Zone (EEZ), principally from Cape Hatteras northward (Figure 1). In a recent 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 waters represent a major feeding ground 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). 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. Despite the suggested similarity in patterns of seasonal occurrence with humpback whales, the U.S. currently recognizes one stock of fin whales in the western North Atlantic.

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). 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 abundance estimate available for the western North Atlantic fin whale stock is 7,418 (CV=0.25). This estimate is the sum of the 2016 NOAA shipboard and aerial surveys and the 2016 Canadian Northwest Atlantic International Sightings Survey (NAISS). 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. 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, the 2016 survey estimates in U.S. waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected.

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 of 1,595 (CV=0.33) fin whales was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines that were in waters offshore of North Carolina 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 double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the multiple-covariate distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).The abundance estimates of fin whales include a percentage of the estimate of animals identified as fin/sei whales (the two species being sometimes hard to distinguish).The percentage used is the ratio of positively identified fin whales to the total number of positively identified fin whales and positively identified sei whales; the CV of the abundance estimate includes the variance of the estimated fraction.

An abundance estimate of 23 (CV=0.87) fin whales was generated from a shipboard survey conducted concurrently (June–August 2011; Garrison 2016) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25× bigeye binoculars. A total of 4,445 km of tracklines was surveyed. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 3,006 (CV=0.61) 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.

The Department of Fisheries and Oceans, Canada (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 (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 over the Newfound/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 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 western North Atlantic fin whales with month, year, and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun-Aug 2011	Central Virginia to lower Bay of Fundy	1,595	0.33
Jun-Aug 2011	Central Florida to Central Virginia	23	0.76
Jun-Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	1,618	0.33
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)	7,418	0.25

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 7,418 (CV=0.25). The minimum population estimate for the western North Atlantic fin whale is 6,029.

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). 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 6,029. 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 12.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2013 through 2017, the minimum annual rate of human-caused mortality and serious injury to fin whales was 2.35 per year. This value includes incidental fishery interaction records, 1.55 (0 U.S./ 0.95 unknown but first reported in U.S. waters/0.6 Canadian waters); and records of vessel collisions, 0.8 (all U.S.) (Table 2a; Henry *et al.* 2020). 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.

Fishery-Related Serious Injury and Mortality

U.S.

No confirmed fishery-related mortalities or serious injuries of fin whales have been reported in the NMFS Sea Sampling bycatch database. A review of the records of stranded, floating, or injured fin whales for the period 2013 through 2017 on file at NMFS found no records with substantial evidence of fishery interactions causing mortality in U.S. waters (Table 2a; Henry *et al.* 2020). Serious injury determinations from fishery interaction records yielded a value of 4.75 over five years, for an annual average of 0.95 (Table 2a; Henry *et al.* 2020). 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 last five years are reported in Table 2b. Three records with substantial evidence of fishery interactions causing mortality or serious injury were reported for the 2013–2017 period, resulting in a 5-year annual average of 0.6 animals.

Table 2a. Confirmed human-caused mortality and serious injury records of fin whales (Balaenoptera physalus) first reported in U.S. waters or attributed to U.S. where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2013–2017[a]

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
13-Jan-13	Mortality	-	East Hampton, NJ	VS	1	US	-	Fracturing of left cranium with associated hematoma
12-Apr-14	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.
23-Jun-14	Prorated Injury	-	off Chatham, MA	EN	0.75	XU	NR	Free-swimming, trailing 200ft of line. Attachment point(s) unknown. No resights.
20-Aug-14	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.
05-Oct-14	Mortality	-	off Manasquan, NJ	VS	1	US	-	Large area of hemorrhage along dorsal, ventral, and right lateral surfaces consistent with blunt

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
								force trauma.
06-Jun-15	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.
06-Jul-16	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.
08-Jul-16	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.
14-Dec-16	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.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
05/30/17	Mortality		Port Newark, NJ	VS	1	US	-	Fresh carcass on bow of 656 ft vessel. Speed at strike unknown.

Assigned Cause	5-Year mean (US/XU)
Vessel strike	0.8 (0.8/ 0.0)
Entanglement	0.95(0/ 0.95)

a. For more details on events please see Henry *et al.* 2020.
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.
e. H=hook, GN=gillnet, GU=gear unidentifiable, MF=monofilament, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir.

Table 2b. Confirmed human-caused mortality and serious injury records of fin whales (Balaenoptera physalus) first reported in Canadian waters or attributed to Canada where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2013–2017[a]

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
6/6/13	Serious Injury	Capitaine Crochet	St. Lawrence Marine Park, Quebec	EN	1	CN	PT	Pot resting on upper jaw w/ bridle lines embedding in mouth; health decline: emaciation
5/13/14	Mortality	-	Rocky Harbour, NL	EN	1	CN	PT	Fresh carcass hog-tied in gear.
8/25/17	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.

Assigned Cause	5-Year mean (CN/XC)
Vessel strike	0
Entanglement	0.6 (0.6/ 0.0)

a. For more details on events please see Henry *et al.* 2020.
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, 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

After reviewing NMFS records for 2013 through 2017, 4 were found that had sufficient information to confirm the cause of death as collisions with vessels (Table 2a; Henry *et al.* 2020). These records constitute an annual rate of serious injury or mortality of 0.8 fin whales from vessel collisions in U.S. waters.

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. The total level of human-caused mortality and serious injury is unknown. NMFS records represent coverage of only a portion of the area surveyed for the population estimate for the stock. The total U.S. 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.

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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.

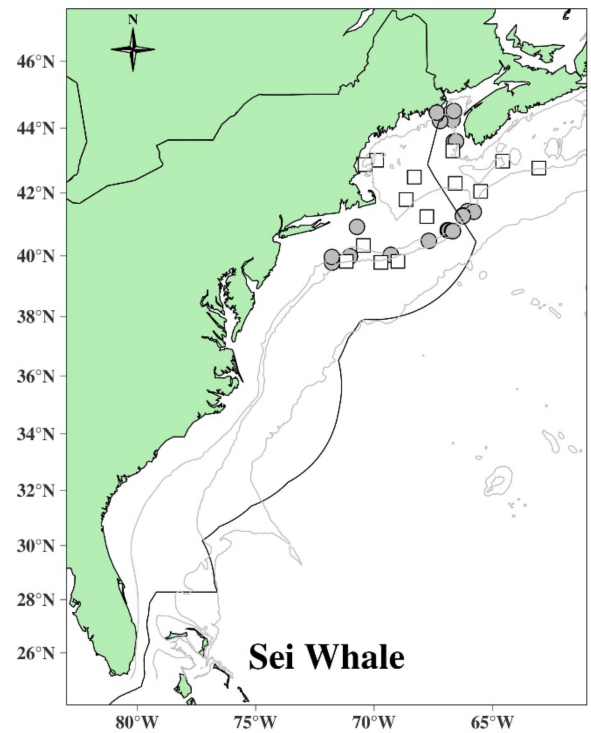

Figure 1. Distribution of sei whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, and 2016 and DFO’s 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-m, 1000-m and 4000-m depth contours.

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, 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 (R.D. Kenney, pers. comm.; 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 abundance estimate of 357 (CV=0.52) sei whales was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters from north of New Jersey from the coastline to the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines that were in waters offshore of 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 double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the multiple-covariate distance sampling (MCDS) option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009). The abundance estimates of sei whales include a percentage of the estimate of animals identified as fin/sei whales (the two species being sometimes hard to distinguish). The percentage used is the ratio of positively identified sei whales to the total of positively identified fin whales and positively identified sei whales; the CV of the abundance estimate includes the variance of the estimated fraction. Although this is the best estimate available for this stock, it should be noted that the abundance survey from which it was derived excluded waters off the Scotian Shelf, an area encompassing a large portion of the stated range of the stock.

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 (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 (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 less 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 (Nbest) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun−Aug 2011	Central Virginia to lower Bay of Fundy	357	0.52
Mar−May 2010−2013	Halifax, Nova Scotia to Florida	6,292	1.015
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
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.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2013 through 2017, the minimum annual rate of human-caused mortality and serious injury to sei whales was 1.0. This value includes incidental fishery interaction records, 0.2, and records of vessel collisions, 0.8 (Table 2; Henry *et al*. 2020). 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.

Fishery-Related Serious Injury and Mortality

No confirmed fishery-related mortalities or serious injuries of sei whales have been reported in the NMFS Sea Sampling bycatch database. A review of the records of stranded, floating, or injured sei whales for the period 2013 through 2017 on file at NMFS found 1 record with substantial evidence of fishery interaction causing serious injury or mortality (Table 2), which results in an annual serious injury and mortality rate of 0.2 sei whales from fishery interactions.

Table 2. Confirmed human-caused mortality and serious injury records of sei whales (Balaenoptera borealis borealis) where the cause was assigned as either an entanglement (EN) or a vessel strike (VS): 2013–2017 [a]

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
5/4/2014	Mortality		Hudson River, NY	VS	1	US	-	Fresh carcass on bow of vessel. Extensive skeletal fractures w/ associated hemorrhage along right side.
5/7/2014	Mortality		Delaware River, PA	VS	1	US	-	Fresh carcass on bow of vessel.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
8/14/2014	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.
Five-year averages		Shipstrike (US/CN/XU/XC)			0.80 (0.80/ 0.00/ 0.00/ 0.00)			
		Entanglement (US/CN/XU/XC)			0.20 (0.00/ 0.00/ 0.20/ 0.00)			
a. For more details on events please see Henry *et al.* 2020.								
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								

Other Mortality

For the period 2013 through 2017 files at NMFS included four records with substantial evidence of vessel collision causing serious injury or mortality (Table 2), which resulted in an annual rate of serious injury and mortality of 0.8 sei whales from vessel collisions.

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.

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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).

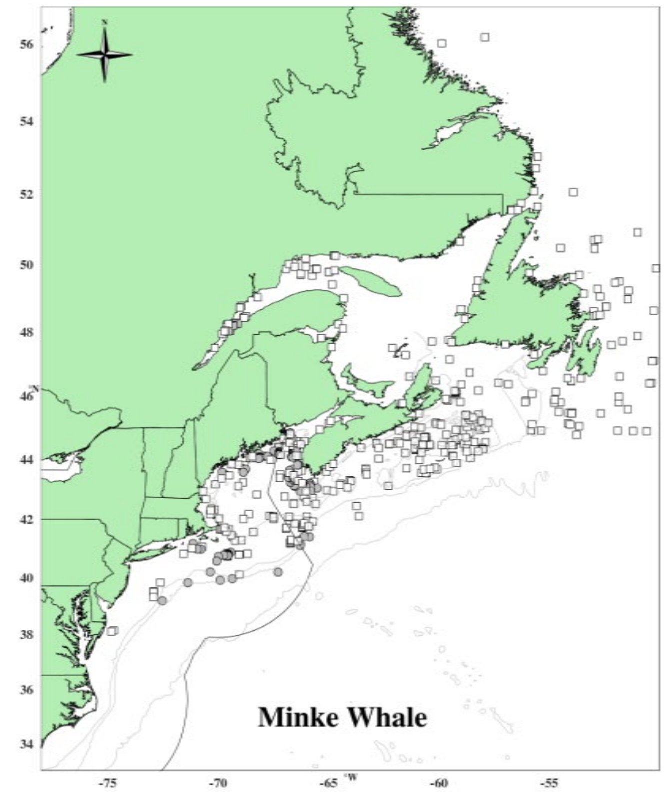

Figure 1. Distribution of minke whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-mm 1000-m and 4000-m depth contours. Circle 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 animals' 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 2,591 (CV=0.81) common minke whales was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour through the U.S. and Canadian Gulf of Maine, and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 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 double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the visually detected species (Laake and Borchers, 2004). Estimation of the abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the multiple-covariate distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 5,036 (CV=0.68) 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_{best}) and and coefficient of variation. (CV).

Month/Year	Area	N_{best}	CV
Jul–Aug 2011	Central Virginia to lower Bay of Fundy	2,591	0.81
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	5,036	0.68
Aug–Sep 2016	Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf	6,158	0.40

Month/Year	Area	N_{best}	CV
Aug–Sep 2016	Newfoundland/Labrador	13,008	0.46
Jun–Sep 2016	Central Virginia to Labrador – COMBINED	24,202	0.30

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 24,202 animals (CV=0.30). The minimum population estimate is 18,902 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 18,902. 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 189.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

During 2013 to 2017, the average annual minimum detected human-caused mortality and serious injury was 8.2 minke whales per year, which is the sum of 6.8 (2.7 U.S./2.3 Canada/1.45 unassigned but first reported in the U.S./0.35 unassigned but first reported in Canada) minke whales per year (unknown CV) from U.S. and Canadian fisheries using strandings and entanglement data, 1.0 (0.8 U.S./0.2 Canada) per year from vessel strikes, 0.2 takes in observed U.S. fishing gear, and 0.2 non-fishery entanglement takes.

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 2 while those recorded by the Observer or At-Sea Monitor Programs are shown in Table 3.

A key uncertainty in the estimate of the annual human-caused mortality and serious injury for this stock, along with other large whales, is due to using strandings and entanglement data as the primary data source. Detected interactions in the strandings and entanglement data should not be considered an unbiased representation of human-caused mortality. Detections are haphazard and not the result of a designed sampling scheme. As such they represent a minimum estimate, which is almost certainly biased low.

Fishery Information

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for information on historical takes.

U.S.

Mid-Atlantic Gillnet

In December 2016 one minke whale mortality was observed in mid-Atlantic gillnet gear. Annual average estimated minke whale mortality and serious injury from the mid-Atlantic sink gillnet fishery during 2013 to 2017 was 0.2. This value was not expanded like other observed bycaught species (see Orphanides 2020) due to the low sample size.

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 2. Data recorded during 2013 to 2017, as determined from stranding and entanglement records confirmed to be of U.S. origin or first sighted in U.S. waters, yielded a minimum detected average annual mortality and serious injury of 3.95common minke whales per year in U.S. fisheries (Table 2a). One of the serious injury entanglement cases reported in Table 2a was a non-fishery interaction (strapping) and so 0.2 was subtracted from the total entanglement 5-year average of 4.15. 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 an unknown Canadian fishery are detailed in Table 2b. During 2013 to 2017, as determined from stranding and entanglement records confirmed to be of Canadian origin or first sighted in Canadian waters, the minimum detected average annual mortality and serious injury was 2.65 minke whales per year in Canadian fisheries (Table 2b; prorated value).

Table 2a. Confirmed human-caused mortality and serious injury records of common minke whales (Balaenoptera acutorostrata acutorostrata) first reported in U.S. waters or attributed to U.S.: 2013–2017[a]

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
7/23/2013	Prorated Injury	-	off Newport, RI	EN	0.75	XU	NR	Full configuration unknown
8/17/2013	Serious Injury	-	off Newburyport, MA	EN	1	XU	NR	Constricting rostrum wrap cutting into upper lip

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
10/04/2013	Prorated Injury	-	off Seal Harbor, ME	EN	0.75	US	NR	Anchored, partially disentangled, final configuration unknown
6/9/2014	Mortality	-	off Truro, MA	EN	1	US	PT	Fresh carcass anchored, hog-tied in gear. COD: peracute underwater entrapment.
7/10/2014	Prorated Injury	-	S of Bristol, ME	EN	0.75	XU	NR	Free-swimming, trailing 2 buoys. Attachment point(s) unknown.
7/12/2014	Serious Injury	-	South Shinnecock Inlet, NY	EN	1	XU	NR	Free-swimming with yellow plastic strapping cutting into top and sides of rostrum. No trailing gear.
7/17/2014	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.
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.
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.
09/01/2015	Mortality	-	Gloucester, MA	EN	1	US	NP	Evidence of extensive, constricting gear with associated

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
								hemorrhaging. No gear present.
8/15/2016	Mortality	-	off Seguin Island, ME	EN	1	US	NR	Line exiting mouth leading to weighted/anchored gear.
4/27/2017	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.
7/6/2017	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.
7/22/2017	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.
8/9/2017	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.
8/11/2017	Prorated Injury	-	off York, ME	EN	0.75	US	NR	Partially disentangled from anchoring gear. Final configuration unknown.
8/12/2017	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.
8/14/2017	Mortality	-	Pt. Judith, RI	EN	1	US	NP	Evidence of constricting entanglement along left side with associated hemorrhaging. Found floating in

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
								stationary offshore fishing trap, but not entangled in trap gear. No gear present on animal.
8/17/2017	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.
8/28/2017	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.
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.
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.

Assigned Cause	5-Year mean (US/XU)
Vessel strike (US/ XU)	0.8 (0.8/ 0.00)
Entanglement (US/ XU)	4.15 (2.7/ 1.45)

a. For more details on events please see Henry *et al.* 2020
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.
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).

Table 2b. Confirmed human-caused mortality and serious injury records of minke whales (Balaenoptera acutorostrata acutorostrata) first reported in Canadian waters or attributed to Canada: 2013–2017a

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
8/31/2013	Mortality	-	Miminegash, PEI	EN	1	CN	NP	Fresh carcass w/ evidence of extensive, constricting gear
7/2/2014	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.
7/29/2014	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)
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

Date[b]	Injury determination	ID	Location [b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
								abnormalities noted.
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.
09/21/2015	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.
5/3/2016	Mortality	-	Biddeford, ME	EN	1	US	PT	Carcass in gear. Line through mouth and evidence of constricting wraps on ventral

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
								pleats and peduncle with associated hemorrhaging.
7/21/2016	Serious Injury	-	Digby, NS	EN	1	XC	GU	Free-swimming with netting deeply embedded in rostrum. Disentangled, but significant health decline.
8/15/2016	Mortality	-	off Seguin Island, ME	EN	1	US	NR	Line exiting mouth leading to weighted/anchored gear.
11/2/2016	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.
8/30/2017	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.
9/4/2017	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.

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
9/17/2017	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.
9/26/2017	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.

Assigned Cause	5-Year mean (CN/XC)
Vessel strike	0.20 (0.20/ 0.00)
Entanglement	2.65 (2.30/ 0.35)

a. For more details on events please see Henry *et al.* 2020.
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, 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).

Table 3. From observer program data, summary of the incidental mortality of the Canadian East Coast stock of minke whales (Balaenoptera acutorostrata) by commercial fishery including the years sampled, the type of data used, the annual observer coverage,

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury [c]	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Combined Serious Injury	Estimated CVs	Mean Annual Combined Mortality
Mid-Atlantic Gillnet	2013	Obs. Data, Weighout	0.03	0	0	0	0	0	0	0.2 (na)
	2014		0.05	0	0	0	0	0	0	
	2015		0.06	0	0	0	0	0	0	
	2016		0.08	0	1	0	1	1	na	
	2017		0.09	0	0	0	0	0	0	
TOTAL	-	-	-	-	-	-	-	-	-	0.2 (na)

a. Observer data (Obs. Data), used to measure bycatch rates, are collected within the Northeast 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. Mandatory Vessel Trip Reports (VTR) (Trip Logbook) are used to determine the spatial distribution of landings and fishing effort in the sink gillnet, bottom trawl and mid-water trawl fisheries. In addition, the Trip Logbooks are the primary source of the measure of total effort (tow duration) in the mid-water and bottom trawl fisheries.
b. Observer coverage for the U.S. mid-Atlantic coastal gillnet fisheries is based on tons of fish landed.
c. Serious injuries were evaluated since 2011 using new guidelines and include both at-sea monitor and traditional observer data (Josephson *et al.* 2019).

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 harvested.

U.S.

Common minke whales inhabit coastal waters during much of the year and are thus susceptible to collision with vessels. In 2014, a confirmed vessel strike resulted in a mortality off Dam Neck, Virginia. In 2015, a fresh carcass of a common minke whale was reported off Coney Island, New York with wounds consistent with vessel strike. In 2017 there are 2 records of minke whale mortalities as a result of vessel strikes. Thus, during 2013–2017, as determined from stranding and entanglement records, the minimum detected annual average was 0.8 common minke whales per year struck by vessels in U.S. waters or first seen in U.S. waters (Table 2a; Henry *et al.* 2020).

One entanglement interaction reported in Table 2a involved strapping, not fishing gear, so while counted as a human-caused mortality, was not included in the fishery interaction total.

An Unusual Mortality Event was established for minke whales in January 2017 due to elevated stranding along the Atlantic coast (https://www.fisheries.noaa.gov/national/marine-life-distress/ 2017-2018-minke-whale-unusual-mortality-event-along-atlantic-coast). Anthropogenic mortalities and serious injuries that occurred in 2017 are included in Tables 1a and 1b.

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). Starting in 1997, 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. The events that were determined to be human-caused serious injury or mortality are included in Table 2b.

The Whale Release and Strandings program reported the following common minke whale stranding mortalities in Newfoundland and Labrador for the time period of this report: 0 in 2013, 1 in 2014, 2 in 2015, 0 in 2016 and 2 in 2017. Those that have been determined to be human-caused serious injury or mortality are included in Table 2b (Ledwell and Huntington 2013, 2014, 2015, 2017, 2018).

During 2013–2017, as determined from stranding and entanglement records, the minimum detected annual average was 0.2 common minke whales per year struck by vessels in Canadian waters or first seen in Canadian waters (Table 2b; Henry *et al.* 2020).

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.

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.

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 (189).

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BLUE WHALE (*Balaenoptera musculus musculus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The distribution of the blue whale, *Balaenoptera musculus musculus,* in the western North Atlantic generally extends from the Arctic to at least mid-latitude waters. Blue whales are most frequently sighted in the waters off eastern Canada, with the majority of records from the Gulf of St. Lawrence (Sears *et al.* 1987). The species was hunted around Newfoundland in the first half of the 20th century (Sergeant 1966). The present Canadian distribution, broadly described, is spring, summer, and fall in the Gulf of St. Lawrence, especially along the north shore from the St. Lawrence River estuary to the Strait of Belle Isle and off eastern Nova Scotia. The species occurs in winter off southern Newfoundland and also in summer in Davis Strait (Mansfield 1985). Individual identification has confirmed the movement of a blue whale between the Gulf of St. Lawrence and western Greenland (Sears and Larsen 2002), although the extent of exchange between these two areas remains unknown. Similarly, a blue whale photographed on the Scotian Shelf by a NMFS large whale survey in August 1999 had previously been observed in the Gulf of St. Lawrence in 1985 (R. Sears and P. Clapham, unpublished data) and there have been additional photographic resightings between the Gulf of Maine, Scotian Shelf and Gulf of St. Lawrence (R. Sears, pers. comm.).

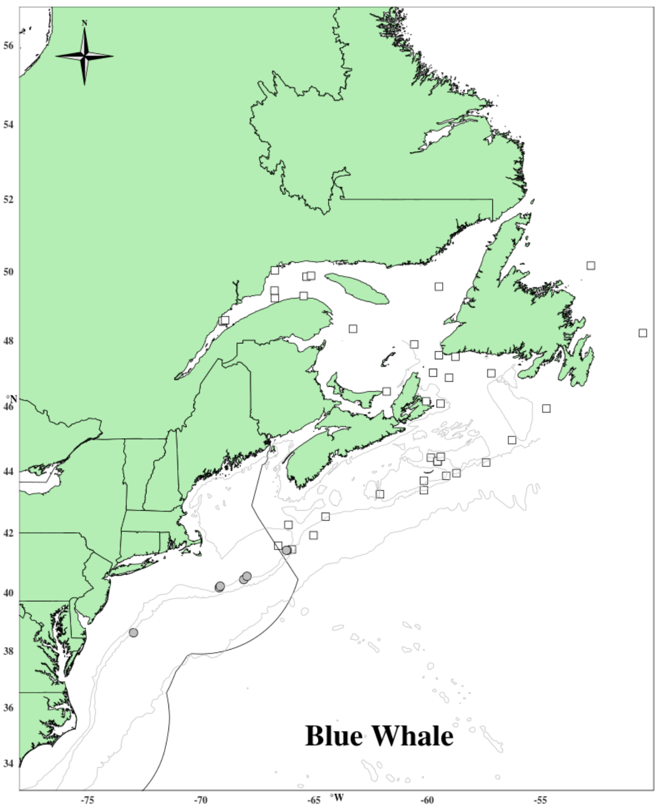

Figure 1: Distribution of blue whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2010, 2011, 2013, and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

The blue whale is best considered as an occasional visitor in U.S. Atlantic Exclusive Economic Zone (EEZ) waters, which may represent the current southern limit of its feeding range (CETAP 1982; Wenzel *et al.* 1988). All of the five sightings described in the foregoing two references were in August. There were three blue whale sighting by whale-watchers south of Montauk Point, New York, between about the 30- and 50-m isobaths, over a one-week period in July and August 1990, believed to be the same animal each time (Kenney and Vigness-Raposa 2010). Yochem and Leatherwood (1985) summarized records that suggested an occurrence of this species south to Florida and the Gulf of Mexico, although the actual southern limit of the species' range is unknown.

Using the U.S. Navy's SOSUS program, blue whales have been detected and tracked acoustically in much of the North Atlantic, including in subtropical waters north of the West Indies and in deep water east of the U.S. Atlantic EEZ, indicating the potential for long-distance movements (Clark 1995). Most of the acoustic detections were around the Grand Banks area of Newfoundland and west of the British Isles. Recordings made in 2006 and 2007 in the Gully Marine Protected Area at the outer edge of the Scotian Shelf had blue whale vocalizations on 17% of the summer recordings and 4% of winter recordings (Marotte and Moors-Murphy 2015). A 2008 study detected blue whale calls in offshore areas of the New York Bight on 28 out of 258 days of recordings, mostly during winter (Muirhead *et al.*

2018). Historical blue whale observations collected by Reeves *et al*. (2004) show a broad longitudinal distribution in tropical and warm temperate latitudes during the winter months, with a narrower, more northerly distribution in summer. Sigurjónsson and Gunnlaugsson (1990) note that North Atlantic blue whales appear to have been depleted by commercial whaling to such an extent that they remain rare in some formerly important habitats, notably in the northern and northeastern North Atlantic.

Photo-identification in eastern Canadian waters indicates that blue whales from the St. Lawrence, Newfoundland, Nova Scotia, New England and Greenland all belong to the same stock, while blue whales photographed off Iceland and the Azores appear to be part of a separate population (Wenzel *et al.* 1988; Sears and Calambokidis 2002; Sears and Larsen 2002).

POPULATION SIZE

Little is known about the population size of blue whales except for the Gulf of St. Lawrence area. From 1980 to the summer of 2008, a total of 402 blue whales was photo-identified, mainly in the St. Lawrence estuary and northwestern Gulf of St. Lawrence (Ramp and Sears 2013). Biopsies have been taken on nearly 40% of this population (R. Sears, pers. comm.). Each year, from 20 to 105 blue whales are identified in this region. Approximately 40% of the identified blue whales return frequently to the study area, while the others have been observed during fewer than three seasons between 1979 and 2002, which suggests that these individuals range mostly outside the St. Lawrence, possibly in the waters at the edge of the continental shelf, from the Labrador Sea and Davis Strait in the north, east to the Flemish Cap and south to New England (Sears and Calambokidis 2002). Photo-identification data from outside the estuary and Gulf of St. Lawrence are limited. A few blue whales have been photographed along the coast of Newfoundland, on the Scotian Shelf and in the Gulf of Maine, and some are not included among the 402 blue whales that have been identified in the estuary and northwest part of of the Gulf of St. Lawrence (Sears and Calambokidis 2002; Ramp and Sears 2013; J. Lawson, pers. comm.). Ramp *et al*. (2006) estimated the survival rate at 0.975 and the gender ratio of the 139 biopsy sampled individuals at 79 males for 67 females (Sears 2003). Given the small proportion of the distribution range that has been sampled and considering the low number of blue whales encountered and photographed, the current data, based on photo-identification, do not allow for an estimate of abundance of this species in the Northwest Atlantic with a minimum degree of certainty (Sears *et al.* 1987, 1990; Hammond *et al.* 1990; Sears and Calambokidis 2002; Beauchamp *et al*. 2009). Mitchell (1974) estimated that the blue whale population in the western North Atlantic may number only in the low hundreds. R. Sears (pers. comm.) suggests that 400 to 600 individuals may be found in the western North Atlantic.

An abundance estimate of 39 (CV=0.64) blue 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 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 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. Because this estimate is only for the U.S. portion of the stock, the above catalogue count of 402 is considered the best estimate.

Table 1. Summary of recent abundance estimates for western North Atlantic blue whales (Balaenoptera musculus) by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
1980–2008	Gulf of Saint Lawrence Catalogue	402	-
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	39	0.64

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.

Minimum Population Estimate

The catalogue count of 402 recognizable individuals from the Gulf of St. Lawrence is considered to be a minimum population estimate for the western North Atlantic stock.

Current Population Trend

There are insufficient data to determine population trends for this species. Off western and southwestern Iceland, an increasing trend of 4.9% a year was reported for the period 1969–1988 (Sigurjónsson and Gunnlaugsson 1990). Pike *et al*. (2009) conducted ship surveys in the Central and Northeast Atlantic in 1987, 1989, 1995 and 2001. Blue whales were most commonly sighted off western Iceland, and to a lesser extent northeast of Iceland. They were very rare or absent in the Northeast Atlantic. Sightings were combined over all surveys to estimate the detection function using standard line-transect methodology, with the addition of a covariate to account for differences between surveys. Total abundance was highest in 1995 (979, 95% CI 137–2,542) and lowest in 1987 (222, 95% CI 115–440). Uncertainty in species identity had little effect on estimates of abundance. There was a significant positive trend in abundance northeast of Iceland and in the total survey area. These estimates should be treated with caution given the effort biases underlying the sightings data on which they were based.

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 402. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor, which accounts for stocks which are endangered, depleted, or threatened or of unknown status relative to optimum sustainable population (OSP), is assumed to be 0.10 because the blue whale is listed as endangered under the Endangered Species Act (ESA). PBR for the Western North Atlantic stock of blue whale is 0.8.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Threats for North Atlantic blue whales are poorly known, but may include ship strikes, pollution, entanglement in fishing gear, and long-term changes in climate (which could affect the abundance of their zooplankton prey). During winter and early spring, ice-related strandings and entrapments have been documented on the southwestern and eastern coasts of Newfoundland (Sears and Calambokidis 2002). There are no recent confirmed records of anthropogenic mortality or serious injury to blue whales in the U.S. Atlantic EEZ or in Atlantic Canadian waters (Henry *et al.* 2020). However, in March 1998 a dead 20-m (66-ft) male blue whale was brought into Rhode Island waters on the bow of a tanker. The cause of death was determined to be ship strike. Although it appears likely that the vessel concerned was responsible, the necropsy revealed some injuries that were difficult to explain in this context. The location of the strike was not determined; given the known rarity of blue whales in U.S. Atlantic waters, and the vessel's port of origin (Antwerp), it seems reasonable to suppose that the whale died somewhere to the north or east of the U.S. Atlantic EEZ.

Fishery Information

No fishery information is presented because there are no observed fishery-related mortalities or serious injury.

HABITAT ISSUES

Anthropogenic noise associated with seismic projects has been shown to affect blue whale acoustic activity in the Saint Lawrence Estuary, Canada (Iorio and Clark 2009). A 2016 DFO study performed a risk-mapping analysis of shipping-noise impacts on blue whales in the Saint Lawrence Estuary (Aulanier *et al.* 2016). It was determined that there was no area correlated with injury risk, and that while a large part of the study area had a low probability of experiencing shipping noise levels exceeding behavioral response thresholds, the risk of behavioral-level impacts near the shipping lanes might be present up to 30% of the time.

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

The status of this stock relative to OSP in the U.S. Atlantic EEZ is unknown, but the species is listed as endangered under the ESA. There are insufficient data to determine population trends for blue whales. The total level of human-caused mortality and serious injury is unknown, but it is believed to be insignificant and approaching a zero mortality and serious injury rate. This is a strategic stock because the blue whale is listed as an endangered species under the ESA. A draft of a revised Recovery Plan was published in October of 2018 (NMFS 2018).

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SPERM WHALE (*Physeter macrocephalus*): North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The distribution of the sperm whale in the U.S. Exclusive Economic Zone (EEZ) occurs on the continental shelf edge, over the continental slope, and into mid-ocean regions (Figure 1). Waring *et al.* (1993, 2001) suggested that this offshore distribution is more commonly associated with the Gulf Stream edge and other features. However, the sperm whales that occur in the eastern U.S. Atlantic EEZ likely represent only a fraction of the total stock. The nature of linkages of the U.S. habitat with those to the south, north, and offshore is unknown. Historical whaling records compiled by Schmidly (1981) suggested an offshore distribution off the southeast U.S., over the Blake Plateau, and into deep ocean waters. In the southeast Caribbean, both large and small adults, as well as calves and juveniles of different sizes are reported (Watkins *et al.* 1985). Whether the northwestern Atlantic population is discrete from northeastern Atlantic is currently unresolved. The International Whaling Commission recognizes one stock for the North Atlantic. Based on reviews of many types of stock studies (i.e., tagging, genetics, catch data, mark-recapture, biochemical markers, etc.), Reeves and Whitehead (1997) and Dufault *et al.* (1999) suggested that sperm whale populations have no clear geographic structure. Ocean-wide genetic studies (Lyrholm and Gyllensten 1998; Lyrholm *et al.* 1999) indicated low genetic diversity, but strong differentiation between potential social (matrilineally related) groups. Further, Englehaupt *et al.* (2009) found no differentiation for mtDNA between samples from the western North Atlantic and from the North Sea, but significant differentiation between samples from the Gulf of Mexico and from the Atlantic Ocean just outside the Gulf of Mexico. These ocean-wide findings, combined with observations from other studies, indicate stable social groups, site fidelity, and latitudinal range limitations in groups of females and juveniles (Whitehead 2002). In contrast, males migrate to polar regions to feed and move among populations to breed (Whitehead 2002, Englehaupt 2009). There exists one tag return of a male tagged off Browns Bank (Nova Scotia) in 1966 and returned from Spain in 1973 (Mitchell 1975). Another male taken off northern Denmark in August 1981 had been wounded the previous summer by whalers off the Azores (Reeves and Whitehead 1997). Steiner *et al*. (2012) reported on resightings of photographed individual male sperm whales between the Azores and Norway. In U.S. Atlantic EEZ waters, there appears to be a distinct seasonal cycle (CETAP 1982; Scott and Sadove 1997). In winter, sperm whales are concentrated east and northeast of Cape Hatteras. In spring, the center of distribution shifts northward to east of Delaware and Virginia, and is widespread throughout the central portion of the mid-Atlantic bight and the southern portion of Georges Bank. This is supported by acoustic studies in which detection of sperm whale vocalizations had a winter peak off Cape Hatteras, with the peak shifting farther north in the spring (Stanistreet *et al.* 2018). In summer, the distribution is similar but now also includes the area east and north of Georges Bank and into

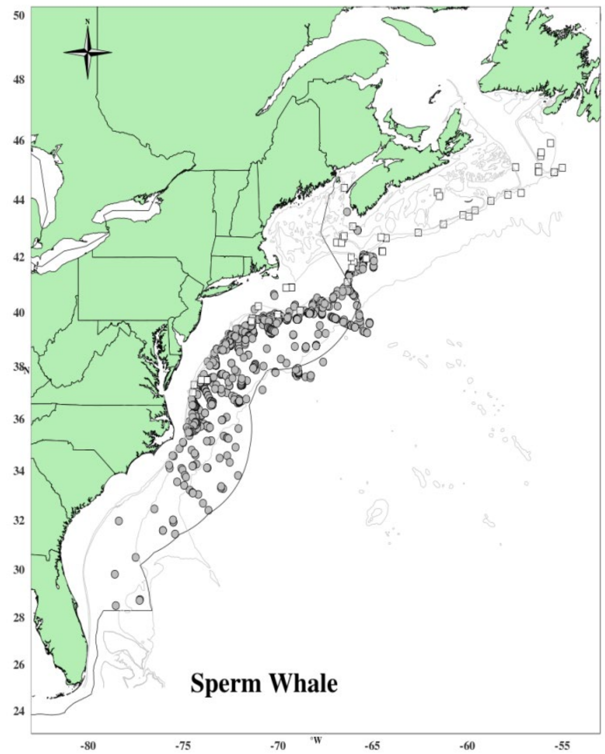

Figure 1. Distribution of sperm whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summer in 1998, 1999, 2002, 2004, 2006, 2011 and 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100m, 1,000m, and 4,000m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

the Northeast Channel region, as well as the continental shelf (inshore of the 100-m isobath) south of New England. In the fall, sperm whale occurrence south of New England on the continental shelf is at its highest level, and there remains a continental shelf edge occurrence in the mid-Atlantic bight. Similar inshore (<200 m) observations have been made on the southwestern (R.D. Kenney, pers. comm.) and eastern Scotian Shelf, particularly in the region of "the Gully" (Whitehead *et al.* 1991).

Geographic distribution of sperm whales may be linked to their social structure and their low reproductive rate, and both of these factors have management implications. Several basic groupings or social units are generally recognized—nursery schools, harem or mixed schools, juvenile or immature schools, bachelor schools, bull schools or pairs, and solitary bulls (Best 1979; Whitehead *et al.* 1991; Christal *et al.* 1998). These groupings have distinct geographical distributions, with females and juveniles generally based in tropical and subtropical waters, and males more wide-ranging and occurring in higher latitudes. Male sperm whales are present off and sometimes on the continental shelf along the entire east coast of Canada south of Hudson Strait, whereas, females rarely migrate north of the southern limit of the Canadian EEZ (Reeves and Whitehead 1997; Whitehead 2002). Off the northeastern U.S., Cetacean and Turtle Assessment Program (CETAP) and NEFSC sightings in shelf-edge and off-shelf waters included many social groups with calves/juveniles (CETAP 1982; Waring *et al.* 1992, 1993). The basic social unit of the sperm whale appears to be the mixed school of adult females plus their calves and some juveniles of both sexes, normally numbering 20–40 animals in all. There is evidence that some social bonds persist for many years (Christal *et al.* 1998).

POPULATION SIZE

Several estimates from selected regions of sperm whale habitat exist for select time periods, however, at present there is no reliable estimate of total sperm whale abundance for the entire North Atlantic. Sightings have been almost exclusively in the continental shelf edge and continental slope areas (Figure 1), however there has been little or no survey effort beyond the slope. The best recent abundance estimate for sperm whales is the sum of the 2016 surveys—4,349 (CV=0.28).

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. Due to changes in survey methodology these historical data should not be used to make comparisons to more current estimates.

Recent surveys and abundance estimates

An abundance estimate of 1,593 (CV=0.36) sperm whales was generated from a shipboard and aerial survey conducted during Jun–Aug 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portioned covered 3,107 km of tracklines that were in waters offshore of 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 double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers, 2004). Shipboard data were inspected to determine if there was significant responsive movement to the ship (Palka and Hammond 2001). Because there was an insignificant amount of responsive movement for this species, the estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 695 (CV=0.39) sperm whales was generated from a shipboard survey conducted concurrently (June-August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed the double-platform methodology searching with 25x bigeye binoculars. A total of 4,445 km of tracklines was surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Abundance estimates of 3,321 (CV=0.35),and 1,028 (CV=0.35) sperm whales 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 (Central Virginia) 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 isobath 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 abundance estimates for the western North Atlantic sperm whale (Physeter macrocephalus). Month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	Central Virginia to lower Bay of Fundy	1,593	0.36
Jun–Aug 2011	Central Florida to Central Virginia	695	0.39
Jun–Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	2,288	0.28
Jun–Aug 2016	Central Virginia to lower Bay of Fundy	3,321	0.35
Jun–Aug 2016	Central Florida to Virginia	1, 028	0.35
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	4,349	0.28

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 sperm whales is 4,349 (CV=0.28). The minimum population estimate for the western North Atlantic sperm whale is 3,451.

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. While more is probably known about sperm whale life history in other regions, some life history and vital rates information is available for the Northwest Atlantic. These include: calving interval is 4–6 years; lactation period is 24 months; gestation period is 14.5–16.5 months; births occur mainly in July to November; length at birth is 4.0 m; length at sexual maturity 11.0–12.5 m for males and 8.3–9.2 m for females; mean age at sexual maturity is 19 years for males and 9 years for females; and mean age at physical maturity is 45 years for males and 30 years for females (Best 1974; Best *et al.* 1984; Lockyer 1981; Rice 1989).

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 3,451. 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 0.10 because the sperm whale is listed as endangered under the Endangered Species Act (ESA). PBR for the western North Atlantic sperm whale is 6.9.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

There are no documented reports of fishery-related mortality or serious injury to this stock in the U.S. EEZ during 2013–2017.

Fishery Information

Detailed fishery information is reported in Appendix III.

Other Mortality

During 2013–2017, 12 sperm whale strandings were documented along the U.S. Atlantic coast within the EEZ (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018). None of these strandings were classified as human interactions.

Table 2. Sperm whale (Physeter macrocephalus) reported strandings along the U.S. and Canada Atlantic coast 2013–2017.

Stranding State or Province	2013	2014	2015	2016	2017	**Total**
Newfoundland/Labrador[a]	1	2	1	1	0	**5**
Nova Scotiab	1	0	0	0	0	**1**
Massachusetts	0	0	0	0	1	**1**
Virginia	0	0	0	1	0	**1**
North Carolina	1	0	0	0	1	**2**
Florida	1	5	0	1	1	**8**
TOTAL U.S.	**2**	**5**	**0**	**2**	**3**	**12**

a. Data provided by Whale Release and Strandings, Tangly Whales Inc. Newfoundland, Canada (Ledwell and Huntington 2013, 2014, 2015, 2017, 2018).

b. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.).

Mass strandings have been reported in many oceanic regions (Rice *et al*. 1986; Kompanje and Reumer 1995; Evans *et al*. 2002; Fujiwara *et al.* 2007; Pierce *et al.* 2007; Mazzariol *et al*. 2011). Reasons for the strandings are unknown, although multiple causes (e.g., topography, changes in geomagnetic field, solar cycles, ship strikes, global changes in water temperature and prey distribution, and pollution) have been suggested (Kirschvink *et al*. 1986; Brabyn and Frew 1994; Holsbeek *et al.* 1999; Mazzariol *et al*. 2011).

Ship strikes are another source of human-caused mortality (McGillivary *et al*. 2009; Carrillo and Ritter 2010). In May 1994 a ship-struck sperm whale was observed south of Nova Scotia (Reeves and Whitehead 1997), in May 2000 a merchant ship reported a strike in Block Canyon, and in 2001 the U.S. Navy reported a ship strike within the EEZ (NMFS, unpublished data). In 2006, a sperm whale was found dead from ship-strike wounds off Portland, Maine. In spring, the Block Canyon region is part of a major pathway for sperm whales entering southern New England continental shelf waters in pursuit of migrating squid (CETAP 1982; Scott and Sadove 1997). A 2012 Florida stranding mortality was classified as a vessel strike mortality.

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 sperm whales 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

This is a strategic stock because the species is listed as endangered under the ESA. 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 a zero mortality and serious injury rate. The status of this stock relative to OSP in U.S. Atlantic EEZ is unknown. There are insufficient data to determine population trends. The current stock abundance estimate was based upon a small portion of the known stock range. A Recovery Plan for sperm whales was finalized in 2010 (NMFS 2010).

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DWARF SPERM WHALE (*Kogia sima*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The dwarf sperm whale (*Kogia sima*) is distributed worldwide in temperate to tropical waters (Caldwell and Caldwell 1989; McAlpine 2009). Pygmy sperm whales and dwarf sperm whales (*K. sima*) are difficult to differentiate at sea (Caldwell and Caldwell 1989; Bloodworth and Odell 2008; McAlpine 2009), and sightings of either species are often categorized as *Kogia* sp. Sightings of *Kogia* whales in the western North Atlantic occur in oceanic waters along the continental shelf break and slope from Canada to Florida (Figure 1; Mullin and Fulling 2003; Roberts *et al.* 2015). In addition, stranding records for *Kogia* spp. are common from Canada to Florida (Bloodworth and Odell 2008; Berini *et al.* 2015). Based on the results of passive acoustic monitoring, Hodge *et al.* (2018) reported that *Kogia* are common in the western North Atlantic in continental shelf break and slope waters between Virginia and Florida, and more common than suggested by visual surveys.

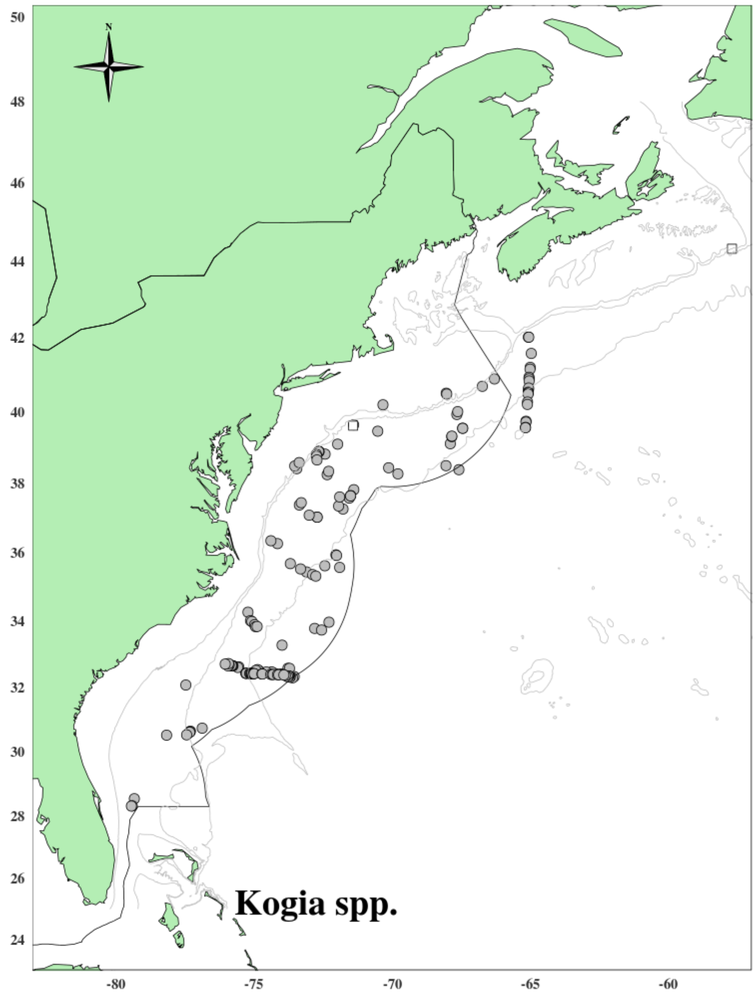

Figure 1. Distribution of Kogia spp. sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 200m, 1,000m and 4,000m depth contours. The darker line indicates the U.S. EEZ.

In addition to similarities in appearance, dwarf sperm whales and pygmy sperm whales demonstrate similarities in their foraging ecology as well as their acoustic signals. Staudinger *et al.* (2014) conducted diet and stable isotope analyses on stranded pygmy and dwarf sperm whales from the mid-Atlantic coast and found that the two species shared the same primary prey (cephalopods, primarily squid) and fed in similar habitats. The acoustic signals of dwarf and pygmy sperm whales cannot be distinguished from each other at this time because the signals of the two species are too similar to each other and to other species with narrow-band, high-frequency clicks (Merkens *et al.* 2018).

Across its geographic range, including the western North Atlantic, the population biology of dwarf sperm whales is inadequately known (Staudinger *et al.* 2014). Dwarf sperm whales in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. 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 western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding *et al.* 2007; Moore and

Merrick 2011). Within the western North Atlantic, the range of *Kogia* sightings traverses multiple marine ecoregions (Spalding *et al.* 2007) and crosses Cape Hatteras, a known biogeographic break for other marine species, so it is possible that multiple demographically independent populations exist within the western North Atlantic stock. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

POPULATION SIZE

Total numbers of dwarf sperm whales off the U.S. Atlantic coast are unknown. Because *K. sima* and *K. breviceps* are difficult to differentiate at sea, the reported abundance estimates are for both species of *Kogia* combined. The best estimate for *Kogia* spp. in the western North Atlantic is 7,750 (CV=0.38; Table 1; Garrison 2020; Palka 2020). This estimate is from summer 2016 surveys covering waters from central Florida to the lower Bay of Fundy. This estimate is almost certainly negatively biased. One component of line transect estimates is *g(0)*, the probability of seeing an animal on the transect line. Estimating *g(0)* is difficult because it consists of accounting for both perception bias (i.e., at the surface but missed) and availability bias (i.e., below the surface while in range of the observers), and many uncertainties (e.g., group size and diving behavior) can confound both (Marsh and Sinclair 1989; Barlow 1999). The long dive times of *Kogia* spp. contribute to a lower probability that animals will be available at the surface and therefore more negative bias. The best estimate was corrected for perception bias (see below) but not availability bias and is therefore an underestimate.

Earlier abundance estimates

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

Recent surveys and abundance estimates

An abundance estimate of 1,783 (CV=0.62) *Kogia* spp. was generated from aerial and shipboard surveys conducted during June–August 2011 between central Virginia and the lower Bay of Fundy (Palka 2012). The aerial portion covered 6,850 km of tracklines over waters north of New Jersey between the coastline and the 100-m depth contour through the U.S. and Canadian Gulf of Maine, and up to and including the lower Bay of Fundy. The shipboard portion covered 3,811 km of trackline between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a double-platform data collection procedure, which allowed estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 2,002 (CV=0.69) *Kogia* spp. was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida (Garrison 2016). This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25x bigeye binoculars. A total of 4,445 km of trackline were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Abundance estimates of 4,548 (CV=0.49) and 3,202 (CV=0.59) *Kogia* spp. were generated from two non-overlapping vessel surveys conducted in 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 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 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 abundance estimates for the western North Atlantic* Kogia *spp. with month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	central Virginia to lower Bay of Fundy	1,783	0.62
Jun–Aug 2011	central Florida to central Virginia	2,002	0.69
Jun–Aug 2011	central Florida to lower Bay of Fundy (COMBINED)	3,785	0.47
Jun–Aug 2016	New Jersey to lower Bay of Fundy	4,548	0.49
Jun–Aug 2016	central Florida to New Jersey	3,202	0.59
Jun–Aug 2016	central Florida to lower Bay of Fundy (COMBINED)	7,750	0.38

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 *Kogia* spp. is 7,750 (CV=0.38). The minimum population estimate for *Kogia* spp. is 5,689 animals.

Current Population Trend

There are three available coastwide abundance estimates for *Kogia* spp. from the summers of 2004, 2011, and 2016. Each of these is derived from vessel surveys with similar survey designs and all three used the two-team independent observer approach to estimate abundance. The resulting estimates were 395 (CV=0.4) in 2004, 3,785 (CV=0.47) in 2011, and 7,750 (CV=0.38) in 2016 (Garrison 2020; Palka 2020). While there is an apparent increasing trend in these population estimates, a generalized linear model did not indicate a statistically significant (p=0.071) trend. The high level of uncertainty in these estimates limits the ability to detect a statistically significant trend. In addition, interpretation of trends is complicated by two methodological factors. First, the ability to detect *Kogia* spp. visually is highly dependent upon weather and visibility conditions which may contribute to differences between estimates. Second, during 2016 both surveys did not use scientific echosounders during some survey periods. Changing the use of echosounders may affect the surfacing/diving patterns of the animals and thus have an influence on the availability of animals to the visual survey teams. Finally, a key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock. Therefore, the possible increasing trend should be interpreted with caution.

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 for *Kogia* spp. is 5,689. 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 western North Atlantic *Kogia* spp. is 46.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to dwarf sperm whales or *Kogia* spp. in the western North Atlantic.

Fishery Information

The commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean are the Category I Atlantic Highly Migratory Species longline and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively.

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of dwarf sperm whales or *Kogia* sp. within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery. During 2013–2017, there were no observed mortalities or serious injuries of dwarf sperm whales or *Kogia* spp. by this fishery (Garrison and Stokes 2014; 2016; 2017; 2019; 2020). Historically, observed takes of *Kogia* spp. have been rare, and the most recent observed take occurred in 2011. Please see Appendix V for historical estimates of annual mortality and serious injury for *Kogia* spp. by this fishery.

Other Mortality

During 2013–2017, 46 dwarf sperm whales were reported stranded along the U.S. Atlantic coast from New York to Florida (Table 2; Northeast Regional Marine Mammal Stranding Network, Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER)). It could not be determined whether there was evidence of human interaction for 20 of these strandings, and for 26 strandings, no evidence of human interaction was detected. In addition, there were 12 records of unidentified stranded *Kogia*. It could not be determined whether there was evidence of human interaction for 10 of these strandings; for one, no evidence of human interaction was detected; and for the remaining stranding, evidence of human interaction was self-reported by a citizen who transported the animal.

Stranding data probably 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

Table 2. Dwarf and pygmy sperm whale (**Kogia sima** ***(Ks),*** **Kogia breviceps** ***(Kb) and*** **Kogia** ***sp. (Sp)) strandings along the Atlantic coast, 2013–2017. Strandings that were not reported to species have been reported as*** **Kogia** ***sp. 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	2013			2014			2015			2016			2017			TOTALS		
	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp
Massachusetts	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0
Rhode Island	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0
New York	0	2	0	0	1	0	0	0	0	0	2	0	2	1	0	2	6	0
New Jersey	1	1	0	0	1	0	0	0	0	0	1	0	0	3	0	1	6	0
Delaware	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0
Maryland	0	0	0	0	1	0	0	2	0	0	0	0	0	0	0	0	3	0
Virginia	1	2	0	1	2	0	0	0	0	0	0	0	0	2	0	2	6	0
North Carolina	3	4	0	3	4	1	12	4	0	2	2	0	0	2	1	20	16	2
South Carolina	2	2	0	0	3	0	1	8	0	0	2	0	1	3	0	4	18	0
Georgia	0	5	1	5	1	0	0	3	0	0	3	0	0	2	0	5	14	1
Florida	0	9	6	0	9	0	5	12	2	4	9	0	3	7	1	12	46	9
TOTALS	7	27	7	9	25	1	18	29	2	6	19	0	6	20	2	46	120	12

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018). Bryan *et al.* (2012) examined liver and kidney samples from stranded pygmy sperm whales from the U.S. Atlantic and Gulf of Mexico and found that all samples contained mercury concentrations in excess of the USEPA action limits, potentially levels hazardous to the health of whales and putting them at greater risk of disease. Because animals are exposed to mercury through the consumption of their prey, and the foraging ecology of dwarf sperm whales is similar to that of pygmy sperm whales (Staudinger *et al.* 2014), dwarf sperm whales are likely also experiencing potentially hazardous levels of mercury. Reed *et al.* (2015) examined metal concentrations in dwarf sperm whales stranded along the South Carolina coast, and found that levels of mercury for all adults and cadmium for most adults, exceeded FDA historical levels of concern, while concentrations of some metals were low.

Harmful algal blooms have been responsible for large-scale marine mammal mortality events as well as chronic, harmful health effects and reproductive failure (Fire *et al.* 2009). Diatoms of the genus *Pseudo-nitzschia* produce domoic acid, a neurotoxin. Fire *et al.* (2009) sampled pygmy and dwarf sperm whales stranded along the U.S. East Coast from Virginia to Florida, and more than half (59%) of the samples tested positive for domoic acid, indicating year-round, chronic exposure, whereas other cetaceans stranded in the same area had no detectable domoic acid. Harmful algal blooms may be occurring in offshore areas not currently being monitored, and the detection only in *Kogia* species suggests a possible unknown, unique aspect of their foraging behavior or habitat utilization (Fire *et al.* 2009).

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; Pinsky *et al*. 2013; Poloczanska *et al.* 2013; 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

Dwarf sperm whales 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. 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 U.S. Atlantic EEZ relative to OSP is unknown. There was no statistically significant trend in population size for *Kogia* spp.; however, there are key methodological issues and uncertainty that limit the ability to evaluate trend.

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PYGMY SPERM WHALE (*Kogia breviceps*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pygmy sperm whale (*Kogia breviceps*) is distributed worldwide in temperate and tropical waters (Caldwell and Caldwell 1989; McAlpine 2009). Pygmy sperm whales and dwarf sperm whales (*K. sima*) are difficult to differentiate at sea (Caldwell and Caldwell 1989; Bloodworth and Odell 2008; McAlpine 2009), and sightings of either species are often categorized as *Kogia* sp. Sightings of the two *Kogia* species in the western North Atlantic occur in oceanic waters along the continental shelf break and slope from Canada to Florida (Figure 1; Mullin and Fulling 2003; Roberts *et al.* 2015). In addition, stranding records for *Kogia* spp. are common from Canada to Florida (Bloodworth and Odell 2008; Berini *et al.* 2015). Based on the results of passive acoustic monitoring, Hodge *et al.* (2018) reported that *Kogia* are common in the western North Atlantic in continental shelf break and slope waters between Virginia and Florida, and more common than suggested by visual surveys.

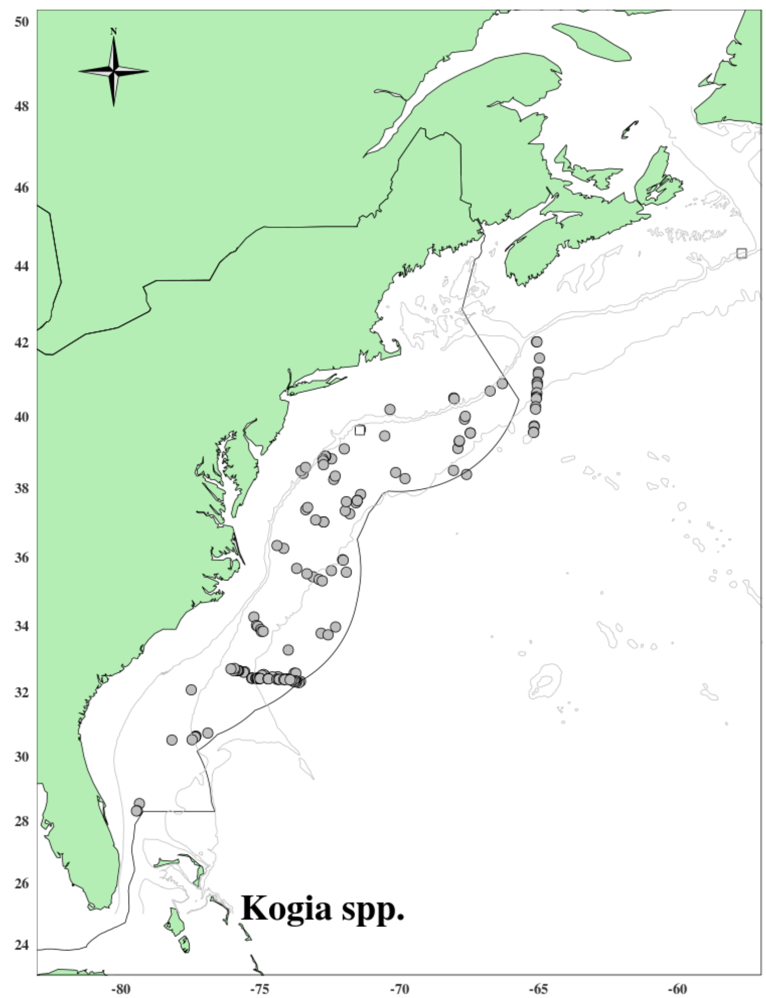

Figure 1. Distribution of Kogia spp. sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 200m, 1,000m and 4,000m depth contours. The darker line indicates the U.S. EEZ.

In addition to similarities in appearance, dwarf sperm whales and pygmy sperm whales demonstrate similarities in their foraging ecology as well as their acoustic signals. Staudinger *et al.* (2014) conducted diet and stable isotope analyses on stranded pygmy and dwarf sperm whales from the mid-Atlantic coast and found that the two species shared the same primary prey and fed in similar habitats. The acoustic signals of dwarf and pygmy sperm whales cannot be distinguished from each other at this time because the signals of the two species are too similar to each other and to other species with narrow-band, high-frequency clicks (Merkens *et al.* 2018).

Across its geographic range, including the western North Atlantic, the population biology of pygmy sperm whales is inadequately known (Staudinger *et al.* 2014). Pygmy sperm whales in the western North Atlantic Ocean are managed separately from those in the northern Gulf of Mexico. 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 western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding *et al.* 2007; Moore and Merrick 2011). Within the western North Atlantic, the range of *Kogia* sightings traverses multiple marine ecoregions (Spalding *et al.* 2007) and crosses Cape Hatteras, a known biogeographic break for other marine species, so it is possible that multiple demographically independent populations exist within the western North Atlantic stock. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

POPULATION SIZE

Total numbers of pygmy sperm whales off the U.S. Atlantic coast are unknown. Because *K. breviceps* and *K. sima* are difficult to differentiate at sea, the reported abundance estimates are for both species of *Kogia* combined. The best abundance estimate for *Kogia* spp. in the western North Atlantic is 7,750 (CV=0.38; Table 1; Garrison 2020; Palka 2020). This estimate is from summer 2016 surveys covering waters from central Florida to the lower Bay of Fundy. This estimate is almost certainly negatively biased. One component of line transect estimates is *g(0)*, the probability of seeing an animal on the transect line. Estimating *g(0)* is difficult because it consists of accounting for both perception bias (i.e., at the surface but missed) and availability bias (i.e., below the surface while in range of the observers), and many uncertainties (e.g., group size and diving behavior) can confound both (Marsh and Sinclair 1989; Barlow 1999). The long dive times of *Kogia* spp. contribute to a lower probability that animals will be available at the surface and therefore more negative bias. The best estimate was corrected for perception bias (see below) but not availability bias and is therefore an underestimate.

Earlier abundance estimates

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

Recent surveys and abundance estimates

An abundance estimate of 1,783 (CV=0.62) *Kogia* spp. was generated from aerial and shipboard surveys conducted during June–August 2011 between central Virginia and the lower Bay of Fundy (Palka 2012). The aerial portion covered 6,850 km of trackline over waters north of New Jersey between the coastline and the 100-m depth contour through the U.S. and Canadian Gulf of Maine, and up to and including the lower Bay of Fundy. The shipboard portion covered 3,811 km of trackline between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a double-platform data collection procedure, which allowed estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 2,002 (CV=0.69) *Kogia* spp. was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida (Garrison 2016). This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams. A total of 4,445 km of trackline were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Abundance estimates of 4,548 (CV=0.49) and 3,202 (CV=0.59) *Kogia* spp. were generated from two non-overlapping 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 included 5,354 km of on-effort trackline along the shelf break and offshore to the outer edge 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 edge of the U.S. EEZ from 30 June to 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 (Thomas *et al.* 2009). Estimates from the two surveys were combined and CVs pooled to produce a species abundance estimate for the stock area.

Table 1. Summary of abundance estimates for the western North Atlantic* Kogia *spp. with month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	central Virginia to lower Bay of Fundy	1,783	0.62
Jun–Aug 2011	central Florida to central Virginia	2,002	0.69
Jun–Aug 2011	central Florida to lower Bay of Fundy (COMBINED)	3,785	0.47
Jun–Aug 2016	New Jersey to lower Bay of Fundy	4,548	0.49
Jun–Aug 2016	central Florida to New Jersey	3,202	0.59
Jun–Aug 2016	central Florida to lower Bay of Fundy (COMBINED)	7,750	0.38

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 *Kogia* spp. is 7,750 (CV=0.38). The minimum population estimate for *Kogia* spp. is 5,689 animals.

Current Population Trend

There are three available coastwide abundance estimates for *Kogia* spp. from the summers of 2004, 2011, and 2016. Each of these is derived from vessel surveys with similar survey designs and all three used the two-team independent observer approach to estimate abundance. The resulting estimates were 395 (CV=0.4) in 2004, 3,785 (CV=0.47) in 2011, and 7,750 (CV=0.38) in 2016 (Garrison and Palka 2018). While there is an apparent increasing trend in these population estimates, a generalized linear model did not indicate a statistically significant (p=0.071) trend. The high level of uncertainty in these estimates limits the ability to detect a statistically significant trend. In addition, interpretation of trends is complicated by two methodological factors. First, the ability to detect *Kogia* spp. visually is highly dependent upon weather and visibility conditions which may contribute to differences between estimates. Second, during 2016 both surveys did not use scientific echosounders during some survey periods. Changing the use of echosounders may affect the surfacing/diving patterns of the animals and thus have an influence on the availability of animals to the visual survey teams. Finally, a key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock. Therefore, the possible increasing trend should be interpreted with caution.

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 for *Kogia* spp. is 5,689. 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 western North Atlantic *Kogia* spp. is 46.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to pygmy sperm whales or *Kogia* spp. in the western North Atlantic.

Fishery Information

The commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean are the Category I Atlantic Highly Migratory Species longline and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively.

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of pygmy sperm whales or *Kogia* sp. within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ. Pelagic swordfish, tunas and billfish are the targets of the large pelagics longline fishery. During 2013–2017, there were no observed mortalities or serious injuries of pygmy sperm whales or *Kogia* spp. by this fishery (Garrison and Stokes 2014; 2016; 2017; 2019; 2020). Historically, observed takes of *Kogia* spp. have been rare, and the most recent observed take occurred in 2011. Please see Appendix V for historical estimates of annual mortality and serious injury for *Kogia* spp. by this fishery.

Other Mortality

During 2013–2017, 120 pygmy sperm whales were reported stranded along the U.S. Atlantic coast from Massachusetts to Florida (Table 2; Northeast Regional Marine Mammal Stranding Network, Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER)). It could not be determined whether there was evidence of human interaction for 51 of these strandings, and for 59 strandings, no evidence of human interaction was detected. For the remaining ten pygmy sperm whale strandings, evidence of human interaction was detected. Six of the ten with evidence of human interaction had ingested plastic debris. In addition, there were 12 records of unidentified *Kogia*. It could not be determined whether there was evidence of human interaction for ten of these strandings; for one, no evidence of human interaction was detected; and for the remaining stranding, human interaction was self-reported by a citizen who transported the animal to a new location.

Stranding data probably 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

Table 2. Dwarf and pygmy sperm whale (**Kogia sima** ***(Ks),*** **Kogia breviceps** ***(Kb) and*** **Kogia** ***sp. (Sp)) strandings along the Atlantic coast, 2013–2017. Strandings that were not reported to species have been reported as*** **Kogia** ***sp. 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	2013			2014			2015			2016			2017			TOTALS		
	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp	Ks	Kb	Sp
Massachusetts	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0
Rhode Island	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0
New York	0	2	0	0	1	0	0	0	0	0	2	0	2	1	0	2	6	0
New Jersey	1	1	0	0	1	0	0	0	0	0	1	0	0	3	0	1	6	0
Delaware	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0
Maryland	0	0	0	0	1	0	0	2	0	0	0	0	0	0	0	0	3	0
Virginia	1	2	0	1	2	0	0	0	0	0	0	0	0	2	0	2	6	0
North Carolina	3	4	0	3	4	1	12	4	0	2	2	0	0	2	1	20	16	2
South Carolina	2	2	0	0	3	0	1	8	0	0	2	0	1	3	0	4	18	0
Georgia	0	5	1	5	1	0	0	3	0	0	3	0	0	2	0	5	14	1
Florida	0	9	6	0	9	0	5	12	2	4	9	0	3	7	1	12	46	9
TOTALS	7	27	7	9	25	1	18	29	2	6	19	0	6	20	2	46	120	12

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018). Bryan *et al.* (2012) examined liver and kidney samples from stranded pygmy sperm whales from the U.S. Atlantic and Gulf of Mexico and found that all samples contained mercury concentrations in excess of the USEPA action limits, potentially levels hazardous to the health of whales and putting them at greater risk of disease.

Harmful algal blooms have been responsible for large-scale marine mammal mortality events as well as chronic, harmful health effects and reproductive failure (Fire *et al.* 2009). Diatoms of the genus *Pseudo nitzschia* produce domoic acid, a neurotoxin. Fire *et al.* (2009) sampled pygmy and dwarf sperm whales stranded along the U.S. east coast from Virginia to Florida, and more than half (59%) of the samples tested positive for domoic acid, indicating year-round, chronic exposure, whereas other cetaceans stranded in the same area had no detectable domoic acid. Harmful algal blooms may be occurring in offshore areas not currently being monitored, and the detection only in *Kogia* species suggests a possible unknown, unique aspect of their foraging behavior or habitat utilization (Fire *et al.* 2009).

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; Pinsky *et al*. 2013; Poloczanska *et al.* 2013; 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

Pygmy sperm whales 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. 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 U.S. Atlantic EEZ relative to OSP is unknown. There was no statistically significant trend in population size for *Kogia* spp.; however, there are key methodological issues and uncertainty that limit the ability to evaluate trend.

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PYGMY KILLER WHALE (*Feresa attenuata*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pygmy killer whale is distributed worldwide in tropical and sub-tropical waters (Jefferson *et al.* 1994). However, sightings of this species in the western North Atlantic are extremely rare and stranding records are also sparse, probably due to the natural rarity of the species (Baird 2018; Braulik 2018). In the western North Atlantic, strandings are recorded from primarily South Carolina and Georgia, with two from North Carolina and one from Massachusetts, and there have been two sighting during NMFS vessel surveys from 1992 to 2016. In the Hawaiian Islands, there is evidence for limited movement of individuals and for island-associated populations (Baird 2018), and the author suggested it is likely that there is population structure within the species elsewhere. Pygmy killer whales in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. 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 (Baird 2018) and is further supported because the two stocks occupy distinct marine ecoregions (Spalding *et al*. 2007; Moore and Merrick 2011). Due to the paucity of sightings in the western North Atlantic, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

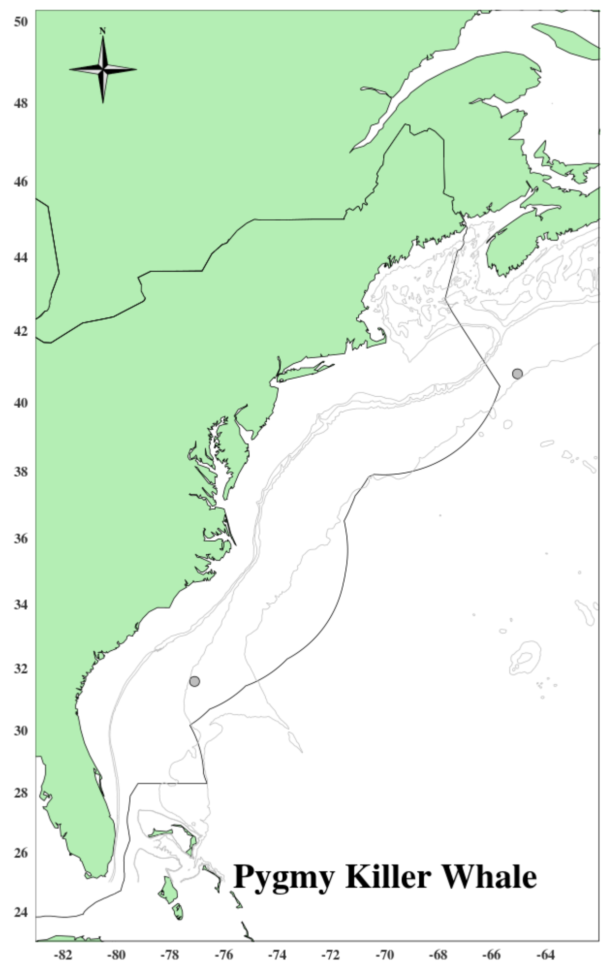

Figure 1. Distribution of pygmy killer whale sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1992, 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2013 and 2016. Isobaths are the 100m, 200m, 1,000m and 4,000m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The number of pygmy killer whales off the U.S. Atlantic coast is unknown since it was rarely seen in any surveys. A single group of six pygmy killer whales was sighted in waters ~1500 m deep off Georgia during a 1992 NMFS winter vessel survey (Hansen *et al.* 1994), and a single pygmy killer whale was sighted in waters ~4000 m deep far offshore of Long Island, New York, during a 2013 NMFS summer vessel survey (NEFSC and SEFSC 2013). Abundances have not been estimated from these single sightings. Several cruises—a winter 2002 cruise, a summer 2005 cruise, and a summer 2016 cruise—each had one or two sightings of pygmy killer or melon-headed whales (identity was not confirmed), and these groups were recorded off Cape Hatteras or off the North Carolina/South Carolina border.

Minimum Population Estimate

Present data are insufficient to calculate a minimum population estimate for this stock.

Current Population Trend

There are insufficient data to determine the population trends for this stock because no estimates of population size are available.

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 history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal level (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 is unknown. 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 this stock is of unknown status. PBR for the western North Atlantic stock of pygmy killer whales is unknown.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to pygmy killer whales in the western North Atlantic.

Fishery Information

The commercial fishery that could potentially interact with this stock in the Atlantic Ocean is the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery. Percent observer coverage (percentage of sets observed) for this fishery for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively. There were no observed mortalities or serious injuries to pygmy killer whales by this fishery in the Atlantic Ocean during 2013–2017 (Garrison and Stokes 2014; 2016; 2017; 2019; 2020). Detailed fishery information is reported in Appendix III.

There has historically been some take of this species in small cetacean fisheries in the Caribbean (Caldwell and Caldwell 1971).

Other Mortality

Three strandings of pygmy killer whales were reported along the U.S. East Coast during 2013–2017 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER)). All three strandings occurred in Virginia during 2013. For two strandings, it could not be determined if there was evidence of human interaction, and for the remaining stranding, no evidence of human interaction was detected. Stranding data probably 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.Table 1. Pygmy killer whale (Feresa attenuata) reported strandings

HABITAT ISSUES

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.

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for this stock 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; 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

Pygmy killer whales 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. No fishery-related mortality or serious injury has been observed during 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 western U.S. Atlantic EEZ relative to OSP is unknown. There are insufficient data to determine the population trends for this species.

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FALSE KILLER WHALE (*Pseudorca crassidens*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The false killer whale is distributed worldwide throughout warm temperate and tropical oceans (Jefferson *et al.* 2008). This species is usually sighted in offshore waters but in some cases inhabits waters closer to shore, particularly around oceanic islands (e.g., Hawaii, Baird *et al.* 2013). While sightings from the U.S. western North Atlantic have been uncommon (Figure 1), the combination of sighting, stranding and bycatch records indicates that this species routinely occurs in the western North Atlantic. False killer whales have been sighted in U.S. Atlantic waters from southern Florida to Maine (Schmidly 1981). There are periodic records (primarily stranding) from southern Florida to Cape Hatteras dating back to 1920 (Schmidly 1981). Most of the records are from the southern half of Florida and include a mass stranding in 1970 that may have numbered as many as 175 individuals (Caldwell *et al*. 1970; Schmidly 1981).

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 western North Atlantic are managed separately from those in the northern Gulf of Mexico. 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 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). Given the paucity of sightings, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

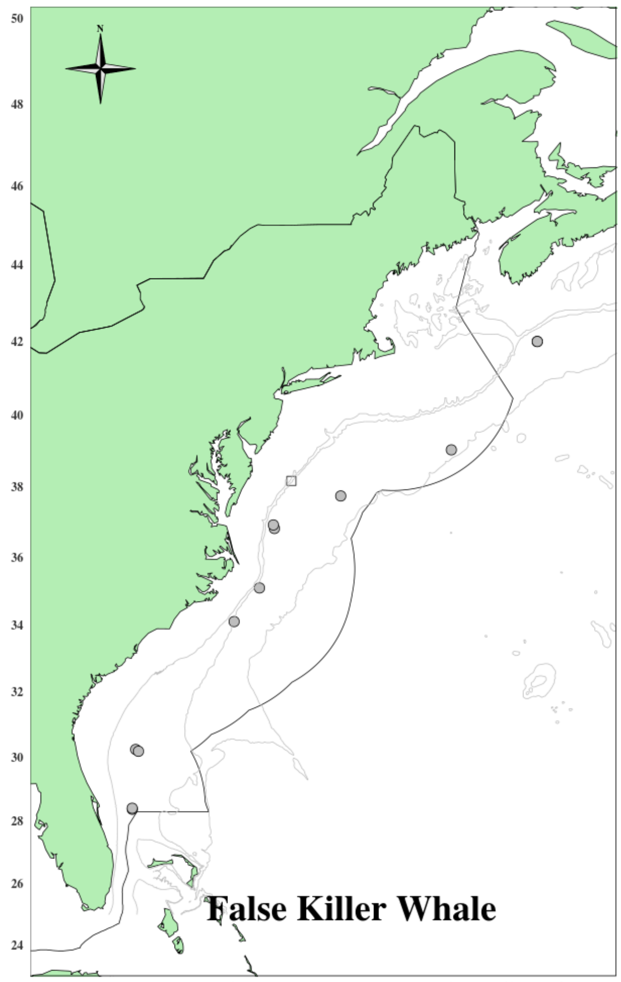

Figure 1. Distribution of false killer whale sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 200m, 1,000m, and 4,000m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The best available abundance estimate for western North Atlantic false killer whales is 1,791 (CV=0.56; Table 1; Garrison 2020; Palka 2020). This estimate is from summer 2016 surveys covering waters from central Florida to the lower Bay of Fundy.

Recent surveys and abundance estimates

There were no sightings of false killer whales during aerial and shipboard surveys conducted during June-August 2011 from central Virginia to the lower Bay of Fundy. The aerial portion covered 6,850 km of tracklines over waters north of New Jersey between the coastline and the 100-m depth contour through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,811 km of tracklines between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a double-platform data collection procedure.

An abundance estimate of 442 (CV=1.06; Table 1) false killer whales based one sighting was generated from a shipboard survey conducted concurrently (June-August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25x bigeye binoculars. A total of 4,445 km of tracklines was surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Abundance estimates of 1,182 (CV=0.63) and 609 (CV=1.08) false killer whales 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 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 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. It should be noted that the abundance estimate from the second vessel survey was based on a single sighting and therefore has a very high uncertainty.

Table 1. Summary of abundance estimates for the western North Atlantic false killer whale* (Pseudorca crassidens) *by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun-Aug 2011	central Virginia to lower Bay of Fundy	0	0-
Jun-Aug 2011	central Florida to central Virginia	442	1.06
Jun-Aug 2011	central Florida to lower Bay of Fundy (COMBINED)	442	1.06
Jun–Aug 2016	New Jersey to lower Bay of Fundy	1,182	0.63
Jun–Aug 2016	central Florida to New Jersey	609	1.08
Jun–Aug 2016	central Florida to lower Bay of Fundy (COMBINED)	1,791	0.56

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 false killer whales is 1,791 (CV=0.56). The minimum population estimate for false killer whales is 1,154.

Current Population Trend

False killer whales are rarely sighted during abundance surveys, and the resulting estimates of abundance are both highly variable between years and highly uncertain. The rare encounter rates limit the ability to assess or interpret trends in population size.

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 history (Barlow *et al.* 1995).

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 (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size is 1,154. 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 this stock is of unknown status. PBR for the western North Atlantic false killer whale stock is 12.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to false killer whales in the western North Atlantic.

Fishery Information

The commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean are the Category I Atlantic Highly Migratory Species longline and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively.

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of false killer whales within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to false killer whales by this fishery in the Atlantic Ocean during 2013–2017 (Garrison and Stokes 2014; 2016; 2017; 2019; 2020).

Other Mortality

There was one reported stranding of a false killer whale in the U.S. Atlantic Ocean during 2013–2017 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER)). This stranding occurred off Florida in 2013, and it could not be determined if there was evidence of human interaction. Historically, there have been intermittent false killer whale strandings. From 1990 through 2012, the following seven false killer whale strandings occurred: one animal in 2009 and one in 2002 in North Carolina; two in Florida in 1997; one in Massachusetts in 1997; one in Georgia in 1996; and one in Florida in 1995. Stranding data probably 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

HABITAT ISSUES

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.

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., Schwacke *et al.* 2002; Jepson *et al.*

2016; Hall *et al.* 2018), but research on contaminant levels for this stock 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; 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

False killer whales 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. While no fishery-related mortality or serious injury has been observed in the last five years, there was a recorded interaction with the pelagic longline fishery in 2011. False killer whale interactions with longline fisheries in the Pacific are of considerable concern, but little is known about interactions in the Atlantic. Thus, insufficient information is 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 false killer whales in the U.S. EEZ relative to OSP is unknown. There are insufficient data to determine population trends for this stock.

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CUVIER'S BEAKED WHALE (*Ziphius cavirostris*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The distribution of Cuvier's beaked whales is poorly known, and is based mainly on stranding records (Leatherwood *et al.* 1976). Strandings have been reported from Nova Scotia along the eastern U.S. coast south to Florida, around the Gulf of Mexico, and within the Caribbean (Leatherwood *et al.* 1976; CETAP 1982; Heyning 1989; Houston 1990; MacLeod *et al.* 2006; Jefferson *et al.* 2008). Acoustic presence has been demonstrated from recordings collected from North Carolina to Nova Scotia (Stanistreet 2018).

Stock structure in the North Atlantic is unknown. A study of 20 Cuvier's beaked whales satellite-tagged offshore of Cape Hatteras, North Carolina, between 2014 and 2017 suggested that these animals have very restricted movements and could be a resident population (Foley 2018). Because the current stock spans multiple eco-regions (Longhurst 2007; Spalding *et al.* 2007), it is plausible that the stock could actually contain multiple demographically independent populations that should themselves be stocks.

Cuvier's beaked whale sightings have occurred principally along the continental shelf edge in the Mid-Atlantic region off the northeast U.S. coast (CETAP 1982; Waring *et al.* 1992; Waring *et al.* 2001; Hamazaki 2002; Palka 2006). Monthly aerial surveys conducted off Cape Hatteras between 2011 and 2015 recorded Cuvier's beaked whales sighted during every month of the year (McLellan *et al.* 2018) and acoustic recordings confirm consistent year-round presence (Stanistreet *et al.* 2017).

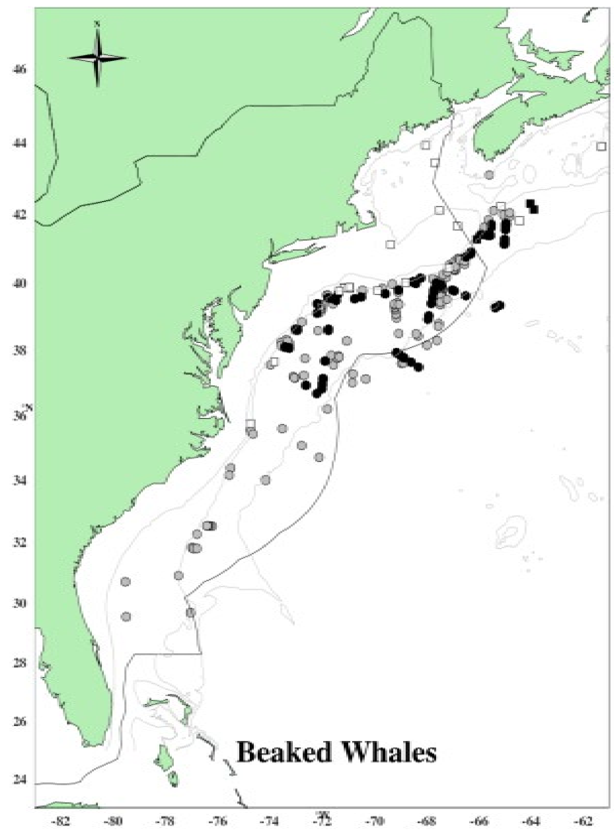

Figure 1. Distribution of beaked whale (includes Ziphius and Mesoplodon spp.) sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, and 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings. Black symbols are sightings identified as Cuvier's beaked whales.

POPULATION SIZE

The best abundance estimate for undifferentiated beaked whales is sum of the northeast and southeast 2016 surveys—5,744 (CV=0.36). 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 two surveys were added together and the CVs pooled using a delta method to produce an abundance estimate for the stock area.

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, and should not be used for PBR determinations. Further, due to changes in survey methodology these data should not be used to make comparisons to more current estimates.

Recent surveys and abundance estimates

Abundance estimates of 3,897 (CV=0.47) and 1,847 (CV=0.49) Cuvier's beaked whales (not including Mesoplodon spp.) 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 isobath 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 an abundance estimate for the stock area, yielding an combined total of 5,744 Cuvier's beaked whales (CV=0.36). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE, and these numbers likely include an unknown number of Cuvier's beaked whales.

An abundance estimate of 4,962 (CV=0.37) Cuvier's beaked whales (not including Mesoplodon spp.) was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines that were in water offshore of North Carolina to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the U.S. EEZ). Both sighting platforms used a double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers, 2004). Shipboard data were inspected to determine if there was significant responsive movement to the ship (Palka and Hammond 2001). Because there was an insignificant amount of responsive movement for this species, the estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling (MRDS) option in the computer program Distance (version 6.0, release 2, Thomas et al. 2009).

An abundance estimate of 1,570 (CV=0.65) Cuvier's beaked whales (not including Mesoplodon spp.) was also generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25× bigeye binoculars. A total of 4,445 km of tracklines were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas et al. 2009).

Table 1. Summary of abundance estimates for the wester North Atlantic stock of Cuvier's beaked whales. Month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	**Area**	N_{best}	**CV**
Jul−Aug 2011	central Virginia to lower Bay of Fundy	4,962	0.37
Jun−Aug 2011	central Virginia to central Florida	1,570	0.65
Jun−Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	6,532	0.32
Jun−Sep 2016	Central Virginia to lower Bay of Fundy	3,897	0.47
Jun−Aug 2016	Central Florida to Virginia	1,847	0.49
Jun−Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	5,744	0.36

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 undifferentiated beaked whales is 5,744 (CV=0.36). The minimum population estimate for undifferentiated beaked whales in the western North Atlantic is 4,282.

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. Life history parameters that could be used to estimate net productivity include: length at birth is 2 to 3 m, length at sexual maturity is 6.1m for females, and 5.5 m for males, maximum age for females were 30 growth layer groups (GLG's) and for males was 36 GLG's, which may be annual layers (Mitchell 1975; Mead 1984; Houston 1990).

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 for undifferentiated beaked whales is 4,282. 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. PBR for Cuvier's beaked whales is 43.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2013–2017 minimum annual rate of human-caused mortality of Cuvier's beaked whales averaged 0.2 animals per year. This is from 1 stranding record that reported signs of human interaction (plastic ingestion; Table 2).

Fishery Information

Detailed U.S. fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

Other Mortality

During 2013–2017, 7 Cuvier's beaked whales stranded along the U.S. Atlantic coast (Table 2; NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). One animal showed evidence of a human interaction.

Several unusual mass strandings of beaked whales throughout their worldwide range have been associated with naval activities (Cox *et al*. 2006; D'Amico *et al.* 2009; Fernandez *et al*. 2005; Filadelfo *et al.* 2009). During the mid- to late 1980s multiple mass strandings of Cuvier's beaked whales (4 to about 20 per event) and small numbers of Gervais' beaked whale and Blainville's beaked whale occurred in the Canary Islands (Simmonds and Lopez-Jurado 1991). Twelve Cuvier's beaked whales that live stranded and subsequently died in the Mediterranean Sea on 12-13 May 1996 were associated with low frequency acoustic sonar tests conducted by the North Atlantic Treaty Organization (Frantzis 1998; D'Amico *et al*. 2009; Filadelfo *et al*. 2009). In March 2000, 14 beaked whales live stranded in the Bahamas; 6 beaked whales (5 Cuvier's and 1 Blainville's) died (Balcomb and Claridge 2001; NMFS

2001; Cox *et al.* 2006). Four Cuvier's, 2 Blainville's and 2 unidentified beaked whales were returned to sea. The fate of the animals returned to sea is unknown, since none of the whales have been resighted. Necropsies of 6 dead beaked whales revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand. Subsequently, the animals died due to extreme physiologic stress associated with the physical stranding (i.e., hyperthermia, high endogenous catecholamine release) (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).

Table 2. Cuvier's beaked whale (Ziphius cavirostris) strandings along the U.S. Atlantic coast.

State	**2013**	**2014**	**2015**	**2016**	**2017**	**Total**
New York	0	1	1	0	0	**2**
North Carolina	0	0	0	1	1	**2**
Florida[a]	1	1	0	0	1	**3**
Total	**1**	**2**	**1**	**1**	**2**	**7**

a. Animal in Florida in 2014 had plastic bags and line in first stomach chamber.

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 beaked whales 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

The western North Atlantic stock of Cuvier's beaked whale is not a strategic stock because average annual human-related mortality and serious injury does not exceed PBR. The total U.S. fishery mortality and serious injury for this group of species 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 Cuvier's beaked whale relative to OSP in the U.S. Atlantic EEZ is unknown. This species is not listed as threatened or endangered under the Endangered Species Act.

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BLAINVILLE'S BEAKED WHALE (*Mesoplodon densirostris*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Within the genus *Mesoplodon*, there are four species of beaked whales that reside in the northwest Atlantic. These include True's beaked whale, *M. mirus*; Gervais' beaked whale, *M. europaeus*; Blainville's beaked whale, *M. densirostris*; and Sowerby's beaked whale, *M. bidens* (Mead 1989). These species are difficult to identify to the species level at sea; therefore, much of the available characterization for beaked whales is to genus level only. Stock structure for each species is unknown. Thus, it is plausible that the stock could actually contain multiple demographically independent populations that should themselves be stocks, because the current stock spans multiple eco-regions (Longhurst 2007; Spalding *et al.* 2007).

The distributions of *Mesoplodon* spp. in the Northwest Atlantic are known principally from stranding records (Mead 1989; Nawojchik 1994; Mignucci-Giannoni *et al.* 1999; MacLeod *et al.* 2006; Jefferson *et al.* 2008). Off the U.S. Atlantic coast, beaked whale (*Mesoplodon* spp.) sightings have occurred principally along the shelf-edge and in deeper oceanic waters (Figure 1; CETAP 1982; Waring et al. 1992, 2001; Tove 1995; Hamazaki 2002; Palka 2006). Most sightings were in late spring and summer, which corresponds to survey effort. Blainville's beaked whales have been reported from southwestern Nova Scotia to Florida, and are believed to be widely but sparsely distributed (Leatherwood et al. 1976; Mead 1989; MacLeod et al. 2006; Jefferson et al. 2008). There are two records of strandings in Nova Scotia which probably represent strays from the Gulf Stream (Mead 1989). They are considered rare in Canadian waters (Houston 1990).

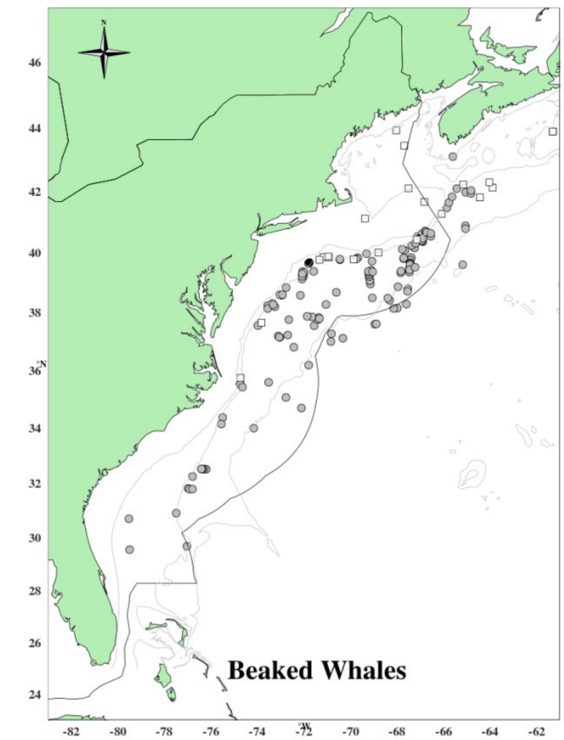

Figure 1. Distribution of beaked whale (includes Ziphius and Mesoplodon spp.) sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, and 2007, 2008, 2010, 2011 and 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings. Black symbols are sightings identified as Blainsville's beaked whales.

POPULATION SIZE

The best abundance estimate for *Mesoplodon* beaked whales is the sum of the 2016 survey estimates – 10,107 (CV=0.27). 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 two surveys were added together and the CVs pooled using a delta method to produce an abundance estimate for the stock area.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. Due to changes in survey methodology these historical data should not be used to make comparisons to more current estimates.

Recent surveys and abundance estimates

Abundance estimates of 6,760 (CV=0.37) and 3,347 (CV=0.29) *Mesoplodon* spp. beaked whales (not including *Ziphius*) 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 isobath 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 an abundance estimate for the stock area, yielding a combined total of 10,107 *Mesoplodon* beaked whales (CV=0.27). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE, and these numbers likely include an unknown number of *Mesoplodon* beaked whales.

An abundance estimate of 5,500 (CV=0.67) *Mesoplodon* spp. beaked whales (not including *Ziphius*) was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,017 km of tracklines that were in water offshore of North Carolina 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 double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers, 2004). Shipboard data were inspected to determine if there was significant responsive movement to the ship (Palka and Hammond 2001). Because there was an insignificant amount of responsive movement for this species, the estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 1,570 (CV=0.65) *Mesoplodon* spp. beaked whales (not including *Ziphius*) was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25× bigeye binoculars. A total of 4,445 km of tracklines were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Table 1. Summary of abundance estimates for the Mesoplodon beaked whales, , month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun-Aug 2011	Central Virginia to lower Bay of Fundy	5,500	0.67
Jun-Aug 2011	Central Florida to Central Virginia	1,592	0.67
Jun-Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	7,092	0.54
Jun-Sep 2016	Central Virginia to lower Bay of Fundy	6,760	0.37
Jun-Aug 2016	Central Florida to Virginia	3,347	0.29
Jun-Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	10,107	0.27

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 undifferentiated beaked whales is 10,107 (CV=0.27). The minimum population estimate for undifferentiated beaked whales in the western North Atlantic is 8,085.

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. *Mesoplodon* densirostris life history parameters that could be used to estimate net productivity include: length at birth of up to 1.9 m, maximum reported adult length of 4.7, and minimum reported age at sexual maturity of 9 growth layer groups (GLG's), which may be annual layers (Mead 1984).

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 for undifferentiated beaked whales is 10,107. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor, which accounts for endangered, depleted, or threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) is assumed to be 0.5. PBR for undifferentiated beaked whales in the western North Atlantic is 81.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2013–2017 total average estimated annual mortality of Blainville's beaked whales in fisheries in the U.S. Atlantic EEZ is 0.2 based on one stranded animal likely killed in 2017 by plastic injestion (Table 3).

Fishery Information

Total fishery-related mortality and serious injury cannot be estimated separately for each beaked whale species because of the uncertainty in species identification by fishery observers. The Atlantic Scientific Review Group advised adopting the risk-averse strategy of assuming that any beaked whale stock which occurred in the U.S. Atlantic EEZ might have been subject to the observed fishery-related mortality and serious injury.

Estimated annual average fishery-related mortality or serious injury of this stock in 2013–2017 in U.S. fisheries was 0.

Earlier Interactions

See Appendix V for more information on historical takes.

Other Mortality

From 2013–2017, a total of 4 Blainville's beaked whales stranded along the U.S. Atlantic coast between Florida and Massachusetts (NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). One animal in 2017 that stranded in Florida was classified as a human interaction due to plastic ingestion.

Several unusual mass strandings of beaked whales throughout their worldwide range have been associated with naval activities (D'Amico *et al.* 2009; Filadelfo *et al.* 2009). During the mid- to late 1980s multiple mass strandings of Cuvier's beaked whales (4 to about 20 per event) and small numbers of Gervais' beaked whale and Blainville's beaked whale occurred in the Canary Islands (Simmonds and Lopez-Jurado 1991). Twelve Cuvier's beaked whales that live stranded and subsequently died in the Mediterranean Sea on 12–13 May 1996 were associated with low-frequency sonar tests conducted by the North Atlantic Treaty Organization (Frantzis 1998; D'Amico *et al.* 2009; Filadelfo *et al.* 2009). In March 2000, 14 beaked whales live stranded in the Bahamas; 6 beaked whales (5 Cuvier's and 1 Blainville's) died (Balcomb and Claridge 2001; NMFS 2001; Cox *et al.* 2006). Four Cuvier's, 2 Blainville's, and 2 unidentified beaked whales were returned to sea. The fate of the animals returned to sea is unknown, since none of the whales have been resighted. Necropsy of 6 dead beaked whales revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand. Subsequently, the animals died due to extreme physiologic stress associated with the physical stranding (i.e., hyperthermia, high endogenous catecholamine release) (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 (Martin *et al.* 2004; Fernandez *et al.* 2005; Cox *et al.* 2006). 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).

Table 2. Blainville's beaked whale (Mesoplodon densirostris) strandings along the U.S. Atlantic coast.

State	2013	2014	2015	2016	2017	Total
New Jersey	0	0	0	1	0	**1**
Virginia	0	1	0	0		**1**
North Carolina	0	0	0	1	0	**1**
Florida	0	0	0	0	1	**1**
Total	**0**	**1**	**0**	**2**	**1**	**4**

a. Animal in Florida in 2017 is classified as a human interaction due to plastic chips found in forestomach.

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 beaked whales 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

Blainville's beaked whales are not listed as threatened or endangered under the Endangered Species Act and the western North Atlantic stock of Blainville's beaked whale is not considered strategic under the Marine Mammal Protection Act. There are insufficient data to determine the population size or trends, and, while a PBR value has been calculated for undifferentiated beaked whales, PBR cannot be calculated for this species independently. The permanent closure of the pelagic drift gillnet fishery has eliminated the principal known source of incidental fishery mortality, and a single 2017 stranding record was the only human-related mortality and serious injury observed during the recent 5-year (2013–2017) period. Therefore, total U.S. fishery-related mortality and serious injury rate can be considered to be insignificant and approaching zero. The status of Blainville's beaked whales relative to OSP in U.S. Atlantic EEZ is unknown.

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GERVAIS' BEAKED WHALE (*Mesoplodon europaeus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Within the genus *Mesoplodon*, there are four species of beaked whales that reside in the northwest Atlantic. These include True's beaked whale, *Mesoplodon mirus*; Gervais' beaked whale, *M. europaeus*; Blainville's beaked whale, *M. densirostris*; and Sowerby's beaked whale, *M. bidens* (Mead 1989). These species are difficult to identify to the species level at sea; therefore, much of the available characterization for beaked whales is to genus level only. Stock structure for each species is unknown. Thus, it is plausible the stock could actually contain multiple demographically independent populations that should themselves be stocks, because the current stock spans multiple eco-regions (Longhurst 1998; Spalding *et al*. 2007).

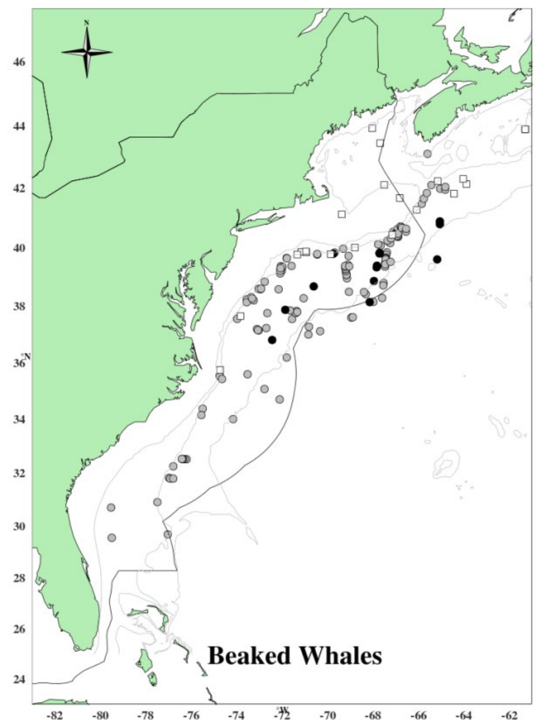

Figure 1. Distribution of beaked whale (includes Ziphius and Mesoplodon spp.) sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, and 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings. Black symbols are sightings identified as Gervais' beaked whales.

The distribution of *Mesoplodon* spp. in the northwest Atlantic is known principally from stranding records (Mead 1989; Nawojchik 1994; Mignucci-Giannoni *et al.* 1999; MacLeod *et al.* 2006; Jefferson *et al*. 2008). Off the U.S. Atlantic coast, beaked whale (*Mesoplodon* spp.) sightings have occurred principally along the shelf-edge and deeper oceanic waters (Figure 1; CETAP 1982; Waring *et al.* 1992; Tove 1995; Waring *et al.* 2001; Hamazaki 2002; Palka 2006). Most sightings were in late spring and summer, which corresponds to survey effort.

Gervais' beaked whales are believed to be principally oceanic, and strandings have been reported from Cape Cod to Florida, into the Caribbean and the Gulf of Mexico (NMFS unpublished data; Leatherwood *et al.* 1976; Mead 1989; Moore *et al.* 2005; MacLeod *et al.* 2006; Jefferson *et al*. 2008; McLellan *et al.* 2018). This is the most common species of *Mesoplodon* to strand along the U.S. Atlantic coast.

POPULATION SIZE

The best abundance estimate for *Mesoplodon* beaked whales is the sum of the 2016 survey estimates – 10,107 (CV=0.27). 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 two surveys were added together and the CVs pooled using a delta method to produce an abundance estimate for the stock area.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. Due to changes in survey methodology these historical data should not be used to make comparisons to more current estimates.

Recent surveys and abundance estimates

Abundance estimates of 6,760 (CV=0.37) and 3,347 (CV=0.29) undifferentiated beaked whales (*Ziphius* and

Mesoplodon spp.) 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 isobath 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 an abundance estimate for the stock area, yielding a combined total of 10,107 *Mesoplodon* beaked whales (CV=0.27). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE, and these numbers likely include an unknown number of *Mesoplodon* beaked whales.

An abundance estimate of 5,500 (CV=0.67) *Mesoplodon* spp. beaked whales (not including *Ziphius*) was generated from a shipboard and aerial survey conducted during June–August 2011(Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,017 km of tracklines that were in water offshore of North Carolina 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 double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers, 2004). Shipboard data were inspected to determine if there was significant responsive movement to the ship (Palka and Hammond 2001). Because there was an insignificant amount of responsive movement for this species, the estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 1,570 (CV=0.65) *Mesoplodon* spp. beaked whales (not including *Ziphius*) was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25× bigeye binoculars. A total of 4,445 km of tracklines were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Table 1. Summary of abundance estimates for Mesoplodon beaked , month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	Central Virginia to lower Bay of Fundy	5,500	0.67
Jun–Aug 2011	Central Florida to Central Virginia	1,592	0.67
Jun–Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	7,092	0.54
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	6,760	0.37
Jun–Aug 2016	Central Florida to Virginia	3,347	0.29
Jun–Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	10,107	0.27

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 undifferentiated beaked whales is 10,107 (CV=0.27). The minimum population estimate for undifferentiated beaked whales in the western North Atlantic is 8,085.

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. *Mesoplodon europaeus* life history parameters that could be used to estimate net productivity include: estimated mean length at birth of 2.1 m, length at sexual maturity of up to 5.2 m for females and up to 4.6 m for males, and maximum age of 27 growth layer groups (GLG's), which may be annual layers (Mead 1984).

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 for undifferentiated beaked whales is 8.085. 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. PBR for undifferentiated beaked whales in the western North Atlantic is 81.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2013–2017 total average estimated annual mortality of Gervais' beaked whales in observed fisheries in the U.S. Atlantic EEZ is zero.

Fishery Information

Total fishery-related mortality and serious injury cannot be estimated separately for each beaked whale species because of the uncertainty in species identification by fishery observers. The Atlantic Scientific Review Group advised adopting the risk-averse strategy of assuming that any beaked whale stock which occurred in the U.S. Atlantic EEZ might have been subject to the observed fishery-related mortality and serious injury.

Estimated annual average fishery-related mortality or serious injury of this stock in 2013–2017 in U.S. fisheries was zero. Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

Other Mortality

During 2013–2017, 12 Gervais' beaked whales stranded along the U.S. Atlantic coast (Table 2; NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). Three of these animals displayed signs of human interaction due to trash ingestion.

Several unusual mass strandings of beaked whales in North Atlantic marine environments have been associated with naval activities (D'Amico *et al*. 2009; Filadelfo *et al*. 2009. During the mid- to late 1980's multiple mass strandings of Cuvier's beaked whales (4 to about 20 per event) and small numbers of Gervais' beaked whale and Blainville's beaked whale occurred in the Canary Islands (Simmonds and Lopez-Jurado 1991). Twelve Cuvier's beaked whales that live stranded and subsequently died in the Mediterranean Sea on 12-13 May 1996 was associated with low frequency acoustic sonar tests conducted by the North Atlantic Treaty Organization (Frantzis 1998; A'Amico *et al*. 2009; Filadelfo *et al*. 2009). In March 2000, 14 beaked whales live stranded in the Bahamas; 6 beaked whales (5 Cuvier's and 1 Blainville's) died (Balcomb and Claridge 2001; NMFS 2001; Cox *et al.* 2006). Four Cuvier's, 2

Blainville's, and 2 unidentified beaked whales were returned to sea. The fate of the animals returned to sea is unknown, since none of the whales have been resighted. Necropsy of 6 dead beaked whales revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand. Subsequently, the animals died due to extreme physiologic stress associated with the physical stranding (i.e., hyperthermia, high endogenous catecholamine release) (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).

Table 3. Gervais' beaked whale (Mesoplodon europaeus) strandings along the U.S. Atlantic coast.

State	2013	2014	2015	2016	2017	Total
North Carolina[a]	1	0	2	0	2	**5**
South Carolina	0	0	0	1	0	**1**
Florida[b]	2	1	0	3	0	**6**
Total	**3**	**1**	**2**	**4**	**2**	**12**

a. North Carolina stranding in 2013 deemed human interaction due to plastic ingestion.
b. Florida strandings in 2013 and 2016 deemed HI due to human trash ingestion (yellow cap, piece of corn cob)

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 beaked whales 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

Gervais' beaked whales are not listed as threatened or endangered under the Endangered Species Act and the western North Atlantic stock of Gervais' beaked whale is not considered strategic under the Marine Mammal Protection Act. There are insufficient data to determine the population size or trends, and, while a PBR value has been calculated for the undifferentiated beaked whales, PBR cannot be calculated for this species independently. The permanent closure of the pelagic drift gillnet fishery has eliminated the principal known source of incidental fishery mortality, and no fishery-related mortality and serious injury has been observed during the recent 5-year (2013–2017) period. Therefore, the total U.S. fishery mortality and serious injury rate can be considered to be insignificant and approaching zero. The status of Gervais' beaked whales relative to OSP in U.S. Atlantic EEZ is unknown.

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SOWERBY'S BEAKED WHALE (*Mesoplodon bidens*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Within the genus *Mesoplodon*, there are four species of beaked whales that reside in the northwest Atlantic. These include True's beaked whale, *M. mirus*; Gervais' beaked whale, *M. europaeus*; Blainville's beaked whale, *M. densirostris*; and Sowerby's beaked whale, *M. bidens* (Mead 1989). These species are difficult to identify to the species level at sea; therefore, much of the available characterization for beaked whales is to genus level only. Stock structure for each species is unknown. Thus, it is plausible the stock could actually contain multiple demographically independent populations that should themselves be stocks, because the current stock spans multiple eco-regions (Longhurst 1998; Spalding *et al*. 2007).

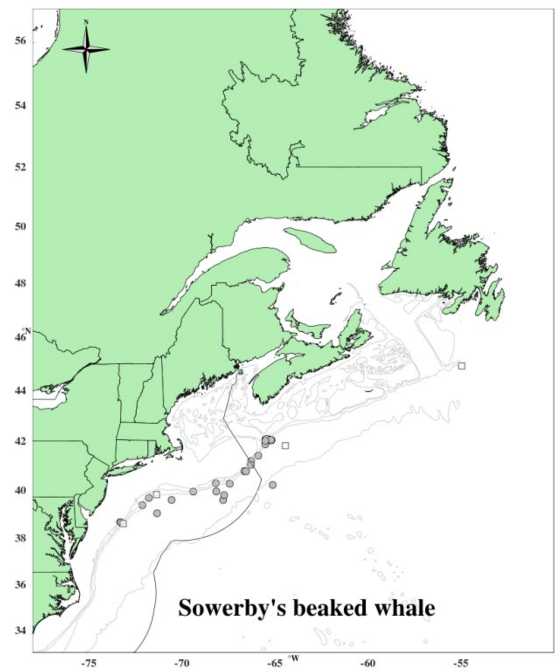

Figure 1: Distribution of beaked whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS survey. Isobaths are the 1000-m and 4000-m depth contours. Circle symbols represent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

The distributions of *Mesoplodon* spp. in the northwest Atlantic are known principally from stranding records (Mead 1989; Nawojchik 1994; Mignucci-Giannoni *et al.* 1999; MacLeod *et al.* 2006). Off the U.S. Atlantic coast, beaked whale (*Mesoplodon* spp.) sightings have occurred principally along the shelf-edge and deeper oceanic waters (Figure 1; CETAP 1982; Waring *et al.* 1992; Tove 1995; Waring *et al.* 2001; Hamazaki 2002; Palka 2006). Most sightings were in late spring and summer, which corresponds to survey effort. The distributions of Sowerby's beaked whales are also known from acoustical surveys (Cholewiak *et al.* 2013, Stanistreet *et al.* 2018) and bycatch confirmed genetically to be *M. bidens* (Wenzel *et al*. 2013).

Sowerby's beaked whales have been reported from New England waters north to the ice pack (e.g., Davis Strait), and individuals are seen along the Newfoundland coast in summer (Leatherwood *et al.* 1976; Mead 1989; MacLeod *et al.* 2006; Jefferson *et al*. 2008). Furthermore, a single stranding occurred off the Florida west coast (Mead 1989). This species is considered rare in Canadian waters (Lien *et al.* 1990) and has been designated as "Special Concern" by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Whitehead (2013) reports that in the 23 years of cetacean observations in the Gully Marine Protected Area, on the edge of the Scotian Shelf, Nova Scotia, Canada, they have observed a significant increase in sightings of Sowerby's.

POPULATION SIZE

The best abundance estimate for undifferentiated beaked whales is the sum of the 2016 survey estimates–10,107 (CV=0.27). 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 two surveys were added together and the CVs pooled using a delta method to produce an abundance estimate for the stock area.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. Due to changes in survey methodology these historical data should not be used to make comparisons to more current estimates.

Recent surveys and abundance estimates

Abundance estimates of 6,760 (CV=0.37) and 3,347 (CV=0.29) *Mesoplodon* spp. beaked whales (not including *Ziphius*) 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 isobath 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 an abundance estimate for the stock area, yielding a combined total of 10,107 *Mesoplodon* beaked whales (CV=0.27). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE, and these numbers likely include an unknown number of *Mesoplodon* beaked whales.

An abundance estimate of 5,500 (CV=0.67) *Mesoplodon* spp. beaked whales (not including *Ziphius*) was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey and shallower than the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines that were in waters offshore of 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-simultaneous team data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers, 2004). Shipboard data were inspected to determine if there was significant responsive movement to the ship (Palka and Hammond 2001). Because there was an insignificant amount of responsive movement for this species, the estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 1,570 (CV=0.65) *Mesoplodon* spp. beaked whales (not including *Ziphius*) was also generated from a shipboard survey conducted during June–August 2011 between central Florida and Virginia. The survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25x bigeye binoculars. A total of 4,445 km of survey effort were accomplished with 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Table 1. Summary of abundance estimates for Mesoplodon spp. , month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	Central Virginia to lower Bay of Fundy	5,500	0.67
Jun–Aug 2011	Central Florida to Central Virginia	1,592	0.67
Jun–Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	7,092	0.54
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	6,760	0.37
Jun–Aug 2016	Central Florida to Central Virginia	3,347	0.29
Jun–Sep 2016	Central Florida to lower Bay of Fundy (COMBINED)	10,107	0.27

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 undifferentiated beaked whales is 10,107 (CV=0.27). The minimum population estimate for undifferentiated beaked whales in the western North Atlantic is 8,085.

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. *Mesoplodon bidens* life history parameters that could be used to estimate net productivity include: length at birth of up to 2.4 m and maximum length of 5 m for females and 5.5 m for males (Mead 1984).

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 for undifferentiated beaked whales is 8,085. 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. PBR for undifferentiated beaked whales in the western North Atlantic is 81.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2013–2017 total average estimated annual mortality of Sowerby's beaked whales in observed fisheries in the U.S. Atlantic EEZ is zero.

Fishery Information

Estimated annual average fishery-related mortality or serious injury of this stock in 2013–2017 in U.S. fisheries was zero. Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

Other Mortality

During 2013–2017 3 Sowerby's beaked whales stranded along the U.S. Atlantic coast (Table 2; NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). One of these animals was recorded as a human interaction due to plastic injestion.

Several unusual mass strandings of beaked whales throughout their worldwide range have been associated with naval activities (D'Amico *et al*. 2009; Filadelfo *et al*. 2009). During the mid- to late 1980s multiple mass strandings of Cuvier's beaked whales (4 to about 20 per event) and small numbers of Gervais' beaked whale and Blainville's beaked whale occurred in the Canary Islands (Simmonds and Lopez-Jurado 1991). Twelve Cuvier's beaked whales that live stranded and subsequently died in the Mediterranean Sea on 12-13 May 1996 were associated with low frequency acoustic sonar tests conducted by the North Atlantic Treaty Organization (Frantzis 1998; D'Amico *et al*. 2009; Filadelfo *et al*. 2009). In March 2000, 14 beaked whales live stranded in the Bahamas; 6 beaked whales (5 Cuvier's and 1 Blainville's) died (Balcomb and Claridge 2001; NMFS 2001; Cox *et al.* 2006). Four Cuvier's, 2 Blainville's, and 2 unidentified beaked whales were returned to sea. The fate of the animals returned to sea is unknown, since none of the whales have been resighted. Necropsy of 6 dead beaked whales revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand. Subsequently, the animals died due to extreme physiologic stress associated with the physical stranding (i.e., hyperthermia, high endogenous catecholamine release) (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).

Table 3. Sowerby's beaked whale (Mesoplodon bidens) strandings along the U.S. Atlantic coast.

State	2013	2014	2015	2016	2017	Total
Maine	0	1	0	0	0	1
Massachusetts[a]	0	0	2	0	0	2
Total	0	1	2	0	0	3

a. One animal in Massachusetts classified as human interaction due to plastic ingestion.

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 beaked whales 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

While Sowerby's beaked whales are not listed as threatened or endangered under the Endangered Species Act they have been listed as a species of Special Concern by both COSEWIC and SARA (the Species at Risk Act) in Canada (COSEWIC 2006). The western North Atlantic stock of Sowerby's beaked whale is not considered strategic under the Marine Mammal Protection Act. No habitat issues are known to be of concern for this species, but questions have been raised regarding potential effects of human-made sounds on deep-diving cetacean species such as Sowerby's beaked whales (Richardson et al. 1995). There are insufficient data to determine the population size or trends, and, while a PBR value has been calculated for undifferentiated beaked whales, PBR cannot be calculated for this species independently. The permanent closure of the pelagic drift gillnet fishery has eliminated the principal known source of incidental fishery mortality, and no fishery-related mortality and serious injury has been observed during the recent 5-year (2013–2017) period. Therefore, the total U.S. fishery mortality and serious injury rate can be considered to be insignificant and approaching zero. The status of Sowerby's beaked whales relative to OSP in U.S. Atlantic EEZ is unknown.

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TRUE'S BEAKED WHALE (*Mesoplodon mirus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Within the genus *Mesoplodon*, there are four species of beaked whales that reside in the Northwest Atlantic. These include True's beaked whale, *M. mirus*; Gervais' beaked whale, *M. europaeus*; Blainville's beaked whale, *M. densirostris*; and Sowerby's beaked whale, *M. bidens* (Mead 1989). These species are difficult to identify to the species level at sea; therefore, much of the available characterization for beaked whales is to genus level only. Stock structure for each species is unknown. Thus, it is plausible that the stock could actually contain multiple demographically independent populations that should themselves be stocks, because the current stock spans multiple eco-regions (Longhurst 2007; Spalding *et al.* 2007).

The distributions of *Mesoplodon* spp. in the Northwest Atlantic are known principally from stranding records (Mead 1989; Nawojchik 1994; Mignucci-Giannoni *et al.* 1999; MacLeod *et al.* 2006; Jefferson *et al.* 2008). Off the U.S. Atlantic coast, beaked whale (*Mesoplodon* spp.) sightings have occurred principally along the shelf-edge and in deeper oceanic waters (Figure 1; CETAP 1982; Waring *et al.* 1992, 2001; Tove 1995; Hamazaki 2002; Palka 2006; NEFSC and SEFSC 2018). Most sightings were in late spring and summer, which corresponds to survey effort.

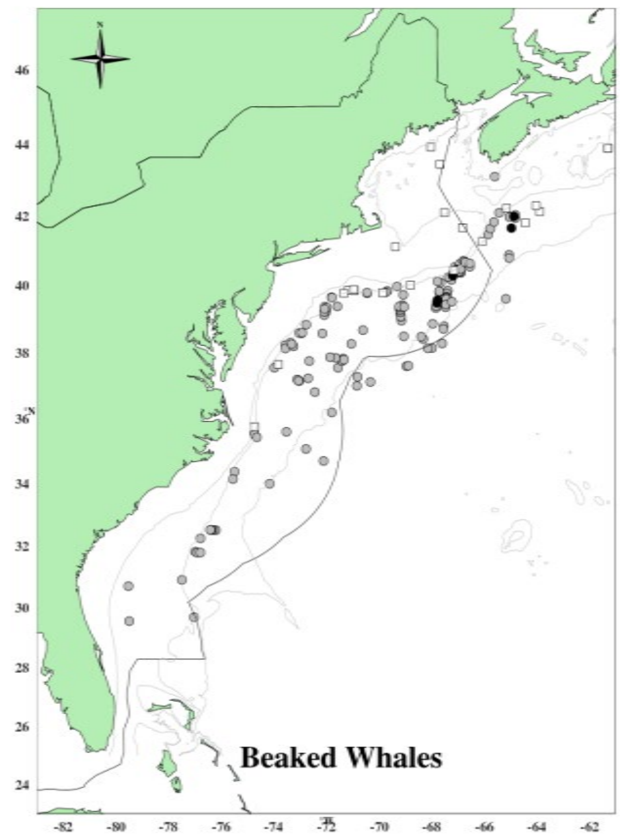

Figure 1: Distribution of beaked whale (includes Ziphius and Mesoplodon spp.) sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings. Black symbols are the sightings identified as True's beaked whales.

True's beaked whale is a temperate-water species that has been reported from Cape Breton Island, Nova Scotia, to the Bahamas (Leatherwood *et al.* 1976; Mead 1989; MacLeod *et al.* 2006; Jefferson *et al.* 2008).

POPULATION SIZE

The best abundance estimate for undifferentiated beaked whales is the sum of the 2016 survey estimates—10,107 (CV=0.27). 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 two 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 some of the 2016 survey estimates were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected.

Earlier abundance estimates

Please see Appendix IV for a summary of abundance estimates, including earlier estimates and survey descriptions. Due to changes in survey methodology these historical data should not be used to make comparisons to more current estimates.

Recent surveys and abundance estimates

Abundance estimates of 6,760 (CV=0.37) and 3,347 (CV=0.29) *Mesoplodon* spp. beaked whales (not including *Ziphius*.) 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 isobath 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 an abundance estimate for the stock area, yielding a combined total of 10,107 *Mesoplodon* beaked whales (CV=0.27). These estimates are known to be biased low due to the fact that unidentified Ziphiidae abundance was estimated at 3,755 (CV=0.42) in the NE and at 2,812 (CV=0.43) in the SE, and these numbers likely include an unknown number of *Mesoplodon* beaked whales.

An abundance estimate of 5,500 (CV=0.67) *Mesoplodon* spp. beaked whales was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 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 double-platform collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers, 2004). Shipboard data were inspected to determine if there was significant responsive movement to the ship (Palka and Hammond 2001). Because there was an insignificant amount of responsive movement for this species, the estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 1,570 (CV=0.65) *Mesoplodon* spp. beaked whales was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25× bigeye binoculars. A total of 4,445 km of tracklines were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Table 1. Summary of abundance estimates for Mesoplodon spp.[a], month, year, area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun-Aug 2011	Central Virginia to lower Bay of Fundy	5,500	0.67
Jun-Aug 2011	Central Florida to Central Virginia	1,592	0.67
Jun-Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	7,092	0.54
Jun-Sep 2016	Central Virginia to lower Bay of Fundy	6,760	0.37
Jun-Aug 2016	Central Florida to Virginia	3,347	0.29
Jun-Aug 2016	Central Florida to lower Bay of Fundy (COMBINED)	10,107	0.27

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 undifferentiated beaked whales is 21,818 (CV= 0.15). The minimum population estimate for undifferentiated. beaked whales in the western North Atlantic is 19,243.

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. Life history parameters that could be used to estimate net productivity include: length at birth is 2 to 3 m, length at sexual maturity 6.1 m for females, and 5.5 m for males, maximum age for females were 30 growth layer groups (GLG's) and for males was 36 GLG's, which may be annual layers (Mead 1984).

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 for undifferentiated beaked whales is 419,243. 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. PBR for undifferentiated beaked whales in the western North Atlantic is 192.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The 2013–2019 total average estimated annual mortality of True’s beaked whales in observed fisheries in the U.S. Atlantic EEZ is zero. One True’s beaked whale found stranded in New York in 2015 was classified as a fishery interaction (Table 2), resulting in a total annual average fishery-related mortality or serious injury during 2013–2017 of 0.2 animals.

Fishery Information

Total fishery-related mortality and serious injury cannot be estimated separately for each beaked whale species because of the uncertainty in species identification by fishery observers. The Atlantic Scientific Review Group advised adopting the risk-averse strategy of assuming that any beaked whale stock which occurred in the U.S. Atlantic EEZ might have been subject to the observed fishery-related mortality and serious injury.

Estimated annual average fishery-related mortality or serious injury of this stock in 2013-2017 in observed U.S. fisheries was zero. Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

Other Mortality

During 2013–2017, 6 True’s beaked whales stranded along the U.S. Atlantic coast (Table 2; NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). One of these animals showed evidence of a fishery interaction.

Several unusual mass strandings of beaked whales throughout their worldwide range have been associated with naval activities (D'Amico *et al.* 2009; Filadelfo *et al.* 2009). During the mid- to late 1980's multiple mass strandings of Cuvier's beaked whales (4 to about 20 per event) and small numbers of Gervais' beaked whale and Blainville's beaked whale occurred in the Canary Islands (Simmonds and Lopez-Jurado 1991). Twelve Cuvier's beaked whales that live stranded and subsequently died in the Mediterranean Sea on 12–13 May 1996 were associated with low-frequency sonar tests conducted by the North Atlantic Treaty Organization (Frantzis 1998; D'Amico *et al.* 2009; Filadelfo *et al.* 2009). In March 2000, 14 beaked whales live stranded in the Bahamas; 6 beaked whales (5 Cuvier's and 1 Blainville's) died (Balcomb and Claridge 2001; NMFS 2001; Cox *et al.* 2006). Four Cuvier's, 2 Blainville's, and 2 unidentified beaked whales were returned to sea. The fate of the animals returned to sea is unknown, since none of the whales have been resighted. Necropsy of 6 dead beaked whales revealed evidence of tissue trauma associated with an acoustic or impulse injury that caused the animals to strand. Subsequently, the animals died due to extreme physiologic stress associated with the physical stranding (i.e., hyperthermia, high endogenous catecholamine release) (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 (Martin *et al.* 2004; Fernandez *et al.* 2005; Cox *et al.* 2006). 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).

Table 2. True's beaked whale (Mesoplodon mirus) strandings along the U.S. Atlantic coast.

State	2013	2014	2015	2016	2017	Total
New York	0	2	2	0	0	**4**
Virginia	0	0	0	0	1	**1**
Georgia	0	0	0	1	0	**1**
Total	**0**	**2**	**2**	**1**	**1**	**6**

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 sperm whales 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; 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

True's beaked whales 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. There are insufficient data to determine the population size or trends, and, while a PBR value has been calculated for undifferentiated beaked whales, PBR cannot be calculated for this species independently. The permanent closure of the pelagic drift gillnet fishery has eliminated the principal known source of incidental fishery mortality, and only one fishery-related mortality and serious injury has been reported during the recent 5-year (2013–2017) period. Therefore, total U.S. fishery-related mortality and serious injury rate can be considered to be insignificant and approaching zero. The status of True's beaked whales relative to OSP in U.S. Atlantic EEZ is unknown.

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MELON-HEADED WHALE (*Peponocephala electra*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The melon-headed whale is distributed worldwide in tropical and sub-tropical waters (Jefferson *et al.* 1994). However, sightings of this species in the western North Atlantic are extremely rare. Most stranding records are from Florida and South Carolina, with a few from Virginia and one from New Jersey. There have been two sightings during NMFS vessel surveys between 1992 and 2016. Melon-headed whales in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. 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). Due to the paucity of sightings in the western North Atlantic, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

POPULATION SIZE

The number of melon-headed whales off the U.S. Atlantic coast is unknown because they were rarely seen in any surveys. A single group of melon-headed whales was sighted off of Cape Hatteras, North Carolina, in waters >2500 m deep during both a summer 1999 (20 whales) and a winter 2002 (80 whales) vessel survey of the western North Atlantic (Figure 1; NMFS 1999; NMFS 2002). Abundances have not been estimated from these single sightings. Therefore the population size of melon-headed whales is unknown. No confirmed sightings of melon-headed whales have been observed in any other NMFS surveys. Several cruises, a winter 2002 cruise, a summer 2005 cruise, and a summer 2016 cruise, each had one or two sightings of pygmy killer or melon-headed whales (identity was not confirmed), and these groups were recorded off Cape Hatteras or off the North Carolina/South Carolina border.

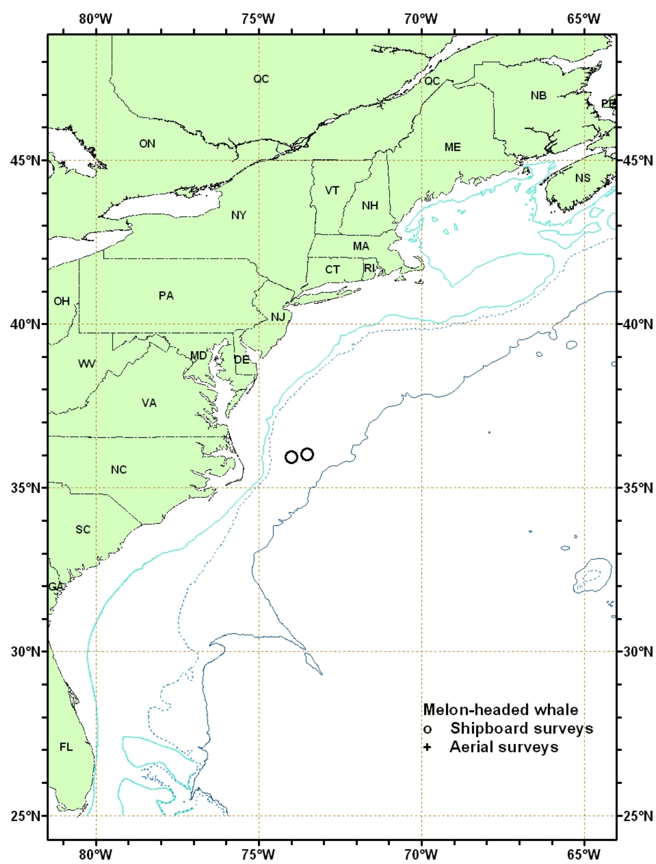

Figure 1. Distribution of melon-headed whale sightings from NEFSC and SEFSC shipboard and aerial surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 100m, 1,000m and 4,000m depth contours.

Minimum Population Estimate

Present data are insufficient to calculate a minimum population estimate for this stock.

Current Population Trend

There are insufficient data to determine the population trends for this stock because no estimates of population size are available.

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 history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal level (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 is unknown. 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 this stock is of unknown status. PBR for the western North Atlantic stock of melon-headed whales is unknown because the minimum population size is unknown.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to melon-headed whales in the western North Atlantic.

Fishery Information

The commercial fishery that could potentially interact with this stock in the Atlantic Ocean is the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery (Appendix III). Pelagic swordfish, tunas and billfish are the targets of the longline fishery. Percent observer coverage (percentage of sets observed) for this fishery for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively. There were no observed mortalities or serious injuries to melon-headed whales by this fishery in the Atlantic Ocean during 2013–2017 (Garrison and Stokes 2014; 2016; 2017; 2019; 2020).

Other Mortality

There were three reported strandings of melon-headed whales in the U.S. Atlantic Ocean during 2013–2017 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER)). All three occurred off Florida during 2015. For two of the three strandings, no evidence of human interaction was detected, but for one stranding, evidence of human interaction was detected in the form of an ingested plastic bag. Stranding data probably 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

HABITAT ISSUES

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.

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for this stock 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; 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

Melon-headed whales 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. No fishery-related mortality or serious injury has been observed during 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 western U.S. Atlantic EEZ relative to OSP is unknown. There are insufficient data to determine the population trends for this species.

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RISSO'S DOLPHIN (*Grampus griseus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Risso's dolphins are distributed worldwide in tropical and temperate seas (Jefferson *et al*. 2008, 2014), and in the Northwest Atlantic occur from Florida to eastern Newfoundland (Leatherwood *et al*. 1976; Baird and Stacey 1991). Off the northeastern U.S. coast, Risso's dolphins are distributed along the continental shelf edge from Cape Hatteras northward to Georges Bank during spring, summer, and autumn (CETAP 1982; Payne *et al*. 1984) (Figure 1). In winter, the range is in the mid-Atlantic Bight and extends outward into oceanic waters (Payne *et al*. 1984). In general, the population occupies the mid-Atlantic continental shelf edge year round, and is rarely seen in the Gulf of Maine (Payne *et al*. 1984). During 1990, 1991 and 1993, spring/summer surveys conducted along the continental shelf edge and in deeper oceanic waters sighted Risso's dolphins associated with strong bathymetric features, Gulf Stream warm-core rings, and the Gulf Stream north wall (Waring *et al*. 1992, 1993; Hamazaki 2002). Sightings during 2016 surveys were concentrated along the shelf break (NEFSC and SEFSC 2018; Figure 1).

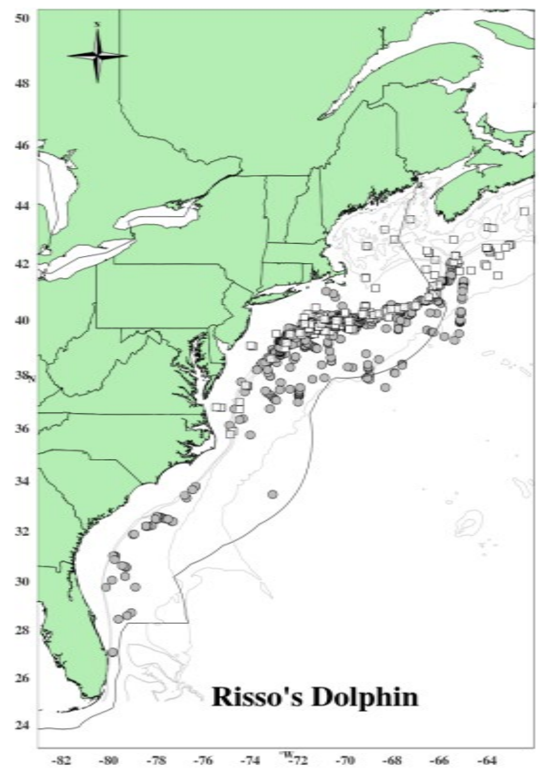

Figure 1. Distribution of Risso's dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008 2010, 2011 and 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100m, 1,000m, and 4,000m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

There is no information on stock structure of Risso's dolphin in the western North Atlantic, or to determine if separate stocks exist in the Gulf of Mexico and Atlantic. Thus, it is plausible that the stock could actually contain multiple demographically independent populations that should themselves be stocks, because the current stock spans multiple eco-regions (Longhurst 1998; Spalding *et al*. 2007). In 2006, a rehabilitated adult male Risso's dolphin stranded and released in the Gulf of Mexico off Florida was tracked via satellite-linked tag to waters off Delaware (Wells *et al*. 2009). The Gulf of Mexico and Atlantic stocks are currently being treated as two separate stocks.

POPULATION SIZE

The best abundance estimate for Risso's dolphins is the sum of the estimates from the 2016 NEFSC and Department of Fisheries and Oceans Canada (DFO) surveys—35,493 (CV=0.19: Table 1). Because the survey areas did not overlap, the estimates from the two surveys were added together and the CV's 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 (Table 1).

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 and abundance estimates

An abundance estimate of 15,197 (CV=0.55) Risso's dolphins was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour, through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 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 double-platform data-collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers, 2004). Shipboard data were inspected to determine if there was significant responsive movement to the ship (Palka and Hammond 2001). Because there was evidence of responsive (evasive) movement of this species to the ship, estimation of the abundance was based on Palka and Hammond (2001) and the independent-observer approach assuming full independence (Laake and Borchers 2004), and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 3,053 (CV=0.44) Risso's dolphins was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida (Garrison 2016). This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed the double-platform methodology searching with 25×150 "bigeye" binoculars. A total of 4,445 km of tracklines was surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

The Department of Fisheries and Oceans, Canada (DFO) generated Risso's dolphin 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 (Lawson and Gosselin 2018). A total of 29,123 km of effort were flown over the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf strata and 21,037 over the Newfound/Labrador strata. The Bay of Fundy/Scotian shelf portion of the Risso's dolphin population was estimated as 6,073 (CV=0.445).

Abundance estimates of 75,079 (CV=0.38) and 7,245 (CV=0.44) Risso's 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 the western North Atlantic Risso's dolphin (Grampus griseus), by month, year, and area covered during each abundance survey, resulting abundance estimate (Nbest) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	Central Virginia to lower Bay of Fundy	15,197	0.55
Jun–Aug 2011	Central Florida to Central Virginia	3,053	0.44
Jun–Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	18,250	0.46
Jun–Sep 2016	Central Florida to Central Virginia	7,245	0.44
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	22,175	0.23
Aug–Sep 2016	Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf	6,073	0.445
Jun–Sep 2016	Central Florida to Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf - COMBINED	35,493	0.19

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 Risso's dolphins is 35,493 (CV=0.19), obtained from the 2016 surveys. The minimum population estimate for the western North Atlantic Risso's dolphin is 30,289.

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 strata.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. Due to uncertainties about the stock-specific life history parameters, 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).

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 30,289. The maximum productivity rate is 0.04, the default value for cetaceans (Barlow *et al.* 1995). The recovery factor is 0.5, the default value for stocks of unknown status relative to OSP, and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic stock of Risso's dolphin is 303.

ANNUAL HUMAN-CAUSED MORTALITY

Total annual estimated average human-caused mortality or serious injury to this stock during 2013–2017 was 54.3 Risso's dolphins, derived from 2 components: 1) 53.9 estimated mortalities in observed fisheries (CV=0.24; Table 2) and 2) 0.4 from average 2013–2017 non-fishery related, human interaction stranding mortalities (NMFS unpublished data). Key uncertainties include the potential that the observer coverage was not representative of the fishery during all times and places.

Fishery Information

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

Pelagic Longline

Pelagic longline bycatch estimates of Risso's dolphins for 2013–2017 are documented in Garrison and Stokes (2014, 2016, 2017, 2019, 2020). Most of the estimated marine mammal bycatch was from U.S. Atlantic EEZ waters between South Carolina and Cape Cod. 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 2 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

One Risso's dolphin was observed taken in northeast bottom trawl fisheries in 2014 and 2 in 2016 (Table 2). Annual Risso's dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos *et al.* 2020). See Table 2 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

Risso's dolphins have been observed taken in mid-Atlantic bottom trawl fisheries (Table 2). Annual Risso's dolphin mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos *et al.* 2020). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Northeast Sink Gillnet

In the northeast sink gillnet fishery, Risso's dolphin interactions have historically been rare, but in 2013 one animal was observed in the waters south of Massachusetts (Hatch and Orphanides 2015, 2016; Orphanides 2019, 2020). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Table 2. Summary of the incidental serious injury and mortality of Risso's dolphin (Grampus griseus) by commercial fishery including the years sampled, the type of data used, the annual observer coverage, the observed mortalities and serious injuries recorded by on-board observers, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury, the estimated CV of the combined estimates and the mean of the combined estimates (CV in parentheses).

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury[c]	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
Pelagic Longline	2013	Obs. Data, Logbook	0.09	1	0	1.9	0	1.9	1	6.9 (0.39)
	2014		0.10	1	0	7.7	0	7.7	1	
	2015		0.12	2	0	8.4	0	8.4	0.71	
	2016		0.15	1	1	10.5	5.6	16.1	0.57	
	2017		0.12	1	0	0.2	0	0.2	1	
Northeast Sink Gillnet	2013	Obs. Data, Trip Logbook, Allocated Dealer Data	0.11	0	1	0	23	23	1	5.8 (0.79)
	2014		0.18	0	0	0	0	0	0	
	2015		0.14	0	0	0	0	0	0	
	2016		0.10	0	0	0	0	0	0	
	2017		0.12	0	0	0	0	0	0	
Northeast Bottom Trawl	2013	Obs. Data, Weighout	0.15	0	0	0	0	0	0	4.2 (0.73)
	2014		0.17	0	1	0	4.2	4.2	0.91	
	2015		0.19	0	0	0	0	0	0	
	2016		0.12	0	2	0	17	17	0.88	
	2017		0.16	0	0	0	0	0	0	

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury[c]	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
Mid-Atlantic Bottom Trawl	2013 2014 2015 2016 2017	Obs. Data, Dealer Data	0.06 0.08 0.09 0.10 0.10	0 0 2 0 2	4 2 1 4 5	0 0 27 0 12	42 21 13 39 31	42 21 40 39 31	0.71 0.93 0.63 0.56 0.51	37 (.29)
TOTAL	-	-	-	-	-	-	-	-	-	53.9 (0.24)

[a] Observer data (Obs. Data) are used to measure bycatch rates and the data are collected within the Northeast Fisheries Observer Program. NEFSC collects landings data (unallocated Dealer Data and 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. Total landings are used as a measure of total effort for the coastal gillnet fishery.
[b] The observer coverages for the northeast and mid-Atlantic sink gillnet fishery are ratios based on tons of fish landed. Northeast bottom trawl, mid-Atlantic bottom trawl, northeast mid-water and mid-Atlantic mid-water trawl fishery coverages are ratios based on trips. Total observer coverage reported for gillnet and bottom trawl gear include samples collected from traditional fisheries observers in addition to fishery at-sea monitors through the Northeast Fisheries Observer Program (NEFOP).
[c] Serious injuries were evaluated for the 2013–2017 period and include both at-sea monitor and traditional observer data (Josephson *et al*. 2019).

Other Mortality

From 2013 to 2017, 38 Risso's dolphin strandings were recorded along the U.S. Atlantic coast (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018). Three animals had indications of human interaction, none of which were classified as fishery interactions. Indications of human interaction are not necessarily the cause of death (Table 3).

Table 3. Risso's dolphin (Grampus griseus) reported strandings along the U.S. Atlantic coast and Puerto Rico, 2013–2017.

STATE	2013	2014	2015	2016	2017	TOTALS
Massachusetts[b]	3	2	1	2	14	22
Rhode Island	0	0	0	0	1	1
New York	2	0	2	0	0	4
New Jersey	0	0	0	0	1	1
Maryland	1	0	0	0	0	1
Virginia[c]	0	1	0	0	0	1
North Carolina	1	1	0	0	1	3
Florida[a]	2	0	0	2	1	5
TOTAL	**9**	**4**	**4**	**4**	**4**	**38**

a. One animal in 2013 classified as human interaction due to linear wound on face.
b. One animal in 2014 was classified as CBD for human interaction due to signs of ear trauma.
c. One animal in 2014 classified as HI due to plastic ingestion.

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.

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., Storelli and Macrotrigiano 2000; 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 Risso's dolphins 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

Risso's 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 2013–2017 average annual human-related mortality does not exceed PBR. The total U.S. fishery 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 Risso's dolphins relative to OSP in the U.S. Atlantic EEZ is unknown. Population trends for this species have not been investigated. Based on the low levels of uncertainties described in the above sections, it expected these uncertainties will have little effect on the designation of the status of this stock.

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LONG-FINNED PILOT WHALE (*Globicephala melas melas*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

There are two species of pilot whales in the western Atlantic—the long-finned pilot whale, *Globicephala melas melas*, and the short-finned pilot whale, *G. macrorhynchus*. These species are difficult to differentiate at sea and cannot be reliably visually identified during either abundance surveys or observations of fishery mortality without high-quality photographs (Rone and Pace 2012); therefore, the ability to separately assess the two species in U.S. Atlantic waters is complex and requires additional information on seasonal spatial distribution. The long-finned pilot whale is distributed from North Carolina to North Africa (and the Mediterranean) and north to Iceland, Greenland and the Barents Sea (Sergeant 1962; Leatherwood *et al.* 1976; Abend 1993; Bloch *et al.* 1993; Abend and Smith 1999). The stock structure of the North Atlantic population is uncertain (ICES 1993; Fullard *et al.* 2000). Morphometric (Bloch and Lastein 1993) and genetic (Siemann 1994; Fullard *et al.* 2000) studies have provided little support for stock separation across the Atlantic (Fullard *et al.* 2000). However, Fullard *et al.* (2000) have proposed a stock structure that is related to sea-surface temperature: 1) a cold-water population west of the Labrador/North Atlantic current, and 2) a warm-water population that extends across the Atlantic in the Gulf Stream.

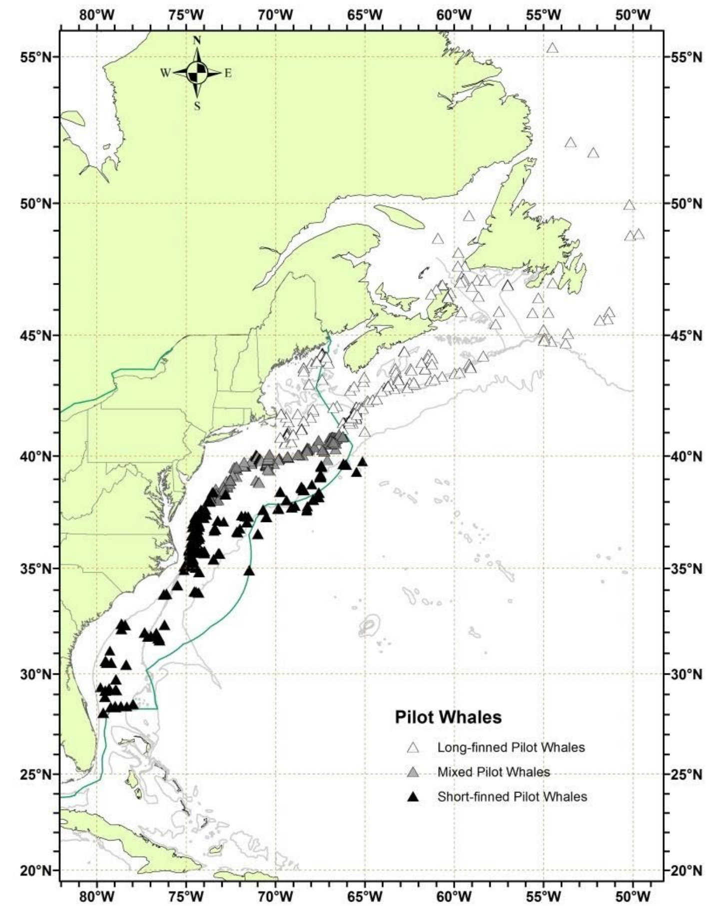

Figure 1. Distribution of long-finned (open symbols), short-finned (black symbols), and possibly mixed (gray symbols; could be either species) pilot whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2011, and 2016 and the Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. The inferred distribution of the two species is preliminary and is valid for June-August only. Isobaths are the 1,000-m and 3,000-m depth contours. The U.S. EEZ is also displayed in green.

In U.S. Atlantic waters, pilot whales (*Globicephala* sp.) are distributed principally along the continental shelf edge off the northeastern U.S. coast in winter and early spring (CETAP 1982; Payne and Heinemann 1993; Abend and Smith 1999; Hamazaki 2002). In late spring, pilot whales move onto Georges Bank and into the Gulf of Maine and more northern waters, and remain in these areas through late autumn (CETAP 1982; Payne and Heinemann 1993). Pilot whales tend to occupy areas of high relief or submerged banks. They are also associated with the Gulf Stream wall and thermal fronts along the continental shelf edge (Waring *et al.* 1992). Long-finned and short-finned pilot whales overlap spatially along the mid-Atlantic shelf break between Delaware and the southern flank of Georges Bank (Payne and Heinemann 1993; Rone and Pace 2012). Long-finned pilot whales have occasionally been observed stranded as far south as Florida, and short-finned pilot whales have occasionally been observed stranded as far north as Massachusetts. The latitudinal ranges of the two species therefore remain uncertain, although south of Cape Hatteras, most pilot whale sightings are expected to be short-finned pilot whales, while north of ~42°N most pilot whale sightings are expected to be long-finned pilot whales (Figure 1).

POPULATION SIZE

The best available estimate for long-finned pilot whales in the western North Atlantic is 39,215 (CV=0.30; Table 1; Garrison 2020; Palka 2020; Lawson and Gosselin 2018). This estimate is the sum of the estimates generated from the northeast U.S. summer 2016 surveys covering U.S. waters from central Virginia to Maine and the Department of Fisheries and Oceans Canada summer 2016 survey covering Canadian waters from the U.S. to Labrador. 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. 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, the 2016 survey estimates in U.S. waters were corrected for availability bias (due to diving behavior), whereas the 2011 estimates were not corrected. These survey data have been combined with an analysis of the spatial distribution of the 2 species based on genetic analyses of biopsy samples to derive separate abundance estimates (Garrison and Rosel 2017).

Key uncertainties in the population size estimate include the uncertain separation between the short-finned and long-finned pilot whales; the small negative bias due to the lack of an abundance estimate in the region between the US and the Newfoundland/Labrador survey area; and the uncertainty due to the unknown precision and accuracy of the availability bias correction factor that was applied.

Earlier 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. Due to changes in survey methodology, these historical data should not be used to make comparisons with more current estimates.

Recent surveys and abundance estimates for *Globicephala* sp.

An abundance estimate of 11,865 (CV=0.57) *Globicephala* sp. was generated from aerial and shipboard surveys conducted during June–August 2011 between central Virginia and the lower Bay of Fundy (Palka 2012). The aerial portion covered 6,850 km of tracklines over waters north of New Jersey between the coastline and the 100-m depth contour through the U.S. and Canadian Gulf of Maine, and up to and including the lower Bay of Fundy. Pilot whales were not observed during the aerial portion of the survey. The shipboard portion covered 3,811 km of tracklines between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. Exclusive Economic Zone (EEZ). Both sighting platforms used a double-platform data-collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009). The vessel portion of this survey included habitats where both short-finned and long-finned pilot whales occur. A logistic regression (see next section) was used to estimate the abundance of long-finned pilot whales from this survey as 5,636 (CV=0.63).

An abundance estimate of 16,946 (CV=0.43) *Globicephala* sp. was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida (Garrison 2016). This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25· bigeye binoculars. A total of 4,445 km of tracklines was surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break north of Cape Hatteras, North Carolina, with a lower number of sightings over the continental slope in the southern portion of the survey. Estimation of pilot whale abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009). This survey included habitats where only short-finned pilot whales are expected to occur.

Abundance estimates of 8,166 (CV=0.31) and 25,114 (CV=0.27) *Globicephala* sp. were generated from vessel surveys conducted in the northeast and southeast U.S., respectively, during the summer of 2016. The Northeast survey was conducted during 27 June–25 August and consisted of 5,354 km of on-effort trackline. The majority of the survey was conducted in waters north of 38ºN latitude and included trackline along the shelf break and offshore to the U.S. EEZ. Pilot whale sightings were concentrated along the shelf-break between the 1,000-m and 2,000-m isobaths and along Georges Bank (NMFS 2017). The Southeast vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort. Pilot whales were observed in high densities along the shelf-break between Cape Hatteras and New Jersey and also in waters further offshore in the mid-Atlantic and off the coast of Florida (NMFS 2017; Garrison and Palka 2018). Both the Northeast and Southeast 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. A logistic regression model was used to estimate the abundance of long-finned pilot whales from these surveys. For the northeast survey, this resulted in an abundance estimate of 10,997 (CV=0.51) long-finned pilot whales. In the southeast, the model indicated that this survey included habitats expected to exclusively contain short-finned pilot whales so no estimate for long-finned pilot whales was generated.

An abundance estimate of 28,218 (CV=0.36) long-finned pilot whales from the Newfoundland/Labrador region was 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 were 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. An availability bias correction factor, which was based on the cetaceans' surface intervals, was also applied. The Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf survey detected 10 pilot whale groups, however, no abundance estimate was produced.

Spatial Distribution and Abundance Estimates for *Globicephala melas*

Biopsy samples from pilot whales were collected during summer months (June–August) from South Carolina to the southern flank of Georges Bank between 1998 and 2007. These samples were identified to species using phylogenetic analysis of mitochondrial DNA sequences Stranded specimens that were morphologically identified to species were used to assign clades in the phylogeny to species and thereby identify all samples. The probability of a sample being from a long-finned (or short-finned) pilot whale was evaluated as a function of sea-surface temperature, latitude, and month using a logistic regression. This analysis indicated that the probability of a sample coming from a long-finned pilot whale was near 1 at water temperatures <22°C, and near 0 at temperatures >25°C. The probability of a long-finned pilot whale also increased with increasing latitude. Spatially, during summer months, this regression model predicted that all pilot whales observed in offshore waters near the Gulf Stream are most likely short-finned pilot whales. The area of overlap between the two species occurs primarily along the shelf break off the coast of New Jersey between 38°N and 40°N latitude (Garrison and Rosel 2017).

This model was used to partition the abundance estimates from surveys conducted during the summers of 2011 and 2016. The sightings from the southeast shipboard surveys covering waters from Florida to New Jersey were predicted to consist entirely of short-finned pilot whales. The aerial portion of the northeast surveys covered the Gulf of Maine and the Bay of Fundy and surveys where the model predicted that only long-finned pilot whales would occur. The vessel portion of the northeast surveys recorded a mix of both species along the shelf break, and the sightings in offshore waters near the Gulf Stream were predicted to consist predominantly of short-finned pilot whales (Garrison and Rosel 2017).

Table 1. Summary of recent abundance estimates for the western North Atlantic long-finned pilot whale (Globicephala melas melas) by month, year, and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	central Virginia to Lower Bay of Fundy	5,636	0.63
Jun–Aug 2016	central Virginia to Lower Bay of Fundy	10,997	0.51
Aug–Sep 2016	Newfoundland/Labrador	28,218	0.36
Jun–Sep 2016	Central Virginia to Labrador -COMBINED	39,215	0.30

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 western North Atlantic long-finned pilot whales is 39,215 animals (CV=0.30). The minimum population estimate for long-finned pilot whales is 30,627.

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 for long-finned pilot whales is 30,627. The maximum productivity rate is 0.04, the default value for cetaceans. The "recovery" factor is 0.5 because this stock is of unknown status relative to optimum sustainable population (OSP) and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic long-finned pilot whale is 306.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual observed average fishery-related mortality or serious injury during 2013–2017 was 21 long-finned pilot whales (CV=0.22; Table 2). In bottom trawls and mid-water trawls and in the gillnet fisheries, mortalities were more generally observed north of 40°N latitude and in areas expected to have only long-finned pilot whales. Takes in these fisheries were therefore attributed to the long-finned pilot whales. Takes in the pelagic longline fishery were partitioned according to a logistic regression model (Garrison and Rosel 2017).

Fishery Information

The commercial fisheries that could potentially interact with this stock in the Atlantic Ocean are the Category I northeast sink gillnet and the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries; and the Category II northeast bottom trawl and northeast mid-water trawl (including pair trawl) fisheries. Detailed fishery information is reported in Appendix III.

Earlier Interactions

Historically, fishery interactions have been documented with pilot whales in the Atlantic pelagic drift gillnet fishery, Atlantic tuna pair trawl and tuna purse seine fisheries, northeast and mid-Atlantic gillnet fisheries, northeast and mid-Atlantic bottom trawl fisheries, northeast midwater trawl fishery, and the pelagic longline fishery. See Appendix V for more information on historical takes.

Longline

Most of the estimated marine mammal bycatch in the U.S. pelagic longline fishery was recorded in U.S. Atlantic EEZ waters between South Carolina and Cape Cod (Garrison 2017). During 2010–2013, all observed interactions and estimated bycatch in the pelagic longline fishery was assigned to the short-finned pilot whale stock because the observed interactions all occurred at times and locations where available data indicated that long-finned pilot whales were very unlikely to occur. Specifically, the highest bycatch rates of undifferentiated pilot whales were observed during September–November along the mid-Atlantic coast (south of 40°N; Garrison 2007), and biopsy data collected in this area during October–November 2011 indicated that only short-finned pilot whales occurred in this region (Garrison and Rosel 2017). Similarly, all genetic data collected from interactions in the pelagic longline fishery have indicated interactions with short-finned pilot whales. During 2014–2017, pilot whale interactions (all serious injuries) were apportioned between the short-finned and long-finned pilot whale stocks according to a logistic regression model (described above in 'Spatial Distribution and Abundance Estimates for *Globicephala melas*') (Garrison and Rosel 2017). See Table 2 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

Fishery-related bycatch rates for years 2013–2017 were estimated using an annual stratified ratio-estimator (Lyssikatos *et al.* 2020). See Table 2 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 (Including Pair Trawl)

Three pilot whales were taken in the northeast mid-water trawl fishery in 2013 near the western edge of Georges Bank. Four were taken in 2014 and 3 during 2016. Using model-based predictions and at-sea identification, these takes have all been assigned as long-finned pilot whales. Expanded estimates of fishery mortality for 2013- 2017 are not available, and so for those years the raw number is provided. See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

CANADA

Unknown numbers of long-finned pilot whales have been taken in Newfoundland, Labrador, Scotian shelf and Bay of Fundy groundfish gillnets; Atlantic Canada and Greenland salmon gillnets; and Atlantic Canada cod traps (Read 1994).

Table 2. Summary of the incidental mortality and serious injury of long-finned pilot whales (Globicephala melas melas.) by U.S. commercial fisheries including the years sampled (Years), the type of data used (Data Type), the annual observer coverage coverage (Observer Coverage), the observed mortalities and serious injuries recorded by on-board observers, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury (Estimated Combined Mortality), the estimated CV of the combined estimates (Est. CVs) and the mean of the combined estimates (CV in parentheses). These are minimum observed counts as expanded estimates are not available.

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury[e]	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
Northeast Bottom Trawl	2013 2014 2015 2016 2017	Obs. Data, Logbook	0.15 0.17 0.19 0.12 0.16	0 1 0 0 0	4 5 0 4 0	0 6 0 0 0	16 25 0 29 0	16 32 0 29 0	0.42 0.44 na 0.58 na	15 (0.30)
Northeast Mid-Water Trawl - Including Pair Trawl[c]	2013 2014 2015 2016 2017	Obs. Data, Dealer Data, VTR Data	0.37 0.42 0.08 0.27 0.16	0 0 0 0 0	3 4 0 3 0	0 0 0 0 0	3 4 0 3 0	3 4 0 3 0	na na na na na	2.0 (na)
Pelagic Longline Fishery	2013 2014 2015 2016 2017	Obs. Data, Logbook Data	0.09 0.1 0.12 0.15 0.12	0 1 1 1 1	0 0 0 0 0	0 9.6 2.2 1.1 3.3	0 0 0 0	0 9.6 2.2 1.1 3.3	na 0.43 0.49 0.6 .98	3.2 (0.33)
TOTAL	-	-	-	-	-	-	-	-	-	21 (0.22)

[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 (unallocated Dealer Data and Allocated Dealer Data) which are used as a measure of total landings. Mandatory Vessel Trip Reports (VTR) (Trip Logbook) are used to determine the spatial distribution of landings and fishing effort. Total landings are used as a measure of total effort for the coastal gillnet fishery.
[b] The observer coverages for the northeast sink gillnet fishery are ratios based on tons of fish landed. Northeast bottom trawl and northeast mid-water trawl fishery coverages are ratios based on trips.

[c] Expanded estimates are not available for this fishery.
[d] Serious injuries were evaluated for the 2013–2017 period and include both at-sea monitor and traditional observer data (Josephson *et al*. 2019).

Other Mortality

Pilot whales have a propensity to mass strand throughout their range, but the role of human activity in these events is unknown. From 2013 to 2017, 16 long-finned pilot whales (*Globicephala melas melas*) were reported stranded between Maine and Florida, including the EEZ (Table 3; NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018).

Long-finned pilot whales have been reported stranded as far south as Florida, where 2 long-finned pilot whales were reported stranded in November 1998, though their flukes had been apparently cut off, so it is unclear where these animals actually may have died. One additional long-finned pilot whale stranded in South Carolina in 2003, though the confidence in the species identification at the time was only moderate. A genetic sample from this animal has subsequently been sequenced and mitochondrial DNA analysis supports the long-finned pilot whale identification.

During 2013–2017, 1 human interaction was documented in stranded pilot whales within the U.S. EEZ. One long-finned pilot whale in 2014 in Maine was classified as a human interaction.

Table 3. Pilot whale (Globicephala melas melas) strandings along the Atlantic coast, 2013–2017. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded pilot whales to species, reports to specific species should be viewed with caution.

STATE	2013	2014	2015	2016	2017	TOTAL-
Nova Scotia[a]	15	0	21	12	12	60
Newfoundland and Labrador[b]	1	0	0	0	1	2
Maine[c]	0	3	0	1	1	5
Massachusetts	3	1	0	1	1	6
New York	2	1	0	0	0	3
New Jersey	1	0	0	0	0	1
Maryland	1	0	0	0	0	1
TOTAL U.S.	**7**	**5**	**0**	**2**	**2**	**16**

[a] Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.). Strandings in 2013 include one fishery entanglement (bait net) and one mass stranding of 4 animals.
[b] (Ledwell and Huntington 2013, 2014, 2015, 2017, 2018).
[c] 2016 animal released alive.

Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury, particularly for offshore species such as pilot whales, 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). 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.

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018). Moderate levels of these contaminants have been found in pilot whale blubber (Taruski *et al.* 1975; Muir *et al.* 1988; Weisbrod *et al.* 2000). Weisbrod *et al.* (2000) examined polychlorinated biphenyl and chlorinated pesticide concentrations in bycaught and stranded pilot whales in the western North Atlantic. Contaminant levels were similar to or lower than levels found in other toothed whales in the western North Atlantic, perhaps because they are feeding further offshore than other species (Weisbrod *et al.* 2000). Dam and Bloch (2000) found very high

PCB levels in long-finned pilot whales in the Faroes. Also, high levels of toxic metals (mercury, lead, cadmium) and selenium were measured in pilot whales harvested in the Faroe Island drive fishery (Nielsen *et al.* 2000). However, the population effect of the observed levels of such contaminants on this stock 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.

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; 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

The long-finned pilot whale is not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not considered strategic under the MMPA because the mean annual human-caused mortality and serious injury does not exceed PBR. Total U.S. fishery-related mortality and serious injury for long-finned pilot whales 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 this stock relative to OSP in the U.S. Atlantic EEZ is unknown. A population trend analysis for this stock has not been conducted.

Based on the low levels of uncertainty described in the above sections, it expected these uncertainties will have little effect on the designation of the status of this stock.

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SHORT-FINNED PILOT WHALE (*Globicephala macrorhynchus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

There are two species of pilot whales in the western North Atlantic - the long-finned pilot whale, *Globicephala melas melas*, and the short-finned pilot whale, *G. macrorhynchus*. These species are difficult to differentiate at sea and cannot be reliably visually identified during either abundance surveys or observations of fishery mortality without high-quality photographs (Rone and Pace 2012). Pilot whales (*Globicephala* sp.) in the western North Atlantic occur primarily along the continental shelf break from Florida to the Nova Scotia Shelf (Mullin and Fulling 2003). Long-finned and short-finned pilot whales overlap spatially along the mid-Atlantic shelf break between Delaware and the southern flank of Georges Bank (Payne and Heinemann 1993; Rone and Pace 2012). Long-finned pilot whales have occasionally been observed stranded as far south as Florida, and short-finned pilot whales have occasionally been observed stranded as far north as Massachusetts (Pugliares *et al.* 2016). The exact latitudinal ranges of the two species remain uncertain. However, south of Cape Hatteras most pilot whale sightings are expected to be short-finned pilot whales, while north of approximately 42°N most pilot whale sightings are expected to be long-finned pilot whales (Figure 1; Garrison and Rosel 2017). Short-finned pilot whales are also documented in the wider Caribbean (Bernard and Riley 1999) and along the continental shelf and continental slope in the northern Gulf of Mexico (Mullin and Fulling 2004; Maze-Foley and Mullin 2006).

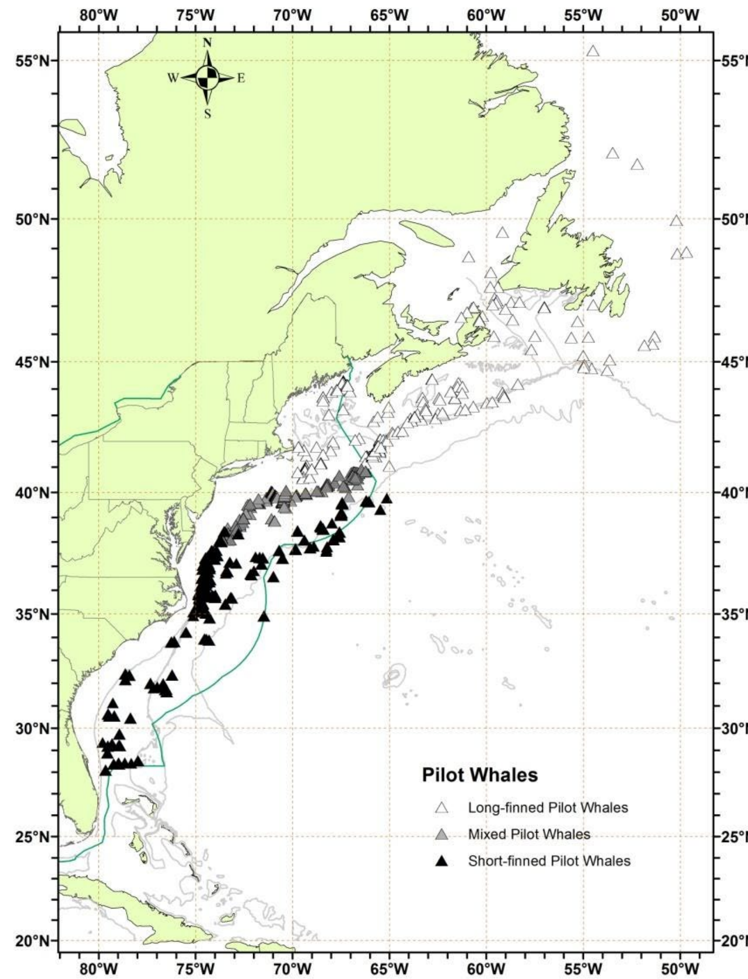

Figure 1. Distribution of long-finned (open symbols), short-finned (black symbols), and possibly mixed (gray symbols; could be either species) pilot whale sightings from NEFSC and SEFSC shipboard and aerial surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016, and DFO's 2007 TNASS and 2016 NAISS surveys. The inferred distribution of the two species is preliminary and is valid for June-August only. Isobaths are the 200m, 1,000m and 4,000m depth contours. The green line indicates the U.S. EEZ.

Thorne *et al.* (2017) tracked 33 short-finned pilot whales off Cape Hatteras in 2014 and 2015 using satellite-linked telemetry tags. Kernel density estimates of habitat use by whales during tracking were concentrated along the continental shelf break from Cape Hatteras north to Hudson Canyon, but whale distribution also included shelf break waters south of Cape Lookout, shelf break waters off Nantucket Shoals, and deeper offshore waters of the Gulf Stream east and north of Cape Hatteras, reinforcing that the continental shelf break is an important foraging habitat for short-finned pilot whales in the western North Atlantic.

Finally, short-finned pilot whales that have stranded alive along the U.S. Atlantic coast and subsequently were released and tracked via satellite telemetry have travelled hundreds of kilometers from their release sites to other areas of the U.S. Atlantic and to the Caribbean (e.g., Irvine *et al.* 1979; Wells *et al.* 2013). Whether these movements are representative of normal species' patterns is unknown because they were generated from animals that stranded.

An analysis of stock structure within the western North Atlantic Stock has not been completed so there are insufficient data to determine whether there are multiple demographically-independent populations within this stock. Continued studies to evaluate genetic population structure in short-finned pilot whales throughout the region will improve understanding of stock structure. Pending these results, the *Globicephala macrorhynchus* population occupying U.S. Atlantic waters is managed separately from both the northern Gulf of Mexico stock and the Puerto Rico and U.S. Virgin Islands stock.

POPULATION SIZE

The best available estimate for short-finned pilot whales in the western North Atlantic is 28,924 (CV=0.24; Table 1; Palka 2012; Garrison 2016; Garrison and Rosel 2017; Garrison and Palka 2018). This estimate is from summer 2016 shipboard surveys covering waters from central Florida to the lower Bay of Fundy and is considered the best available abundance estimate because it is based on the most recent surveys covering the full range of short-finned pilot whales in U.S. Atlantic waters. Because long-finned and short-finned pilot whales are difficult to distinguish at sea, sightings data were reported as *Globicephala* sp. Pilot whale sightings from these surveys were strongly concentrated along the continental shelf break; however, pilot whales were also observed over the continental slope in waters associated with the Gulf Stream (Figure 1). These survey data have been combined with an analysis of the spatial distribution of the two pilot whale species based on genetic analyses of biopsy samples to derive separate abundance estimates for each species (Garrison and Rosel 2017).

Earlier Estimates

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

Recent surveys and abundance estimates for *Globicephala* sp.

For waters between central Virginia and the lower Bay of Fundy, an abundance estimate of 11,865 (CV=0.57) *Globicephala* sp. was generated from aerial and shipboard surveys conducted during June–August 2011 (Palka 2012). The aerial portion covered 6,850 km of trackline over waters north of New Jersey between the coastline and the 100-m depth contour through the U.S. and Canadian Gulf of Maine, and up to and including the lower Bay of Fundy. Pilot whales were not observed during the aerial portion of the survey. The shipboard portion covered 3,811 km of trackline between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. Exclusive Economic Zone (EEZ). Estimation of abundance was based on the independent observer approach, which allows estimation of abundance corrected for perception bias of the detected species, assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009). The vessel portion of this survey included habitats where both short-finned and long-finned pilot whales occur. Short-finned pilot whales are not predicted to occur north of Georges Bank. A logistic regression (see next section) was used to estimate the abundance of short-finned pilot whales from this survey as 4,569 (CV=0.57).

For waters between central Virginia and central Florida, an abundance estimate of 16,946 (CV=0.43) *Globicephala* sp. was generated from a shipboard survey conducted during June–August 2011 (Garrison 2016). This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25x150 "bigeye" binoculars. A total of 4,445 km of trackline was surveyed. The majority of pilot whale sightings occurred along the continental shelf break north of Cape Hatteras, North Carolina, with a lower number of sightings over the continental slope in the southern portion of the survey. Estimation of pilot whale abundance was based on the independent observer approach, which allows estimation of abundance corrected for perception bias of the detected species, assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009). A logistic regression (see next section) was used to estimate the abundance of short-finned pilot whales from this survey. The regression indicated this survey included habitats expected to exclusively contain short-finned pilot whales resulting in an abundance estimate of 16,946 (CV=0.43) short-finned pilot whales from this survey.

Abundance estimates of 3,810 (CV=0.42) and 25,114 (CV=0.27) *Globicephala* sp. were generated from vessel surveys conducted in the northeast and southeast U.S., respectively, during the summer of 2016. The northeast survey was conducted during 27 June–25 August and consisted of 5,354 km of on-effort trackline. The majority of the survey was conducted in waters north of 38ºN latitude and included trackline along the shelf break and offshore to the U.S. EEZ. Pilot whale sightings were concentrated along the shelf-break between the 1,000-m and 2,000-m isobaths and along Georges Bank (NMFS 2017). The southeast vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and the U.S. EEZ during 30 June–19 August. A total of 4,399 km of trackline was covered on effort. Pilot whales were observed in high densities along the shelf-break between Cape Hatteras and New Jersey and also in waters further offshore in the mid-Atlantic and off the coast of Florida (NMFS 2017; Garrison and Palka 2018). Both the northeast and southeast 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. A logistic regression model (see next section) was used to estimate the abundance of short-finned pilot whales from these surveys. For the northeast survey, this resulted in an abundance estimate of 3,810 (CV=0.42) short-finned pilot whales. In the southeast, the model indicated that this survey included habitats expected to exclusively contain short-finned pilot whales resulting in an abundance estimate of 25,114 (CV=0.27).

Spatial Distribution and Abundance Estimates for *Globicephala macrorhynchus*

Pilot whale biopsy samples were collected during summer months (June–Augst) from South Carolina to the southern flank of Georges Bank between 1998 and 2007. These samples were identified to species using phylogenetic analysis of mitochondrial DNA sequences. Samples from stranded specimens that were morphologically identified to species were used to assign clades in the phylogeny to species and thereby identify all survey samples. The probability of a sample being from a short-finned (or long-finned) pilot whale was evaluated as a function of sea surface temperature, latitude, and month using a logistic regression. This analysis indicated that the probability of a sample coming from a short-finned pilot whale was near zero at water temperatures <22°C, and near one at temperatures >25°C. The probability of being a short-finned pilot whale also decreased with increasing latitude. Spatially, during summer months, this regression model predicted that all pilot whales observed in offshore waters near the Gulf Stream are most likely short-finned pilot whales. The area of overlap between the two species occurs primarily along the shelf break between 38°N and 40°N latitude (Garrison and Rosel 2017). This model was used to partition the abundance estimates from surveys conducted during the summers of 2011 and 2016. The sightings from the shipboard surveys covering waters from Florida to New Jersey were predicted to consist entirely of short-finned pilot whales. The vessel portion of the northeast surveys from New Jersey to the southern flank of Georges Bank included waters along the shelf break and waters further offshore extending to the U.S. EEZ. Pilot whales were observed in both areas during the survey. Along the shelf break, the model predicted a mixture of both species, but the sightings in offshore waters near the Gulf Stream were again predicted to consist predominantly of short-finned pilot whales (Garrison and Rosel 2017). The best abundance estimate for short-finned pilot whales is thus the sum of the southeast survey estimate (25,114; CV=0.27) and the estimated number of short-finned pilot whales from the northeast vessel survey (3,810; CV=0.42). The best available abundance estimate is thus 28,924 (CV=0.24).

Table 1. Summary of recent abundance estimates for the western North Atlantic short-finned pilot whale (**Globicephala macrorhynchus*****) by month, year, and area covered during each abundance survey, and resulting abundance estimate (Nbest) and coefficient of variation (CV). Estimates for the entire stock area (COMBINED) include pooled CVs.***

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	central Virginia to lower Bay of Fundy	4,569	0.57
Jun–Aug 2011	central Florida to central Virginia	16,946	0.43
Jun–Aug 2011	central Florida to lower Bay of Fundy (COMBINED)	21,515	0.37
Jun–Aug 2016	New Jersey to lower Bay of Fundy	3,810	0.42
Jun–Aug 2016	central Florida to New Jersey	25,114	0.27
Jun–Aug 2016	central Florida to lower Bay of Fundy (COMBINED)	28,924	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 western North Atlantic *Globicephala macrorhnychus* is 28,924 animals (CV=0.24). The minimum population estimate is 23,637.

Current Population Trend

There are three available coastwide abundance estimates for short-finned pilot whales from the summers of 2004, 2011, and 2016. Each of these is derived from vessel surveys with similar survey designs and all three used the two-team independent observer approach to estimate abundance. The southeast component of these surveys all were expected to contain exclusively short-finned pilot whales, and the logistic regression model was used to partition pilot whale sightings from the northeast portion of the survey between the short-finned and long-finned species based upon habitat characteristics. The resulting estimates were 24,674 (CV=0.52) in 2004, 21,515 (CV=0.36) in 2011, and 28,924 (CV=0.24) in 2016 (Garrison and Palka 2018). A generalized linear model indicated no significant trend in these abundance estimates. The key uncertainty is the assumption that the logistic regression model accurately represents the relative distribution of short-finned vs. long-finned pilot whales in each year.

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 for short-finned pilot whales is 23,637. 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 and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic short-finned pilot whale is 236.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for this stock during 2013–2017 is unknown. The estimated mean annual fishery-related mortality and serious injury during 2013–2017 due to the pelagic longline fishery was 160 short-finned pilot whales (CV=0.12; Table 2). Uncertainty in this estimate arises because it incorporates a logistic regression model to predict the species of origin (long-finned or short-finned pilot whale) for each bycaught whale. The statistical uncertainty in the assignment to species is incorporated into the abundance estimates; however, the analysis assumes that the collected biopsy samples adequately represent the distribution of the two species and that the resulting model correctly predicts shifts in distribution in response to changes in environmental conditions. In addition to observed takes in the pelagic longline fishery, there was a self-reported take in 2013 in the unobserved hook and line fishery. This unobserved take renders the estimate of total annual fishery-caused mortality and serious injury an underestimate.

In bottom trawl, mid-water trawl, and gillnet fisheries, pilot whale mortalities were observed north of 40°N latitude in areas expected to have only long-finned pilot whales. Takes and bycatch estimates for these fisheries are therefore attributed to the long-finned pilot whale stock.

Fishery Information

There are three commercial fisheries that interact, or that potentially could interact, with this stock in the Atlantic Ocean. These include two Category I fisheries (the Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline and the Atlantic Highly Migratory Species longline fisheries) and one Category III fishery (the Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery). All recent gillnet and trawl interactions have been assigned to long-finned pilot whales using model-based predictions. Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for information on historical takes.

Pelagic Longline

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. The estimated annual average serious injury and mortality attributable to the Atlantic Ocean large pelagics longline fishery for the five-year period from 2013 to 2017 was 160 short-finned pilot whales (CV=0.12; Table 2). During 2013–2017, 92 serious injuries were observed in the following fishing areas of the North Atlantic: Florida East Coast, Mid-Atlantic Bight, Northeast Coastal, and South Atlantic Bight. During 2013–2017, one mortality was observed (in 2016) in the Florida East Coast fishing area (Garrison and Stokes 2014; 2016; 2017; 2019; 2020).

Prior to 2014, estimated bycatch in the pelagic longline fishery was assigned to the short-finned pilot whale stock because the observed interactions all occurred at times and locations where available data indicated that long-finned pilot whales were very unlikely to occur. Specifically, the highest bycatch rates of undifferentiated pilot whales were observed during September–November along the mid-Atlantic coast (south of 38°N; Garrison 2007), and biopsy data collected in this area during October–November 2011 indicated that only short-finned pilot whales occurred in this region (Garrison and Rosel 2017). Similarly, all genetic data collected from interactions in the pelagic longline fishery have indicated interactions with short-finned pilot whales. However, during 2014–2016, pilot whale interactions (including serious injuries) were observed further north and along the southern flank of Georges Bank. Therefore, the logistic regression model (described above in 'Spatial Distribution and Abundance Estimates for *Globicephala macrorhynchus*') was applied to estimate the probability that these interactions were from short-finned vs. long-finned pilot whales (Garrison and Rosel 2017). Due to high water temperatures (ranging from 22 to 25ºC) at the time of the observed takes, these interactions were estimated to have a >90% probability of coming from short-finned pilot whales. The estimated probability was used to apportion the estimated serious injury and mortality from 2014 to 2016 in the pelagic longline fishery between the short-finned and long-finned pilot whale stocks (Garrison and Stokes 2016; 2017; 2019).

Between 1992 and 2004, most of the marine mammal bycatch in the U.S. pelagic longline fishery was recorded in U.S. Atlantic EEZ waters between South Carolina and Cape Cod (Garrison 2007). From January to March, observed bycatch was concentrated on the continental shelf edge northeast of Cape Hatteras, North Carolina. During April–June, bycatch was recorded in this area as well as north of Hydrographer Canyon in water over 1,000 fathoms (1830 m) deep. During the July–September period, observed takes occurred on the continental shelf edge east of Cape Charles, Virginia, and on Block Canyon slope in over 1,000 fathoms of water. October–December bycatch occurred between the 20- and 50-fathom (37- and 92-m) isobaths between Barnegat Bay, New Jersey, and Cape Hatteras, North Carolina.

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of short-finned pilot whales within high seas waters of the Atlantic Ocean have been observed or reported thus far.

See Table 2 for bycatch estimates and observed mortality and serious injury for the current five-year period, and Appendix V for historical estimates of annual mortality and serious injury.

Table 2. Summary of the incidental mortality and serious injury of short-finned pilot whales (*Globicephala macrorhynchus*) by the pelagic longline commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), 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 (Estimated Combined Mortality), the estimated CV of the combined annual mortality estimates (Est. CVs) and the mean of the combined mortality estimates (CV in parentheses).

Fishery	Years	Vessels[a]	Data Type[b]	Percent Observer Coverage[c]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
Pelagic Longline	2013	879	Obs. Data, Logbook	79	113	0	124	00	124	0.32	160 (0.12)
	2014	78		10	19	0	233	0	233	0.24	
	2015	74		12	32	0	200	0	200	0.24	
	2016	60		15	14	1	106	5.1	111	0.31	
	2017	65		12	14	0	133	0	133	0.29	

[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 (NEFOP) and the Southeast Pelagic Longline Observer Program.
[c] Percentage of sets observed

Hook and Line

During 2013–2017, there was one self-reported take (in 2013) in which a short-finned pilot whale was hooked and entangled by a charterboat fisherman. The animal was released alive but considered seriously injured (Maze-Foley and Garrison 2016).

Other Mortality

Pilot whales have a propensity to mass strand throughout their range, but the role of human activity in these events is unknown. Between two and 168 pilot whales have stranded annually, either individually or in groups, along the eastern U.S. seaboard since 1980 (NMFS 1993; NOAA National Marine Mammal Health and Stranding Response Database unpublished data). During 2013–2017, 14 short-finned pilot whales (*Globicephala macrorhynchus*) and one pilot whale not specified to the species level (*Globicephala* sp.) were reported stranded between Massachusetts and Florida (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 13 June 2018 (SER) and 8 June 2018 (NER)). One short-finned pilot whale stranding was reported as far north as Cape Cod, Massachusetts (2016); the remaining strandings occurred from North Carolina southward (Table 3). It could not be determined whether there was evidence of human interaction for six of these strandings, and for eight, no evidence of human interaction was detected. Evidence of human interaction was detected for one stranded animal which had ingested a fishing hook (Table 3).

Table 3. Short-finned pilot whale (*Globicephala macrorhynchus *[SF] and* Globicephala *sp. [Sp]) strandings along the Atlantic coast, 2013–2017. Strandings which were not reported to species have been reported as* Globicephala *sp. The level of technical expertise among stranding network personnel varies, and given the potential difficulty in correctly identifying stranded pilot whales to species, reports to specific species should be viewed with caution. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER). EEZ=U.S. Exclusive Economic Zone (offshore U.S. waters).

STATE	2013-SF	2013-Sp	2014-SF	2014-Sp	2015-SF	2015-Sp	2016-SF	2016-Sp	2017-SF	2017-Sp	TOTAL-SF	TOTAL-Sp
EEZ	0	0	0	1[a]	0	0	0	0	0	0	0	1
Massachusetts	0	0	0	0	0	0	1[b]	0	0	0	1	0
North Carolina	0	0	3	0	2	0	0	0	1	0	6	0
South Carolina	1	0	2	0	0	0	0	0	0	0	3	0
Georgia	0	0	0	0	1	0	0	0	1	0	2	0
Florida	0	0	0	0	2	0	0	0	0	0	2	0
TOTALS	**1**	**0**	**5**	**1**	**5**	**0**	**1**	**0**	**2**	**0**	**14**	**1**

a. This animal was found offshore, 90 NM east of Cape Cod.
b. This animal had evidence of an interaction with fishing gear (the animal ingested a fishing hook).

Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury, particularly for offshore species such as pilot whales, 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018). Moderate levels of these contaminants have been found in pilot whale blubber (Taruski *et al.* 1975; Muir *et al.* 1988; Weisbrod *et al.* 2000). Weisbrod *et al.* (2000) examined polychlorinated biphenyl and chlorinated pesticide concentrations in bycaught and stranded pilot whales in the western North Atlantic. Contaminant levels were similar to or lower than levels found in other toothed whales in the western North Atlantic, perhaps because they are feeding further offshore than other species (Weisbrod *et al.* 2000). Dam and Bloch (2000) found very high PCB levels in long-finned pilot whales in the Faroes. Also, high levels of toxic metals (mercury, lead, cadmium) and selenium were measured in pilot whales harvested in the Faroe Island drive fishery (Nielsen *et al.* 2000). However, the population effect of the observed levels of such contaminants on this stock 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.

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; 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

The short-finned pilot whale is not listed as threatened or endangered under the Endangered Species Act, and the western North Atlantic stock is not a strategic stock under the MMPA because the mean annual human-caused mortality and serious injury does not exceed PBR. The status of this stock relative to OSP in the U.S. Atlantic EEZ is unknown. Total U.S. fishery-related mortality and serious injury attributed to short-finned pilot whales exceeds 10% of the calculated PBR and therefore cannot be considered to be insignificant and approaching zero mortality and serious injury rate. There is no evidence for a trend in population size for this stock. Should there be multiple demographically-independent stocks within this stock's range, the geographically-concentrated nature of the fishery-related mortality and serious injury could mean that the mortality is impacting one stock more than the other.

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ATLANTIC WHITE-SIDED DOLPHIN (*Lagenorhynchus acutus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The dolphin genus *Lagenorhynchus* is currently proposed to be revised (Vollmer *et al.* 2019); though until the revision is officially accepted, the previous definitions will be used. White-sided dolphins are found in temperate and sub-polar waters of the North Atlantic, primarily in continental shelf waters to the 100-m depth contour. In the western North Atlantic the species inhabits waters from multiple marine ecoregions (Spalding 2007) within the region from central West Greenland to North Carolina (about 35˚N) and perhaps as far east as 29˚W in the vicinity of the mid-Atlantic Ridge (Evans 1987; Hamazaki 2002; Doksaeter *et al.* 2008; Waring *et al.* 2008). Distribution of sightings, strandings and incidental takes suggest the possible existence of three population units: Gulf of Maine, Gulf of St. Lawrence and Labrador Sea populations (Palka *et al.* 1997). Evidence for a separation between the population in the southern Gulf of Maine and the Gulf of St. Lawrence population comes from the reduced density of summer sightings along the Atlantic side of Nova Scotia. This was reported in Gaskin (1992), is evident in Smithsonian stranding records and in Canadian/west Greenland bycatch data (Stenson *et al.* 2011), and was obvious during summer abundance surveys that covered waters from Virginia to the Gulf of St. Lawrence and during the Canadian component of the Trans-North Atlantic Sighting Survey in the summer of 2007 (Lawson and Gosselin 2009, 2011). White-sided dolphins were seen frequently in Gulf of Maine waters and in waters at the mouth of the Gulf of St. Lawrence, but only a relatively few sightings were recorded between these two regions. This gap has been less obvious since 2007 and could be related to an increasing number of animals being distributed more northwards due to climatic/ecosystem changes that are occurring in the Gulf of Maine (Nye *et al*. 2009; Head *et al*. 2010; Pinsky *et al*. 2013; Hare *et al*. 2016; Grieve *et al.* 2017). No comparative genetic analyses of samples from U.S. waters and the Gulf of St. Lawrence and/or Newfoundland have been made.

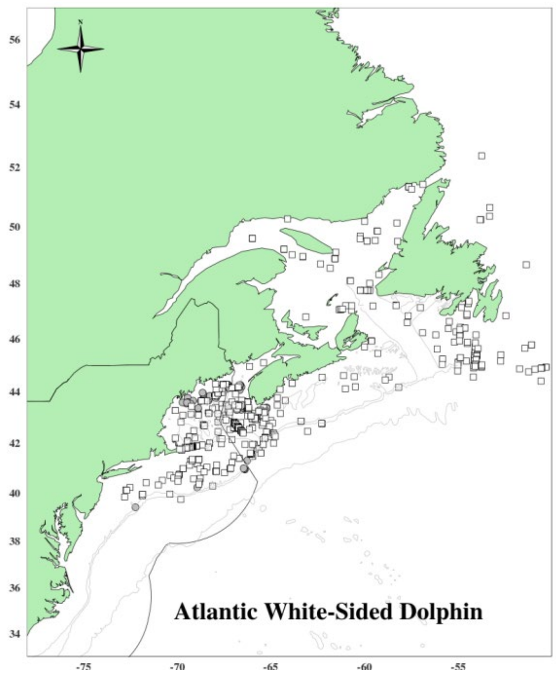

Figure 1. Distribution of white-sided dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 200-m, 1000-m and 4000-m depth contours.

The Gulf of Maine population of white-sided dolphins is most common in continental shelf waters from Hudson Canyon (approximately 39˚N) to Georges Bank, and in the Gulf of Maine and lower Bay of Fundy. Sighting data indicate seasonal shifts in distribution (Northridge *et al.* 1997). During January to May, low numbers of white-sided dolphins are found from Georges Bank to Jeffreys Ledge (off New Hampshire), with even lower numbers south of Georges Bank, as documented by a few strandings collected on beaches of Virginia to South Carolina. From June through September, large numbers of white-sided dolphins are found from Georges Bank to the lower Bay of Fundy. From October to December, white-sided dolphins occur at intermediate densities from southern Georges Bank to southern Gulf of Maine (Payne and Heinemann 1990). Sightings south of Georges Bank, particularly around Hudson Canyon, occur year-round but at low densities. The Virginia and North Carolina observations appear to represent the southern extent of the species' range during the winter months. On 4 May 2008 a stranded 17-year old male white-sided dolphin with severe pulmonary distress and reactive lymphadenopathy stranded in South Carolina (Powell *et al.* 2012). In the absence of additional strandings or sightings, this stranding seems to be an out-of-range anomaly. The seasonal spatial distribution of this species appears to be changing during the last few years. There is evidence for an earlier distributional shift during the 1970s, from primarily offshore waters into the Gulf of Maine, hypothesized to be related to shifts in abundance of pelagic fish stocks resulting from depletion of herring by foreign distant-water fleets (Kenney *et al*. 1996).

Stomach-content analysis of both stranded and incidentally caught white-sided dolphins in U.S. waters determined that the predominant prey were silver hake (*Merluccius bilinearis*), spoonarm octopus (*Bathypolypus bairdii*) and haddock (*Melanogrammus aeglefinus*). Sand lances (*Ammodytes* spp.) were only found in the stomach of one stranded white-sided dolphin. Seasonal variation in diet was indicated; pelagic Atlantic herring (*Clupea harengus*) was the most important prey in summer, but was rare in winter (Craddock *et al.* 2009).

Within the Gulf of Maine population a genetic analysis comparing samples from Maine to samples from Massachusetts found no significant differentiation (Banguera-Hinestroza *et al.* 2014). Abrahams (2014) compared samples collected between Connecticut and Maine to those collected between New York and North Carolina and found no evidence for genetic differentiation between these two regions. Sample sizes in these studies in some cases were low, and the potential for seasonal movement, as suggested by Northridge *et al.* (1997), has the potential to confound these studies if season was not considered in the sampling scheme.

As a consequence of these distribution patterns and genetic analyses, this report assumes white-sided dolphins in U.S. waters are from the Gulf of Maine population, which is separate from the neighboring Gulf of St. Lawrence population. In summary, the Western North Atlantic stock of white-sided dolphins may contain multiple demographically-independent populations, where the animals in U.S. waters are part of the Gulf of Maine population. However, further research is necessary to support this hypothesis and eliminate the uncertainties.

POPULATION SIZE

The best available current abundance estimate for white-sided dolphins in the western North Atlantic stock is 93,233 (CV= 0.71), resulting from the June–September 2016 surveys conducted by the U.S. and Canada that ranged from Labrador to the U.S. east coast, which covered nearly the entire western North Atlantic stock: all of the Gulf of Maine and Gulf of St. Lawrence populations and part of the Labrador population. Because the survey areas did not overlap, the estimates from the 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 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 to determine the current PBR.

Recent surveys and abundance estimates

An abundance estimate of 31,912 (CV=0.61) U.S. Gulf of Maine white-sided dolphins 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 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 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 61,321 (CV=1.04) white-sided dolphins from the Canadian side of the Gulf of Maine population and the entire Gulf of St. Lawrence population was generated from an aerial survey conducted by the Department of Fisheries and Oceans, Canada (DFO). No white-sided dolphins were detected on the east side of Labrador in the Labrador population. 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 Newfound/Labrador stratum using a DeHavilland Twin Otter. The 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 the cetaceans' surface intervals, was also applied.

Table 1. Summary of recent abundance estimates for western North Atlantic stock of white-sided dolphins (Lagenorhynchus acutus), by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	Central Virginia to lower Bay of Fundy	48,819	0.61
Jun–Sep 2016	Central Virginia to Maine (US part of Gulf of Maine population)	31,912	0.61
Aug–Sep 2016	Bay of Fundy to Gulf of St. Lawrence (Canadian part of Gulf of Maine and all of Gulf of St. Lawrence population)	61,321	1.04
Aug–Sep 2016	Newfoundland and Labrador (part of the Labrador population)	0	0
Jun–Sep 2016	Central Virginia to Labrador – COMBINED	93,233	0.710

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 western North Atlantic stock of white-sided dolphins is 93,233 (CV=0.71). The minimum population estimate for these white-sided dolphins is 54,443.

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. Life history parameters that could be used to estimate net productivity include: calving interval is 2–3 years; lactation period is 18 months; gestation period is 10–12 months and births occur from May to early August, mainly in June and July; length at birth is 110 cm; length at sexual maturity is 230–240 cm for males, and 201–222 cm for females; age at sexual maturity is 8–9 years for males and 6–8 years for females; mean adult length is 250 cm for males and 224 cm for females (Evans 1987); and maximum reported age for males is 22 years and for females, 27 years (Sergeant *et al.* 1980).

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 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 54,443. 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 the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic stock of white-sided dolphin is 544.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated average fishery-related mortality or serious injury to this stock during 2013–2017 was 26 (CV=0.20) white-sided dolphins (Table 2).

Key uncertainties include 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 in some times and areas (0.02–0.10). The effect of this is unknown.

There are no major known sources of unquantifiable human-caused mortality or serious injury for the U.S. portion of the Gulf of Maine population. When considering the entire western North Atlantic stock, mortality in Canadian Atlantic waters is largely unquantified.

Fishery Information

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

U.S.

Northeast Sink Gillnet

White-sided dolphin bycatch has been rare in this fishery, but when it occurred it was in both the Gulf of Maine and southern New England regions and mostly in non-summer (May–August) months. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Table 2; Hatch and Orphanides 2015, 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for long-term bycatch information.

Northeast Bottom Trawl

White-sided dolphins have been bycaught year-round in the Gulf of Maine, where most occurred outside of summer (May–August) and offshore near the outer edge of the EEZ. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos *et al.* 2020). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for long-term bycatch information.

Mid-Atlantic Bottom Trawl

White-sided dolphin bycatch has been rare in this fishery, but when it occurred it was usually in the winter (January–April) and around Hudson Canyon. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos *et al.* 2020). See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for long-term bycatch information.

Table 2. Summary of the incidental mortality of western North Atlantic stock of white-sided dolphins (Lagenorhynchus acutus) 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 estimated CV of the combined annual mortality and the mean annual mortality (CV in parentheses).

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
Northeast Sink Gillnet	2013	Obs. Data, Weighout, Trip Logbook	0.11	0	1	0	4	4	1.03	2.8 (0.56)
	2104		0.18	0	2	0	10	10	.66	
	2015		0.14	0	0	0	0	0	0	
	2016		0.10	0	0	0	0	0	0	
	2017		0.12	0	0	0	0	0	0	
Northeast Bottom Trawl	2013	Obs. Data, Trip Logbook	0.15	0	8	0	33	33	.31	21 (0.21)
	2104		0.17	0	3	0	16	16	.5	
	2015		0.19	0	3	0	15	15	.52	
	2016		0.12	0	3	0	28	28	.46	

Fishery	Years	Data Type [a]	Observer Coverage [b]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality
	2017		0.16	1	1	7.4	7.4	14.8	.64	
Mid-Atlantic Bottom Trawl	2012 2013 2104 2015 2016 2017	Obs. Data,Trip Logbook	0.06 0.08 0.09 0 10 0.10	0 0 0 0 0	0 1 0 0 0	0 0 0 0 0	0 9.67 0 0 0	0 9.67 0 0 0	0 .94 0 0 0	1.9 (0.94)
Total										26 (0.20)

a Observer data (Obs. Data), used to measure bycatch rates, are collected within the Northeast 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. Mandatory Vessel Trip Reports (VTR) (Trip Logbook) are used to determine the spatial distribution of landings and fishing effort in the sink gillnet, bottom trawl and mid-water trawl fisheries. In addition, the Trip Logbooks are the primary source of the measure of total effort (tow duration) in the mid-water and bottom trawl fisheries.

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. Total observer coverage reported for bottom trawl and gillnet gear includes samples collected from the at-sea monitoring program in addition to traditional observer coverage through the Northeast Fisheries Observer Program (NEFOP).

c Serious injuries were evaluated for the 2013–2017 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2019).

CANADA

There is little information available that quantifies fishery interactions involving white-sided dolphins in Canadian waters. Two white-sided dolphins were reported caught in groundfish gillnet sets in the Bay of Fundy during 1985 to 1989, and 9 were reported taken in West Greenland between 1964 and 1966 in the now non-operational salmon drift nets (Gaskin 1992). Several (number not specified) were also taken during the 1960s in now non-operational Newfoundland and Labrador groundfish gillnets. A few (number not specified) were taken in an experimental drift gillnet fishery for salmon off West Greenland that took place from 1965 to 1982 (Read 1994).

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 25–40% of large Canadian fishing vessels (greater than 100 feet long), and on approximately 5% of smaller Canadian fishing vessels. Bycaught marine mammals were noted as weight in kilos rather than by the numbers of animals caught. Thus the number of individuals was estimated by dividing the total weight per species per trip by the maximum recorded weight of each species. During 1991 through 1996, an estimated 6 white-sided dolphins were observed taken. One animal was from a longline trip south of the Grand Banks (43º 10'N 53º 08'W) in November 1996 and the other 5 were taken in the bottom trawl fishery off Nova Scotia in the Atlantic Ocean; 1 in July 1991, 1 in April 1992, 1 in May 1992, 1 in April 1993, 1 in June 1993 and 0 in 1994 to 1996.

Estimation of small cetacean bycatch for Newfoundland fisheries using data collected during 2001 to 2003 (Benjamins *et al.* 2007) indicated that, while most of the estimated 862 to 2,228 animals caught were harbor porpoises, a few were white-sided dolphins caught in the Newfoundland nearshore gillnet fishery and offshore monkfish/skate gillnet fisheries.

Other Mortality

U.S.

Recent Atlantic white-sided dolphin strandings on the U.S. Atlantic coast are documented in Table 3 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018). Sixteen of these animals were released alive. Human interaction was indicated in 4 records during this period. None of these were classified as fishery interactions.

Mass strandings involving up to a hundred or more animals at one time are common for this species. The causes of these strandings are not known. Because such strandings have been known since antiquity, it could be presumed that recent strandings are a normal condition (Gaskin 1992). It is unknown whether human causes, such as fishery interactions and pollution, have increased the number of strandings. In an analysis of mortality causes of stranded marine mammals on Cape Cod and southeastern Massachusetts between 2000 and 2006, Bogomolni *et al.* (2010) found 69% (46 of 67) of stranded white-sided dolphins were involved in mass-stranding events with no significant cause determined, and 21% (14 of 67) were classified as disease-related.

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.

CANADA

The Nova Scotia Stranding Network documented whales and dolphins stranded on the coast of Nova Scotia during 1991 to 1996 (Hooker *et al.* 1997). Researchers with Dept. of Fisheries and Oceans, Canada documented strandings on the beaches of Sable Island during 1970 to 1998 (Lucas and Hooker 2000). More recently whales and dolphins stranded on the coast of Nova Scotia have been recorded by the Marine Animal Response Society and the Nova Scotia Stranding Network (Table 3; Marine Animal Response Society, pers. comm.). In addition, stranded white-sided dolphins in Newfoundland and Labrador are being recorded by the Whale Release and Strandings Program (Table 3; Ledwell and Huntington 2013, 2014, 2015, 2017, 2018).

Table 3. Atlantic white-sided dolphin (Lagenorhynchus acutus) reported strandings along the U.S. and Canadian Atlantic coast, 2013-2017.

Area	**2013**	**2014**	**2015**	**2016**	**2017**	**Total**
Maine[b]	1	2	1	0	0	4
Massachusetts[a,b]	10	4	3	27	8	52
Rhode Island	1	0	0	0	0	1
Connecticut	0	0	0	1	1	1
New York	2	0	0	0	0	2
TOTAL US	**14**	**6**	**4**	**28**	**28**	**60**
Nova Scotia[c]	7	12	11	11	8	49
Newfoundland and Labrador[d]	0	0	0	13	1	14
GRAND TOTAL	**21**	**23**	**`15**	**38**	**38**	**123**

[a] Records of mass strandings in Massachusetts during this period are: April 2013 - 2 animals (1 released alive); December 2013 - 3 animals (all released alive); March 2016 - 2 animals (1 released alive), July 2016 – 2 animals (1 released alive), 3 animals (all released alive); September 2016 - 17 animals (all released alive) .

[b]In 2014, 1 animal in Massachusetts was classified as human interaction due to attempts by public to return animal to sea. In 2014, 1 animal in Maine was classified as human interaction due to plastics ingestion. In 2016, 2 animals (one of which was released alive) in Massachusetts were classified as human interaction due to intervention on the beach.

[c] Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.). 2014 data include a mass stranding of 7 animals all released alive and a single animal released alive. 2015 data include a mass stranding of 5 animals.

[d] Ledwell and Huntington (2013, 2014, 2015, 2017, 2018).

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 Atlantic white-sided dolphins 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

White-sided dolphins are not listed as threatened or endangered under the Endangered Species Act. The Western North Atlantic stock of white-sided dolphins is not considered strategic under the Marine Mammal Protection Act. The estimated average annual human-related mortality does not exceed PBR and is less than 10% of the calculated PBR; therefore, it is considered to be insignificant and approaching zero mortality and serious injury rate. The status of white-sided dolphins, relative to OSP, in the U.S. Atlantic EEZ is unknown. A trend analysis has not been conducted for this species.

Even with the levels of uncertainties regarding the stock structure within the western North Atlantic white-sided dolphin stock described above, it is expected these uncertainties will have little effect on the designation of the status of this population.

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WHITE-BEAKED DOLPHIN (*Lagenorhynchus albirostris*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The dolphin genus *Lagenorhynchus* is currently proposed to be revised (Vollmer *et al.* 2019); though until the revision is officially accepted, the previous definitions will be used. White-beaked dolphins are the more northerly of the two species of *Lagenorhynchus* in the northwest Atlantic (Leatherwood *et al.* 1976). The species is found in waters from southern New England to southern Greenland and Davis Straits (Leatherwood *et al.*1976; CETAP 1982), across the Atlantic to the Barents Sea and south to at least Portugal (Reeves *et al.* 1999). Differences in skull features indicate that there are at least two separate stocks, one in the eastern and one in the western North Atlantic (Mikkelsen and Lund 1994). No genetic analyses have been conducted to corroborate this stock structure.

In waters off the northeastern U.S. coast, white-beaked dolphin sightings are concentrated in the western Gulf of Maine and around Cape Cod (CETAP 1982). The limited distribution of this species in U.S. waters has been attributed to opportunistic feeding (CETAP 1982). Prior to the 1970's, white-sided dolphins (*L. acutus*) in U.S. waters were found primarily offshore on the continental slope, while white-beaked dolphins were found on the continental shelf. During the 1970's, there was an apparent switch in habitat use between these two species. This shift may have been a result of the increase in sand lance in the continental shelf waters (Katona *et al.* 1993; Kenney *et al.* 1996).

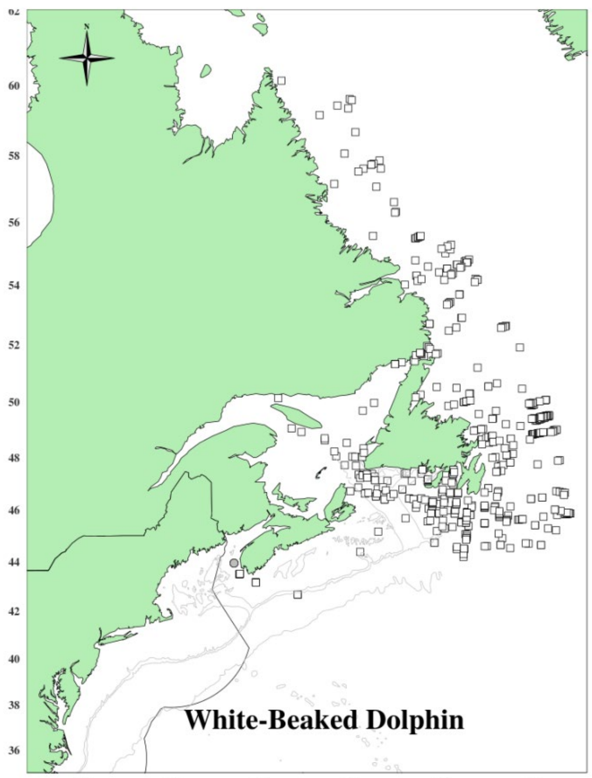

Figure 1. Distribution of white-beaked dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004 and 2006, 2011 and 2016 and DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100m. 200m, 1000m and 4000m depth contours.

POPULATION SIZE

The best abundance estimate for the western North Atlantic white-beaked dolphin is 536,016 (CV=0.31), an estimate derived from aerial survey data collected in during the Canadian Northwest Atlantic International Sightings Survey (NAISS) survey in the summer of 2016.

As recommended in the GAMMS Workshop Report (Wade and Angliss 1997), estimates older than eight years are deemed unreliable and should not be used for PBR determinations.

Recent surveys and abundance estimates

An abundance estimate of 530,538 (CV=0.39; Table 1) white-beaked dolphins in Atlantic Canadian waters was 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 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 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 to all platform estimates. An availability bias correction factor, which was based on published records of the cetaceans' surface intervals, was also applied.

No white-beaked dolphins were seen on the summer 2016 U.S. surveys.

Table 1. Summary of abundance estimates for western North Atlantic white-beaked dolphins. Month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Aug–Sep 2016	Bay of Fundy/Scotian Shelf	5,478	0.495
Aug–Sep 2016	Newfoundland/Labrador	530,538	0.314
Aug–Sep 2016	Canadian Atlantic waters (COMBINED)	536,016	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 western North Atlantic stock of white-beaked dolphins is 536,016 (CV=0.31). The minimum population estimate for these white-beaked dolphins is 415,344.

Current Population Trend

There are insufficient data to determine population trends for this species. The change in abundance estimates between the DFO 2007 and 2016 aerial surveys in Canadian waters could not have resulted from reproduction alone so immigration from other areas of the north Atlantic likely occurred.

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 (Wade and Angliss 1997). The minimum population size of white-beaked dolphins is 415,344. 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 this stock is of unknown status. PBR for the western North Atlantic white-beaked dolphin is 4,153.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

There are no documented reports of fishery-related mortality or serious injury to this stock in the U.S. EEZ.

Fishery Information

Because of the absence of observed fishery-related mortality and serious injury to this stock in the U.S. and Canadian waters, no fishery information is provided.

Other Mortality

Recent white-beaked dolphin strandings on the U.S. Atlantic coast are documented in Table 2 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018). Human interaction was indicated in 2 records during this period, one due to plastic ingestion as well as buckshot found in the blubber (healed) and one due to post-mortem carcass handling. Neither of these were classified as fishery interactions.

Table 2. Summary of number of stranded white-beaked dolphins during 2013 to 2017, by year and area within U.S. and Canada.

Area	Year					**Total**
	2013	2014	2015	2016	2017	
Massachusetts	0	4	0	0	0	4
North Carolinaa	0	0	1	0	0	1
TOTAL US	**0**	**4**	**0**	**1**	**0**	**5**
Nova Scotia[b]	0	2	0	0	0	2
Newfoundland/Labrador[c]	0	68	6	0	11	85
GRAND TOTAL	**2**	**3**	**0**	**1**	**1**	**7**

a. North Carolina stranding was a new southerly record for this species (Thayer *et al.* 2018).
b. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.).
c. Data supplied by the Newfoundland and Labrador Whale Release and Strandings Program (Ledwell and Huntington 2013, 2014, 2015, 2017, 2018). Includes animals released alive.

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., Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for the western north Atlantic stock of white-beaked dolphins 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

The status of white-beaked dolphins, relative to OSP, in U.S. Atlantic coast waters is unknown. The species is not listed as threatened or endangered under the Endangered Species Act. There are insufficient data to determine population trends for this species. The total documented U.S. fishery-related mortality and serious injury for this stock (0) is less than 10% of the calculated PBR (4.153) and, therefore, is considered to be insignificant and at zero mortality and serious injury rate. This is a non-strategic stock because the 2013-2017 estimated average annual human related mortality does not exceed PBR.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

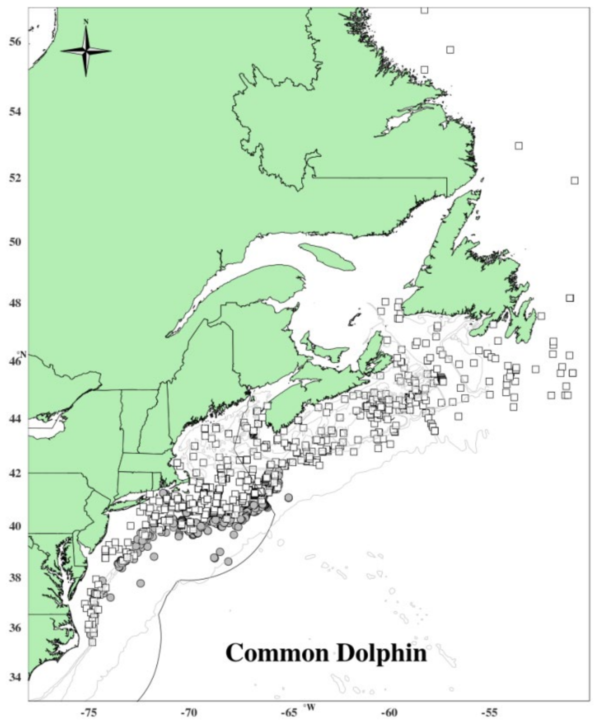

Figure 1. Distribution of common dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2010 ,2011, 2016 and Department of Fisheries and Oceans Canada 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100-m, 1000-m and 4000-m depth 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 (32º N) (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 172,825 (CV=0.21) which is the total of Canadian and U.S. surveys conducted in 2016. 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 (Table 1).

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

An abundance estimate of 67,191 (CV=0.29) common dolphins was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a double-platform data-collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling (MRDS) option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 2,993 (CV=0.87) common dolphins was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed a double-platform visual team procedure searching with 25·150 "bigeye" binoculars. A total of 4,445 km of tracklines was surveyed. Estimation of the abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the MRDS option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009) (Table 1).

Abundance estimates of 48,574 (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 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_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jul–Aug 2011	Central Virginia to lower Bay of Fundy	67,191	0.29
Jun–Aug 2011	Central Florida to Central Virginia	2,993	0.87
Jun–Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	70,184	0.28
June–Sep 2016	Central Virginia to lower Bay of Fundy	80,227	0.31
June–Aug 2016	Florida to Central Virginia	900	0.57
June–Sep 2016	Newfoundland/Labrador	48,574	0.48
June–Sep 2016	Bay of Fundy/Scotian Shelf/Gulf of St. Lawrence	43,124	0.28
June–Sep 2016	Florida to Newfoundland/Labrador (COMBINED)	172,825	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 172,825 animals (CV=0.21), derived from the 2016 aerial and shipboard surveys. The minimum population estimate for the western North Atlantic common dolphin is 145,091.

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.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Average annual estimated fishery-related mortality or serious injury to this stock during 2013–2017 was 419 (CV=0.10) common dolphins from estimated annual bycatch in observed fisheries plus 0.2 from research takes, for a total of 419.2.

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.

Fishery information

Detailed fishery information is reported in Appendix III. Earlier Interactions

Historically, U.S. fishery interactions have been documented with common dolphins in the northeast and mid-Atlantic gillnet fisheries, northeast and mid-Atlantic bottom trawl fisheries, northeast and mid-Atlantic mid-water trawl fishery, and the pelagic longline fishery. See Appendix V for more information on historical takes.

Northeast Sink Gillnet

Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Hatch and Orphanides 2015, 2016; Orphanides and Hatch 2017*,* Orphanides 2019, 2020). See Table 2 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 2015, 2016; Orphanides and Hatch 2017, Orphanides 2019, 2020). See Table 2 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.* 2020). See Table 2 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.* 2020). See Table 2 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 2013–2017 were documented in Garrison and Stokes (2014, 2016, 2017, 2020. 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 2 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 to Mystic Aquarium, Mystic, Connecticut; NOAA GARFO Office, Gloucester, Massachusetts; NOAA law enforcement; and NOAA Protected Species Branch, Woods Hole, Massachusetts. A complete necropsy was conducted at the Wood Hole Oceanographic Institution, Woods Hole, Massachusetts.

Table 2. 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	2013	Obs. Data, Trip Logbook, Allocated Dealer Data	0.11	0	5	0	104	104	0.46	97 (.19)
	2014		0.18	0	11	0	111	111	0.47	
	2015		0.14	0	3	0	55	55	0.54	
	2016		0.10	0	8	0	80	80	0.38	
	2017		0.12	0	20	0	133	133	0.28	
Mid-Atlantic Gillnet	2013	Obs. Data, Weighout	0.03	0	2	0	62	62	0.67	18 (.25)
	2014		0.05	0	1	0	17	17	0.86	
	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	
Northeast Bottom Trawl [c]		Obs. Data, Logbook		0		0				14 (.25)
	2013		0.15	0	4	0	17	17	0.54	
	2014		0.17	0	3	0	17	17	0.53	
	2015		0.19	0	4	0	22	22	0.45	
	2016		0.12	0	2	0	16	16	0.46	
	2017		0.16		0		0	0	0	
Mid-Atlantic Bottom Trawl [c]	2013	Obs. Data, Dealer Data	0.06	0	24	0	254	254	0.29	278(.13)
	2014		0.08	3	38	24	305	329	0.29	
	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	
Pelagic Longline	2013	Obs. Data, Logbook Data	0.09	0	0	0	0	0	0	2.8 (.74)
	2014		0.10	0	0	0	0	0	0	
	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	
TOTAL	-	-	-	-	-	-	-	-	-	419 (.10)

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. Beginning in May 2010 total observer coverage reported for bottom trawl and gillnet gear includes samples collected from the at-sea monitoring program in addition to traditional observer coverage through the Northeast Fisheries Observer Program (NEFOP).

c. Fishery related bycatch rates for years 2013-2017 were estimated using an annual stratified ratio-estimator (Lyssikatos *et al.* 2020).

d. Serious injuries were evaluated for the 2013–2017 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2019)

Other Mortality

From 2013 to 2017, 608 common dolphins were reported stranded between Maine and Florida (Table 3; (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018). The total includes mass-stranded common dolphins in Massachusetts during 2013 (a total of 9 in 3 events), 2014 (a total of 14 in 4 events), 2015 (a total of 37 in 13 events), and 2016 (a total of 35 animals in 9 events), and 2 mass strandings in Virginia in 2013 (a total of 6 in 2 events). Animals released or last sighted alive include13 animals in 2013, 12 in 2014, 9 in 2015, 23 in 2016 and 70 in 2017. In 2013, 10 cases were classified as human interaction, 4 of which were fishery interactions. In 2014, 5 cases were classified as human interaction, 1 of which was a fishery interaction. In 2015, 2 cases were classified as human interactions, both in Rhode Island. Seven cases in 2016 were coded as human interaction, 1 of which was a fishery interaction. Six cases in 2017 were coded as human interaction, 2 of which were classified as fishery interactions and 1 of which was classified as a boat collision. 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 no common dolphins stranded in 2013, 3 in 2014, 2 in 2015, 5 in 2016 and 5 in 2017 (Tonya Wimmer/Andrew Reid, pers. comm.).

Table 3. Common dolphin (Delphinus delphis delphis) reported strandings along the U.S. Atlantic coast, 2013-2017.

STATE	2013	2014	2015	2016	2017	**TOTALS**
New Hampshire	0	0	1	1	2	**4**
Massachusetts[a, b]	48	38	40	67	166	**359**
Rhode Island[b]	6	6	7	4	5	**28**
Connecticut	0	0	2	1	1	**4**
New York [b]	24	7	3	3	15	**56**
New Jersey	19	8	3	5	0	**35**
Delaware	3	0	2	0	0	**5**
Maryland	3	0	1	0	0	**4**
Virginia[a]	13	9	2	0	1	**25**
North Carolina	9	6	4	1	0	**20**
TOTALS	**125**	**74**	**65**	**82**	**190**	**540**

a. Massachusetts mass strandings (2013–4, 3 2, 2014 – 2, 2, 5, 5, 2015–2, 2, 2, 2, 2, 2, 2, 3, 3, 3, 4, 4, 4, 4), 2016–8,5,4,4,4,3,3,2,2, 2017–2x5, 3x3, 4x4, 5x5, 7x3, 14x1). Two mass strandings in Virginia in April 2013 - a group of 4 and a group of 2.

b. Ten records with indications of human interactions in 2013 (3 in New York, 1 in Rhode Island and 6 in Massachusetts), 4 of which (1 in Massachusetts and 3 in New York) were classified as fishery interactions. Five records of human interaction in 2014 (1 fisheries interaction in Rhode Island, 2 other human interactions in Massachusetts and 2 in Rhode Island). Two of the human interactions in 2014 (1 Massachusetts and 1 Rhode Island) involved live animals. Two records of HI in 2015, both in Rhode Island. Seven HI cases in 2016 (6 in Massachusetts and 1 in Rhode Island), 5 of which were relocation responses to live animals. Of the 2 dead HI, 1 in Massachusetts was coded as a fishery interaction and 1 in Rhode Island had unauthorized public intervention prior to euthanasia by stranding responders. Six HI cases in 2017 (1 in Rhode Island and 5 in Massachusetts), 2, of which were classified as fishery interactions (1 in Rhode Island and 1 in Massachusetts). One of the Massachusetts HI cases was classified as a boat collision.

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.

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 2013–2017 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. Population trends for this species have not been investigated.

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ATLANTIC SPOTTED DOLPHIN (*Stenella frontalis*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Atlantic spotted dolphins are distributed in tropical and warm temperate waters of the western North Atlantic (Leatherwood *et al.* 1976). Their distribution ranges from southern New England, south through the Gulf of Mexico and the Caribbean to at least Venezuela (Leatherwood *et al.* 1976; Perrin *et al.* 1994). Atlantic spotted dolphins regularly occur in continental shelf and continental slope waters (Figure 1; Payne *et al.* 1984; Mullin and Fulling 2003). Sightings have also been made along the north wall of the Gulf Stream and warm-core ring features (Waring *et al.* 1992).

The Atlantic spotted dolphin occurs in two forms or ecotypes, which may be distinct sub-species (Perrin *et al.* 1987, 1994; Rice 1998): a large, heavily spotted form that inhabits the continental shelf and is usually found inside or near the 200 m isobath in continental shelf waters south of Cape Hatteras; and a smaller, less spotted island and offshore form which occurs in the western North Atlantic in continental slope waters particularly north of Cape Hatteras (Mullin and Fulling 2003). Where they co-occur, the offshore ecotype of the Atlantic spotted dolphin and the pantropical spotted dolphin, *Stenella attenuata*, can be difficult to differentiate at sea.

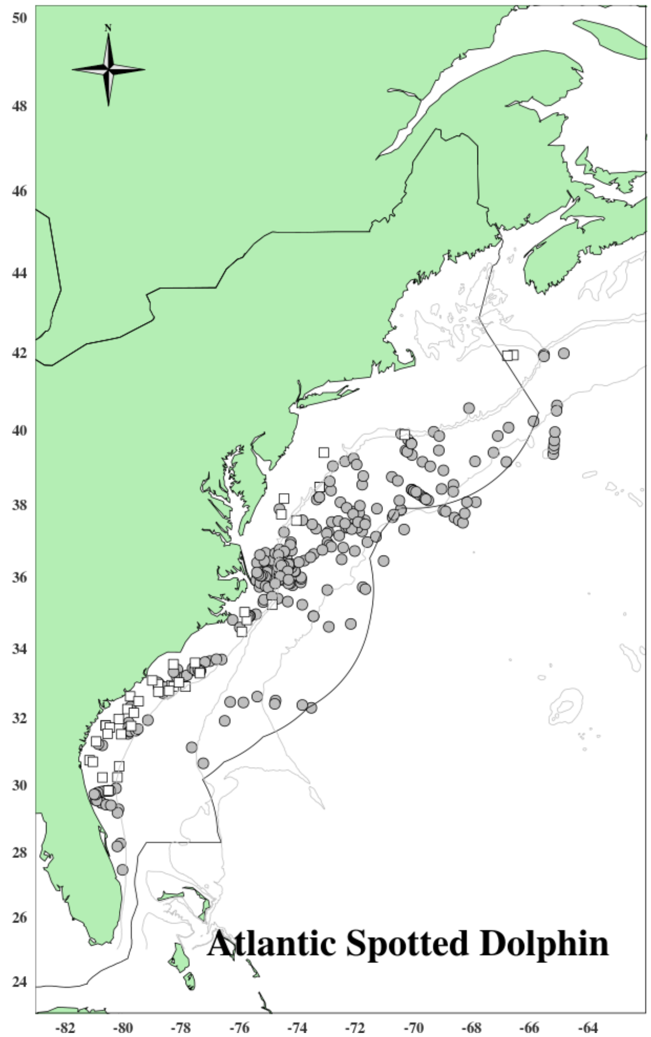

Figure 1. Distribution of Atlantic spotted dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 200m, 1,000m, and 4,000m depth contours. The darker line indicates the U.S. EEZ.

Genetic analyses of mtDNA and microsatellite DNA data from samples collected in the Gulf of Mexico and the western North Atlantic revealed significant genetic differentiation between these two areas (Adams and Rosel 2006; Viricel and Rosel 2014), supporting delimitation of a demographically independent population for each area. In addition, the genetic data provided evidence for separation of dolphins within the western North Atlantic, suggesting the Western North Atlantic stock of Atlantic spotted dolphins may comprise multiple demographically independent populations (Adams and Rosel 2006; Viricel and Rosel 2014). One population consists of the smaller, pelagic form and occupies waters over the continental slope and deeper. The second population is restricted to continental shelf waters at and south of Cape Hatteras. The two genetically-identified populations correspond with the two morphological forms identified by Perrin *et al.* (1987), and the level of genetic differentiation between them indicates they are independent evolutionary pathways with dispersal rates of less than 0.3% per generation (Viricel and Rosel 2014).

POPULATION SIZE

The best abundance estimate available for Atlantic spotted dolphins in the western North Atlantic is 39,921 (CV=0.27; Table 1; Garrison 2020; Palka 2020). This estimate is from summer 2016 surveys covering waters from central Florida to the lower Bay of Fundy. Distinction between the two Atlantic spotted dolphin ecotypes has not regularly been made during surveys, and at their November 1999 meeting, the Atlantic SRG recommended that without a genetic determination of stock structure for the two ecotypes, the abundance estimates for the coastal and offshore forms should be combined. The abundance estimate provided here is a species-specific estimate combining both ecotypes of Atlantic spotted dolphins.

Earlier abundance estimates

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

Recent surveys and abundance estimates

An abundance estimate of 26,798 (CV=0.66) Atlantic spotted dolphins was generated from aerial and shipboard surveys conducted during June-August 2011 between central Virginia and the lower Bay of Fundy. The aerial portion covered 6,850 km of tracklines over waters north of New Jersey between the coastline and the 100-m depth contour through the U.S. and Canadian Gulf of Maine, and up to and including the lower Bay of Fundy. The shipboard portion covered 3,811 km of tracklines between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a double-platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 17,917 (CV=0.42) Atlantic spotted dolphins was generated from a shipboard survey conducted concurrently (June-August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25x bigeye binoculars. A total of 4,445 km of tracklines were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Abundance estimates of 8,247 (CV=0.24) and 31,674 (CV=0.33) Atlantic spotted 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 edge 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 edge 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 abundance estimates for the western North Atlantic spotted dolphins,* Stenella frontalis*, by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	central Virginia to lower Bay of Fundy	26,798	0.66
Jun–Aug 2011	central Florida to central Virginia	17,917	0.42
Jun–Aug 2011	central Florida to lower Bay of Fundy (COMBINED)	44,715	0.43

Jun–Aug 2016	New Jersey to Bay of Fundy	8,247	0.24
Jun–Aug 2016	central Florida to New Jersey	31,674	0.33
Jun–Aug 2016	central Florida to Bay of Fundy (COMBINED)	39,921	0.27

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 abundance estimate is 39,921 (CV=0.27). The minimum population estimates based on the 2016 abundance estimates is 32,032.

Current Population Trend

There are three available coastwide abundance estimates for Atlantic spotted dolphins from the summers of 2004, 2011, and 2016. Each of these is derived from vessel surveys with similar survey designs and all three used the two-team independent observer approach to estimate abundance. The resulting estimates were 50,978 (CV=0.42) in 2004, 44,715 (CV=0.43) in 2011, and 39,921 (CV=0.27) in 2016 (Garrison and Palka 2018). A generalized linear model indicated a statistically significant (p=0.011) linear decrease in these abundance estimates. A key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

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 for the Atlantic spotted dolphin is 32,032. 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 set to 0.5 because this stock is of unknown status. PBR for the combined offshore and coastal forms of Atlantic spotted dolphins is 320.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to Atlantic spotted dolphins in the western North Atlantic.

Fishery Information

The commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean are the Category I Atlantic Highly Migratory Species longline and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively.

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of Atlantic spotted dolphins within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to Atlantic spotted dolphins by this fishery in the Atlantic Ocean during 2013–2017 (Garrison and Stokes 2014; 2016; 2017; 2019; 2020).

Total fishery-related mortality and serious injury cannot be estimated separately for the two species of spotted dolphins in the U.S. Atlantic EEZ because of the uncertainty in species identification by fishery observers. The Atlantic Scientific Review Group advised adopting the risk-averse strategy of assuming that either species might have been subject to the observed fishery-related mortality and serious injury.

Other Mortality

During 2013–2017, 21 Atlantic spotted dolphins were reported stranded between North Carolina and Florida (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER)). It could not be determined whether there was evidence of human interaction for 9 of these strandings, and for 12 dolphins, no evidence of human interaction was detected. 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

Table 2. Atlantic spotted dolphin (**Stenella frontalis*****) reported strandings along the U.S. Atlantic coast, 2013–2017. Data are from the NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER).***

STATE	2013	2014	2015	2016	2017	TOTALS
North Carolina	2	4	2	5	1	14
South Carolina	0	0	1	1	0	2
Georgia	0	0	1	0	0	1
Florida	0	1	1	2	0	4
TOTALS	2	5	5	8	1	21

HABITAT ISSUES

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.

Offshore wind development in the U.S. Atlantic may also pose a threat to this stock, particularly south of Cape Hatteras where it comes closer to shore. Activities associated with development include geophysical and geotechnical surveys, installation of foundations and cables, and operation, maintenance and decommissioning of facilities (BOEM 2018). The greatest threat from these activities is likely underwater noise, however other potential threats include vessel collision due to increased vessel traffic, benthic habitat loss, entanglement due to increased fishing around structures, marine debris, dredging, and contamination/degradation of habitat (BOEM 2018).

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for this stock is lacking. Méndez-Fernandez *et al.* (2018) examined persistent organic pollutant (POP) concentrations (PCBs, DDTs, PBDEs, chlordanes, mirex, and HCB) in Atlantic spotted dolphins from different parts of the Atlantic Ocean, including the Azores, Canary Islands, São Paulo (southeastern Brazil), and Guadalupe Island (Caribbean Sea). Their findings indicated POP concentrations and accumulation patterns varied by location, so dolphins in different geographical areas were subjected to different types of contamination. When PCB concentrations were compared to established toxicity thresholds, 33.9% of animals sampled from all locations exceeded the lowest threshold (9µg/g lw). It was suggested two of the populations examined, from São Paulo and Canary Islands, should be considered vulnerable given the results of the POP concentrations (Méndez-Fernandez *et al.* (2018).

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; 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

Atlantic spotted 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. No fishery-related mortality or serious injury has been observed during 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 Atlantic spotted dolphins in the U.S. Atlantic EEZ relative to OSP is unknown. Available abundance estimates indicate a decline in population size for this species between 2004 and 2016.

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PANTROPICAL SPOTTED DOLPHIN (*Stenella attenuata attenuata*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The pantropical spotted dolphin is distributed worldwide in tropical and some sub-tropical oceans (Perrin *et al.* 1987; Perrin and Hohn 1994). There are two species of spotted dolphin in the Atlantic Ocean, the Atlantic spotted dolphin, *Stenella frontalis,* and the pantropical spotted dolphin, *S. attenuata* (Perrin *et al.* 1987). Where they co-occur in pelagic waters, the Atlantic spotted dolphin and the pantropical spotted dolphin can be difficult to differentiate at sea.

Sightings during surveys in the Atlantic north of Cape Hatteras have been along the continental slope while in waters south of Cape Hatteras sightings were recorded over the Blake Plateau and in deeper offshore waters of the mid-Atlantic (Figure 1).

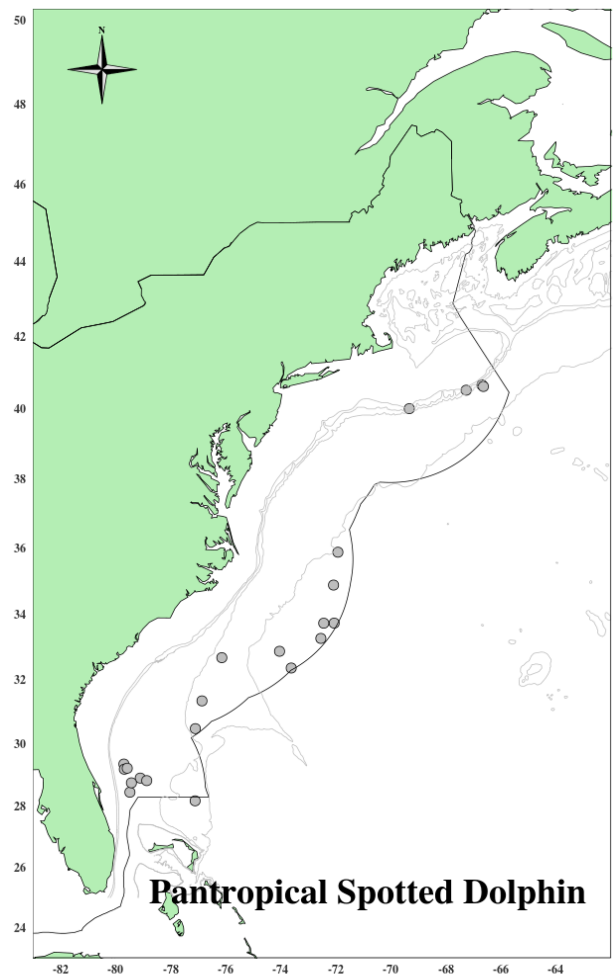

Figure 1. Distribution of pantropical spotted dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 100m, 200m, 1,000m, and 4,000m depth contours. The darker line indicates the U.S. EEZ.

Pantropical spotted dolphins in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. 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). Due to the paucity of sightings, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

POPULATION SIZE

The best abundance estimate available for western North Atlantic pantropical spotted dolphins is 6,593 (CV=0.52; Table 1; Garrison 2020; Palka 2020). This estimate is from summer 2016 surveys covering waters from central Florida to the lower Bay of Fundy.

Earlier abundance estimates

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

Recent surveys and abundance estimates

There were no sightings of pantropical spotted dolphins during aerial and shipboard surveys conducted during June-August 2011 from central Virginia to the lower Bay of Fundy. The aerial portion covered 6,850 km of tracklines over waters north of New Jersey between the coastline and the 100-m depth contour through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,811 km of tracklines between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a double-platform data collection procedure.

An abundance estimate of 3,333 (CV=0.91) pantropical spotted dolphins was generated from a shipboard survey conducted concurrently (June-August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25x bigeye binoculars. A total of 4,445 km of tracklines were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Abundance estimates of 0 and 6,593 (CV=0.52) pantropical spotted dolphins were generated from two non-overlapping 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 included 5,354 km of on-effort trackline along the shelf break and offshore to 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 U.S. EEZ from 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 (Thomas *et al.* 2009).

Table 1. Summary of abundance estimates for the western North Atlantic pantropical spotted dolphin* (Stenella attenuata) *by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	central Virginia to lower Bay of Fundy	0	0
Jun–Aug 2011	central Florida to central Virginia	3,333	0.91
Jun–Aug 2011	central Florida to lower Bay of Fundy (COMBINED)	3,333	0.91
Jun–Aug 2016	New Jersey to lower Bay of Fundy	0	-
Jun–Aug 2016	central Florida to New Jersey	6,593	0.52
Jun–Aug 2016	central Florida to lower Bay of Fundy (COMBINED)	6,593	0.52

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 pantropical spotted dolphins is 6,593 (CV=0.52). The minimum population estimate for pantropical spotted dolphins is 4,367.

Current Population Trend

There are three available coastwide abundance estimates for pantropical spotted dolphins from the summers of 2004, 2011, and 2016. Each of these is derived from vessel surveys with similar survey designs and all three used the two-team independent observer approach to estimate abundance. The resulting estimates were 4,439 (CV=0.49) in 2004, 3,333 (CV=0.91) in 2011, and 6,593 (CV=0.52) in 2016 (Garrison and Palka 2018). A generalized linear model indicated no statistically significant (p=0.645) linear trend in these abundance estimates. The high uncertainty in these estimates limits the ability to detect a population trend. In addition, a key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

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 for pantropical spotted dolphins is 4,367. 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 this stock is of unknown status. PBR for pantropical spotted dolphins is 44.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to pantropical spotted dolphins in the western North Atlantic.

Fishery Information

The commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean are the Category I Atlantic Highly Migratory Species longline and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively.

Detailed fishery information is reported in Appendix III. The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of pantropical spotted dolphins within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to pantropical spotted dolphins by this fishery in the Atlantic Ocean during 2013–2017 (Garrison and Stokes 2014; 2016; 2017; 2019; 2020).

Total fishery-related mortality and serious injury cannot be estimated separately for the two species of spotted dolphins in the U.S. Atlantic EEZ because of the uncertainty in species identification by fishery observers. The Atlantic Scientific Review Group advised adopting the risk-averse strategy of assuming that either species might have been subject to the observed fishery-related mortality and serious injury.

Other Mortality

During 2013–2017, five pantropical spotted dolphins were reported stranded on the U.S. East Coast, all occurring in Florida during 2015 (n=4) and 2016 (n=1) (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER). It could not be determined whether there was evidence of human interaction for one of these strandings, and for the other four strandings, no evidence of human interaction was detected. Stranding data probably 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

HABITAT ISSUES

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.

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for this stock 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; 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

Pantropical spotted 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. No fishery-related mortality or serious injury has been observed during 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 pantropical spotted dolphins in the western U.S. Atlantic EEZ relative to OSP is unknown. There was no statistically significant trend in population size for this species.

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STRIPED DOLPHIN (*Stenella coeruleoalba*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The striped dolphin, *Stenella coeruleoalba*, is distributed worldwide in warm-temperate to tropical seas (Archer and Perrin 1997; Archer 2002). Striped dolphins are found in the western North Atlantic from Nova Scotia south to at least Jamaica and in the Gulf of Mexico. In general, striped dolphins appear to prefer continental slope waters offshore to the Gulf Stream (Leatherwood *et al.* 1976; Perrin *et al.* 1994; Schmidly 1981). There is very little information concerning striped dolphin stock structure in the western North Atlantic (Archer and Perrin 1997).

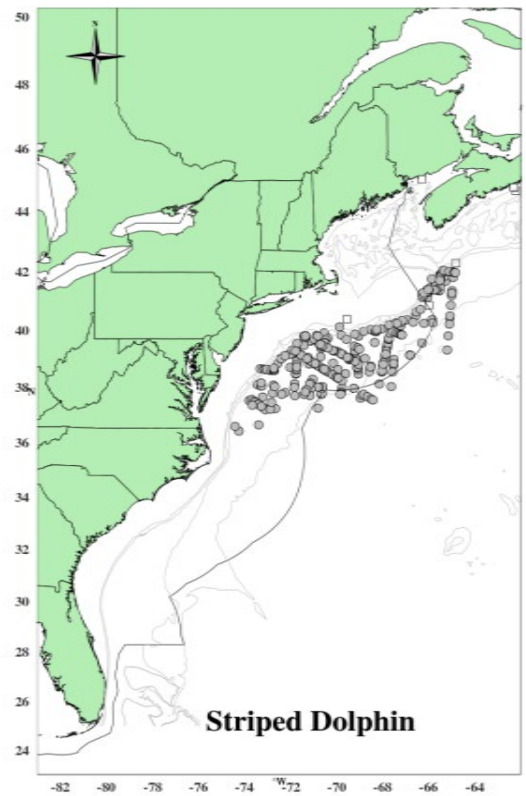

Figure 1: Distribution of striped dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1998, 1999, 2002, 2004, 2006, 2007, 2010, 2011 and 2016. Isobaths are the 100-m, 200-m, 1000-m and 4000-m depth contours. Circle symbols represent shipboard sightings and squares are aerial sightings.

In waters off the northeastern U.S. coast, striped dolphins are distributed along the continental shelf edge from Cape Hatteras to the southern margin of Georges Bank, and also occur offshore over the continental slope and rise in the mid-Atlantic region (CETAP 1982; Mullin and Fulling 2003). Continental shelf edge sightings in this program were generally centered along the 1,000 m depth contour in all seasons (CETAP 1982). During 1990 and 1991 cetacean habitat-use surveys, striped dolphins were associated with the Gulf Stream north wall and warm-core ring features (Waring *et al.* 1992). Striped dolphins seen in a survey of the New England Sea Mounts (Palka 1997) were in waters that were between 20˚and 27˚C and deeper than 900 m.

Although striped dolphins are considered to be uncommon in Canadian Atlantic waters (Baird *et al.* 1997), summer sightings (2-125 individuals) in the deeper and warmer waters of the Gully (submarine canyon off eastern Nova Scotia shelf) suggest that this region may be an important part of their range (Gowans and Whitehead 1995; Baird *et al.* 1997). A July 2017 live stranding of a striped dolphin is the first stranding record of this species in Newfoundland and Labrador (Ledwell *et al*. 2018).

POPULATION SIZE

Several abundance estimates from selected regions are available for striped dolphins for select time periods. Sightings are almost exclusively in the continental shelf edge and continental slope areas west of Georges Bank (Figure 1). The best abundance estimate for striped dolphins is the sum of the 2016 survey estimates—67,036 (CV=0.29).

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 and abundance estimates

An abundance estimate of 46,882 (CV=0.33) striped dolphins was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 km of tracklines that were in waters offshore of Virginia to Massachusetts (waters that were deeper than the 100-m depth contour out to beyond the U.S. EEZ). Both sighting platforms used adouble platform data collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers, 2004). Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling (MRDS) option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 7,925 (CV=0.66) striped dolphins was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25× bigeye binoculars. A total of 4,445 km of tracklines were surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Abundance estimates of 42,783(CV=0.25) and 24,163 (CV=0.66) striped dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison in 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 abundance estimates for western North Atlantic striped dolphins. Month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	Central Virginia to lower Bay of Fundy	46,882	0.33
Jun–Aug 2011	Central Florida to Central Virginia	7,925	0.66
Jun–Aug 2011	Central Florida to lower Bay of Fundy (COMBINED)	54,807	0.3
Jun–Sep 2016	Central Virginia to lower Bay of Fundy	42,783	0.25
Jun–Sep 2016	Florida to Central Virginia	24,163	0.66
Jun–Sep 2016	Florida to lower Bay of Fundy (COMBINED)	67,036	0.29

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 striped dolphins is 67,036 (CV=0.29), obtained from the 2016 surveys. The minimum population estimate for the western North Atlantic striped dolphin is 52,939.

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 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 52,939. 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 0.5 because this stock is of unknown status. PBR for the western North Atlantic striped dolphin is 529.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated average fishery-related mortality to this stock during 2013-2017 was zero striped dolphins.

Fishery Information

Detailed fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

Other Mortality

A total of 22 striped dolphins were reported stranded along the U.S. Atlantic coast between 2013 and 2017 (Table 3; NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). This includes one record of a mass stranding of 12 animals in North Carolina in 2005.

In eastern Canada, 17 strandings were reported between 2013 and 2017. As noted above, 2017 marked the first time a striped dolphin stranding was reported in Newfoundland and Labrador.

Table 3. Striped dolphin reported strandings along the U.S. Atlantic and Canadian coast 2013-2017.

Area	2013	2014	2015	2016	2017	Total
Massachusetts[a]	0	0	1	0	1	2
New York[b]	3	1	1	0	0	5
New Jersey	1	2	0	0	0	7
Maryland	0	0	1	0	1	1
North Carolina	2	2	0	0	0	4
South Carolina	0	0	1	0	0	1
Florida	0	0	0	0	2	2
U.S. TOTAL	7	3	5	0	7	22
Nova Scotia/Prince Edward Island[c,d]	1	1	2	3	9	16
Newfoundland and New	0	0	0	0	1	1

Brunswick[e]						
GRAND TOTAL	8	4	7	3	17	39

a. 2015 animal was released alive.
b. 2013 animal classified as human interaction with signs of vessel strike. 2015 animal classified as a fishery interaction
c. Three of the 2017 animals released alive.
d. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.).
e. Ledwell *et al*. 2018.

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., Storelli and Macrotrigiano 2000; 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 striped dolphins 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

Striped 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. Average annual human-related mortality and serious injury does not exceed the PBR. The total U.S. fishery-related mortality and serious injury for this stock is less than 10% of the calculated PBR, therefore can be considered to be insignificant and approaching zero mortality and serious injury rate. The status of striped dolphins, relative to OSP, in the U.S. Atlantic EEZ is unknown. There are insufficient data to determine the population trends for this species.

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FRASER'S DOLPHIN (*Lagenodelphis hosei*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Fraser's dolphin is distributed worldwide in tropical waters (Perrin *et al.* 1994), and has 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 of this species are rare, and in fact there has been only a single sighting on NMFS surveys in the western North Atlantic (Figure 1). Sightings in the more extensively surveyed northern Gulf of Mexico are uncommon but occur on a regular basis in oceanic waters (>200m) and in all seasons (Leatherwood *et al.* 1993; Hansen *et al.* 1996; Mullin and Hoggard 2000; Mullin and Fulling, 2004). Fraser's dolphins in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. 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 western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding *et al.* 2007; Moore and Merrick 2011). Due to the paucity of sightings in the western North Atlantic, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

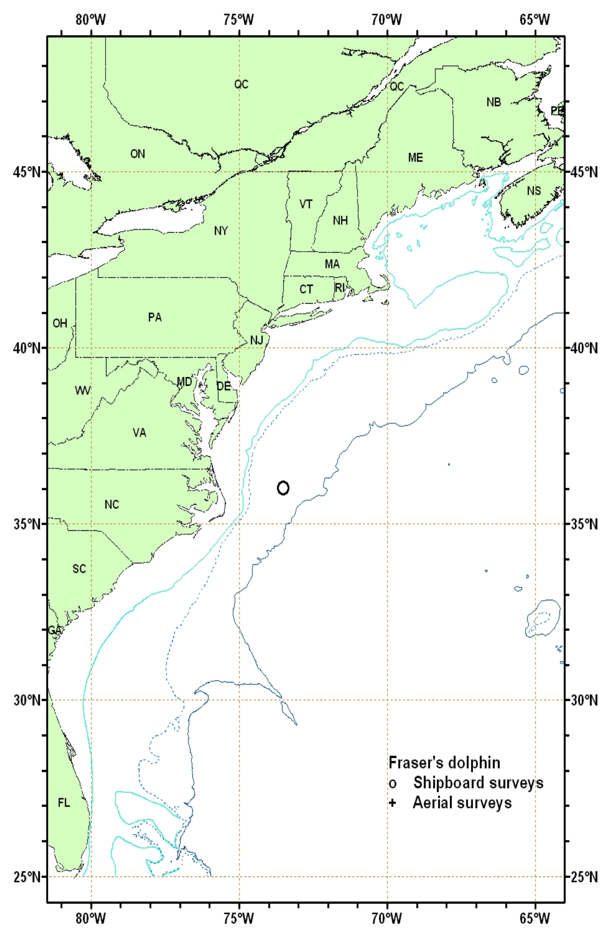

Figure 1. Location of a Fraser's dolphin sighting from a SEFSC vessel survey during summer 1999. NEFSC and SEFSC shipboard and aerial surveys were conducted during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 100m, 1,000m and 4,000m depth contours.

POPULATION SIZE

The numbers of Fraser's dolphins off the U.S. or Canadian Atlantic coast are unknown since it was rarely seen in any surveys. A group of an estimated 250 Fraser's dolphins was sighted in waters 3300 m deep in the western North Atlantic off Cape Hatteras during a 1999 vessel survey (Figure 1; NMFS 1999). Abundances have not been estimated from the 1999 vessel survey in western North Atlantic (NMFS 1999) because the sighting was not made during line- transect sampling effort. Therefore, the population size of Fraser's dolphins is unknown. No Fraser's dolphins have been observed in any other NMFS surveys.

Minimum Population Estimate

Present data are insufficient to calculate a minimum population estimate for this stock.

Current Population Trend

There are insufficient data to determine the population trends for this stock because no estimates of population size are available.

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 history (Barlow *et al.* 1995).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal level (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 is unknown. 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 this stock is of unknown status. PBR for the western North Atlantic Fraser's dolphin stock is unknown because the minimum population size is unknown.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to Fraser's dolphins in the western North Atlantic.

Fishery Information

The commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean are the Category I Atlantic Highly Migratory Species longline and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively.

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of Fraser's dolphins within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to Fraser's dolphins by this fishery in the Atlantic Ocean during 2013–2017 (Garrison and Stokes 2014; 2016; 2017; 2019; 2020).

Other Mortality

There were no reported strandings of a Fraser's dolphin in the U.S. Atlantic Ocean during 2013–2017 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER).

HABITAT ISSUES

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.

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for this stock 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; 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

Fraser's 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. No fishery-related mortality or serious injury has been observed during 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 western U.S. Atlantic EEZ relative to OSP is unknown. There are insufficient data to determine the population trends for this species.

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CLYMENE DOLPHIN (*Stenella clymene*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The clymene dolphin is endemic to tropical and sub-tropical waters of the Atlantic (Jefferson and Curry 2003). Clymene dolphins have been commonly sighted in the Gulf of Mexico since 1990 (Mullin *et al.* 1994; Fertl *et al.* 2003). Sightings of this species in the western North Atlantic along the U.S. East Coast are rare; there have been only ten survey sightings since 1995. These sightings, plus stranding records (Fertl *et al.* 2003), indicate that this species routinely occurs in the western North Atlantic. Nara *et al.* (2017) analyzed mitochondrial DNA sequence data from 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. Given the paucity of sightings, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. However, those sightings do encompass multiple marine ecoregions (Spalding *et al.* 2007), and include Cape Hatteras, a known biogeographic break for other marine species, so it is possible that multiple demographically independent populations of *S. clymene* exist within this stock. Additional morphological, acoustic, genetic and/or behavioral data are needed to further delineate population structure in this region.

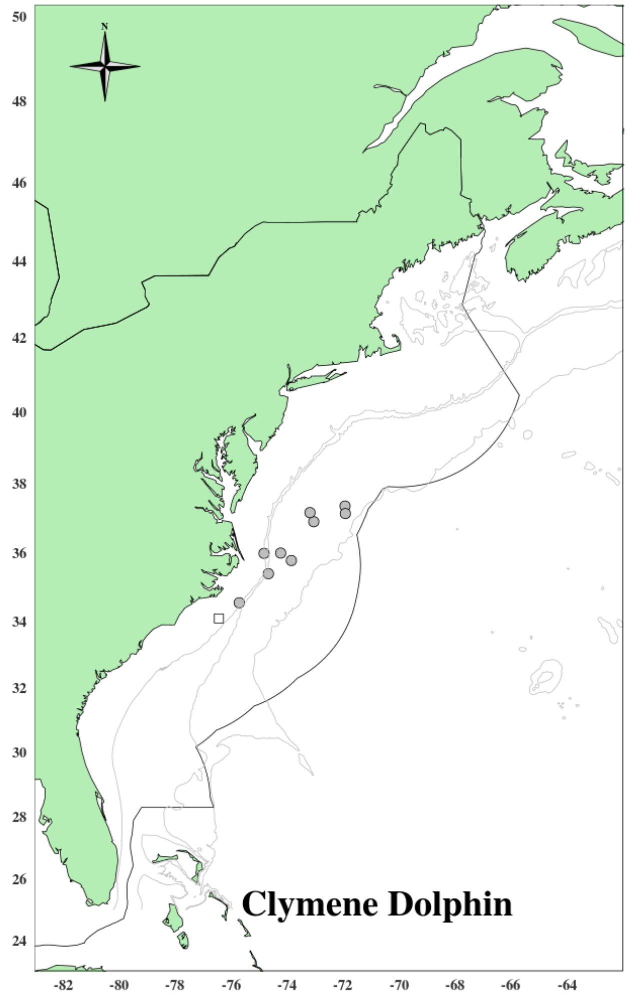

Figure 1. Distribution of Clymene dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011, and 2016. Isobaths are the 200m, 1,000m and 4,000m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The best abundance estimate available for clymene dolphins in the western North Atlantic is 4,237 (CV=1.03; Garrison 2020; Palka 2020). This estimate is from summer 2016 surveys covering waters from central Florida to the lower Bay of Fundy, and is the first estimate since a survey conducted in summer of 1998 (Mullin and Fulling 2003). Clymene dolphins were not sighted during surveys of the U.S. Atlantic coast conducted in the summers of 2004 and 2011.

Abundance estimates of 0 and 4,237 (CV=1.03) clymene dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (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 U.S. EEZ (NEFSC and SEFSC 2018). No clymene dolphins were observed during this survey. Clymene dolphins were observed in the second vessel survey, which covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and 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; Garrison 2020; Palka 2020). 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.

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 abundance estimate is 4,237 (CV=1.03). The minimum population estimates based on the 2016 abundance estimates is 2,071.

Current Population Trend

Clymene dolphins are rarely sighted during abundance surveys, and the resulting estimates of abundance are both highly variable between years and highly uncertain. The rare encounter rates limit the ability to assess or interpret trends in population size.

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 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 clymene dolphin is 2,071. 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 set to 0.5 because this stock is of unknown status. PBR for the western North Atlantic stock of clymene dolphins is 21.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to clymene dolphins in the western North Atlantic.

Fishery Information

The commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean are the Category I Atlantic Highly Migratory Species longline and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively.

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of clymene dolphins within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to clymene dolphins by this fishery in the Atlantic Ocean during 2013–2017 (Garrison and Stokes 2014; 2016; 2017; 2019; 2020).

Other Mortality

One stranding of a clymene dolphin was reported for the U.S. Atlantic Ocean during 2013–2017 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER)). This animal stranded in New Jersey in 2013. No evidence of human interaction was detected for this stranding.

There may be some uncertainty in the identification of this species due to similarities with other *Stenella* species. Stranding data probably 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for this stock 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; Pinsky *et al*. 2013; Poloczanska *et al.* 2013; 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

Clymene 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. 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 U.S. EEZ relative to OSP is unknown. There are insufficient data to determine population trends for this stock.

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SPINNER DOLPHIN (*Stenella longirostris longirostris*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Spinner dolphins are distributed in tropical oceanic and coastal waters worldwide (Leatherwood *et al.* 1976). The species is found in offshore, deep-waters (Schmidly 1981; Perrin and Gilpatrick 1994) but island associated populations are documented in the Pacific (Karczmarski *et al.* 2005; Andrews *et al.* 2010) and the Indian Ocean (Oremus *et al.* 2007; Viricel *et al.* 2016) where they often use shallower waters for resting during the day. Restricted levels of gene flow have been documented among some island populations (Oremus *et al.* 2007; Viricel *et al.* 2016) and among pelagic populations in eastern tropical Pacific (Leslie and Morin 2016). The species' distribution in the western North Atlantic is very poorly known. Spinner dolphin sightings have occurred almost exclusively in deeper (>2,000 m) oceanic waters (CETAP 1982; Waring *et al.* 1992) off the northeast U.S. coast. There was one sighting during summer 2011 in oceanic waters off North Carolina, and two additional sightings during summer 2016 in oceanic waters off Virginia (Figure 1). They are more commonly sighted in the Gulf of Mexico than the western North Atlantic. Stranding records exist from North Carolina, South Carolina, Florida, and Puerto Rico in the Atlantic, and in Texas, Louisiana, Alabama, and Florida in the Gulf of Mexico.

Spinner dolphins in the western North Atlantic are managed separately from those in the northern Gulf of Mexico. 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). Due to the paucity of sightings, there are insufficient data to determine whether the western North Atlantic stock comprises multiple demographically independent populations. Additional morphological, acoustic, genetic, and/or behavioral data are needed to further delineate population structure within the western North Atlantic and across the broader geographic area.

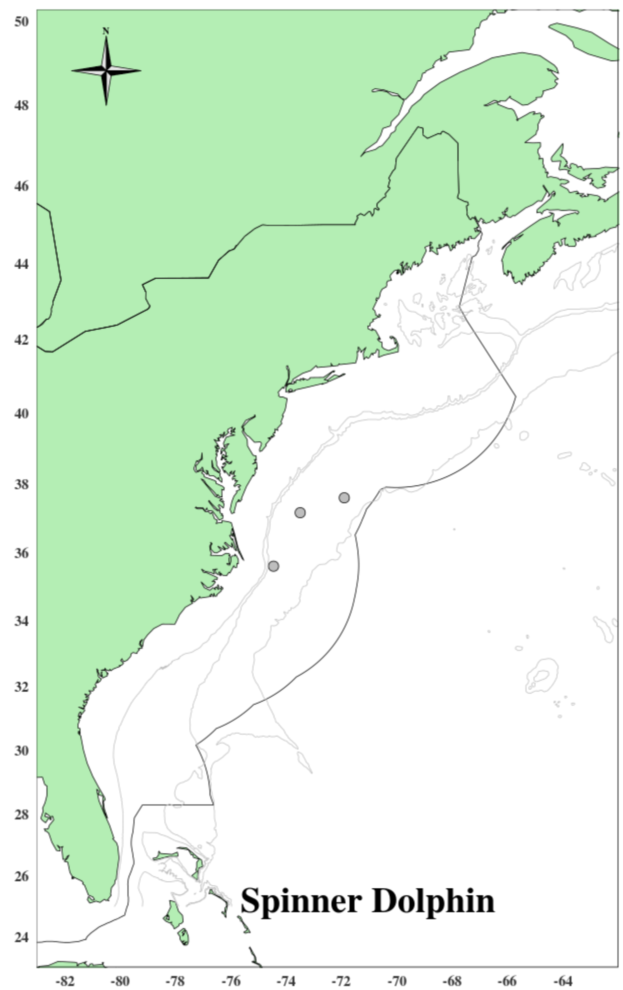

Figure 1. Distribution of spinner dolphin sightings from NEFSC and SEFSC shipboard (circles) and aerial (squares) surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 200m, 1,000m and 4,000m depth contours. The darker line indicates the U.S. EEZ.

POPULATION SIZE

The best abundance estimate available for spinner dolphins in the western North Atlantic is 4,102 (CV=0.99; Garrison 2020; Palka 2020). This estimate is from summer 2016 surveys covering waters from central Florida to the lower Bay of Fundy. The number of spinner dolphins off the U.S. Atlantic coast has not previously been estimated because there have only been three sightings during recent NMFS surveys.

Recent surveys and abundance estimates

Abundance estimates of 160 (CV=0; based on a single sighting) and 3,942 (CV=1.03) spinner dolphins were generated from vessel surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (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 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 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.

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 spinner dolphins is 4,102 (CV=0.99). The minimum population estimate for spinner dolphins is 2,045.

Current Population Trend

There are insufficient data to determine the population trends for this stock because only one estimate of population size is available.

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 unknown. 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 this stock is of unknown status. PBR for the western North Atlantic spinner dolphin is 20

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated fishery-related mortality and serious injury to this stock during 2013–2017 was presumed to be zero, as there were no reports of mortalities or serious injuries to spinner dolphins in the western North Atlantic.

Fishery Information

The commercial fisheries that interact, or that could potentially interact, with this stock in the Atlantic Ocean are the Category I Atlantic Highly Migratory Species longline and Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fisheries (Appendix III). Percent observer coverage (percentage of sets observed) for these longline fisheries for each year during 2013–2017 was 9, 10, 12, 15, and 12, respectively.

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of spinner dolphins within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. There were no observed mortalities or serious injuries to spinner dolphins by this fishery in the Atlantic Ocean during 2013–2017 (Garrison and Stokes 2014; 2016; 2017; 2019; 2020).

Other Mortality

During 2013–2017, two spinner dolphins were reported stranded on the U.S. East Coast, both occurring in Florida (one in 2016, one in 2017) (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018 (SER) and 8 June 2018 (NER)). It could not be determined whether there was evidence of human interaction for one of the strandings, and for the other, no evidence of human interaction was detected. Stranding data probably 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, shelf and slope stocks in the western North Atlantic are less likely to strand than nearshore coastal stocks. 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.

HABITAT ISSUES

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.

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for this stock 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; 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

Spinner 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. 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 spinner dolphins in the U.S. western North Atlantic EEZ relative to OSP is unknown. There are insufficient data to determine the population trends for this species.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Western North Atlantic Offshore Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

There are two morphologically and genetically distinct forms of common bottlenose dolphin (Duffield *et al.* 1983; Mead and Potter 1995; Rosel *et al.* 2009) described as the coastal and offshore forms in the western North Atlantic (Hersh and Duffield 1990; Mead and Potter 1995; Curry and Smith 1997; Rosel *et al.* 2009). The two morphotypes are genetically distinct based upon both mitochondrial and nuclear markers (Hoelzel *et al.* 1998; Rosel *et al.* 2009). The offshore form is distributed primarily along the outer continental shelf and continental slope in the Northwest Atlantic Ocean from Georges Bank to the Florida Keys (Figure 1; CETAP 1982; Kenney 1990), where dolphins with characteristics of the offshore type have stranded. However, common bottlenose dolphins have occasionally been sighted in Canadian waters, on the Scotian Shelf (e.g., Baird *et al.* 1993; Gowans and Whitehead 1995), and these animals are thought to be of the offshore form.

North of Cape Hatteras, there is separation of the two morphotypes across bathymetry during summer months. Aerial surveys flown during 1979–1981 indicated a concentration of common bottlenose dolphins in waters < 25 m deep corresponding to the coastal morphotype, and an area of high abundance along the shelf break corresponding to the offshore stock (CETAP 1982; Kenney 1990). Biopsy tissue sampling and genetic analysis demonstrated that common bottlenose dolphins concentrated close to shore were of the coastal morphotype, while those in waters > 25 m depth were from the offshore morphotype (Garrison *et al.* 2003). However, south of Cape Hatteras, North Carolina, the ranges of the coastal and offshore morphotypes overlap to some degree. Torres *et al.* (2003) found a statistically significant break in the distribution of the morphotypes at 34 km from shore based upon the genetic analysis of tissue samples collected in nearshore and offshore waters from New York to central Florida. The offshore morphotype was found exclusively seaward of 34 km and in waters deeper than 34 m. Within 7.5 km of shore, all animals were of the coastal morphotype. More recently, offshore morphotype animals have been sampled as close as 7.3 km from shore in water depths of 13 m (Garrison *et al.* 2003). Systematic biopsy collection surveys were conducted coast-wide during the summer and winter between 2001 and

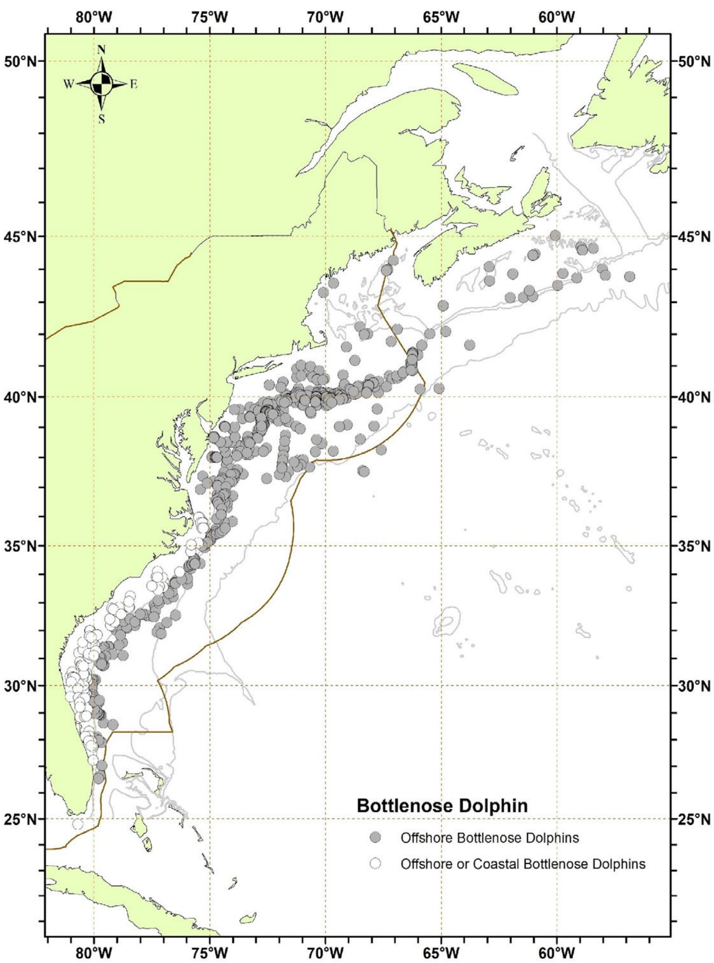

Figure 1. Distribution of bottlenose dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016. Isobaths are the 200m, 1,000m, and 4,000m depth contours. The darker line indicates the U.S. EEZ.

2005 to evaluate the degree of spatial overlap between the two morphotypes. Over the continental shelf south of Cape Hatteras, North Carolina, the two morphotypes overlap spatially, and the probability of a sampled group being from the offshore morphotype increased with increasing depth based upon a logistic regression analysis (Garrison *et al*. 2003). Hersh and Duffield (1990) examined common bottlenose dolphins that stranded along the southeast coast of Florida and found four that had hemoglobin profiles matching that of the offshore morphotype. These strandings suggest the offshore form occurs as far south as southern Florida. The range of the offshore common bottlenose dolphin includes waters beyond the continental slope (Kenney 1990), and also waters beyond the U.S. EEZ, and therefore the offshore stock is a transboundary stock (Figure 1). Offshore common bottlenose dolphins may move between the Gulf of Mexico and the Atlantic (Wells *et al.* 1999).

The western North Atlantic Offshore Stock of common bottlenose dolphins is managed separately from the Gulf of Mexico Oceanic Stock of common bottlenose dolphins. One line of evidence to support this separation 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. In addition, the western North Atlantic and Gulf of Mexico belong to distinct marine ecoregions (Spalding *et al.* 2007). Restricted genetic exchange has been documented among offshore populations in the Gulf of Mexico (Vollmer and Rosel 2016) but analyses to determine whether multiple demographically independent populations exist within the western North Atlantic have not been performed to date.

POPULATION SIZE

The best available estimate for the offshore stock of common bottlenose dolphins in the western North Atlantic is 62,851 (CV=0.23; Table 1; Garrison 2020; Palka 2020). This estimate is from summer 2016 surveys covering waters from central Florida to the lower Bay of Fundy.

Earlier abundance estimates

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

Recent surveys and abundance estimates

An abundance estimate of 26,766 (CV=0.52) offshore common bottlenose dolphins was generated from aerial and shipboard surveys conducted during June–August 2011 between central Virginia and the lower Bay of Fundy (Palka 2012). The aerial portion covered 6,850 km of trackline over waters north of New Jersey between the coastline and the 100-m depth contour through the U.S. and Canadian Gulf of Maine, and up to and including the lower Bay of Fundy. The shipboard portion covered 3,811 km of trackline between central Virginia and Massachusetts in waters deeper than the 100-m depth contour out to beyond the U.S. EEZ. Both sighting platforms used a double-platform data-collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

An abundance estimate of 50,766 (CV=0.55) offshore common bottlenose dolphins was generated from a shipboard survey conducted concurrently (June–August 2011) in waters between central Virginia and central Florida (Garrison 2016). This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed two independent visual teams searching with 25x150 “bigeye” binoculars. A total of 4,445 km of trackline was surveyed, yielding 290 cetacean sightings. The majority of sightings occurred along the continental shelf break with generally lower sighting rates over the continental slope. Estimation of the abundance was based on the independent observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

Abundance estimates of 17,958 (CV=0.33; combined northeast vessel and aerial surveys) and 44,893 (CV=0.29; southeast vessel survey) offshore common bottlenose dolphins were generated from surveys conducted in U.S. waters of the western North Atlantic during the summer of 2016 (Table 1; Garrison 2020; Palka 2020). One vessel survey was conducted from 27 June to 25 August in waters north of 38ºN latitude and included 5,354 km of on-effort trackline along the shelf break and offshore to the U.S. EEZ (NEFSC and SEFSC 2018). A concomitant aerial portion was conducted from 14 August to 28 September and included 11,782 km of trackline that were over waters north of New

Jersey from the coastline to the 100-m depth contour, throughout the U.S. waters (NEFSC and SEFSC 2018). Estimates from these two surveys were combined to provide an abundance estimate for the area north of 38ºN. The second vessel survey covered waters from Central Florida to approximately 38ºN latitude between the 100-m isobaths and 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). All 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 offshore stock of common bottlenose dolphins* (Tursiops truncatus truncatus) *by month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jun–Aug 2011	central Virginia to lower Bay of Fundy	26,766	0.52
Jun–Aug 2011	central Florida to central Virginia	50,766	0.55
Jun–Aug 2011	central Florida to lower Bay of Fundy (COMBINED)	77,532	0.40
Jun–Aug 2016	New Jersey to lower Bay of Fundy	17,958	0.33
Jun–Aug 2016	central Florida to New Jersey	44,893	0.29
Jun–Aug 2016	central Florida to lower Bay of Fundy (COMBINED)	62,851	0.23

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 abundance estimate is 62,851 (CV=0.23). The minimum population estimate for western North Atlantic offshore common bottlenose dolphin is 51,914.

Current Population Trend

There are three available coastwide abundance estimates for offshore common bottlenose dolphins from the summers of 2004, 2011, and 2016. Each of these is derived from surveys with similar survey designs and all three used the two-team independent observer approach to estimate abundance. The resulting estimates were 54,739 (CV=0.24) in 2004, 77,532 (CV=0.40) in 2011, and 62,851 (CV=0.23) in 2016 (Garrison 2020; Palka 2020). A generalized linear model did not indicate a statistically significant (p=0.646) trend in these estimates. The high level of uncertainty in these estimates limits the ability to detect a statistically significant trend. A key uncertainty in this assessment of trend is that interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

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 for offshore common bottlenose dolphins is 51,914. 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 and the CV of the average mortality estimate is less than 0.3 (Wade and Angliss 1997). PBR for the western North Atlantic offshore common bottlenose dolphin is therefore 519.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The estimated mean annual fishery-related mortality and serious injury of offshore common bottlenose dolphins during 2013–2017 was 28 (CV=0.34; Table 2) due to interactions with the northeast sink gillnet, northeast bottom trawl, and mid-Atlantic bottom trawl fisheries.

Fisheries Information

There are seven commercial fisheries that interact, or that potentially could interact, with this stock in the Atlantic Ocean. These include four Category I fisheries (Atlantic Highly Migratory Species longline; Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline; mid-Atlantic gillnet; and northeast sink gillnet), two Category II fisheries (northeast bottom trawl and mid-Atlantic bottom trawl), and the Category III Gulf of Maine, U.S. mid-Atlantic tuna, shark, swordfish hook and line/harpoon fishery. Detailed fishery information is reported in Appendix III.

No interactions have been documented in recent years for the mid-Atlantic gillnet or the U.S. mid-Atlantic tuna, shark, swordfish hook and line/harpoon fishery. See Appendix V for information on historical takes.

Longline

The Atlantic Highly Migratory Species longline fishery operates outside the U.S. EEZ. No takes of common bottlenose dolphins within high seas waters of the Atlantic Ocean have been observed or reported thus far.

The large pelagics longline fishery operates in the U.S. Atlantic (including Caribbean) and Gulf of Mexico EEZ, and pelagic swordfish, tunas and billfish are the target species. During 2013–2017, there were no observed mortalities or serious injuries of common bottlenose dolphins of the offshore stock by this fishery (Garrison and Stokes 2014; 2016; 2017; 2019; 2020). Historically, takes of the offshore stock have been observed occasionally, and the most recent observed take occurred in 2012. During 2013 (2 animals), 2015 (1), and 2017 (1), a total of 4 common bottlenose dolphins were observed entangled and released alive in the Mid-Atlantic Bight and Northeast Coastal regions (Garrison and Stokes 2014; 2016; 2017; 2019; in rpress). These animals were presumed to have no serious injuries.

See Table 2 for observer coverage for the current 5-year period, and Appendix V for historical estimates of annual mortality and serious injury.

Table 2. Summary of the incidental mortality and serious injury of Atlantic Ocean offshore common bottlenose dolphins* (Tursiops truncatus truncatus) *by commercial fishery including the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortalities and serious injuries recorded by on-board observers, the estimated annual mortality and serious injury, the combined annual estimates of mortality and serious injury (Estimated Combined Mortality), the estimated CV of the combined estimates (Estimated CVs) and the mean of the combined estimates (CV in parentheses).

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
Pelagic Longline	2013 2014 2015 2016 2017	Obs. Data Logbook	.09 .10 .12 .15 .12	0 0 0 0 0	0 0 0 0 0	0 0 0 0 0	0 0 0 0 0	0 0 0 0 0	NA NA NA NA NA	0
Northeast Sink Gillnet	2013 2014 2015 2016 2017	Obs. Data Logbook	.11 .18 .14 .10 .12	0 0 0 0 0	1 0 0 0 1	0 0 0 0 0	26 0 0 0 8	26 0 0 0 8	0.95 NA NA NA .92	7 (0.76)
Northeast Bottom Trawl[c]	2013 2014 2015 2016 2017	Obs. Data Logbook	.15 .17 .19 .12 .16	0 0 0 0 0	0 0 3 4 0	0 0 0 0 0	0 0 18.6 33.5 0	0 0 18.6 33.5 0	NA NA 0.65 0.89 NA	10.4 (0.62)
Mid-Atlantic Bottom Trawl[c]	2013 2014 2015 2016 2017	Obs. Data Logbook	.06 .08 .09 .10 .10	0 0 0 0 0	0 3 0 1 3	0 0 0 0 0	0 25 0 7.3 22.1	0 25 0 7.3 22.1	NA 0.66 NA 0.93 0.66	10.9 (0.42)

Fishery	Years	Data Type[a]	Observer Coverage[b]	Observed Serious Injury	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Est. CVs	Mean Annual Mortality
TOTAL	2013–2017	-	-	-	-	-	-	-	-	28 (0.34)

[a] 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).
[b] Proportion of sets observed (for Pelagic Longline).
[c] Fishery related bycatch rates for 2013–2017 were estimated using an annual stratified ratio-estimator following the methodology described in Lyssikatos (2020).

Northeast Sink Gillnet

During 2013–2017, two mortalities were observed (in 2013 and 2017) in the northeast sink gillnet fishery (Hatch and Orphanides 2015, 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020). No takes were observed during 2014–2016. There were no observed injuries of common bottlenose dolphins in the Northeast region during 2013–2017 to assess using new serious injury criteria. See Table 2 for bycatch estimates and observed mortality and serious injury for the current five-year period, and Appendix V for historical bycatch information.

Northeast Bottom Trawl

During 2013–2017, seven mortalities were observed in the northeast bottom trawl fishery (Lyssikatos *et al.* 2020). There were no observed injuries of common bottlenose dolphins in the northeast region during 2013–2017 to assess using new serious injury criteria. See Table 2 for bycatch estimates and observed mortality and serious injury for the current five-year period, and Appendix V for historical bycatch information.

Through the Marine Mammal Authorization Program (MMAP) during 2013–2017, there were four self-reported incidental takes (mortalities) of common bottlenose dolphins off Rhode Island—two in 2014 (single incident involving two animals) and two in 2016. Fishers were trawling for *Illex* and *Loligo* squid.

Mid-Atlantic Bottom Trawl

During 2013–2017, four mortalities were observed in the mid-Atlantic bottom trawl fishery (Lyssikatos *et al.* 2020). There were no observed injuries of common bottlenose dolphins in the mid-Atlantic region during 2013–2017 to assess using new serious injury criteria. See Table 2 for bycatch estimates and observed mortality and serious injury for the current five-year period, and Appendix V for historical bycatch information.

Through the Marine Mammal Authorization Program (MMAP) during 2013–2017, there were three self-reported incidental takes (mortalities) of common bottlenose dolphins off Rhode Island by fishers targeting squid/mackerel/butterfish. All three takes occurred during 2015, and two of those occurred in a single trawling incident.

Other Mortality

Common bottlenose dolphins are among the most frequently stranded small cetaceans along the Atlantic coast. Many of the animals show signs of human interaction (*i.e.*, net marks, mutilation, etc.); however, it is unclear what proportion of these stranded animals is from the offshore stock because most strandings are not identified to morphotype, and when they are, animals of the offshore form are uncommon. For example, only 19 of 185 *Tursiops* strandings in North Carolina were genetically assigned to the offshore form (Byrd *et al.* 2014).

An Unusual Mortality Event (UME) of bottlenose dolphins and other cetaceans occurred along the mid-Atlantic coast from New York to Brevard County, Florida, from 1 July 2013 to 1 March 2015. A total of 1,872 stranded common bottlenose dolphins were recovered in the UME area which stretched from New York to Brevard County, Florida. Morbillivirus was determined to be a primary cause of the event (Morris *et al.* 2015). An assessment of the impacts of the 2013–2015 UME on common bottlenose dolphin stocks in the western North Atlantic is ongoing.

HABITAT ISSUES

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.

Offshore wind development in the U.S. Atlantic may also pose a threat to this stock, particularly south of Cape Hatteras where it comes closer to shore. Activities associated with development include geophysical and geotechnical surveys, installation of foundations and cables, and operation, maintenance and decommissioning of facilities (BOEM 2018). The greatest threat from these activities is likely underwater noise, however other potential threats include vessel collision due to increased vessel traffic, benthic habitat loss, entanglement due to increased fishing around structures, marine debris, dredging, and contamination/degradation of habitat (BOEM 2018).

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., Schwacke *et al.* 2002; Jepson *et al.* 2016; Hall *et al.* 2018), but research on contaminant levels for the offshore stock of bottlenose dolphins 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; 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

The common bottlenose dolphin in the western North Atlantic is not listed as threatened or endangered under the Endangered Species Act, and the offshore stock is not considered strategic under the MMPA. 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 the zero mortality and serious injury rate. The status of this stock relative to OSP in the U.S. Atlantic EEZ is unknown. There was no statistically significant trend in population size for this species; however, the high level of uncertainty in the estimates limits the ability to detect a statistically significant trend.

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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, 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).

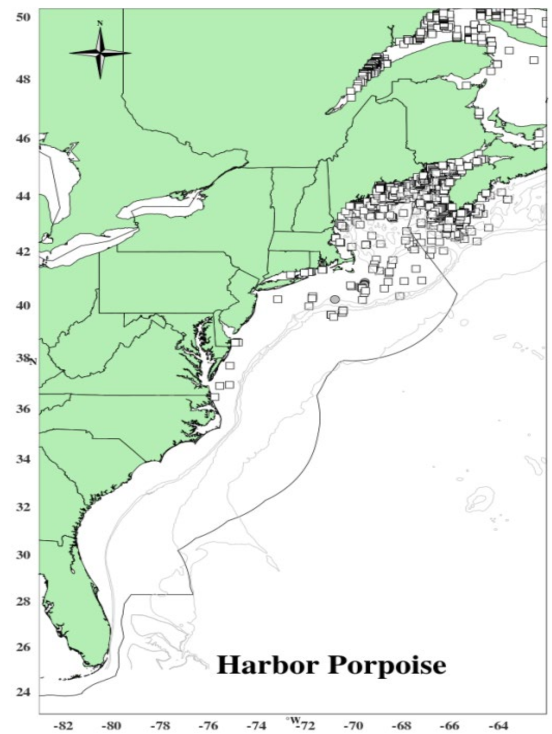

Figure 1. Distribution of harbor porpoises from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, 2011 and 2016 and portions of DFO's 2007 TNASS and 2016 NAISS surveys. Isobaths are the 100m, 200m, 1000m, and 4000m depth contours.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

An abundance estimate of 79,883 (CV=0.32) harbor porpoises was generated from a shipboard and aerial survey conducted during June–August 2011 (Palka 2012). The aerial portion that contributed to the abundance estimate covered 5,313 km of tracklines that were over waters north of New Jersey from the coastline to the 100-m depth contour through the U.S. and Canadian Gulf of Maine and up to and including the lower Bay of Fundy. The shipboard portion covered 3,107 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 double-platform team data-collection procedure, which allows estimation of abundance corrected for perception bias of the detected species (Laake and Borchers 2004). Estimation of the abundance was based on the independent-observer approach assuming point independence (Laake and Borchers 2004) and calculated using the mark-recapture distance sampling option in the computer program Distance (version 6.0, release 2, Thomas et al. 2009).

No harbor porpoises were detected in an abundance survey that was conducted concurrently (June-August 2011) in waters between central Virginia and central Florida. This shipboard survey included shelf-break and inner continental slope waters deeper than the 50-m depth contour within the U.S. EEZ. The survey employed the double-platform methodology searching with 25x150 "bigeye" binoculars. A total of 4,445 km of tracklines was surveyed, yielding 290 cetacean sightings.

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 (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 (Nbest) and coefficient of variation (CV).

Month/Year	Area	Nbest	CV
Jul–Aug 2011	Central Virginia to lower Bay of Fundy	79,883	0.32
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.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual estimated average human-caused mortality and serious injury is 217 harbor porpoises per year (CV=0.15) from U.S. fisheries using observer data. Canadian bycatch information is not available.

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 of the Gulf of Maine/Bay of Fundy population.

Fishery Information

Detailed U.S. fishery information is reported in Appendix III.

Earlier Interactions

See Appendix V for more information on historical takes.

U.S.

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 2015, 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020). See Table 2 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 2015, 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020). See Table 2 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 2 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 2. 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	2013 2014 2015 2016 2017	Obs. Data, Trip Logbook, Allocated Dealer Data	0.11 0.18 0.14 0.10 0.12	0 0 0 0 1	20 28 23 11 18	0 0 0 0 7	399 128 177 125 129	399 128 177 125 136	0.33 0.27 0.28 0.34 0.28	193 (0.16)
Mid-Atlantic Gillnet	2013 2014 2015 2016 2017	Obs. Data, Weighout	0.03 0.05 0.06 0.08 0.09	0 0 2 0 0	1 1 2 2 1	0 0 27 0 0	19 22 33 23 9	19 22 60 23 9	1.06 1.03 1.16 0.64 0.95	21 (0.49)
Northeast Bottom Trawl	2013 2014 2015 2016 2017	Obs. Data, Weighout	0.15 0.17 0.19 0.12 0.16	0 0 0 0 0	1 1 4 0 0	0 0 0 0 0	7.0 5.5 3.7 0 0	7.0 5.5 3.7 0 0	0.98 0.86 0.49 0 0	3.2 (0.53)
TOTAL	-	-	-	-	-	-	-	-	-	217 (0.15)

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 2013–2017 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2019).

Other Mortality

U.S.

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. Further articles describing use of porpoise products for food and other purposes were timed to coincide with ongoing legal action in state court.

Recent harbor porpoise strandings on the U.S. Atlantic coast are documented in Table 3 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018).

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.

Table 3. Harbor porpoise (Phocoena phocoena phocoena) reported strandings along the U.S. and Canadian Atlantic coast, 2013-2017.

Area	2013	2014	2015	2016	2017	Total
Maine[a, b, c, f]	7	5	2	5	8	27

Area	2013	2014	2015	2016	2017	Total
New Hampshire	1	1	0	1	2	5
Massachusetts[a, b, c, f]	40	22	18	8	29	117
Rhode Island [d, e]	3	0	2	2	0	7
Connecticut [b]	1	0	0	0	0	1
New York[a, b, f]	15	1	3	1	12	32
New Jersey [b, c, f]	8	4	2	5	14	33
Delaware	2	0	0	0	6	8
Maryland	3	0	0	0	2	5
Virginia[c, e]	15	3	3	2	5	28
North Carolina[d]	7	11	14	1	1	34
TOTAL U.S.	102	39	44	25	79	297
Nova Scotia/Prince Edward Island[g]	21	9	13	16	22	81
Newfoundland and New Brunswick[h]	3	0	2	0	0	5
GRAND TOTAL	126	48	59	16	101	383

a. Three Massachusetts live strandings were taken to rehab in 2013 and 1 Maine animal was released alive. 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. Ten total HI cases in 2013 (MA-3, ME-2, NY-3, NJ-1, CT-1), including one released alive (ME). Three of these were considered fishery interactions, including one entangled in gear in Maine.
c. 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.
d. Two HI cases in 2015: 1 in Rhode Island and 1 in North Carolina
e. Two HI cases in 2016: 1 in Rhode Island and 1 in Virginia. The Virginia case was coded as a fishery interaction.
f. 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.
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 2013, 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 22013, 2014, 2015, 2017, 2018; Table 3).

HABITAT ISSUES

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 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 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.

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HARBOR SEAL (*Phoca vitulina vitulina*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The harbor seal (*Phoca vitulina vitulina*) is found 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).

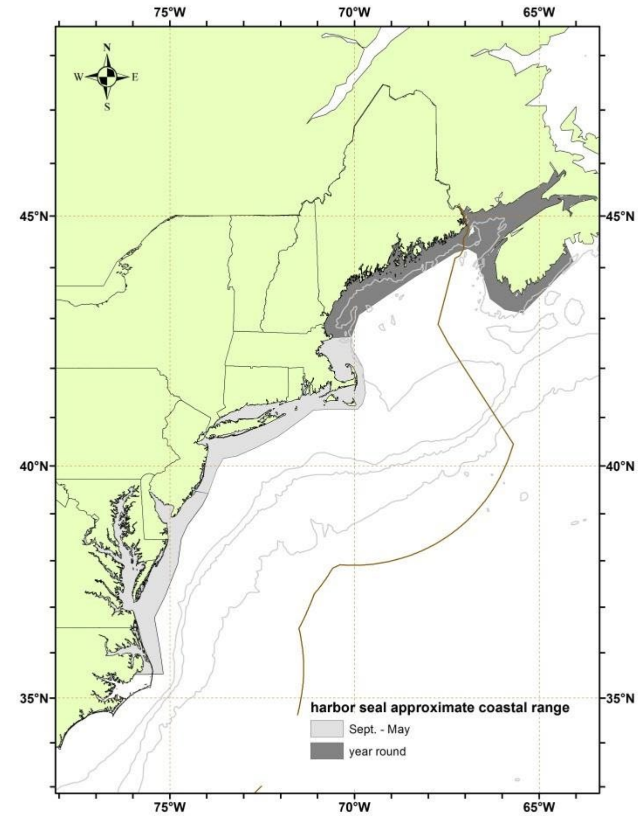

Figure 1. Approximate coastal range of harbor seals. Isobaths are the 100-m, 1000-m, and 4000-m depth contours.

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 9 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 in the process of 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_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	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 has not been possible 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).

Although the 2012 population estimate was lower than the previous estimate of 99,340 obtained from a survey in 2001 (Gilbert *et al.* 2005), Waring *et al.* (2015) did not consider the population to be declining because the two estimates were not significantly different and there was uncertainty over whether some fraction of the population was not in the survey area. This was due to the fact that 31.4% of the count was pups, a percentage that is biologically unlikely. The estimated number of harbor seal pups did not differ significantly between 2001 and 2012. In 2001, there were an estimated 23,722 (CV=0.096) pups in the study area (Gilbert *et al.* 2005); in 2012 there were an estimated 23,830 (CV=0.159) pups in the study area. Therefore some non-pups in the population may not have been available to be counted because they were outside the study area of Coastal Maine. Some seals could have remained farther south in New England, more northerly in Canada, or offshore. Therefore, a decline in the apparent abundance of harbor seals could be explained by changing distributions and/or different survey coverage over time. Other lines of evidence provide support for an apparent decline in abundance and/or changing distributions. 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.

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.

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2013-2017 the total human caused mortality and serious injury to harbor seals is estimated to be 350 per year. The average was derived from two components: 1) 338 (CV=0.12; Table 2) from 2013–2017 observed fisheries; 2) 12 from 2013–2017 non-fishery-related, human interaction stranding mortalities (NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018, and 3) 0.2 from U.S. research mortalities.

Analysis of bycatch rates from fisheries observer program records likely underestimates lethal (Lyle and Willcox 2008), and greatly under-represents sub-lethal, fishery interactions. Reports of seal shootings and other non-fishery-related human interactions are minimums.

Fishery Information

Detailed fishery information is given in Appendix III.

U.S.

Northeast Sink Gillnet:

Harbor seal bycatch is observed year-round, most frequently in the summer in groundfish trips occurring between Boston, Massachusetts, and Maine in coastal Gulf of Maine waters. See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information. Analysis methodology and results can be found in Orphanides (2019, 2020), Hatch and Orphanides (2015, 2016), Orphanides and Hatch (2017), and Josephson *et al.* (2019).

Mid-Atlantic Gillnet

Harbor seal bycatch has been observed in this fishery in waters off Massachusetts and New Jersey and rarely further south. See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information. Analysis methodology and results can be found in Orphanides (2019, 2020), Hatch and Orphanides (2015, 2016), and Orphanides and Hatch (2017).

Northeast Bottom Trawl

Harbor seals are occasionally observed taken in this fishery. See Table 2 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information. Analysis methodology and results can be found in (Lyssikatos *et al*. 2020).

Mid-Atlantic Bottom Trawl

Harbor seals are rarely observed taken in this fishery. Annual harbor seal mortalities were estimated using annual stratified ratio-estimator methods (Lyssikatos *et al*. 2020). See Table 2 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 Fishery (Including Pair Trawl)

Harbor seals are occasionally observed taken in this fishery. An extended bycatch rate has not been calculated for the current 5-year period. Until this bycatch estimate can be developed, the average annual fishery-related mortality and serious injury for 2013–2017 is calculated as 0.8 animals (4 animals/5 years). See Table 2 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. This fishery was not observed until 2003. No mortalities have been observed, but 1 harbor seal was captured and released alive in 2013, and 0 in 2014– 2017. In addition, 0 seals of unknown species were captured and released alive in 2013–2014, 2 in 2015, 1 in 2016, and 0 in 2017. None of the seals captured alive in herring purse seine during 2013-2017 were designated as serious injuries (Josephson *et al.* 2019).

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 2. Summary of the incidental mortality of harbor seals (Phoca vitulina vitulina) by commercial fishery including the years sampled (Years), the number of vessels active within the fishery (Vessels), 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	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Annual Mortality
Northeast Sink Gillnet		Obs. Data, Weighout, Logbooks		0		0				311 (0.13)
	2013		0.11	0	22	0	142	142	0.31	
	2014		0.18	0	59	0	390	390	0.39	
	2015		0.14	0	87	0	474	474	0.17	
	2016		0.10	0	36	0	245	245	0.29	
	2017		0.12		63		298	298	0.18	
Mid-Atlantic Gillnet		Obs. Data, Weighout		0		0				18 (0.41)
	2013		0.03	0	0	0	0	0	0	
	2014		0.05	0	1	0	19	19	1.06	
	2015		0.06	0	5	0	48	48	0.52	
	2016		0.08	0	2	0	18	18	0.95	
	2017		0.09		1		3	3	0.62	
Northeast Bottom Trawl		Obs. Data, Weighout		0		0				3 (.52)
	2013		0.15	0	1	0	4	4	0.89	
	2014		0.17	0	2	0	11	11	0.63	
	2015		0.19	0	0	0	0	0	0	
	2016		0.12	0	0	0	0	0	0	
	2017		0.16		0		0	0	0	
Mid-Atlantic Bottom Trawl		Obs. Data, Dealer		0		0				5.6 (.56)
	2013		0.06	0	1	0	11	11	0.96	
	2014		0.08	0	2	0	10	10	0.95	
	2015		0.09	0	1	0	7	7	1	
	2016		0.10	0	0	0	0	0	0	
	2017		.10		0		0	0	0	
Northeast Mid-water Trawl - Including Pair Trawl		Obs. Data, Weighout, Trip Logbook		0		0				0.8 (na)
	2013		0.37	0	0	0	0	0	0	
	2014		0.42	0	1	0	na	na	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	
TOTAL	-	-	-	-	-	-	-	-	-	338 (0.12)

a. Observer data (Obs. Data) are used to measure bycatch rates, and the data are collected within the Northeast Fisheries Observer Program. 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 2013-2017 includes samples collected from traditional fisheries observers in addition to fishery monitors through the Northeast Fisheries Observer Program (NEFOP).
c. Serious injuries were evaluated for the 2013–2017 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2019)

Other Mortality

U.S.

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. From 2013 to 2017, 1,214 harbor seal stranding mortalities were reported between Maine and Florida (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018). Seventy (5.8%) of the dead harbor seals stranded during this five-year period showed signs of human interaction (15 in 2013, 11 in 2014, 18 in 2015, 16 in 2016, and 10 in 2017), with 10 (0.8%) having some sign of fishery interaction (3 in 2013, 2 in 2014, 2 in 2015, 3 in 2016, and 1 in 2017). Three harbor seals during this period were reported as having been shot. Seven harbor seal mortalities were reported with indications of vessel strike. 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).

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 3. Harbor seal (Phoca vitulina vitulina) stranding mortalities along the U.S. Atlantic coast (2013-2017) with subtotals of animals recorded as pups in parentheses.

State	2013	2014	2015	2016	2017	Total
Maine	99 (74)	127 (94)	73 (47)	76 (58)	120 (84)	495 (357)
New Hampshire	16 (6)	38 (22)	56 (43)	45 (27)	26 (20)	181 (118)
Massachusetts	95 (39)	58 (15)	81 (24)	55 (19)	78 (29)	367 (126)
Rhode Island	9 (3)	7 (1)	8 (0)	5 (1)	9 (3)	38 (8)
Connecticut	2 (1)	0	2 (1)	1 (0)	2 (0)	7 (2)
New York	11 (2)	13 (4)	21 (0)	1 (0)	11 (0)	57 (6)
New Jersey	4 (0)	2 (1)	9 (4)	4 (0)	9 (3)	28 (8)
Delaware	0	3 (0)	1 (0)	1 (1)	1 (0)	6 (1)

State	2013	2014	2015	2016	2017	Total
Maryland	1 (0)	2 (0)	0	0	1 (0)	4 (0)
Virginia	5 (0)	2 (0)	1 (0)	1 (0)	2 (0)	11 (0)
North Carolina	3 (0)	3 (1)	5 (2)	4 (2)	4 (4)	19 (9)
South Carolina	0	1 (0)	0	0	0	1 (0)
Total	**245**	**256**	**257**	**193**	**263**	**1214 (635)**
Unspecified seals (all states)	25	38	31	13	86	193

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 2013–2017 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.

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GRAY SEAL (*Halichoerus grypus atlantica*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The gray seal (*Halichoerus grypus atlantica*) 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 which defines the western North Atlantic stock ranges from New Jersey to Labrador (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) that have overlapping distributions outside the breeding period (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).

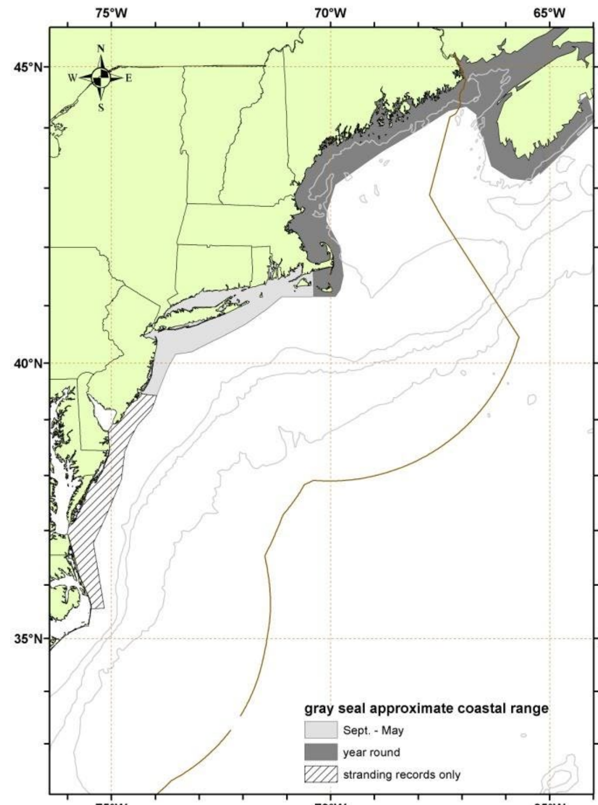

Figure 1. Approximate coastal range of gray seals. Isobaths are the 100-m, 1000-m, and 4000-m depth contours.

In the mid-1980s, small numbers of animals and pupping 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 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 is 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 western 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 to 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).

A minimum of 6,308 of pups were born in 2016 at U.S. breeding colonies, approximately 6% of the total pup production over the entire range of the stock (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. Table 2 summarizes single-day pup counts from U.S. pupping colonies from 2001/2002 to 2015/2016 pupping periods. Aerial survey data from these sites indicate that pup production is increasing (Table 2), although aerial survey quality and coverage has varied significantly among surveys. In U.S. waters, gray seals primarily pup at four established colonies: Muskeget and Monomoy islands in Massachusetts, and Green and Seal islands in Maine. Gray seals have been observed using the historic pupping site on Muskeget Island in Massachusetts since 1988. Pupping has taken place on Seal and Green Islands in Maine since at least the mid-1990s. Since 2010 pupping has also been observed at Noman's Island in Massachusetts and Wooden Ball and Matinicus Rock in Maine. Although white-coated pups have stranded on eastern Long Island beaches in New York, no pupping colonies have been detected in that region.

Using Canadian population models, 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 best population size to pups in Canadian waters (4.3:1). 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). Using this approach, the population estimate in U.S. waters is 27,131 (CV=0.19, 95% CI: 18,768–39,221) animals. The CV and CI around this estimate is 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), and an estimated 28,000–40,000 gray seals in this region in 2015 using correction factors applied to seal counts visible in 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 and 95% confidence interval.

Month/Year	Area	N_{best} [a]	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.

Table 2. Single day pup counts from five U.S. pupping colonies during 2001-2016 from aerial surveys. * = Surveys need further evaluation before reporting. As single day pup counts, these counts do not represent the entire number of pups born in a pupping season.

	Massachusetts			Maine			
Pupping Season	Muskeget Island	Monomoy Island	Nomans Island	Seal Island	Green Island	Wooden Ball	Matinicus Rock
2001-02	883	Not surveyed	Not surveyed	No data	34	Not surveyed	Not surveyed
2002-03	509	Not surveyed	Not surveyed	147	No data	Not surveyed	Not surveyed
2003-04	824	Not surveyed	Not surveyed	150	26	Not surveyed	Not surveyed
2004-05	992	1	Not surveyed	365	33	Not surveyed	Not surveyed
2005-06	868	8	Not surveyed	239	43	Not surveyed	Not surveyed
2006-07	1,704	9	Not surveyed	364	57	Not surveyed	Not surveyed
2007-08	2,095	2	Not surveyed	466	59	Not surveyed	Not surveyed
2008-09	1,104	68	0	*	48	Not surveyed	Not surveyed
2009-10	1,841	154	0	*	51	Not surveyed	Not surveyed
2010-11	3,173	325	1	*	65	Not surveyed	112
2011-12	2,831	80	8	*	41	2	57
2012-13	2,750	633	4	*	Not surveyed	Not surveyed	CIP
2013-14	3,073	507	16	*	30	Not surveyed	201
2014-15	1,633	768	23	*	33	185	182
2015-16	3,787	935	32	1,043	34	284	193

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 of 27,131 (CV=0.19), the minimum population estimate in U.S. waters is 23,158. 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 across all U.S. pupping colonies from 1991-2016 is currently being evaluated. More data on movements of animals between Canada and the U.S. is needed – particularly the number of adult breeding females recruiting into U.S. colonies each year – to separate out intrinsic rates of increase from the overall annual growth rate.

The population in eastern Canada was greatly reduced by hunting and bounty programs, and in the 1950s the gray seal was considered rare (Lesage and Hammill 2001). The Sable Island, Nova Scotia, population was less affected and has been increasing for several decades. Pup production on Sable Island increased exponentially at a rate of 12.8% per year between the 1970s and 1997 (Stobo and Zwanenburg 1990; Mohn and Bowen 1996; Bowen *et al.* 2003; Trzcinski *et al.* 2005; Bowen *et al.* 2007; DFO 2011). 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 been slower (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).

The projected population trends for all Canadian aggregations are still increasing. The model projections in 2016 differed from previous analyses due to changes in adult sex ratio and adult mortality rates (DFO 2017). Uncertainties in the population abundance estimates and mortality could have impacts on the abundance trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. Recent studies estimated the current annual rate of increase at 4.5% for the combined breeding aggregations in Canada (DFO 2014), continuing a decline in the rate of increase (Trzcinski *et al*. 2005; Bowen *et al*. 2007; Thomas *et al*. 2011; DFO 2014). 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).

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 stock in U.S. waters is 23,158. 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 in U.S. waters is 1,389 animals. Uncertainty in the PBR level arises from the same sources of uncertainty in calculating a minimum abundance estimate in U.S. waters.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2013–2017, the average annual estimated human-caused mortality and serious injury to gray seals in the U.S. and Canada was 5,410 (946 U.S./4,464 Canada) per year. The average was derived from six components: 1) 940 (CV=0.09) (Table 3) from the 2013–2017 U.S. observed fisheries; 2) 5.6 from average 2013–2017 non-fishery related, human interaction stranding and shooting mortalities in the U.S.; 3) 0.8 from U.S. research mortalities; 4) 672 from the average 2013–2017 Canadian commercial harvest*;* 5) 55 from the average 2013–2017 DFO scientific collections; and 6) 3,737 removals of nuisance animals in Canada (DFO 2017, Mike Hammill pers. comm).

A source of unquantified human-caused mortality or serious injury for this stock is the fact that 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.* 2019); 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. Canadian reporting of nuisance seal removal is known to be incomplete and there is also limited information on Canadian fishery bycatch (DFO 2017).

Fishery Information

Detailed fishery information is given in Appendix III.

U.S.

Northeast Sink Gillnet

Gray seal bycatch in the northeast sink gillnet fishery was usually observed in the first half of the year in waters to the east and south of Cape Cod, Massachusetts in 12-inch gillnets fishing for skates and monkfish (Hatch and Orphanides 2015, 2016, Orphanides and Hatch 2017; Orphanides 2019*,* 2020). See Table 3 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

Gray seal interactions were first observed in this fishery in 2010, since then, when they are observed, it is usually in waters off New Jersey in gillnets that have mesh sizes ≥ 7 in (Hatch and Orphanides 2015, 2016; Orphanides and Hatch 2017; Orphanides 2019, 2020). See Table 3 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

One gray seal mortality was observed in 2013 in this fishery. An expanded bycatch estimate has not been generated. Until this bycatch estimate can be developed, the average annual fishery-related mortality and serious injury for 2013–2017 is calculated as 0.2 animals (1 animal /5 years). See Table 3 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. This fishery was not observed until 2003, and was not observed in 2006. No mortalities have been observed, but during this time period 1 gray seals was captured and released alive in 2013, 2 in 2014, 0 in 2015, 5 in 2016 and 0 in 2017 In addition, during this time period 2 seals of unknown species were captured and released alive in 2015 and 1 in 2016 (Josephson *et al.* 2019).

Northeast Bottom Trawl

Vessels in the North Atlantic bottom trawl fishery, a Category III fishery under MMPA, were observed in order to meet fishery management, rather than marine mammal management needs. Five gray seal mortalities were observed in this fishery in 2013, 4 in 2014, 4 in 2015, 0 in 2016 and 2 in 2017 (Lyssikatos *et al.* 2020). See Table 3 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

Two gray seal mortalities were observed in this fishery in 2013, 1 in 2014, none in 2015, 3 in 2016 and 5 in 2017 (Lyssikatos *et al.* 2020). See Table 3 for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

CANADA

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 3. Summary of the incidental serious injury and mortality of gray seal (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 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	2013 2014 2015 2016 2017	Obs. Data,Weighout, Trip Logbook	0.11 0.18 0.14 0.10 0.12	0 0 0 0 0	69 159 131 43 158	0 0 0 0 0	1127 917 1021 498 930	1127 917 1021 498 930	0.20 0.14 0.25 0.33 0.16	899 (0.09)
Mid-Atlantic Gillnet	2013 2014 2015 2016 2017	Obs. Data, Trip Logbook, Allocated Dealer Data	0.03 0.05 0.06 0.08 0.09	0 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	9 (0.67)
Northeast Bottom Trawl	2013 2014 2015 2016 2017	Obs. Data,Trip Logbook	0.15 0.17 0.19 0.12 0.16	0 0 0 0 0	5 4 4 0 2	0 0 0 0 0	20 19 23 0 16	20 19 23 0 16	0.37 0.45 0.46 0 0.24	16 (0.20)
Mid-Atlantic Bottom Trawl	2013 2014 2015 2016 2017	Obs. Data,Trip Logbook	0.06 0.08 0.09 0.10 0.10	0 0 0 0 0	2 1 0 3 5	0 0 0 0 0	25 7 0 26 26	25 7 0 26 26	0.67 0.96 0 0.57 0.40	17 (0.30)
Northeast Mid-water Trawl – Incl.Pair Trawl	2013 2014 2015 2016 2017	Obs. Data,Trip Logbook	0.37 0.42 0.08 0.27 0.16	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 0 0 0	0.2 (na) [d]
TOTAL	-	-	-	-	-	-	-	-	-	940 (0.09)

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 2013–2017 period (Josephson *et al.* 2019)

Other Mortality

U.S.

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.

From 2013 to 2017, 603 gray seal stranding mortalities were recorded, extending from Maine to North Carolina (Table 4; NOAA National Marine Mammal Health and Stranding Response Database, accessed 23 October 2018). Most stranding mortalities were in Massachusetts, which is the center of gray seal abundance in U.S. waters. Sixtythree (10%) of the total stranding mortalities showed signs of human interaction (17 in 2013, 8 in 2014, 20 in 2015, 1 in 2016 and 17 in 2017), 35 of which had some indication of fishery interaction (9 in 2013, 2 in 2014, 14 in 2015, 0 in 2016 and 10 in 2017). One gray seal is recorded in the stranding database during the 2013 to 2017 period as having been shot—in Maine in 2015. Another gray seal mortality due to shooting in Maine in 2016 was prosecuted by NOAA law enforcement. 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 2013 and 2017, the average annual human-caused mortality and serious injury to gray seals in Canadian waters from commercial harvest was 672 per year though more are permitted (up to 60,000 seals/year, see http://www.dfo-mpo.gc.ca/decisions/fm-2015-gp/atl-001-eng.htm). This included: 243 in 2013, 82 in 2014,1,381 in 2015, 1,588 in 2016, and 64 in 2017 (DFO 2017, Mike Hammill pers. comm.). In addition, between 2013 and 2017, an average of 3,737 nuisance animals per year were killed. This included, 3,757 in 2013, and 3,732 annually in 2014–2017 (DFO 2017). Nuisance animals in 2017 were not available as of March 2019, so the average number of nuisance animals from 2014-2016 were used for 2017. Lastly, DFO took 58 animals in 2013, 83 animals in 2014, 42 animals in 2015,30 animals in 2016, and 60 animals in 2017 for scientific collections, for an annual average of 55 animals (DFO 2017, Mike Hammill pers. comm).

Table 4. Gray seal (Halichoerus grypus atlantica) stranding mortalities along the U.S. Atlantic coast (2013-2017) with subtotals of animals recorded as pups in parentheses.

State	2013	2014	2015	2016	2017	Total
ME	9 (4)	3 (1)	5	6(0)	14 (1)	37
NH	1 (0)	3 (2)	2	0	3 (0)	9
MA	82 (8)	62 (6)	77 (3)	54(0)	135 (21)	410
RI	11 (2)	8 (1)	7 (1)	4(0)	16 (5)	46
NY	18 (5)	12 (4)	10	1 (1)	57 (0)	57
NJ	7 (2)	7 (6)	7 (6)	3 (1)	4 (3)	28
DE	0	3 (3)	3 (3)	0	1 (0)	7
MD	0	1 (0)	0	0	0	1
VA	0	0	3	0	0	3
NC	0	2 (2)	0	0	0	2
Total	128 (21)	101 (25)	114	68 (2)	192 (30)	603
Unspecified seals (all states)	25	38	31	13	86	193

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 U.S. portion of 2013–2017 average annual human-caused mortality and serious injury in U.S. waters does not exceed the portion of PBR in U.S. waters. 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 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.

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HARP SEAL (*Pagophilus groenlandicus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The harp seal occurs throughout much of the North Atlantic and Arctic Oceans (Ronald and Healey 1981; Lavigne and Kovacs 1988). The world's harp seal population is divided into three separate stocks, each identified with a specific pupping site on the pack ice (Lavigne and Kovacs 1988; Bonner 1990). The largest stock is located off eastern Canada and is divided into two breeding herds. The Front herd breeds off the coast of Newfoundland and Labrador, and the Gulf herd breeds near the Magdalen Islands in the middle of the Gulf of St. Lawrence (Sergeant 1965; Lavigne and Kovacs 1988). The second stock breeds on the West Ice off eastern Greenland (Lavigne and Kovacs 1988), and the third stock breeds on the ice in the White Sea off the coast of Russia. The Front/Gulf stock is equivalent to the western North Atlantic stock. Perry *et al.* (2000) found no significant genetic differentiation between the two Northwest Atlantic whelping areas, though the authors pointed out some uncertainty surrounding that finding due to small sample sizes.

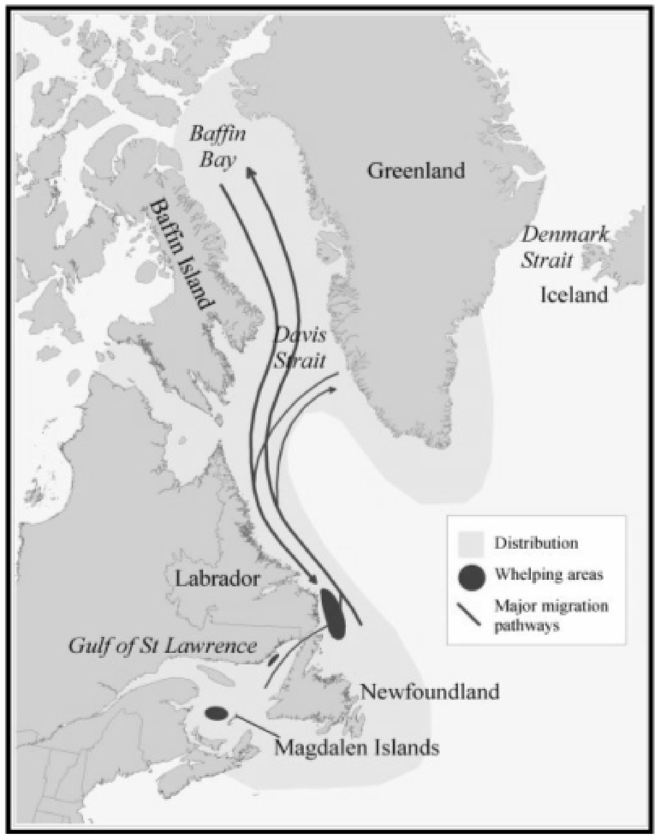

Figure 1: From: Technical Briefing on the Harp Seal Hunt in Atlantic Canada

http://www.dfo-po.gc.ca/misc/seal_briefing_e.htm

Harp seals are highly migratory (Sergeant 1965; Stenson and Sjare 1997). Breeding occurs at different times for each stock between late-February and April. Adults then assemble on suitable pack ice to undergo the annual molt. The migration then continues north to Arctic summer feeding grounds. In late September, after a summer of feeding, nearly all adults and some of the immature animals of the western North Atlantic stock migrate southward along the Labrador coast, usually reaching the entrance to the Gulf of St. Lawrence by early winter. There they split into two groups, one moving into the Gulf and the other remaining off the coast of Newfoundland. The southern limit of the harp seal's habitat extends into the U.S. Atlantic Exclusive Economic Zone (EEZ) during winter and spring.

Since the early 1990s, numbers of sightings and strandings have been increasing off the east coast of the United States from Maine to New Jersey (Katona *et al.* 1993; Rubinstein 1994; Stevick and Fernald 1998; McAlpine 1999; Lacoste and Stenson 2000; Soulen *et al*. 2013). These appearances usually occur in January-May (Harris *et al.* 2002), when the western North Atlantic stock of harp seals is at its most southern point of migration. Concomitantly, a southward shift in winter distribution off Newfoundland was observed during the mid-1990s, which was attributed to abnormal environmental conditions (Lacoste and Stenson 2000).

POPULATION SIZE

Abundance estimates for the western North Atlantic stock are available which use a variety of methods including aerial surveys and mark-recapture (Table 1). These methods involve surveying the whelping concentrations and estimating total population adult numbers from pup production. Roff and Bowen (1983) developed an estimation model to provide a more precise estimate of total abundance. This technique incorporates recent pregnancy rates and estimates of age-specific hunting mortality (CAFSAC 1992). This model has subsequently been updated in Shelton *et al.* (1992, 1996), Stenson (1993), Warren *et al.* (1997), and Hammill and Stenson (2011) to consider struck and loss animals, mortality related to poor ice conditions, and variable reproductive rates. A population model was used to examine changes in the size of the population from 1952-2014 (Hammill *et al.* 2014). The model was fit to 12 estimates of pup production from 1952 to 2012, and to annual estimates of age-specific pregnancy rates between 1954 and 2013. Total population size in 2012 was estimated to be 7,445,000 (95% CI: 6.1 to 8.8 million), and projected to be 7,411,000 (95% CI: 6.1 to 8.7 million) in 2014. The population appears to be relatively stable (Hammill *et al.* 2015), though pup production has become highly variable among years (Stenson *et al.* 2014). A pup survey conducted in March 2017 will provide updated abundance estimates.

Uncertainties not accounted for include variations in reproductive rates as well as changes in mortality due to varying ice conditions.

Table 1. Summary of abundance estimates for western North Atlantic harp seals in Canadian waters. Year and area covered during each abundance survey, resulting abundance estimate (Nbest) and confidence interval (CI).

Month/Year	Area	N_{best}	CI
2012	Front and Gulf	7.4 million	(95% CI 6.1–8.8 million)
2014[a]	Front and Gulf	7.4 million	(95% CI 6.1–8.7 million)

[a] The 2014 abundance estimate is based on model projections from the 2012 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 western North Atlantic harp seals, based on the last 2012 survey, is 7.4 million (CV=0.09, 95% CI 6.1-8.8 million; Hammill *et al.* 2014). The minimum population is 6.9 million. Data are insufficient to calculate the minimum population estimate for U.S. waters due to low sighting rates.

Current population trend

Harp seal pup production in the 1950s was estimated at 645,000, but had decreased to 225,000 by 1970 (Sergeant 1975). Estimated production then began to increase and continued to increase through the late 1990s, reaching 998,000 (CV=0.10) in 1999 (Stenson *et al.* 2003). Estimated pup production in 2008 was 1,630,300 (CV=6.8%), but decreased to 790,000 (SE=69,700, CV=8.8%) in 2012 (Stenson *et al.* 2014). This estimate is approximately half of the estimated number of pups born in 2008, likely due to lower reproductive rates in 2012 compared to 2008 (Stenson *et al.* 2014). Uncertainties in fecundity rates as well as uncertainties in ice conditions (which could impact harp seals' body condition and breeding success) have potentially large impacts on population trends.

The status of the population in U.S. waters is unknown. Recent increases in strandings may not be indicative of population size.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock due to limited understanding of stock specific life history parameters in U.S. waters. Therefore, 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).

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 in U.S. waters is unknown. The maximum productivity rate is 0.12, the default value for pinnipeds. The recovery factor, which accounts for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population (OSP) was set at 1.0 the population is increasing. PBR for the western North Atlantic harp seal in U.S. waters is unknown. The PBR for the stock in U.S. waters is unknown.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2013–2017 the total estimated annual human caused mortality and serious injury to harp seals was 232,422. This is derived from three components: 1) 65 harp seals (CV=0.21) from the observed U.S. fisheries (Table 2a); 2) an average of 2 stranded seals from 2013-2017 that showed signs of non-fishing human interaction; and 3) an average catch of 232,355 seals from 2013-2017 by Canada and Greenland, including bycatch in the lumpfish fishery (Table 2b). Uncertainties in bycatch estimates are small compared to the magnitude of commercial and subsistence harvest in Canada. A potential source of unquantified human-caused mortality is the mortality associated with poor ice conditions due to climate change.

Fishery Information

U.S.

Detailed fishery information is reported in the Appendix III.

Northeast Sink Gillnet:

During 2013–2017, 34 mortalities were observed in the northeast sink gillnet fishery (Hatch and Orphanides 2014; 2015; 2016, Orphanides 2019, 2020). There were no observed injuries of harp seals in the Northeast region during 2013–2017 to assess using new serious injury criteria.

See Table 2a for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Table 2a. Summary of the incidental mortality of harp seal (Pagophilus groenlandicus) 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	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Annual Mortality
				0		0				
Northeast Sink Gillnet	2013	Obs. Data, Weighout, Logbooks	0.11	0	2	0	22	22	0.75	65 (0.21)
	2014		0.18	0	9	0	57	57	0.42	
	2015		0.14	0	12	0	119	119	0.34	
	2016		0.10	0	5	0	85	85	0.50	
	2017		0.12		6		44	44	0.37	
TOTAL	-	-	-	-	-	-	-	-	-	65 (0.21)

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 sink gillnet fishery.
b. The observer coverages for the Northeast sink gillnet fishery and the mid-Atlantic coastal sink gillnet fisheries are ratios based on tons of fish landed. North Atlantic bottom trawl fishery coverages are ratios based on trips.
c. Serious injuries were evaluated for the 2013–2017 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2019).

Other Mortality

U.S.

From 2013 to 2017, 194 harp seal stranding mortalities were reported (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 23 October 2018). Eleven (5.6%) of the mortalities during this five-year period showed signs of human interaction (2 in 2013, 4 in 2014, 2 in 2015, 1 in 2016 and 2 in 2017), 1 of which with some sign of fishery interaction (2013). Harris and Gupta (2006) analyzed NMFS 1996-2002 stranding data and suggested that the distribution of harp seal strandings in the Gulf of Maine was consistent with the species' seasonal migratory patterns in this region.

CANADA

Harp seals have been commercially hunted since the mid-1800s in the Canadian Atlantic (Stenson 1993). Between 2003 and 2010 the harp seal total allowable catch (TAC) in Canada ranged from 270,000 to 330,000 (ICES 2016). In 2011 the TAC was raised to 400,000 and since then, has remained at this level each year. The TAC includes allocations for aboriginal harvesters (6,840), development of new products (20,000), and personal use (2,000). There is no specific allocation or quotas for catches in Arctic Canada. Commercial catches in Canada have remained below 80,000 since 2009 (Table 2b).

Table 2b. Summary of the Canadian directed catch and bycatch mortality of Northwest Atlantic harp seal (Pagophilus groenlandicus) by year.

Fishery	**2013**	**2014**	**2015**	**2016**	**2017[f]**	**Average**
Commercial catches[a]	90,703	54,830	35,304	66,865	66,865	**62,913**
Struck and lost[b]	86,970	66,946	81,609	83,268	83,268	**75,699**
Greenland subsistence catch[c]	80,102	62,147	78,749	78,749	78,749	**75,699**
Canadian Arctic[d]	1,000	1,000	1,000	1,000	1,000	**1,000**
Newfoundland lumpfish[e]	12,330	12,330	12,330	12,330	12,330	**12,330**
Total	**271,105**	**197,253**	**208,992**	**242,212**	**242,212**	**232,355**

a. ICES 2016
b. Animals that are killed but not recovered and reported. Values include seals from both Canada and Greenland (ICES 2016).
c. ICES 2016. Catches in 2015 and 2016 are an average from 2005-2014
d. ICES 2016.
e. Estimates of bycatch levels in the last decade are not available and so the average annual level during the previous decade (12,330) has been assumed (DFO 2014)
f. 2017 statistics are not available. 2016 numbers are reported for 2016 and 2017.

Table 3. Harp seal (Pagophilus groenlandicus) stranding mortalities [a] along the U.S. Atlantic coast (2013–2017) with subtotals of animals recorded as pups in parentheses.

State	**2013**	**2014**	**2015**	**2016**	**2017**	**Total**
Maine	2	2 (1)	1	4	3	12
New Hampshire	1	0	0	2	0	3
Massachusetts	6 (1)	28	17	19 (1)	13 (1)	83
Rhode Island	1	9	4	3	4	21
Connecticut	0	0	0	1	1	2
New York	9	18	12	1	7	47
New Jersey	2	1	3	1	0	7
Delaware	1	0	0	0	0	1
Maryland	0	0	1	0	0	2
Virginia	1	9	4	1	1	6

State	2013	2014	2015	2016	2017	Total
North Carolina	2	1	2	2 (1)	2 (1)	9
Total	**23**	**68**	**44**	**34**	**32**	**194**
Unspecified seals (all states)	25	38	31	13	86	193

a. Mortalities include animals found dead and animals that were euthanized, died during handling, or died in the transfer to, or upon arrival at, rehab facilities.

STATUS OF STOCK

Harp 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 level of human-caused mortality and serious injury in the U.S. Atlantic EEZ is low relative to the total stock size. The status of the harp seal stock, relative to OSP, in the U.S. Atlantic EEZ is unknown, but the stock's abundance appears to have stabilized. The total U.S. fishery-related mortality and serious injury for this stock is very low relative to the stock size and can be considered insignificant and approaching zero mortality and serious injury rate. Based on the low levels of uncertainties described in the above sections, it expected these uncertainties will have little effect on the status of this stock.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*)
West Bay Stock

NOTE – NMFS is in the process of writing individual stock assessment reports for each of the 31 bay, sound and estuary stocks of common bottlenose dolphins in the Gulf of Mexico.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are distributed throughout the bays, sounds, and estuaries (BSE) of the Gulf of Mexico (Mullin 1988). Long-term (year-round, multi-year) residency by at least some individuals has been reported from nearly every estuarine site where photographic identification (photo-ID) or tagging studies have been conducted in the Gulf of Mexico (e.g., Irvine and Wells 1972; Shane 1977; Gruber 1981; Irvine et al. 1981; Wells 1986; Wells et al. 1987; Scott et al. 1990; Shane 1990; Wells 1991; Bräger 1993; Bräger et al. 1994; Fertl 1994; Wells et al. 1996a,b; Wells et al. 1997; Weller 1998; Maze and Würsig 1999; Lynn and Würsig 2002; Wells 2003; Hubard et al. 2004; Irwin and Würsig 2004; Shane 2004; Balmer et al. 2008; Urian et al. 2009; Bassos-Hull et al. 2013; Wells et al. 2017; Balmer et al. 2018). In many cases, residents occur predominantly within estuarine waters, with limited movements through passes to the Gulf of Mexico (Shane 1977; Gruber 1981; Irvine et al. 1981; Shane 1990; Maze and Würsig 1999; Lynn and Würsig 2002; Fazioli et al. 2006; Bassos-Hull et al. 2013; Wells et al. 2017). Genetic data also support the presence of discrete BSE stocks (Duffield and Wells 2002; Sellas et al. 2005). Sellas et al. (2005) examined population subdivision among dolphins sampled in Sarasota Bay, Tampa Bay, and Charlotte Harbor, Florida; Matagorda Bay, Texas; and the coastal Gulf of Mexico (1–12 km offshore) from just outside Tampa Bay to the south end of Lemon Bay, and found evidence of significant population differentiation among all areas on the basis of both mitochondrial DNA control region sequence data and nine nuclear microsatellite loci. Genetic data also indicate restricted genetic exchange between and demographic independence of BSE populations and those occurring in adjacent Gulf coastal waters (Sellas et al. 2005; Rosel et al. 2017). Differences in reproductive seasonality from site to site also suggest genetic-based distinctions among areas (Urian et al. 1996). Photo-ID and genetic data from several inshore areas of the southeastern United States also support the existence of resident estuarine animals and differentiation between animals biopsied along the Atlantic coast and those biopsied within estuarine systems at the same latitude (Caldwell 2001; Gubbins 2002; Zolman 2002; Mazzoil et al. 2005; Litz 2007; Rosel et al. 2009).

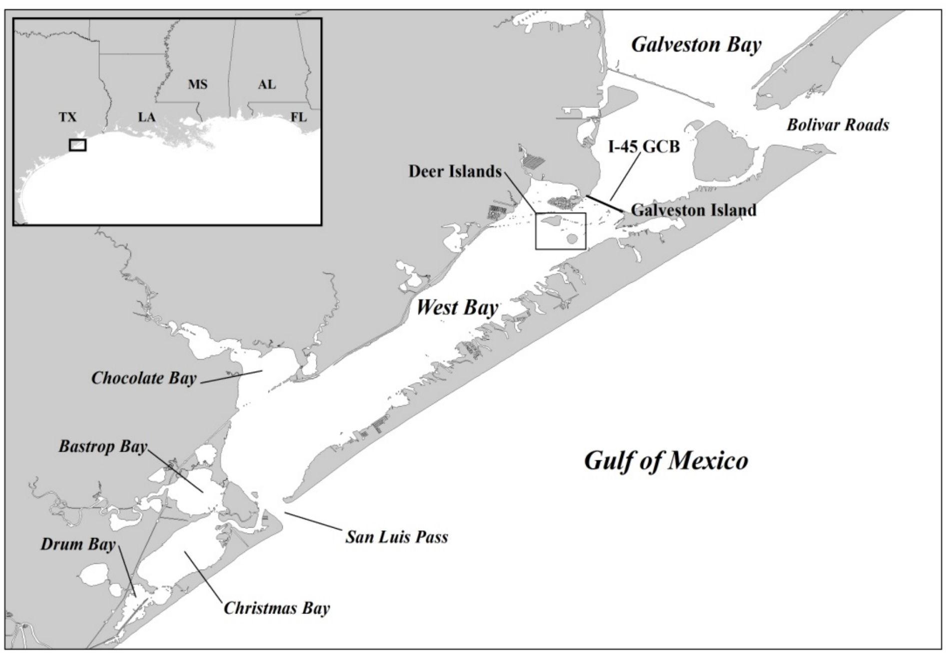

Figure 1. Geographic extent of the West Bay Stock, located within the Galveston Bay Estuary in Texas. I-45 GCB = I-45 Galveston Causeway Bridge.

West Bay, a bay within the Galveston Bay Estuary system, encompasses an area of approximately 180 km2, and is a narrow, long bay averaging 1.2 m in depth (Diener 1975; Phillips and Rosel 2014; Figure 1). It tends to be more saline than Galveston Bay, with an average salinity of 15 to 32 ppt (Pulich and White 1991; Phillips and Rosel 2014). West Bay is separated from the Gulf of Mexico by Galveston Island, and connected to the Gulf via San Luis Pass in the southwest, and connected to Galveston Bay in the northeast via Bolivar Roads. The Galveston Bay Estuary has been selected as an estuary of national significance by the Environmental Protection Agency National Estuary Program (see http://www.gbep.state.tx.us/). Thus, a comprehensive conservation and management plan has been developed and is being implemented through a partnership of local, state, and federal representatives as well as community stakeholders, to restore and protect the estuary (Lester and Gonzalez 2011).

The West Bay Stock was delimited in the first stock assessment reports published in 1995 (Blaylock et al. 1995) and common bottlenose dolphins are present within the bay. The stock boundaries extend from Drum Bay in the southwest to the I-45 Galveston Causeway Bridge in the northeast and includes West Bay, Chocolate Bay, Bastrop Bay, Christmas Bay, Drum Bay, and San Luis Pass (Figure 1). However, Bastrop Bay, Christmas Bay, and Drum Bay are very shallow areas, and dolphins were not sighted there during recent exploratory surveys (Ronje et al. 2018). The area between the Deer Islands and the I-45 Galveston Causeway Bridge is being included in the West Bay Stock due to sightings of two animals that were also seen in southern West Bay (Litz et al. 2019), but this area may serve as a transition zone between the Galveston Bay/East Bay/Trinity Bay Stock and the West Bay Stock. Additional research may result in a revision to the northeastern boundary. Dolphins of this stock also are seen in nearshore coastal waters adjacent to San Luis Pass, where they may be exposed to additional threats. However, the extent to which they use these waters and whether there may be significant seasonality to that usage is unknown. To date, coastal waters approximately 3 km north and south of San Luis Pass and within 1 km of shore are included in the stock area. This coastal range is based on sightings data from a 2014–2015 photo-ID mark recapture survey (see Population Size). The range in coastal waters may be revised as new studies are conducted. Given the small size and relatively homogeneous habitat of West Bay, it is unlikely this stock contains multiple demographically independent populations, but a directed investigation of this question has never been conducted.

POPULATION SIZE

The best available abundance estimate for the West Bay Stock of common bottlenose dolphins is 48 (CV=0.03; 95% CI: 45–50), which is the result of vessel-based capture-recapture photo-ID surveys conducted during winter 2014 and summer 2015 (Litz *et al.* 2019).

Earlier abundance estimates (>8 years old)

Boat-based photo-ID surveys in 1995 and 1996 conducted in southwestern West Bay, Chocolate Bay, San Luis Pass (SLP) and adjacent Gulf coastal waters outside SLP identified 28 year-round residents that utilized the bays, SLP, and nearshore coastal waters adjacent to SLP. During the summer dolphins were most frequently sighted furthest inland, mainly in Chocolate Bay, whereas during winter, sightings were concentrated near San Luis Pass and adjacent Gulf of Mexico coastal waters. In addition to resident animals, transient animals were sighted in Gulf coastal waters only (Maze and Würsig 1999). Additional boat-based surveys were conducted within the same area during 1997–2001 by Irwin and Würsig (2004) to compare three methods of assessing abundance: 1) counts based on photo-ID data; 2) capture-recapture analysis based on photo-ID data; and 3) line-transect surveys to estimate density using the program DISTANCE (Buckland *et al.* 1993). Photo-ID results based on counts yielded 34 resident animals displaying seasonal variation in their habitat use as described above. Capture-recapture analysis estimates of dolphin abundance in each year in warm months ranged from 28 (95% CI: 26–71) in 1998 to a high of 38 (95% CI: 33–55) in 2000. Line-transect density estimates ranged from 0.94 to 1.01 dolphins/km^2, with a warm-month abundance estimate of 108 dolphins (95% CI: 33-358). Irwin and Würsig (2004) suggested their density estimates were positively biased compared to estimates from other locations because the nonrandom distribution of dolphins in the study area makes the area unsuitable for line-transect surveys.

Recent surveys and abundance estimates

Photo-ID capture-recapture surveys were conducted in two seasons (December 2014 and June 2015) with three surveys per season (Ronje *et al.* 2018). The surveys covered the entirety of this stock’s range including West Bay, Chocolate Bay, and San Luis Pass. Christmas Bay was surveyed in the summer but not the winter; there were no sightings in this bay. In addition, two 20-km segments of trackline were surveyed in the coastal waters off San Luis Pass (1 km from shore and 2 km from shore) (Ronje *et al.* 2018). A Poisson-log normal Mark-Resight model was used to estimate abundances for each season (McClintock *et al.* 2009; Litz *et al.* 2019). Six coastal sightings presumed to contain coastal stock animals (primarily 1–2 sightings of each animal and only in coastal waters) were removed from the analyses (Litz *et al.* 2019). The abundance estimate for winter (December 2014) was 51 dolphins (CV=0.04; 95%

CI: 47–56) and the summer (June 2015) estimate was 44 dolphins (CV= 0.03; 95% CI: 43–47), and the mean of the estimates was 48 (CV=0.03; 95% CI: 45–50). The summer and winter estimates were averaged because there were no clear seasonal patterns in sighting distributions (Litz *et al.* 2019; Ronje *et al.* 2018). Capture probabilities were high for both seasons, and resighting data allowed for the exclusion of sightings of coastal stock animals from the abundance estimate. A key uncertainty is the possibility that coastal stock dolphins were present in estuarine waters and therefore could not be completely excluded from the abundance estimate.

Minimum Population Estimate

The minimum population estimate 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 this stock of common bottlenose dolphins is 48 (CV=0.03; 95% CI: 45–50). The minimum population estimate for the West Bay Stock is 46 common bottlenose dolphins.

Current Population Trend

A population trend analysis has not been conducted for this stock. Older abundance estimates exist but data need to be examined for comparability to the 2014–2015 estimate.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this 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 West Bay Stock of common bottlenose dolphins is 46. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because this stock is of unknown status. PBR for this stock of common bottlenose dolphins is 0.5.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the West Bay Stock of common bottlenose dolphins during 2013–2017 is unknown. The mean annual fishery-related mortality and serious injury during 2013–2017 based on strandings and at-sea observations identified as fishery-related was 0.2 (see Shrimp Trawl section for possible additional fishery-related mortality). Additional mean annual mortality and serious injury during 2013–2017 due to other human-caused sources was 0. The minimum total mean annual human-caused mortality and serious injury for this stock during 2013–2017 was therefore 0.2 (Table 1). This is a biased estimate 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, and 4) the estimate of fishery-related interactions includes an actual count of verified fishery-caused deaths and serious injuries and should be considered a minimum (NMFS 2016).

Fishery Information

There are three commercial fisheries that interact, or that potentially could interact, with this stock. These include one Category II fishery (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl) and two Category III fisheries (Gulf of Mexico blue crab trap/pot and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel [hook and line] fisheries). Detailed fishery information is presented in Appendix III.

Shrimp Trawl

Between 1997 and 2014, seven common bottlenose dolphins and seven unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the net, lazy line, turtle excluder device, or tickler chain gear in the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2016). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive without serious injury in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al.* (2015; 2016) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Observer program coverage did not extend into BSE waters, therefore time-area stratified bycatch rates were extrapolated into inshore waters to estimate a five-year unweighted mean mortality estimate for 2010–2014 based on inshore fishing effort (Soldevilla *et al.* 2016). Because the spatial resolution at which fishery effort is modeled is aggregated at the state level (e.g., Nance *et al.* 2008), the mortality estimate covers inshore waters of Texas from Galveston Bay, East Bay, Trinity Bay south to Laguna Madre. The mortality estimate for Texas BSE stocks for the years 2010–2014 was 0 (Soldevilla *et al.* 2016). Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015; 2016).

Blue Crab Trap/Pot

During 2013–2017 there were no documented interactions between commercial blue crab trap/pot gear and the West Bay Stock. There is no systematic observer coverage of crab trap/pot fisheries in the Gulf of Mexico, so it is not possible to quantify total mortality.

Hook and Line (Rod and Reel)

During 2013–2017, one interaction (mortality in 2014) with hook and line gear was documented in the stranding database for the West Bay Stock (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018; Table 2). Available evidence from the stranding data suggested the hook and line gear entanglement contributed to the cause of death, and this animal was included in the annual human-caused mortality and serious injury total for this stock (Table 1).

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., charter boat and 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 in the Gulf of Mexico. The documented interaction in this gear represents a minimum known count of interactions in the last five years.

Other Mortality

NOAA's Office of Law Enforcement has been investigating increased reports from along the northern Gulf of Mexico coast of violence against common bottlenose dolphins, including shootings via guns and bows and arrows, pipe bombs and cherry bombs, and stabbings (Vail 2016). From recent cases that have been prosecuted, it has been shown that fishermen become frustrated and retaliate against dolphins for removing bait or catch, or depredating, their fishing gear. To date there are no records of violent acts for this stock area.

Depredation is a growing problem in Gulf of Mexico coastal and estuary waters and globally, and can lead to serious injury or mortality via ingestion of or entanglement in gear (e.g., Zollett and Read 2006; Read 2008; Powell and Wells 2011; Vail 2016), as well as changes in dolphin activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning, or the illegal feeding, of wild common bottlenose dolphins, may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Provisioning has been documented in the literature in Florida and Texas (Bryant 1994; Samuels and Bejder 2004; Cunningham-Smith *et al.* 2006; Powell and Wells 2011). To date there are no records within the literature of provisioning for this stock area.

All mortalities and serious injuries from known sources for the West Bay Stock are summarized in Table 1.

Table 1. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the West Bay Stock. For the shrimp trawl fishery, the bycatch mortality for the West Bay Stock alone cannot be quantified at this time because mortality estimates encompass all estuarine waters of Texas pooled. However, the estimated mortality for all Texas estuarine waters for 2010–2104 is zero (see Shrimp Trawl section). The remaining fisheries do not have an ongoing, systematic, federal observer program, so counts of mortality and serious injury were based on stranding data, at-sea observations, or fisherman self-reported takes via the Marine Mammal Authorization Program (MMAP). For stranding and at-sea counts, the number reported is a minimum because not all strandings or at-sea cases are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates. NA = not applicable.

Fishery	Years	Data Type	Mean Annual Estimated Mortality and Serious Injury Based on Observer Data	5-year Minimum Count Based on Stranding, At-Sea, and/or MMAP Data
Shrimp Trawl	2010–2014	Observer Data	0	NA
Atlantic Blue Crab Trap/Pot	2013–2017	Stranding Data and At-Sea Observations	NA	0
Hook and Line	2013–2017	Stranding Data and At-Sea Observations	NA	1
Mean Annual Mortality due to commercial fisheries (2013–2017)			**0.2**	
Research Takes (5-year Count)			0	
Other Takes (5-year Count)			0	
Mean Annual Mortality due to research and other takes (2013–2017)			**0**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2013–2017)			**0.2**	

Strandings

During 2013–2017, 10 common bottlenose dolphins were reported stranded within the West Bay area (Table 2; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018). It could not be determined if there was evidence of human interaction for eight of these strandings. For one dolphin, no evidence of human interaction was detected. Evidence of human interactions was detected for the remaining one stranded dolphin, and involved a hook and line fishing gear entanglement (Table 2). Stranding data 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 among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

The West Bay Stock has likely been affected by five common bottlenose dolphin die-offs or Unusual Mortality Events (UMEs). 1) From January through May 1990, a total of 344 common bottlenose dolphins stranded in the northern Gulf of Mexico. Overall this represented a two-fold increase in the prior maximum recorded number of strandings for the same period in the northern Gulf of Mexico. The cause of the 1990 mortality event could not be determined (Hansen 1992), however, morbillivirus may have contributed to this event (Litz *et al.* 2014). One stranding occurred within West Bay and 25 others occurred along the ocean side of Galveston Island, some in the vicinity of West Bay, but the stock origin of those animals is unknown (Phillips and Rosel 2014). 2) In 1993–1994, a UME of common bottlenose dolphins caused by morbillivirus started in the Florida Panhandle and spread west with most of the mortalities occurring in Texas (Lipscomb 1993; Lipscomb *et al.* 1994; Litz *et al.* 2014). From February through April 1994, 236 common bottlenose dolphins were found dead on Texas beaches, of which 67 occurred in a single 10-day period. One stranding occurred within West Bay, and 51 others occurred along the ocean side of Galveston Island and may or may not have involved this stock (Phillips and Rosel 2014). 3) During February and March of 2007 a UME was declared for northeast Texas and western Louisiana involving 64 common bottlenose dolphins and two unidentified dolphins. Decomposition prevented conclusive analyses on most carcasses (Litz *et al.* 2014). Eighteen animals stranded along the ocean side of Galveston Island in the vicinity of West Bay, but the stock origin of the animals is unknown (Phillips and Rosel 2014). 4) During February and March of 2008 a UME was declared in Texas involving 111 common bottlenose dolphin strandings (plus strandings of one unidentified dolphin and one melon-headed whale, *Peponocephala electra*). Most of the animals recovered were in a decomposed state and a direct cause of the mortalities could not be identified. However, there were numerous, co-occurring harmful algal bloom toxins detected during the time period of this UME which may have contributed to the mortalities (Fire *et al.* 2011). Two strandings occurred within West Bay and 35 others occurred along the Gulf side of Galveston Island in the vicinity of West Bay, but the stock origin of the animals is unknown (Phillips and Rosel 2014). 5) A UME occurred from November 2011 to March 2012 across five Texas counties and included 126 common bottlenose dolphin strandings. The strandings were coincident with harmful algal blooms of *K. brevis and Dinophysis sp.* The cause of the bottlenose dolphin UME was determined to be due to biotoxin exposure from brevetoxin and okadaic acid. The additional supporting evidence of fish kills and other species die-offs linked to brevetoxin during the same time and space made a strong case that the harmful algal blooms impacted the dolphins. Three animals from the West Bay Stock were considered to be part of the UME, and an additional 37 strandings occurred along the Gulf side of Galveston Island in the vicinity of West Bay, but the stock origin of the animals is unknown (Phillips and Rosel 2014).

Table 2. Common bottlenose dolphin strandings occurring in the West Bay Stock area from 2013 to 2017, 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. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 13 June 2018). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2013	2014	2015	2016	2017	Total
West Bay Stock	Total Stranded	2	6	0	2	0	10
	HI--Yes	0	1[a]	0	0	0	1
	HI--No	0	0	0	1	0	1
	HI--CBD	2	5	0	1	0	8
[a] This mortality involved an entanglement interaction with hook and line gear.							

HABITAT ISSUES

The estuarine habitat occupied by this stock is adjacent to the highly populated and industrial areas of Houston and Galveston, Texas. The five coastal counties surrounding the Galveston Bay Estuary, which includes West Bay, have a population exceeding 5.4 million people as of January 1, 2018 (TDC 2019). This has been an area of continuous economic growth and development over most of the previous 50 years, with much of this growth attributed to the discovery of oil and the construction of the Houston Ship Channel (Lester and Gonzalez 2011).

There are over 3000 oil and natural gas production platforms in the five counties surrounding Galveston and West Bays, including pipelines for the transport of these products and many refining facilities (Lester and Gonzalez 2011). While most of the platforms are placed on the surrounding land in the West Bay area, several platforms reside in Chocolate Bay and the confluence of Chocolate Bay and West Bay (Lester and Gonzalez 2011). No major oil spills have occurred within West Bay itself, however, repeated spills, from minor to serious in nature, have occurred in the waters of Galveston Bay or in coastal waters off Galveston Island (see Phillips and Rosel 2014 for a summary). A recent oil spill in 2014, referred to as the Texas City Y incident, involved a vessel collision in Galveston Bay near Texas City and the subsequent release of approximately 168,000 gallons of intermediate fuel oil. Through the National Resource Damage Assessment (NRDA) process, impacts of this spill are currently being evaluated and will include impacts to common bottlenose dolphins (NOAA DAARP 2018). No information is currently available on potential impacts to the West Bay Stock. In addition to being known as an area of oil and gas production, the area surrounding Galveston and West Bays produces more than 50% of all chemical products manufactured in the U.S. (Henningsen and Würsig 1991; Lester and Gonzalez 2011).

According to an agricultural census for 2007, over 7,700 farms consisting of >540,000 acres of cropland, were located within the five coastal counties surrounding the Galveston Bay Estuary (Lester and Gonzalez 2011). Raising of livestock is also common in this area. Agricultural impacts on West Bay include the introduction of pesticides, herbicides, and nutrients from crop management, as well as fecal coliform bacteria resulting from livestock waste (Lester and Gonzalez 2011). Due to high levels of fecal coliform bacteria, half of the Galveston Bay Estuary is provisionally or permanently closed to the harvesting of shellfish. Chocolate Bay and Bastrop Bay have been rated as "moderate" for bacterial contamination levels, and West Bay has been rated "good" with fewer than 10% of sampled sites exceeding threshold levels for coliform bacteria (Lester and Gonzalez 2011).

In addition to discharge from the petroleum and chemical refineries and facilities and agricultural sources and sewage, West Bay receives additional pollution from storm water runoff and shipping traffic (Jackson *et al.* 1998; Santschi *et al.* 2001; Lester and Gonzalez 2011; Phillips and Rosel 2014). Analysis of sediment samples from Galveston and West Bays in 2009 and 2010 indicated low concentrations of heavy metals. However, in 2000, two sediment samples from West Bay exceeded safety thresholds for PCBs (lindane and chlordane) (Lester and Gonzalez 2011; Phillips and Rosel 2014). Heavy metal and chemical concentrations in sediments and fish tissues have historically been of concern, and advisories for seafood consumption have often been issued. For example, currently an advisory exists regarding catfish consumption in West Bay and Chocolate Bay due to concerns about dioxins and PCBs (TPWD 2017). Mercury concentrations from samples of blue crab, oysters, and finfish are typically below those considered to be of human health concern, however the second highest concentration of mercury within the Galveston Bay Estuary was measured in a sample of sheepshead collected in West Bay in 1999 (Lester and Gonzalez 2011; Phillips and Rosel 2014). Organic contaminants and trace metals have been monitored in oysters, and the resulting concentration of PCBs has typically surpassed the level for sub-lethal effects (Jackson *et al.* 1998; Phillips and Rosel 2014). The concentrations of lead found in oysters from West Bay and Back Bay (adjacent to West Bay, on the other side of the I-45 Galveston Causeway Bridge) have been higher than those reported from other sampling sites within the Galveston Bay Estuary (Jiann and Presley 1997). Polynuclear aromatic hydrocarbon (PAH) levels in Galveston Bay are higher than national levels and indicate contamination by petroleum products, industrial activities, and urban run-off (Qian *et al.* 2001; Phillips and Rosel 2014). Concentrations of chlorinated hydrocarbons and metals were examined in conjunction with an anomalous mortality event of common bottlenose dolphins in Texas bays (although not West Bay) in 1990 and found to be relatively low in most; however, some had concentrations at levels of possible toxicological concern (Varanasi *et al.* 1992).

Harmful algal blooms and low dissolved oxygen are habitat issues leading to fish kills almost annually in the summers for Galveston and West Bays (McInnes and Quigg 2010). For example, a fish kill occurred in 2005 near Galveston Island due to low dissolved oxygen and a cyanobacteria bloom, killing over 10,000 Gulf menhaden (Phillips and Rosel 2014). In August 2012, a bloom occurred killing approximately one million fish in Galveston and West Bays. Another *K. brevis* bloom occurred along the Texas coast during September 2011–January 2012 resulting in the temporary closure of all shellfish beds in Texas and fish kills in Galveston Bay (Phillips and Rosel 2014). Earlier algal blooms affecting West Bay and resulting in shellfish bed closures occurred in 1972, 1976, 1986, 1996, and 2000 (Magaña *et al.* 2003; Phillips and Rosel 2014). For the 2011–2012 UME mentioned above (Strandings section), the strandings were coincident with a large harmful algal bloom of *K. brevis*. The definitive cause of that event has not been determined, but the algal bloom could have contributed to the mortality event.

Loss of wetland habitat and seagrass beds, and fragmentation of these habitats, within West Bay is another important issue (Lester and Gonzalez 2011; Phillips and Rosel 2014). West Bay has suffered significant loss of wetland habitat since the 1950s, much through the conversion of wetlands to cropland. Subsidence is another leading cause of wetland loss, exacerbated by the removal of petroleum and groundwater in the area (Lester and Gonzalez 2011; Phillips and Rosel 2014). Sea grass beds have been lost due to a complex interaction of causes including shoreline development, dredging, subsidence, boat traffic, and severe storms (Lester and Gonzalez 2011). Conservation partners and resource managers have invested in habitat restoration efforts within West Bay and have begun to restore acres of intertidal marsh and seagrasses (Lester and Gonzalez 2011; Phillips and Rosel 2014).

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act, and the West Bay Stock is not a strategic stock under the MMPA. PBR for the West Bay Stock is 0.5 and so the zero mortality rate goal, 10% of PBR, is 0.05. The documented mean annual human-caused mortality and serious injury for this stock for 2012–2016 was 0.2, which is 40% of the stock's PBR. However, it is likely this estimate is biased low as indicated above (see Annual Human-Caused Mortality and Serious Injury section). 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 and there are insufficient data to determine population trends for this stock.

Although this stock does not meet the criteria to qualify as strategic, NMFS has concerns regarding this stock due to the small stock size and the inability to determine the total human-caused mortality and serious injury.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*)
St. Andrew Bay Stock

NOTE – NMFS is in the process of writing individual stock assessment reports for each of the 31 bay, sound and estuary stocks of common bottlenose dolphins in the Gulf of Mexico.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are distributed throughout the bays, sounds and estuaries of the Gulf of Mexico (Mullin 1988). Long-term (year-round, multi-year) residency by at least some individuals has been reported from nearly every site where photographic identification (photo-ID) or tagging studies have been conducted in the Gulf of Mexico (e.g., Irvine and Wells 1972; Shane 1977; Gruber 1981; Irvine *et al.* 1981; Wells 1986; Wells *et al.* 1987; Scott *et al.* 1990; Shane 1990; Wells 1991; Bräger 1993; Bräger *et al.* 1994; Fertl 1994; Wells *et al.* 1996a,b; Wells *et al.* 1997; Weller 1998; Maze and Würsig 1999; Lynn and Würsig 2002; Wells 2003; Hubard *et al.* 2004; Irwin and Würsig 2004; Shane 2004; Balmer *et al.* 2008; Urian *et al.* 2009; Bassos-Hull *et al.* 2013; Wells *et al.* 2017; Balmer *et al.* 2018). In many cases, residents occur predominantly within estuarine waters, with limited movements through passes to the Gulf of Mexico (Shane 1977; Gruber 1981; Irvine *et al.* 1981; Shane 1990; Maze and Würsig 1999; Lynn and Würsig 2002; Fazioli *et al.* 2006; Bassos-Hull *et al.* 2013; Wells *et al.* 2017). Early studies indicating year-round residency in bays in both the eastern and western Gulf of Mexico led to the delineation of 33 bay, sound and estuary (BSE) stocks, including St. Andrew Bay, with the first stock assessment reports published in 1995.

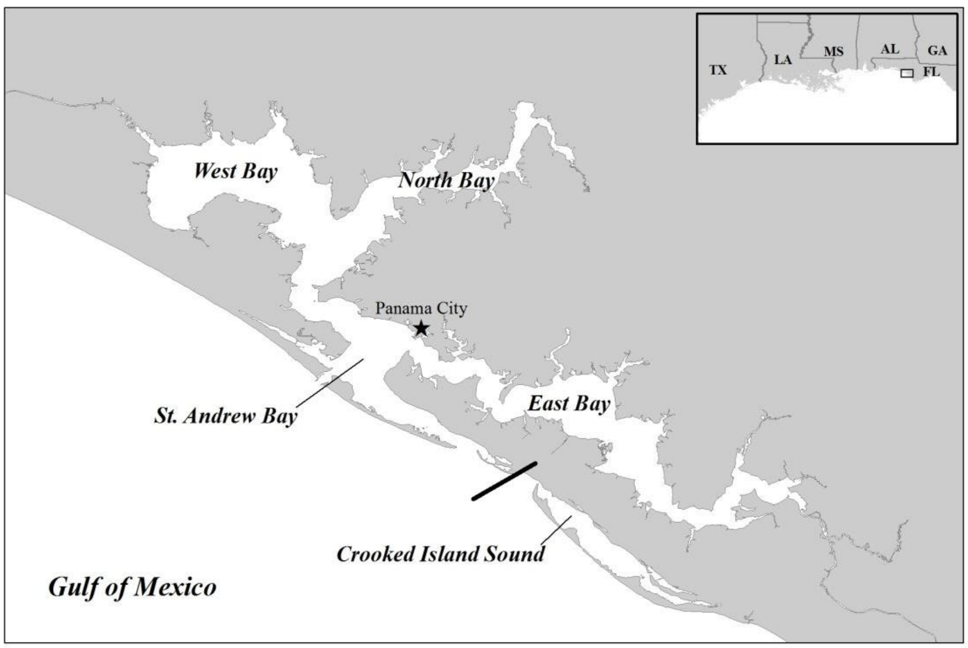

Figure 1. Geographic extent of the St. Andrew Bay Stock, located in the Florida panhandle. The stock includes West Bay, North Bay, East Bay, and St. Andrew Bay. The thick solid line indicates the southeastern boundary of St. Andrew Bay. Crooked Island Sound is part of the St. Joseph Bay Stock to the southeast.

More recently, genetic data also support the concept of discrete BSE stocks (Duffield and Wells 2002; Sellas *et al.* 2005). Sellas *et al.* (2005) examined population subdivision among dolphins sampled in Sarasota Bay, Tampa Bay, Charlotte Harbor, Matagorda Bay, Texas, and the coastal Gulf of Mexico (1–12 km offshore) from just outside Tampa Bay to the south end of Lemon Bay, and found evidence of significant population differentiation among all areas on the basis of both mitochondrial DNA control region sequence data and nine nuclear microsatellite loci. Genetic data also indicate restricted genetic exchange between and demographic independence of BSE populations and those occurring in adjacent Gulf coastal waters (Sellas *et al.* 2005; Rosel *et al.* 2017). Differences in reproductive seasonality from site to site also suggest genetic-based distinctions among areas (Urian *et al.* 1996). Photo-ID and genetic data from several inshore areas of the southeastern United States also support the existence of resident estuarine animals and a differentiation between animals biopsied along the Atlantic coast and those biopsied within estuarine systems at the same latitude (Caldwell 2001; Gubbins 2002;

Zolman 2002; Mazzoil *et al.* 2005; Litz 2007; Rosel *et al*. 2009).

St. Andrew Bay is located in the central panhandle of Florida adjacent to Panama City, and extends approximately 50 km along the coastline (Figure 1). The bay is approximately 243 km^2 in surface area (US EPA 1999). The St. Andrew Bay area is divided up into four smaller bays: West Bay, North Bay, St. Andrew Bay, and East Bay. On average the entire bay is 4 m in depth (US EPA 1999), but West Bay, North Bay, and East Bay are shallower than St. Andrew Bay. St. Andrew Bay is unique among Gulf of Mexico estuaries in that very little fresh water flows into the bay, resulting in high salinities and clear water due to the lack of sedimentation and turbidity (Brim and Handley 2002; Balmer *et al.* 2019). Average salinity is 31 ppt (US EPA 1999). High salinity and clear water facilitate seagrass growth throughout the bay (Brim and Handley 2002). In 1938 the U.S. Army Corps of Engineers excavated through a peninsula to create a rock-jettied inlet which is the main entrance channel (Brim and Handley 2002). St. Andrew Bay has been designated as an aquatic preserve by the state of Florida (Florida DEP 2018).

The St. Andrew Bay Stock boundaries includes all waters of West Bay, North Bay, St. Andrew Bay and East Bay (Figure 1). The boundaries are based on photo-ID studies conducted during 2015–2016 by Balmer *et al.* (2019) which found minimal overlap between animals sighted in BSE waters and those sighted in nearshore coastal waters. The boundaries are subject to change as additional research is conducted. There is strong support from the findings of Balmer *et al.* (2008) to include Crooked Island Sound (also known as St. Andrew Sound) within the St. Joseph Bay Stock, southeast of St. Andrew Bay. However, animals from St. Andrew Bay and surrounding Panama City have also been sighted in Crooked Island Sound, suggesting Crooked Island Sound is an area of overlap for dolphins inhabiting both St. Joseph Bay and St. Andrew Bay (Balmer *et al.* 2010; 2019). Overlap between these stocks primarily occurred at the entrance of Crooked Island Sound and to a lesser degree, at the entrance to St. Andrew Bay.

POPULATION SIZE

The best available abundance estimate for the St. Andrew Bay Stock of common bottlenose dolphins is 199 (95% CI:173–246; CV=0.09), based on an April 2016 vessel-based capture-recapture photo-ID survey (Balmer *et al.* 2019).

Earlier abundance estimates (>8 years old)

Vessel-based capture-recapture photo-ID surveys were conducted during 2004–2007 by Bouveroux *et al.* (2014). The surveys covered a portion of the stock area and included central St. Andrew Bay and nearshore coastal waters. West Bay, North Bay, and East Bay were not surveyed. Seasonal abundance estimates were calculated using robust design population models. Abundance varied seasonally, and ranged from 89 (95% CI=71–161) in March–May 2004 to 183 (95% CI=169–208) in June–July 2007. Because these surveys did not sample all of the estuarine waters where dolphins are known to occur, the estimates of abundance were negatively biased. Overall, the results of Bouveroux *et al.* (2014) indicated a small community of dolphins with high site fidelity utilized the St. Andrew Bay area as well as a large number of transient dolphins that frequently utilized the area.

Recent surveys and abundance estimates

Balmer *et al.* (2019) conducted vessel-based capture-recapture photo-ID surveys during July 2015, October 2015, April 2016, and October 2016 to estimate abundance of common bottlenose dolphins for St. Andrew Bay. Abundance estimates were generated using a robust-design capture-recapture random movement model. Estimates factored in the distinctiveness rate and included animals with distinctive and non-distinctive fins. Abundance ranged from 199 (95% CI=173–246) in April 2016 to 315 (95% CI=274–378) in October 2016. Given the observed seasonal variation in abundance and the possibility that transient animals may occur within estuarine waters (Bouveroux *et al.* 2014), the lowest seasonal abundance estimate (April 2016), 199 (CV=0.09), was used as the best estimate for the St. Andrew Bay Stock as this estimate most likely reflects primarily resident animals. This approach is consistent with that for other BSE stocks where multiple seasonal abundance estimates are available. Key uncertainties in this abundance estimate include movement patterns of individual dolphins between estuarine and coastal waters of St. Andrew Bay. Balmer *et al.* (2019) estimated abundance exclusively within the St. Andrew Bay Stock boundaries but also surveyed coastal waters adjacent to St. Andrew Bay. Although there was minimal crossover of individuals between estuarine and coastal waters (St. Andrew Bay photo-ID catalog: N = 25/353, 7%), and robust capture-recapture models should account for temporary immigration, the abundance estimates from a given sampling period may be biased.

Minimum Population Estimate

The minimum population estimate 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 for the St. Andrew Bay Stock is 199 (CV=0.09).

The resulting minimum population estimate is 185.

Current Population Trend

There are insufficient data to determine the population trends for this stock because only one estimate of population size is available for the entire stock area.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. 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 productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size of the St. Andrew Bay Stock of common bottlenose dolphins is 185. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.4 because the CV of the shrimp trawl mortality estimate for Florida BSE stocks is greater than 0.8 (Wade and Angliss 1997). PBR for this stock of bottlenose dolphins is 1.5.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury to the St. Andrew Bay Stock of common bottlenose dolphins during 2013–2017 is unknown. The mean annual fishery-related mortality and serious injury during 2013–2017 for strandings and at-sea observations identified as fishery-related was 0.2 (see Shrimp Trawl section for additional fishery-related mortality). 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 2013–2017 was 0.2 (Table 1). This is likely a biased estimate and represents several sources of uncertainty 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 of fishery-related interactions includes an actual count of verified fishery-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the estimate does not include shrimp trawl bycatch (see Shrimp Trawl section).

Fishery Information

There are five commercial fisheries that interact, or that potentially could interact, with this stock. These include three Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl; Gulf of Mexico menhaden purse seine; and Southeastern U.S. Atlantic, Gulf of Mexico stone crab trap/pot); and two Category III fisheries (Gulf of Mexico blue crab trap/pot; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

Shrimp Trawl

Between 1997 and 2014, seven common bottlenose dolphins and seven unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the net, lazy line, turtle excluder device, or tickler chain gear in the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2016). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive without serious injury in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al.* (2015; 2016) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Although this fishery operates inside the estuaries of the northern Gulf of Mexico, observer program coverage did not extend into BSE waters; therefore time-area stratified bycatch rates were extrapolated into inshore waters to estimate a five-year unweighted mean mortality estimate for 2010–2014 based on inshore fishing effort (Soldevilla *et al.* 2016). Because the spatial resolution at which fishery effort is modeled is aggregated at the state level (e.g., Nance *et al.* 2008), the mortality estimate covers inshore waters of the Gulf Coast of Florida and thus aggregates all Florida BSE stocks on the west coast, not just the St. Andrew Bay Stock. The mean annual mortality estimate for Florida BSE stocks for the years 2010–2014 was 2.4 (CV=1.6; Soldevilla *et al.* 2016). Because bycatch for the St. Andrew Bay Stock alone cannot be quantified at this time, the shrimp trawl mortality estimate is not included in the annual human-caused mortality and serious injury total for this stock. Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015; 2016).

Menhaden Purse Seine

During 2013–2017 there were no documented interactions between menhaden purse seine gear and the St. Andrew Bay Stock. There are no recent observer program data for the Gulf of Mexico menhaden purse seine fishery. The menhaden fishing effort in this area (Bay County) that corresponds with the St. Andrew Bay Stock fluctuated annually in effort. Number of menhaden fishing trips/year for Bay County during 2013–2017 was as follows: 10 in 2013; 27 in 2014; 25 in 2015; 93 in 2016; and 1 in 2017 (Florida Fish and Wildlife Conservation Commission 2018).

Crab Trap/Pot

During 2013–2017 there were no documented interactions between commercial crab trap/pot gear and the St. Andrew Bay Stock. There is no systematic observer coverage of crab trap/pot fisheries in the Gulf of Mexico, so it is not possible to quantify total mortality.

Hook and Line (Rod and Reel)

During 2013–2017, two live common bottlenose dolphins were observed at-sea (in 2014 and 2015) entangled in hook and line fishing gear. In 2014, a dolphin was observed with a lure hooked to its upper and lower jaw, limiting its ability to open its rostrum. The lure may have come off on its own, and it could not be determined if the animal was seriously injured (Maze-Foley and Garrison 2018). In 2015, another dolphin was sighted with a lure with a treble hook on each end embedded in the upper and lower rostrum, limiting the animal's ability to open its rostrum. This animal could have belonged to either the St. Andrew Bay Stock or the Northern Coastal Stock, and it was considered seriously injured (Maze-Foley and Garrison 2018). The 2015 serious injury was included in the annual human-caused mortality and serious injury total for this stock (Table 1).

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., charter boat and 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 in the Gulf of Mexico. The documented interactions in this gear represents a minimum known count of interactions in the last five years.

Other Mortality

Illegal feeding/provisioning of common bottlenose dolphins has been well documented in the St. Andrew Bay/Panama City area. For many years within certain areas of St. Andrew Bay and adjacent coastal waters, it has been typical to see wild dolphins surrounded by multiple boats, multiple personal watercraft, and multiple swimmers. Studies by Samuels and Bejder (2004) in 1998 and more recently by Powell *et al.* (2018) in 2014 have documented a high rate of unregulated food provisioning and recorded many interactions with humans that put dolphins at risk of injury, illness, or death. In addition to the boaters who regularly feed wild dolphins, there are approximately 25 companies based in Panama City offering dolphin viewing and swim-with opportunities (Powell *et al.* 2018). Dolphins are illegally fed regularly in at least two different locations, one inside St. Andrew Bay at a bait barge, and the other just outside St. Andrew Bay along a coastal beach (Powell *et al.* 2018). Illegal feeding is often performed in conjunction with "swim-with" tourist activities that involve people entering the water to interact with free-ranging dolphins. Research by Powell *et al.* (2018) during 2014 indicated the number of conditioned individual dolphins (conditioned to human interaction by food reinforcement; animals that accepted food handouts from people on a regular basis) tripled (n=21) compared to those documented in 1998 by Samuels and Bejder (2004) (n=7), and that overall the problems of illegal feeding and harassment had increased. Powell *et al.* (2018) found that conditioned dolphins spent the majority of their time approaching boats to beg for food and patrolling among boats and swimmers looking for handouts, which in turn increases their risk of boat strike, entanglement in or hooking by fishing gear, or retaliation by angry fishermen (Wells and Scott 1997; Powell and Wells 2011; Adimey *et al.* 2014; Powell *et al.* 2018).

Depredation is also a growing problem in Gulf of Mexico coastal and estuarine waters and globally, and can lead to serious injury or mortality via ingestion of, hooking by, or entanglement in gear (e.g., Zollett and Read 2006; Read 2008; Powell and Wells 2011; Vail 2016), as well as to changes in dolphin activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that the illegal feeding of wild common bottlenose dolphins may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). For example, in Panama City, two conditioned dolphins previously observed begging, were also sighted patrolling and attempting to depredate from recreational fishermen (Powell *et al.* 2018). There have been two recent cases of dolphins sighted within St. Andrew Bay with fishing lures embedded in their rostrums, limiting the ability of the animals to open their rostrums (see Hook and Line section). These cases of gear entanglement may have been a result of dolphins depredating fishing gear.

All mortalities and serious injuries from known sources for the St. Andrew Bay Stock are summarized in Table 1.

Table 1. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the St. Andrew Bay Stock. For the shrimp trawl fishery, the bycatch mortality for the St. Andrew Bay Stock alone cannot be quantified at this time because mortality estimates encompass all estuarine waters of the Gulf coast of Florida, pooled. Therefore, the Gulf coast mortality estimate for Florida has not been included in the annual human-caused mortality and serious injury total for this stock (see Shrimp Trawl section). The remaining fisheries do not have an ongoing, systematic, federal observer program, so counts of mortality and serious injury were based on stranding data, at-sea observations, or fisherman self-reported takes via the Marine Mammal Authorization Program (MMAP). For stranding and at-sea counts, the number reported is a minimum because not all strandings or at-sea cases are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates. NA = not applicable.

Fishery	Years	Data Type	Mean Annual Estimated Mortality and Serious Injury Based on Observer Data	5-year Minimum Count Based on Stranding, At-Sea, and/or MMAP Data
Shrimp Trawl	2010–2014	Observer Data	Undetermined for this stock (see Shrimp Trawl section)	NA
Menhaden Purse Seine	2013–2017	MMAP fisherman self-reported takes	NA	0
Stone Crab Trap/Pot	2013–2017	Stranding Data and At-Sea Observations	NA	0
Blue Crab Trap/Pot	2013–2017	Stranding Data and At-Sea Observations	NA	0
Hook and Line	2013–2017	Stranding Data and At-Sea Observations	NA	1
Mean Annual Mortality due to commercial fisheries (2013–2017)			**0.2**	
Research Takes (5-year Count)			0	
Other Takes (gunshot wound; 5-year Count)			0	
Mean Annual Mortality due to research and other takes (2013–2017)			**0**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2013–2017)			**0.2**	

Strandings

From 2013 to 2017, 19 common bottlenose dolphins were reported stranded within the St. Andrew Bay Stock area (Table 2; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018). It could not be determined whether there was evidence of human interaction for 18 of these strandings, and for one stranding, signs of human interaction were detected. Stranding data 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 among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Table 2. Common bottlenose dolphin strandings occurring in the St Andrew Bay Stock area from 2013 to 2017, 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. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 13 June 2018). Please note that HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2013	2014	2015	2016	2017	Total
St. Andrew Bay Stock	Total Stranded	6[a]	4[a]	5	2	2	19
	HI--Yes	0	0	1	0	0	1
	HI--No	0	0	0	0	0	0
	HI--CBD	6	4	4	2	2	18

[a] These strandings were part of the Northern Gulf of Mexico UME.

St. Andrew Bay has been affected by four recent unusual mortality events (UMEs). First, between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with *K. brevis* harmful algal blooms and fish kills in the Florida Panhandle. This UME started in the eastern Bays, Apalachicola Bay and St. Joseph Bay, and spread west to St. Andrew Bay and Choctawhatchee Bay, and was concurrent spatially and temporally with a *K. brevis* bloom that spread east to west. There were nine common bottlenose dolphin strandings within the St. Andrew Bay Stock area during this event, and brevetoxin was determined to be the cause (Twiner *et al.* 2012; Litz *et al.* 2014). Second, in March and April 2004, in another Florida Panhandle UME attributed to *K. brevis* blooms, 105 common bottlenose dolphins and two unidentified dolphins stranded dead (Litz *et al.* 2014). This event started in St. Joseph Bay and spread westward. At least two common bottlenose dolphins stranded in the St. Andrew Bay Stock area. Although there was no indication of a *K. brevis* bloom at the time, high levels of brevetoxin were found in the stomach contents of the stranded dolphins (Flewelling *et al.* 2005; Twiner *et al.* 2012). Third, a separate UME was declared in the Florida Panhandle after elevated numbers of dolphin strandings occurred in association with a *K. brevis* bloom in September 2005. Dolphin strandings remained elevated through the spring of 2006 and brevetoxin was again detected in the tissues of most of the stranded dolphins. Between September 2005 and April 2006 when the event was officially declared over, a total of 88 common bottlenose dolphin strandings occurred (plus strandings of five unidentified dolphins), with nine (10%) occurring within the St. Andrew Bay Stock area. Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012; Litz *et al.* 2014). Finally, 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). This UME included cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see Habitat Issues section), 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, although not for strandings in St. Andrew Bay (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section). During 2013–2014, all 10 stranded dolphins from this stock were considered to be part of the UME (see Table 2).

HABITAT ISSUES

The *Deepwater Horizon* MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days up to ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Some heavy to moderate oiling occurred on Alabama and Florida beaches, with the heaviest stretch occurring from Dauphin Island, Alabama, to Gulf Breeze, Florida. Light to trace oil was reported from Gulf Breeze to Panama City, Florida (OSAT-2 2011; Michel *et al.* 2013). The maximum shoreline oiling experienced by the St. Andrew Bay stock area was very light oiling in parts of the stock area (Michel *et al.* 2013) and no deaths in St. Andrew Bay during the spill time period were attributed to oil (DWH NRDAT 2016).

Environmental contaminants have been an issue of concern for common bottlenose dolphins throughout the southeastern U.S. prior to the DWH oil spill (e.g., Kucklick *et al.* 2011), and due to the physical features of St. Andrew Bay, such as the depth, lack of freshwater inflow and resulting high salinity, minimal tidal flushing, and sediment composition, this bay is very vulnerable to contamination and pollution (Brim and Handley 2002). Contaminants cannot be easily flushed out and the sediments in the bay could become reservoirs for contaminants. The Environmental Protection Agency has identified one Superfund hazardous waste site at Tyndall Air Force Base, which borders St. Andrew Bay and East Bay. A Florida state-funded clean-up program includes two additional contaminated sites, and there are four hazardous waste producing facilities in the St. Andrew Bay watershed (Northwest Florida Water Management District 2017).

Storm water runoff and urbanization pose the greatest future threats to the quality of water and sediments in St. Andrew Bay (Brim and Handley 2002). Several common bottlenose dolphin UMEs in St. Andrew Bay (see Strandings section) have been attributed to harmful algal blooms (*K. brevis*), which are a result of eutrophication. For recent UMEs in the Florida Panhandle (1999–2000, 2004, 2005–2006), the site of bloom origin was not known for all, but it is likely none originated in St. Andrew Bay (Twiner *et al.* 2012). However, blooms can be transported by currents from adjacent bays and coastal waters, so eutrophication anywhere along the Florida Panhandle can impact St. Andrew Bay. Other habitat issues for this area include historic loss of seagrasses and damage to seagrasses due to propeller scarring, wetland loss and degradation, and a rapid increase in human population and associated coastal development in the area (Northwest Florida Water Management District 2017).

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act, and the St. Andrew Bay Stock is not a strategic stock under the MMPA. The documented mean annual human-caused mortality for this stock for 2013–2017 is 0.2. However, it is likely that the estimate of annual fishery-caused mortality and serious injury is biased low as indicated above (see Annual Human-Caused Mortality and Serious Injury section). There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. The status of this stock relative to OSP is unknown. There are insufficient data to determine population trends for this stock.

Although this stock does not meet the criteria to qualify as strategic (NMFS 2016), NMFS has concerns regarding this stock due to the small stock size, the high number of common bottlenose dolphin deaths associated with UMEs in the Florida panhandle since 1999, and the high rate of illegal feeding and human interactions.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*)
St. Joseph Bay Stock

NOTE – NMFS is in the process of writing individual stock assessment reports for each of the 31 bay, sound and estuary stocks of common bottlenose dolphins in the Gulf of Mexico.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are distributed throughout the bays, sounds and estuaries of the Gulf of Mexico (Mullin 1988). Long-term (year-round, multi-year) residency by at least some individuals has been reported from nearly every site where photographic identification (photo-ID) or tagging studies have been conducted in the Gulf of Mexico (e.g., Irvine and Wells 1972; Shane 1977; Gruber 1981; Irvine *et al.* 1981; Wells 1986; Wells *et al.* 1987; Scott *et al.* 1990; Shane 1990; Wells 1991; Bräger 1993; Bräger *et al.* 1994; Fertl 1994; Wells *et al.* 1996a,b; Wells *et al.* 1997; Weller 1998; Maze and Würsig 1999; Lynn and Würsig 2002; Wells 2003; Hubard *et al.* 2004; Irwin and Würsig 2004; Shane 2004; Balmer *et al.* 2008; Urian *et al.* 2009; Bassos-Hull *et al.* 2013; Wells *et al.* 2017; Balmer *et al.* 2018). In many cases, residents occur predominantly within estuarine waters, with limited movements through passes to the Gulf of Mexico (Shane 1977; Gruber 1981; Irvine *et al.* 1981; Shane 1990; Maze and Würsig 1999; Lynn and Würsig 2002; Fazioli *et al.* 2006; Bassos-Hull *et al.* 2013; Wells *et al.* 2017). Early studies indicating year-round residency in bays in both the eastern and western Gulf of Mexico led to the delineation of 33 bay, sound and estuary (BSE) stocks, including St. Joseph Bay, with the first stock assessment reports published in 1995.

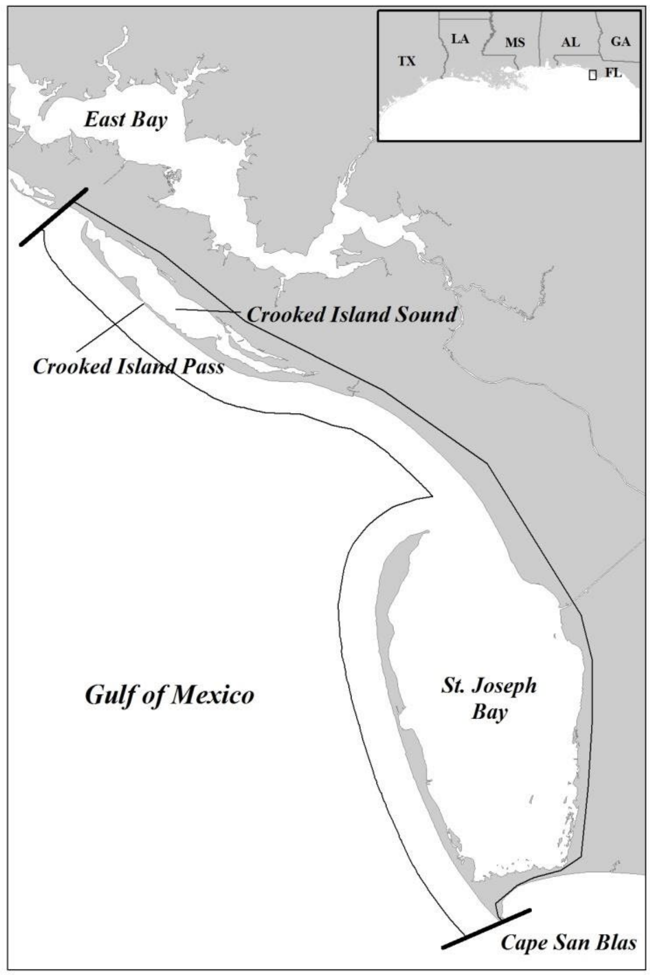

Figure 1. Geographic extent of the St. Joseph Bay Stock, located in the Florida panhandle. The stock boundaries are denoted by solid lines, with the thicker lines denoting the northern and southern boundaries. The stock includes St. Joseph Bay, Crooked Island Sound, and adjacent coastal waters out to 2 km from shore. East Bay is part of the St. Andrew Bay stock to the northwest.

More recently, genetic data also support the concept of discrete BSE stocks (Duffield and Wells 2002; Sellas *et al.* 2005). Sellas *et al.* (2005) examined population subdivision among dolphins sampled in Sarasota Bay, Tampa Bay, Charlotte Harbor, Matagorda Bay, Texas, and the coastal Gulf of Mexico (1–12 km offshore) from just outside Tampa Bay to the south end of Lemon Bay, and found evidence of significant population differentiation among all areas on the basis of both mitochondrial DNA control region sequence data and nine nuclear microsatellite loci. Genetic data also indicate restricted genetic exchange between and demographic independence of BSE populations and those occurring in adjacent Gulf coastal waters (Sellas *et al.* 2005; Rosel *et al.* 2017). Differences in reproductive seasonality from site to site also suggest genetic-based distinctions among areas (Urian *et al.* 1996). Photo-ID and genetic data from several inshore areas of the southeastern United States also support the existence of resident estuarine animals and a differentiation between animals biopsied along the Atlantic coast and those biopsied within estuarine systems at the same latitude (Caldwell 2001; Gubbins 2002; Zolman 2002; Mazzoil *et al.* 2005; Litz 2007; Rosel *et al*. 2009).

St. Joseph Bay is a relatively small embayment of 170 km^2 in area, located just west of Apalachicola in the central panhandle of Florida (Figure 1). The bay is bounded in the south by Cape San Blas, in the west by the St. Joseph Peninsula and opens in the north to the Gulf of Mexico. St. Joseph Bay extends 21 km in length and 10 km in width at its widest point, and is characterized by extensive seagrass beds and salt marshes. The southern quarter of the bay is 1 m or less deep whereas the deepest portions are in the northwest region at approximately 10 m deep. Most of St. Joseph Bay has been designated as an aquatic preserve by the state of Florida. There is minimal freshwater inflow into the bay (U.S. EPA 1999; Balmer 2007; Moretzsohn *et al.* 2010). To the northwest of St. Joseph Bay, Crooked Island Sound (also known as St. Andrew Sound) extends 12 km in length and 2 km in width at its widest point. It varies in depth from 1 m around the margins of the sound to 6–7 m at the sound's entrance (Balmer 2007).

In response to three unusual mortality events along the Florida panhandle, which all impacted the St. Joseph Bay area, Balmer *et al.* (2008) conducted photo-ID surveys from April 2004 to July 2007 to examine seasonal abundance, distribution patterns and site fidelity of common bottlenose dolphins in St. Joseph Bay and along the coast northwest to and inside Crooked Island Sound. In addition, during April 2005 and July 2006, NOAA and the Sarasota Dolphin Research Program along with other partners, conducted health assessments of common bottlenose dolphins in the St. Joseph Bay area. Photo-ID data strongly suggested a movement of dolphins into the St. Joseph Bay region during spring and fall with lower abundance during winter and summer. Dolphins sighted in winter and summer displayed higher site fidelity, whereas the majority of dolphins sighted during spring and fall displayed the lowest site fidelity (Balmer *et al.* 2008). Radio-tracking results supported these findings, with animals tagged in spring 2005 (April) ranging the farthest of all dolphins tagged, extending outside the St. Joseph Bay Stock region. Overall, Balmer *et al.* (2008) found abundance to vary seasonally in the St. Joseph Bay area, and suggested the St. Joseph Bay area supports a resident community of common bottlenose dolphins as well as seasonal visitors during spring and fall seasons. Additional photo-ID surveys were conducted during 2010, 2011, and 2013 to examine abundance, density, and site fidelity during and after the *Deepwater Horizon* (DWH) oil spill (Balmer *et al.* 2018). Abundance was again found to vary seasonally, with the highest abundance during fall and the lowest during the winter. However, summer 2010 data appeared more similar to previous years' spring and fall results, with an increased number of dolphins displaying low site fidelity, higher abundance estimates, and an increase in density in coastal waters. Overall, the more recent data still supported a resident community sighted across seasons and years (Balmer *et al.* 2018).

The St. Joseph Bay Stock boundaries includes St. Joseph Bay, Crooked Island Sound and coastal waters out to 2 km from shore in between St. Joseph Bay and Crooked Island Sound, and coastal waters out to 2 km from shore from Cape San Blas along St. Joseph Peninsula and along Crooked Island (Figure 1). The boundaries of this stock are based on photo-ID and radio-tracking studies conducted during 2004–-2007, and photo-ID studies during 2010, 2011, and 2013 (Balmer 2007; Balmer *et al.* 2008; Balmer *et al.* 2018), which support the inclusion of nearshore coastal waters within the boundaries for this particular stock. The boundaries are subject to change as additional research is conducted. There is strong support from the findings of Balmer *et al.* (2008) to include Crooked Island Sound in the St. Joseph Bay Stock. However, animals from nearby St. Andrew Bay, located to the northwest of St. Joseph Bay (see Figure 1) and surrounding Panama City, have also been sighted in Crooked Island Sound, suggesting Crooked Island Sound is an area of overlap for dolphins inhabiting both St. Joseph Bay and St. Andrew Bay. An example of overlap with St. Andrew Bay is given by Balmer *et al.* (2010), who show the sightings for a particular animal, tracked simultaneously via satellite-linked transmitter and VHF radio transmitter, sighted in both Crooked Island Sound and St. Andrew Bay as well as adjacent coastal waters. Balmer *et al.* (2019) compared St. Joseph Bay (N = 726) and St. Andrew Bay (N = 353) photo-ID catalogs to assess extended movement patterns and stock overlap between these adjacent study areas. A total of 27 matches were made between the St. Andrew Bay (8%) and St. Joseph Bay (4%) catalogs. Overlap between these stocks primarily occurred at the entrance of Crooked Island Sound and to a lesser degree, entrance to St. Andrew Bay.

POPULATION SIZE

The best available abundance estimate for the St. Joseph Bay Stock of common bottlenose dolphins is 142 (95% CI: 92–190; CV=0.17), based on a February 2011 vessel-based capture-recapture photo-ID survey (Balmer *et al.* 2018).

Earlier abundance estimates (>8 years old)

In order to estimate seasonal abundance, Balmer *et al.* (2008) conducted vessel-based capture-recapture photo-

ID surveys across multiple seasons from February 2005 through July 2007 in St. Joseph Bay and along the coast to the northwest including Crooked Island Sound (St. Andrew Sound). Line and contour transects were used to cover the study area, and each survey was only conducted if Beaufort Sea State was 3 or less. Balmer *et al.* (2008) also calculated a distinctiveness rate, which was the proportion of distinctive (marked) dolphins to non-distinctive (un-marked) dolphins, for each survey season. Mark-recapture estimates factored in the distinctiveness rate and included animals with distinctive and non-distinctive fins. Seasonal abundance estimates using the robust 'Markovian Emigration' model ranged from 122 dolphins (CV=0.09) for winter 2006 to 340 dolphins (CV=0.09) for fall 2006. Summer and winter estimates provide the best estimate of the resident population as spring and fall estimates also include transient animals. Therefore, the previous best available abundance estimate for the St. Joseph Bay Stock was the average of the estimates for winter 2005, summer 2005, winter 2006, and summer 2007, which was 146 dolphins (CV=0.18).

Recent surveys and abundance estimates

Using the same field methodology as in previous surveys (Balmer *et al.* 2008), Balmer *et al.* (2018) conducted vessel-based capture-recapture photo-ID surveys during June and August 2010, February 2011, and October 2013 and were able to estimate density and abundance of common bottlenose dolphins for St. Joseph Bay during and after the DWH oil spill. Abundance estimates were generated using a spatially explicit robust-design capture-recapture (SERDCR) model developed by McDonald *et al.* (2017). Estimates factored in the distinctiveness rate and included animals with distinctive and non-distinctive fins. Previously work indicated summer and winter estimates provide the best estimate of the resident population due to an increase in transient animals during spring and fall (Balmer *et al.* 2008). Balmer *et al.* (2018) reported that winter was the optimal season to estimate abundance for the most recent study, and therefore, the best estimate for the St. Joseph Bay Stock is from February 2011, and is 142 (95% CI: 92–190; CV=0.17). Key uncertainties in this abundance estimate include movement patterns of individual dolphins across the boundary between the St. Joseph and St. Andrew Bay Stocks. Balmer *et al.* (2008; 2018) estimated abundance exclusively within the St. Joseph Bay Stock boundaries but telemetry data and comparisons between the St. Joseph Bay and St. Andrew Bay photo-ID catalogs (Balmer *et al.* 2019) suggest some degree of crossover, specifically within Crooked Island Sound. Although robust capture-recapture models should account for temporary immigration, the abundance estimates from a given sampling period may be biased for this stock.

Minimum Population Estimate

The minimum population estimate 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 for the St. Joseph Bay Stock is 142 (CV=0.17). The resulting minimum population estimate is 123.

Current Population Trend

There are three winter abundance estimates from February/March 2005 (212, 95% CI:134–292), February 2006 (150, 95% CI:84–209), and February 2011 (142, 95% CI: 92–190) with overlapping confidence intervals, providing no evidence for a trend in abundance (Balmer *et al.* 2018).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. 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 productivity rate, and a recovery factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997). The minimum population size of the St. Joseph Bay Stock of common bottlenose dolphins is 123. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.4 because the CV of the shrimp trawl mortality estimate for Florida BSE stocks is greater than 0.8 (Wade and Angliss 1997). PBR for this stock of bottlenose dolphins is 1.0.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury to the St. Joseph Bay Stock of common bottlenose dolphins during 20132017 is unknown because this stock may interact with unobserved fisheries (see below), and also because the most current observer data for the shrimp trawl fishery are for 2010–2014 and mortality rates were calculated at the state level (see Shrimp Trawl section). Uncertainties related to human-caused mortality and serious injury include: 1) the estimate does not include shrimp trawl bycatch (see Shrimp Trawl section), 2) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, 3) 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), and 4) cause of death is not (or cannot be) routinely determined for stranded carcasses.

Fishery Information

There are five commercial fisheries that interact, or that potentially could interact, with this stock. These include three Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl; Gulf of Mexico menhaden purse seine; Southeastern U.S. Atlantic, Gulf of Mexico stone crab trap/pot); and two Category III fisheries (Gulf of Mexico blue crab trap/pot; and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line)). Detailed fishery information is presented in Appendix III.

Shrimp Trawl

Between 1997 and 2014, seven common bottlenose dolphins and seven unidentified dolphins, which could have been either common bottlenose dolphins or Atlantic spotted dolphins, became entangled in the net, lazy line, turtle excluder device, or tickler chain gear in the commercial shrimp trawl fishery in the Gulf of Mexico (Soldevilla *et al.* 2016). All dolphin bycatch interactions resulted in mortalities except for one unidentified dolphin that was released alive without serious injury in 2009 (Maze-Foley and Garrison 2016). Soldevilla *et al.* (2015; 2016) provided mortality estimates calculated from analysis of shrimp fishery effort data and NMFS's Observer Program bycatch data. Although this fishery operates inside the estuaries of the northern Gulf of Mexico, observer program coverage did not extend into BSE waters, therefore time-area stratified bycatch rates were extrapolated into inshore waters to estimate a five-year unweighted mean mortality estimate for 20102014 based on inshore fishing effort (Soldevilla *et al.* 2016). Because the spatial resolution at which fishery effort is modeled is aggregated at the state level (e.g., Nance *et al.* 2008), the mortality estimate covers inshore waters of the Gulf Coast of Florida, not just the St. Joseph Bay Stock. The mean annual mortality estimate for Florida BSE stocks for the years 2010–2014 was 2.4 (CV=1.6; Soldevilla *et al.* 2016). Because bycatch for the St. Joseph Bay Stock alone cannot be quantified at this time, the shrimp trawl mortality estimate is not included in the annual human-caused mortality and serious injury total for this stock. Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015; 2016).

Menhaden Purse Seine

During 2013–2017 there were no documented interactions between menhaden purse seine gear and the St. Joseph Bay Stock. There are no recent observer program data for the Gulf of Mexico menhaden purse seine fishery. The menhaden fishing effort in this area (Gulf County) that corresponds with the St. Joseph Bay Stock was limited during 2013–2017. Number of menhaden fishing trips/year for Gulf County was as follows: 23 in 2013; 9 in 2014; 17 in 2015; 33 in 2016; and 13 in 2017 (Florida Fish and Wildlife Conservation Commission 2018).

Crab Trap/Pot

During 2013–2017 there were no documented interactions between commercial crab trap/pot gear and the St. Joseph Bay Stock. There is no systematic observer coverage of crab trap/pot fisheries in the Gulf of Mexico, so it is not possible to quantify total mortality and serious injury.

Hook and Line (Rod and Reel)

During 2013–2017, there were no documented interactions with hook and line gear and the St. Joseph Bay Stock. It is not possible to estimate the total number of interactions with hook and line gear because there is no systematic observer program in the Gulf of Mexico.

Other Mortality

Depredation is a growing problem in Gulf of Mexico coastal and estuarine waters and globally, and can lead to serious injury or mortality via ingestion of or entanglement in gear (e.g., Zollett and Read 2006; Read 2008; Powell and Wells 2011; Vail 2016), as well as changes in dolphin activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning, or the illegal feeding, of wild common bottlenose dolphins, may encourage depredation because provisioning conditions dolphins to approach humans and vessels, where they then may prey on bait and catches (Vail 2016). Illegal feeding/provisioning has been documented in the literature in Florida and Texas (Bryant 1994; Samuels and Bejder 2004; Cunningham-Smith *et al.* 2006; Powell and Wells 2011; Powell *et al.* 2018). Dolphins within the boundaries of this stock, primarily within Crooked Island Sound, have been observed to approach vessels in the area and beg for food (Balmer 2007). Begging behaviors are a result of being illegally fed. It is believed that the animals observed begging within Crooked Island Sound are members of the St. Andrew Bay Stock (the St. Andrew Bay Stock encompasses Panama City, an area where illegal feeding has been documented [Samuels and Bejder 2004; Powell *et al.* 2018]). Three dolphins, which were captured in Crooked Island Sound during the April 2005 health assessment, were observed begging during the three months of subsequent radio tracking (Balmer 2007). Two of these individuals, a mom/calf pair, were sighted exclusively within the boundaries of the St. Andrew Bay Stock during all radio tracking surveys. Both of these individuals were found stranded within two days of each other on 1 November and 3 November 2005 near Panama City and Panama City Beach. The other individual, an adult male, which was documented in Balmer *et al.* (2010), was sighted frequently in the waters from St. Andrew Bay to Crooked Island Sound and in association with individuals from both the St. Andrew Bay and St. Joseph Bay Stocks. Observation of focal common bottlenose dolphin 'X02', examined and freeze-branded during a NMFS 2005 health assessment project in nearby St. Joseph Bay, was documented by Powell *et al.* (2018) being fed repeatedly by the captain of a bait boat off a beach just outside St. Andrew Bay. Thus, the begging behaviors and overlap by individuals of the St. Andrew Bay Stock are likely affecting the behavior of individuals in the St. Joseph Bay Stock. Begging behaviors can be passed through a dolphin population via social learning, thus perpetuating and increasing the prevalence of the problem over time (Wells 2003; Whitehead *et al.* 2004).

All mortalities and serious injuries from known sources for the St. Joseph Bay Stock are summarized in Table 1.

Table 1. Summary of the incidental mortality and serious injury of common bottlenose dolphins (Tursiops truncatus) of the St. Joseph Bay Stock. For the shrimp trawl fishery, the bycatch mortality for the St. Joseph Bay Stock alone cannot be quantified at this time because mortality estimates encompass all estuarine waters of the Gulf coast of Florida pooled. The state-wide mortality estimate for Florida has not been included in the annual human-caused mortality and serious injury total for this stock (see Shrimp Trawl section). The remaining fisheries do not have an ongoing, systematic, federal observer program, so counts of mortality and serious injury were based on stranding data, at-sea observations, or fisherman self-reported takes via the Marine Mammal Authorization Program (MMAP). For stranding and at-sea counts, the number reported is a minimum because not all strandings or at-sea cases are detected. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates. NA = not applicable.

Fishery	Years	Data Type	Mean Annual Estimated Mortality and Serious Injury Based on Observer Data	5-year Minimum Count Based on Stranding, At-Sea, and/or MMAP Data
Shrimp Trawl	2010–2014	Observer Data	Undetermined for this stock (see Shrimp Trawl section)	NA
Menhaden Purse Seine	2013–2017	MMAP fisherman self-reported takes	NA	0
Stone Crab Trap/Pot	2013–2017	Stranding Data and At-Sea Observations	NA	0
Blue Crab Trap/Pot	2013–2017	Stranding Data and At-Sea Observations	NA	0
Hook and Line	2013–2017	Stranding Data and At-Sea Observations	NA	0
Mean Annual Mortality due to commercial fisheries (2013–2017)			**Unknown**	
Research Takes (5-year Count)			0	

Fishery	Years	Data Type	Mean Annual Estimated Mortality and Serious Injury Based on Observer Data	5-year Minimum Count Based on Stranding, At-Sea, and/or MMAP Data
Other Takes (gunshot wound; 5-year Count)			0	
Mean Annual Mortality due to research and other takes (2013–2017)			**0**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2013–2017)			**Unknown**	

Strandings

From 2013 to 2017, 33 common bottlenose dolphins were reported stranded within the St. Joseph Bay Stock area (Table 2; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 June 2018). This particular BSE stock includes nearshore coastal waters within its boundaries, and hence strandings that occurred along the coast within the bounds of this stock are also included in the total. However, because much of the stock area is contiguous, without physical barriers, with the Northern Coastal Stock of common bottlenose dolphins, the stock of origin for animals that strand within the St. Joseph Bay Stock area is uncertain. Nine of the strandings were also included in the stranding total for the Northern Coastal Stock. It could not be determined if there was evidence of human interaction for these strandings. Stranding data 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 among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

Table 2. Common bottlenose dolphin strandings occurring in the St Joseph Bay Stock area from 2013 to 2017, 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. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 13 June 2018). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2013	2014	2015	2016	2017	Total
St. Joseph Bay Stock	Total Stranded	0	4[a]	9	19	1	33
	HI--Yes	-	0	0	0	0	0
	HI--No	-	0	0	0	0	0
	HI--CBD	-	4	9	19	1	33

[a] Three of the four strandings were part of the Northern Gulf of Mexico UME.

St. Joseph Bay has been affected by four recent unusual mortality events (UMEs) and was the geographic focus of a UME in 2004. First, between August 1999 and May 2000, 150 common bottlenose dolphins died coincident with *K. brevis* blooms and fish kills in the Florida Panhandle. This UME started in St. Joseph Bay and was concurrent spatially and temporally with a *K. brevis* bloom that spread east to west. There were 43 common bottlenose dolphin strandings within the St. Joseph Bay Stock area during this event, which accounted for about 29% of the total common bottlenose dolphin strandings for the 1999–2000 UME. Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012; Litz *et al.* 2014). Second, in March and April 2004, in another Florida Panhandle UME attributed to *K. brevis* blooms, 105 common bottlenose dolphins and two unidentified dolphins stranded dead (Litz *et al.* 2014). This event also started in St. Joseph Bay, and 81 (76%) common bottlenose dolphins stranded in the St. Joseph Bay Stock area. Although there was no indication of a *K. brevis* bloom at the time, high levels of brevetoxin were found in the stomach contents of the stranded dolphins (Flewelling *et al.* 2005; Twiner *et al.* 2012). Third, a separate UME was declared in the Florida Panhandle after elevated numbers of dolphin strandings occurred in association with a *K. brevis* bloom in September 2005. Dolphin strandings remained elevated through the spring of 2006 and brevetoxin was again detected in the tissues of most of the stranded dolphins. Between September 2005 and April 2006 when the event was officially declared over, a total of 88 common bottlenose dolphin strandings occurred (plus strandings of five unidentified dolphins), with 12 (13%) occurring within the St. Joseph Bay Stock area. Brevetoxin was determined to be the cause of this event (Twiner *et al.* 2012; Litz *et al.* 2014). Health assessments of dolphins in the stock area found an eosinophilia syndrome, which could over the long-term produce organ damage and alter immunological status and thereby increase vulnerability to other challenges (Schwacke *et al.* 2010). However, the significance of the high prevalence of the syndrome to the observed mortality events in the St. Joseph Bay area is unclear. Finally, 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). This UME included cetaceans that stranded prior to the DWH oil spill (see Habitat Issues section), 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, but strandings in St. Joseph Bay during this time were not attributed to the oil spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015; Colegrove *et al.* 2016; DWH NRDAT 2016; see Habitat Issues section).

HABITAT ISSUES

The *Deepwater Horizon* MC252 drilling platform, located approximately 80 km southeast of the Mississippi River Delta in waters about 1500 m deep, exploded on 20 April 2010. The rig sank, and over 87 days up to ~3.2 million barrels of oil were discharged from the wellhead until it was capped on 15 July 2010 (DWH NRDAT 2016). Some heavy to moderate oiling occurred on Alabama and Florida beaches, with the heaviest stretch occurring from Dauphin Island, Alabama, to Gulf Breeze, Florida. Light to trace oil was reported from Gulf Breeze to Panama City, Florida (OSAT-2 2011; Michel *et al.* 2013). The maximum shoreline oiling experienced by the St. Joseph Bay stock area was very light oiling in parts of the stock area (Michel *et al.* 2013).

A suite of research efforts was conducted after the oil spill. Studies were initiated in Barataria Bay, Chandeleur Sound, Mississippi Sound, and St. Joseph Bay to assess potential injuries to dolphin stocks within the geographic range of the spill. However, after February 2011, NRDA studies in St. Joseph Bay were discontinued due to the minimal oiling in the St. Joseph Bay area (Mullin *et al.* 2017) and no deaths in St. Joseph Bay during the spill time period were attributed to oil (DWH NRDAT 2016).

Environmental contaminants have been an issue of concern for common bottlenose dolphins throughout the southeastern U.S., including St. Joseph Bay, prior to the DWH oil spill. Kucklick *et al.* (2011) examined POPs (PCBs, chlordanes, mirex, DDTs, HCB and dieldrin) and polybrominated diphenyl ether (PBDE) concentrations from common bottlenose dolphin blubber samples collected during 2000–2007 from 14 locations, including St. Joseph Bay, along the U.S. Atlantic and Gulf coasts and Bermuda. Dolphins from both rural and urban estuarine and coastal waters were sampled. Dolphins sampled from St. Joseph Bay had relatively lower concentrations of some pollutants, like PBDEs, mirex, chlordanes, and HCB, and more intermediate concentrations of DDT, dieldrin, and PCBs when compared to dolphins sampled from the other 13 locations (Kucklick *et al.* 2011). The more recent work of Balmer *et al.* (2015), which was in response to the DWH oil spill and involved collecting remote biopsy samples at six northern Gulf study sites with varying levels of oiling during 2010–2011, found similar or lower levels of POPs and PBDEs in St. Joseph Bay when compared to the results of Kucklick *et al.* (2011).

According to the Florida Department of Environmental Protection (FDEP 2008), the greatest habitat concerns for St. Joseph Bay are declining water quality (mainly due to eutrophication), coastal development, loss of seagrass and saltmarsh habitats, and beach erosion. Several common bottlenose dolphin UMEs in St. Joseph Bay (see Strandings section) have been attributed to harmful algal blooms (*K. brevis*), which are a result of eutrophication. For recent UMEs in the Florida Panhandle (1999–2000, 2004, 2005–2006), the site of bloom origin was not known for all, but it is likely two of the UMEs originated in the St. Joseph Bay area (Twiner *et al.* 2012). Blooms can be transported by currents from adjacent bays and coastal waters, so eutrophication anywhere along the Florida Panhandle can impact St. Joseph Bay, and events originating in St. Joseph Bay can impact the entire Panhandle. Loss of seagrass habitat within St. Joseph Bay has been attributed to eutrophication, storms, and an increase in propeller scar damage (FDEP 2008; Wren and Yarbro 2016). The Florida Fish and Wildlife Conservation Commission (FWC) found that seagrass cover, or density, appears to be declining in St. Joseph Bay, and reported propeller scarring to be "extensive" (Wren and Yarbro 2016). Salt marshes in the southeastern U.S. have experienced unparalleled die-offs in recent years (Silliman *et al.* 2005). The shoreline of St. Joseph Bay is bordered by salt marsh habitat, and in the 1990s the salt marsh began showing signs of stress and began dying off. Studies by FWC's Fish and Wildlife Research Institute suggested the die-off resulted from an unidentified pathogen, but also may have been linked to a drought (FDEP

2008). Beginning in 1995 with Hurricane Opal, repetitive damaging storms have eroded beaches of the St. Joseph Peninsula, with Cape San Blas being one of the most severely eroding areas in Florida (FDEP 2008). Coastal development (of residences) is steadily growing along the St. Joseph peninsula and around the bay, which will lead to additional pressure on the area's local natural resources (FDEP 2008).

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act, and the St. Joseph Bay Stock is not a strategic stock under the MMPA. The total human-caused mortality and serious injury for this stock is unknown and there is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. The status of this stock relative to OSP is unknown. There was no evidence of a trend in population size for this stock.

Although this stock does not meet the criteria to qualify as strategic (NMFS 2016), NMFS has concerns regarding this stock due to the small stock size and the high number of common bottlenose dolphin deaths associated with UMEs in the Florida panhandle since 1999.

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APPENDIX I: Estimated serious injury and mortality (SI&M) of Western North Atlantic marine mammals listed by U.S. observed fisheries. Marine mammal species with zero (0) observed SI&M are not shown in this table. (unk = unknown).

Category, Fishery, Species	Yrs. 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	2013-2017	.11, .18, .14, .10, .12	0, 0, 0, 0, 7	399(.33), 128(.27), 177(.28), 125(.34), 129(.28)	193(.16)	851
Atlantic white-sided dolphin	2013-2017	.11, .18, .14, .10, .12	0, 0, 0, 0, 0	4(1.03), 10(.66), 0, 0, 0	2.8 (.56)	544
Common dolphin	2013-2017	.11, .18, .14, .10, .12	0, 0, 0, 0, 0	104(.46), 111(.47), 55(.54), 80(.38), 133(.28)	97(.19)	1,452
Risso's dolphin	2013-2017	.11, .18, .14, .10, .12	0, 0, 0, 0, 0	23(1.0), 0, 0, 0, 0	5.8 (.79)	303
Bottlenose dolphin (offshore)	2013-2017	.11, .18, .14, .10, .12		26(.95), 0, 0, 0, 8	7.0 (.76)	519
Harbor seal	2012-2016	.11, .18, .14, .10, .12	0, 0, 0, 0, 0	142(.31), 390(.39), 474(.17), 245(.29), 298(.18)	311 (.13)	2,006

Gray seal	2013-2017	.11, .18, .14, .10, .12	0, 0, 0, 0, 0	1127(.20), 917(.14), 1021(.25), 498(.33), 930(.16)	899 (.09)	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	2013-2017	.03, .05 .06, .08, .09	0, 0, 27, 0, 0	19(1.06), 22(1.03), 33(1.16), 23(.64), 9(.95)	21(.49)	851
Common dolphin	2013-2017	.03, .05 .06, .08, .09	0, 0, 0, 0, 11	62(.67), 17(.86), 30(.55), 7(.97), 11(.71)	18(.25)	1,452
Harbor seal	2013-2017	.03, .05 .06, .08, .09	0, 0, 0, 0, 0	0, 19(1.06), 48(.52), 18(.95), 3(.18)	18(.41)	2,006
Gray Seal	2013-2017	.03, .05 .06, .08, .09	0, 0, 0, 0, 0	0, 22(1.09), 15(1.04), 7(.93), 0	9 (.67)	1,389
Minke Whale	2013-2017	.03, .05 .06, .08, .09	0, 0, 0, 0, 0	0, 0, 0, 1, 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
Bottlenose dolphin (offshore)	2013-2017	.09, .10, .12 .15, .12	0, 0, 0, 0, 0	0, 0, 0, 0, 0	0	519
Common dolphin	2013-2017	.09, .10, .12, .15, .12	0, 0, 9.05(1), 0, 4.92(1)	0, 0, 0, 0, 0	2.8(.74)	1,452
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	2013-2017	.15, .17 .19, .12, .16	0, 0, 0, 0, 0	4(.96), 11(.63), 0, 0, 0	3 (.52)	2,006
Gray seal	2013-2017	.15, .17 .19, .12, .16	0, 0, 0, 0, 0	20(.37), 19(.45), 23(.46), 0, 16(.24)	16 (.20)	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	0,0,18.6 (0.65) 33.5 (0.89), 0	10.4 (0.62)	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	2013-2017	.15, .17 .19, .12, .16	0, 0, 0, 0, 0	17(.54), 17(.53), 22(.45), 16(.46), 0	14 (.25)	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	2013-2017	.15, .17 .19, .12, .16	0, 0, 0, 0, 0	7(.98), 5.5(.86), 3.7(.49), 0, 0	3.2(.53)	851
Mid-Atlantic Bottom Trawl						
Common dolphin	2013-2017	.06, .08 .09, .10, .10	0, 24, 0, 0, 0	254(.29), 305(.29), 250(.32), 177(.33), 380(.23)	278 (.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), 319.51)	37 (.29)	303
Bottlenose dolphin (offshore)	2013-2017	.06, .08, .09, .10, .10	0, 0, 0, 0, 0	0, 25 (.66), 0, 7.3 (0.93), 22.1 (.66)	10.9(.42)	519
Harbor seal	2013-2017	.06, .08 .09, .10, .10	0, 0, 0, 0, 0	11(.96), 10(.95), 7, 0, 0	5.6 (0.56)	2,006
Gray seal	2013-2017	.06, .08 .09, .10, .10	0, 0, 0, 0, 0	25(.67), 7(.96), 0, 26 (.57), 26(.40)	17 (.30)	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	2013-2017	.37, .42, .08, .27, .16	0, 0, 0, 0, 0	0, na, na, na, 0	0.8 (na)	2,006
Gray seal	2013-2017	.37, .42, .08, .27, .16	0, 0, 0, 0, 0	na, 0, 0, 0, 0	0.2 (na)	1,389

Appendix II: Summary of the confirmed anecdotal human-caused mortality and serious injury (SI) events involving baleen whale stocks along the Gulf of Mexico Coast, U.S. East Coast, and adjacent Canadian Maritimes, 2013–2017, with number of events attributed to entanglements or vessel collisions by year.

Stock	Mean annual mortality and 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 (2013, 2014, 2015, 2016, 2017)	Entanglements Injury value against PBR (2013, 2014, 2015, 2016, 2017)	Vessel Collisions Annual rate (U.S. waters / Canadian waters/unknown first sighted in U.S./unknown first sighted in Canada)	Vessel Collisions Confirmed mortalities (2013, 2014, 2015, 2016, 2017)	Vessel Collisions Injury value against PBR (2013, 2014, 2015, 2016, 2017)
Western North Atlantic right whale *(Eubalaena glacialis)*	6.85 (0.8)	5.55 (0.2/ 1.2/ 2.45/ 1.7)	(0, 2, 0, 2, 4)	(.75, 6, 3.5, 7.5, 2)	1.3 (0.5/ 0.8/ 0.0/ 0.0)	(0, 0, 0, 1, 5)	(0, .52, 0, 0, 0)
Gulf of Maine humpback whale *(Megaptera novaeangliae)*	12.15 (22)	7.75 (2.05/ 0.75/ 4.8/ 0.15)	(0, 2, 2, 1, 3)	(4.75, 1.75, 5.5,7.5, 8, 2)	4.4(4.0/ 0.00/ 0.40/ 0.00)	(0, 2, 0, 4, 5)	(0, 0, 0, 2, 0)
Western North Atlantic fin whale *(Balaenoptera physalus)*	2.35 (12)	1.1(0/ 0.4[2]/ 1.1/ 0)	(0, 1, 0, 0, 2)	(1, 1.5, 1, 2.25, 1)	1.4 (1.4/ 0.00/ 0.00/ 0.00)	(1, 2, 0, 0, 1)	0
Nova Scotian sei whale *(B. borealis)*	1.0 (6.2)	0.2 (0/0/0.2/0)	0	(0, 0, 0, 0, 1)	0.8 (0.80/ 0.00/ 0.00/ 0.00)	(0, 3, 0, 0, 1)	0
Canadian East Coast minke whale *(B. acutorostrata)*	8.00 (189)	7.0 (2.9/ 2.3/ 1.45/ 0.35)	(1, 3, 7, 4, 11)	(2.5, 1.75, 2.5, 1.75, 1.5)	1.0 (0.8/ 0.2/ 0.00/ 0.00)	(0, 2, 1, 0, 1)	0

[1] Potential Biological Removal (PBR)
[2] Not in area covered by abundance estimate so excluded from total.

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 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:

I. frequent incidental mortality or serious injury of marine mammals
II. occasional incidental mortality or serious injury of marine mammals
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*.

IV.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 DWF activities off the Northeast coast of the U.S. With implementation of the Magnuson Fisheries Conservation and Management Act (MFCMA) in that year, 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, 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 vessels included 3, 5, 7, 6, 8, and 8 respectively, Japanese longline vessels. 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: a southern, or winter, stratum and 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 (5 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 35N to 41N and 69W to 72W. Approximately 50% of the total effort was within a one degree square at 39N, 72W, 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 (Goudy 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. 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, Washington, D.C. 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

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 estimates of 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.

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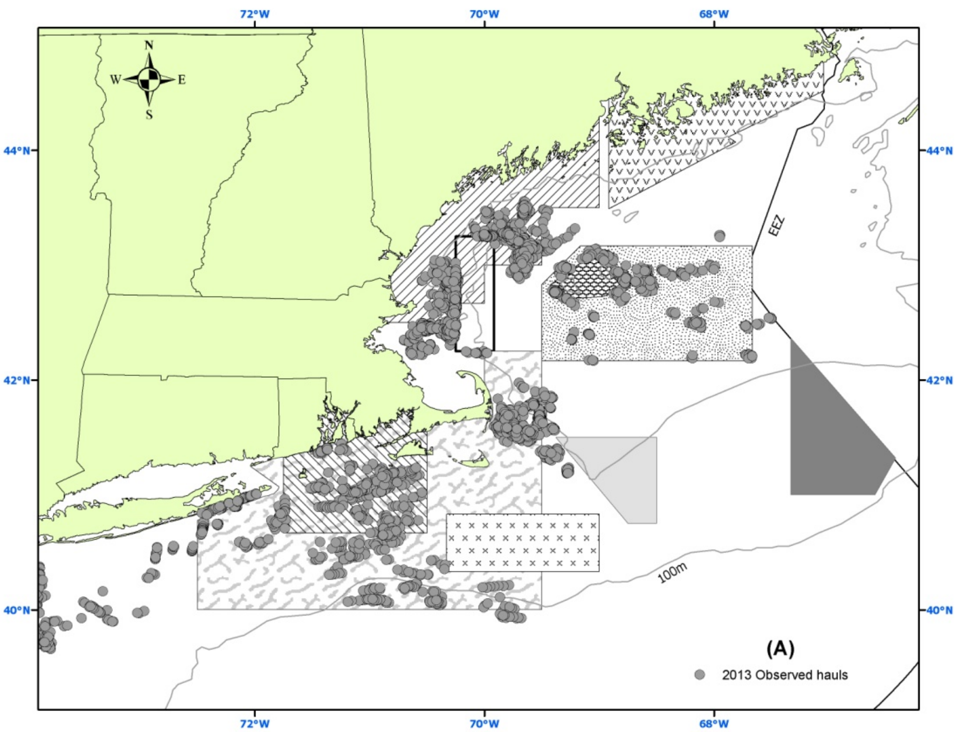
Appendix III: Fishery Descriptions - List of Figures

Figure 1. 2013 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

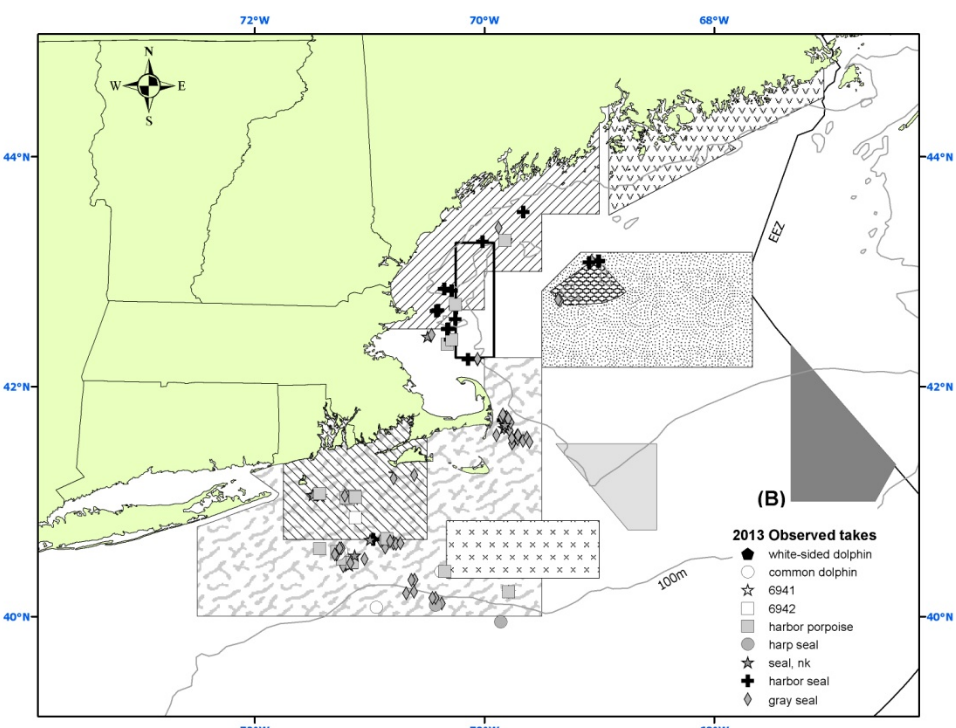


Figure 2. 2014 Northeast sink gillnet observed hauls (A) and observed takes (B).

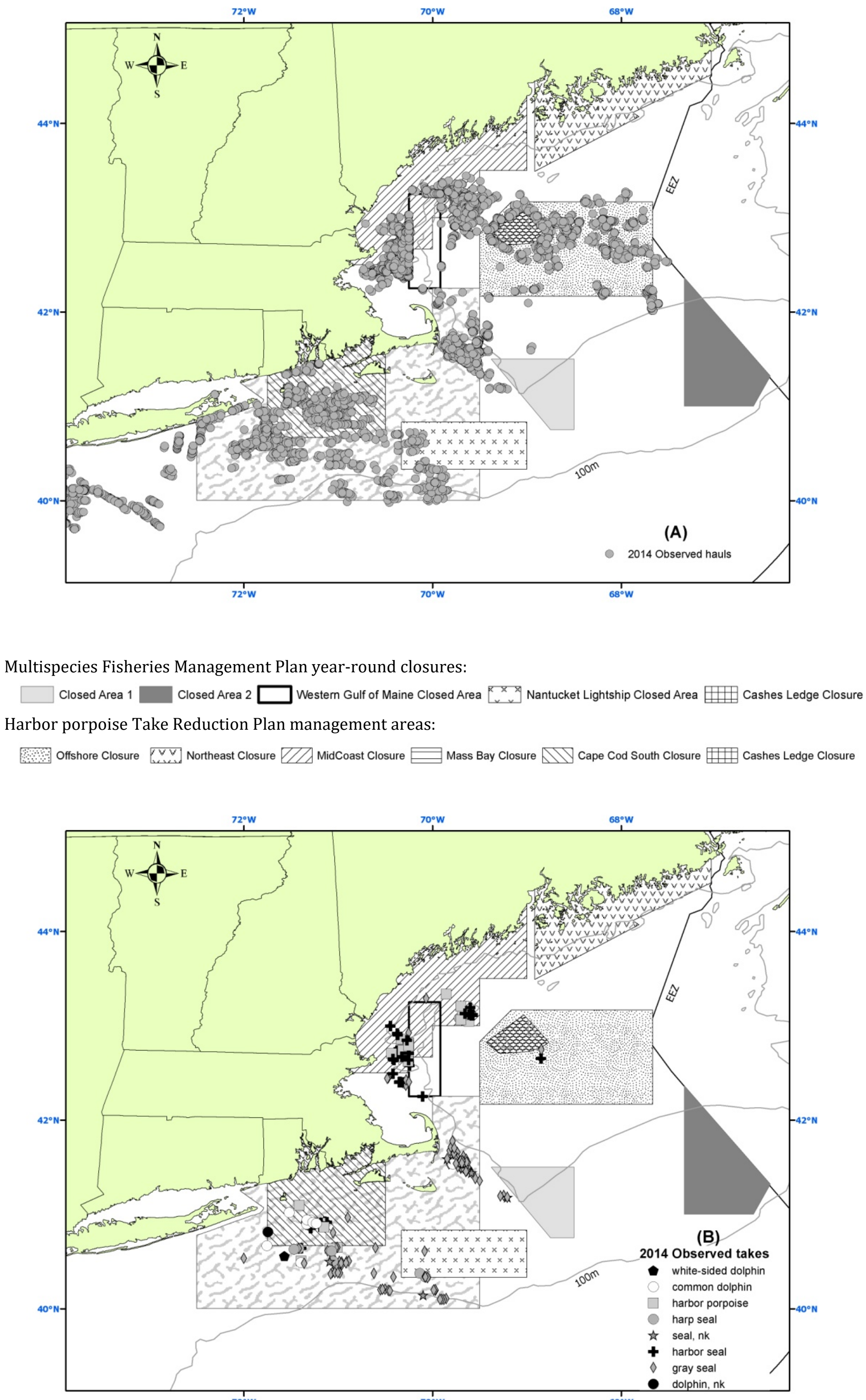


Figure 3. 2015 Northeast sink gillnet observed hauls (A) and observed takes (B).

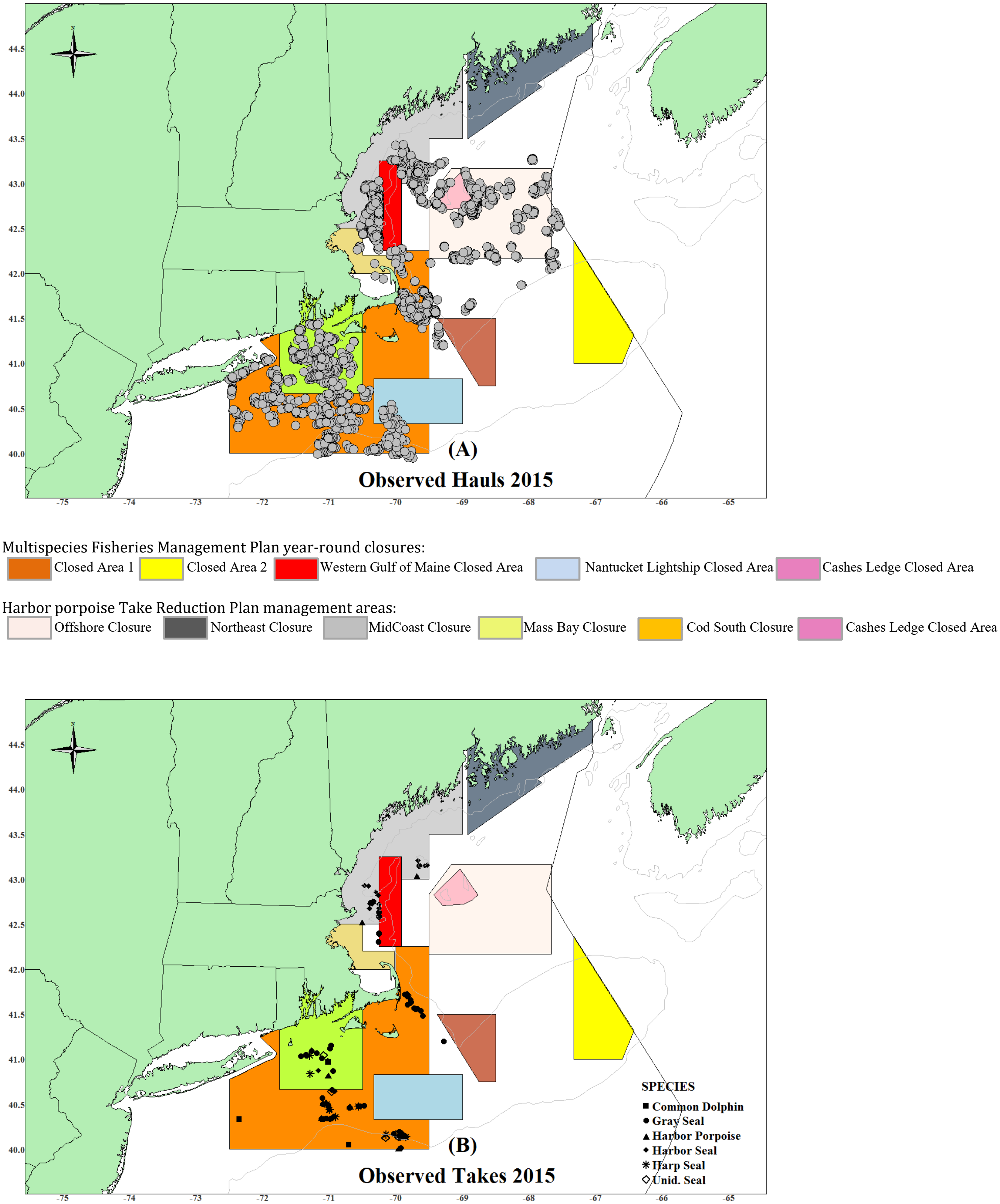


Figure 4. 2016 Northeast sink gillnet observed hauls (A) and observed takes (B).

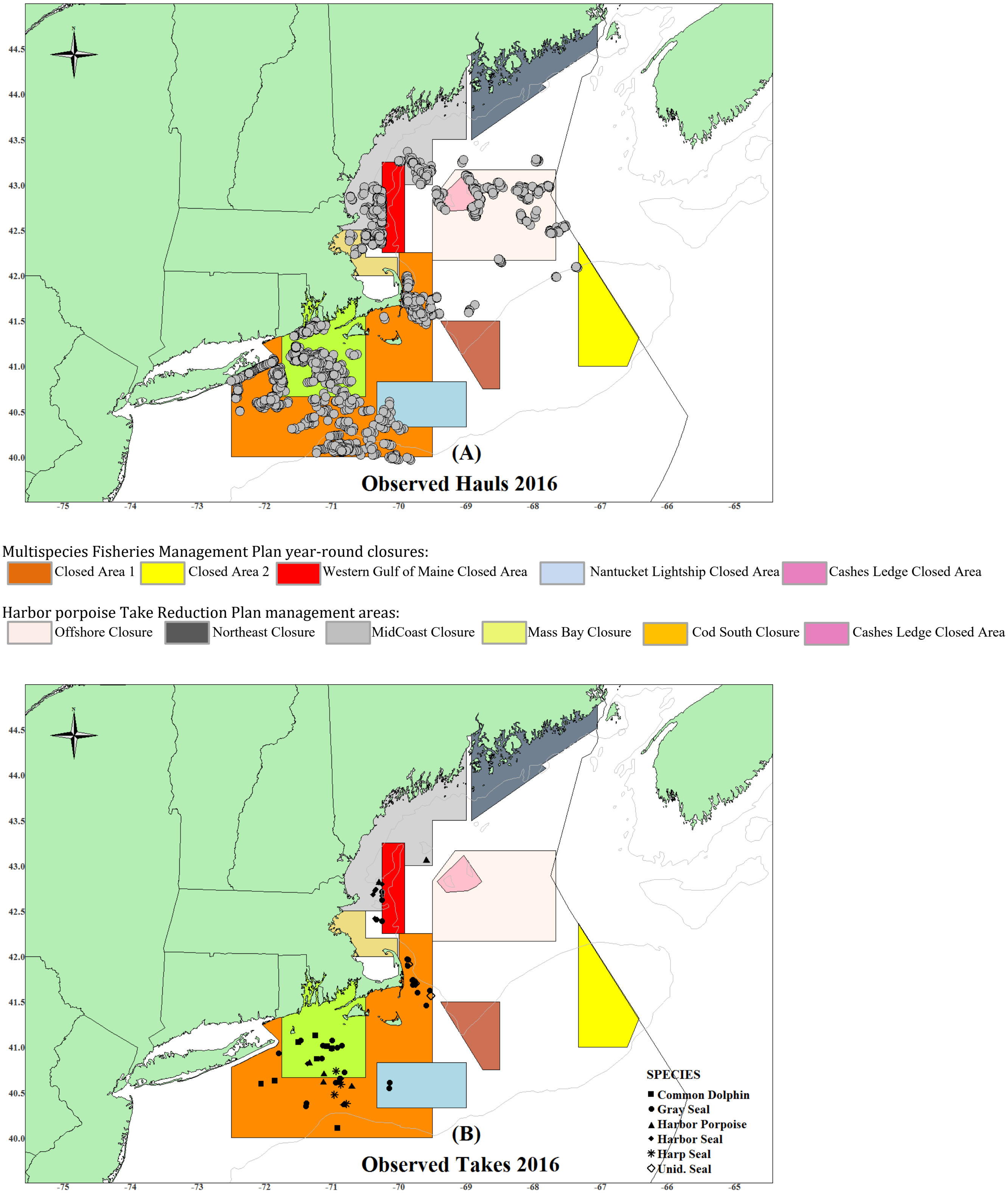
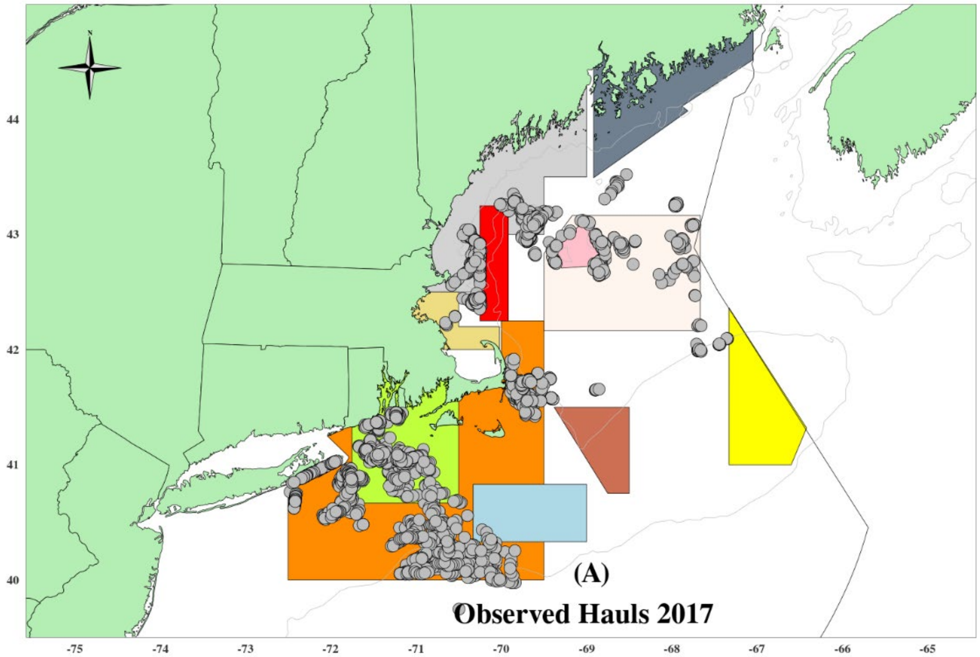


Figure 5. 2017 Northeast sink gillnet observed hauls (A) and observed takes (B).



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

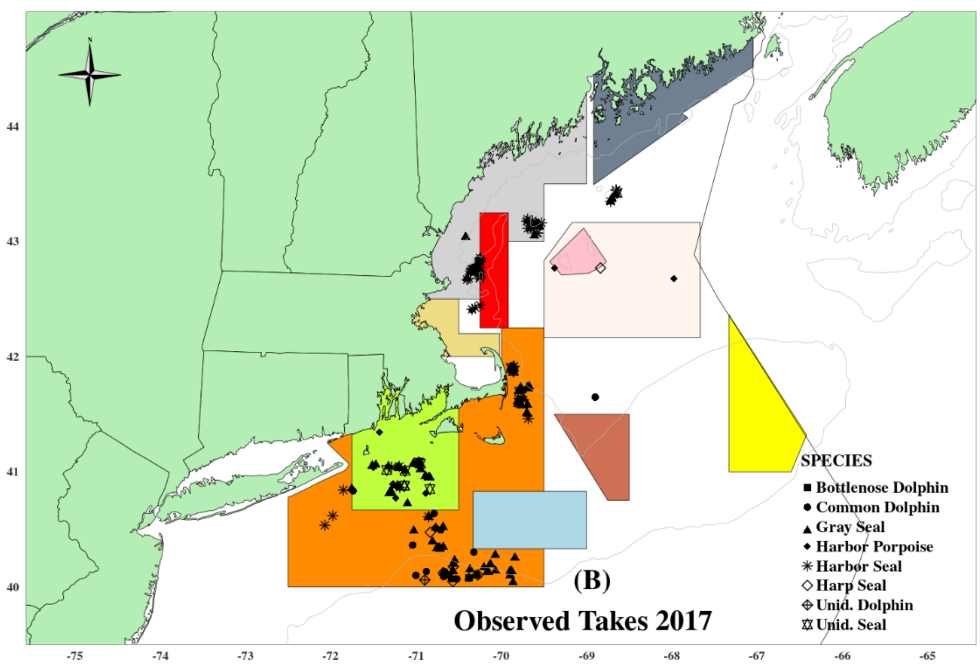
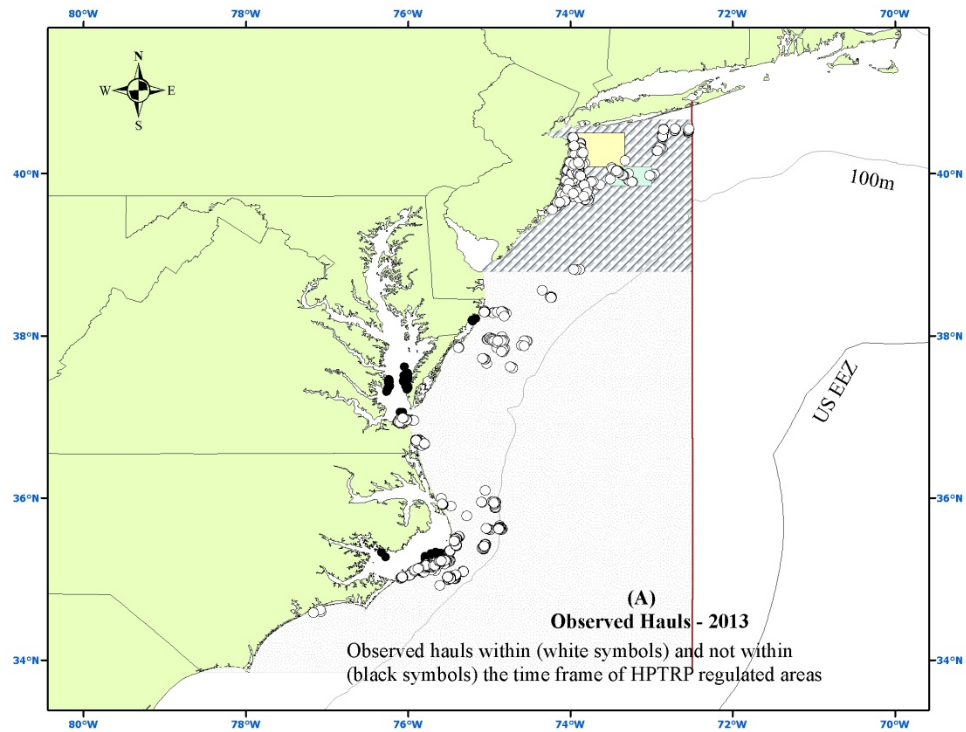


Figure 6. 2013 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

Southern mid-Atlantic waters   New Jersey Mudhole   waters off New Jersey

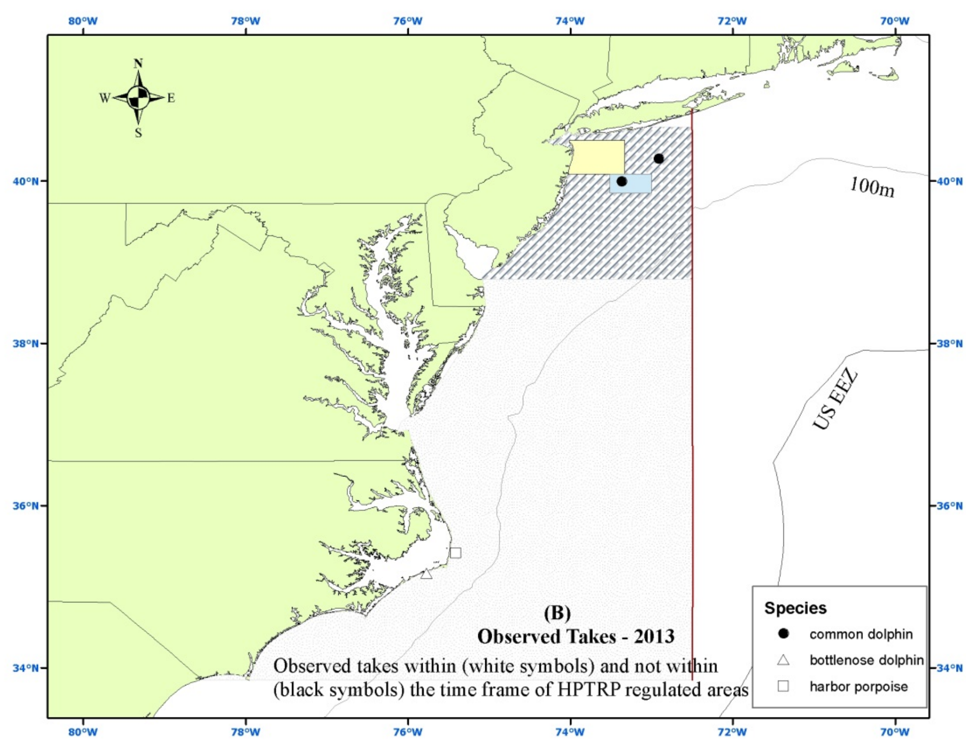
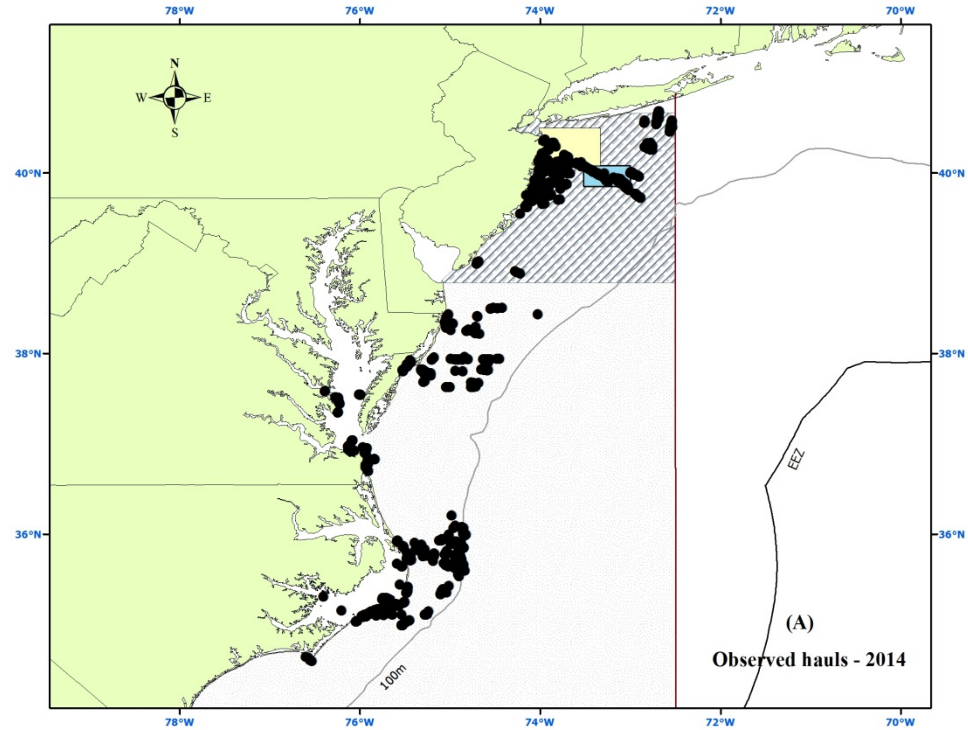


Figure 7. 2014 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

Southern mid-Atlantic waters | New Jersey Mudhole | waters off New Jersey

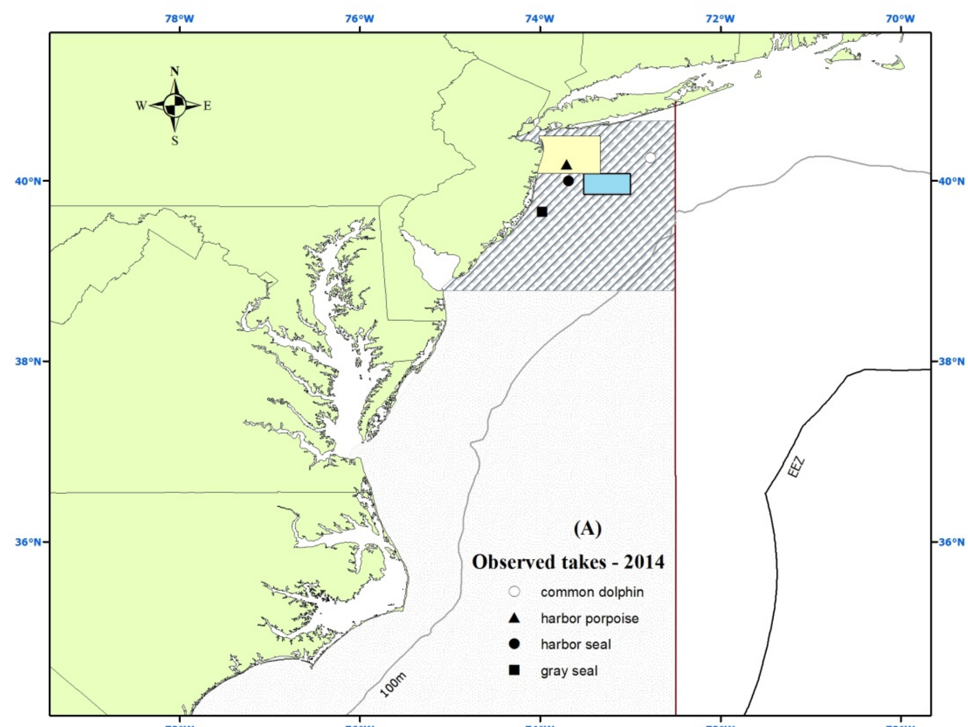
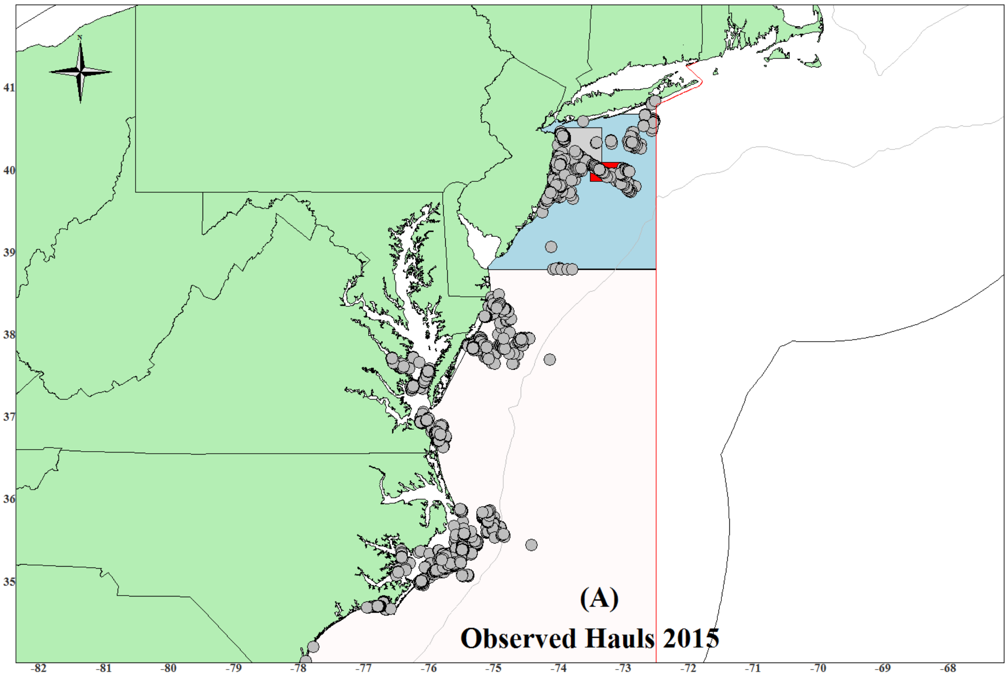


Figure 8. 2015 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters New Jersey Mudhole Mudhole South waters off New Jersey

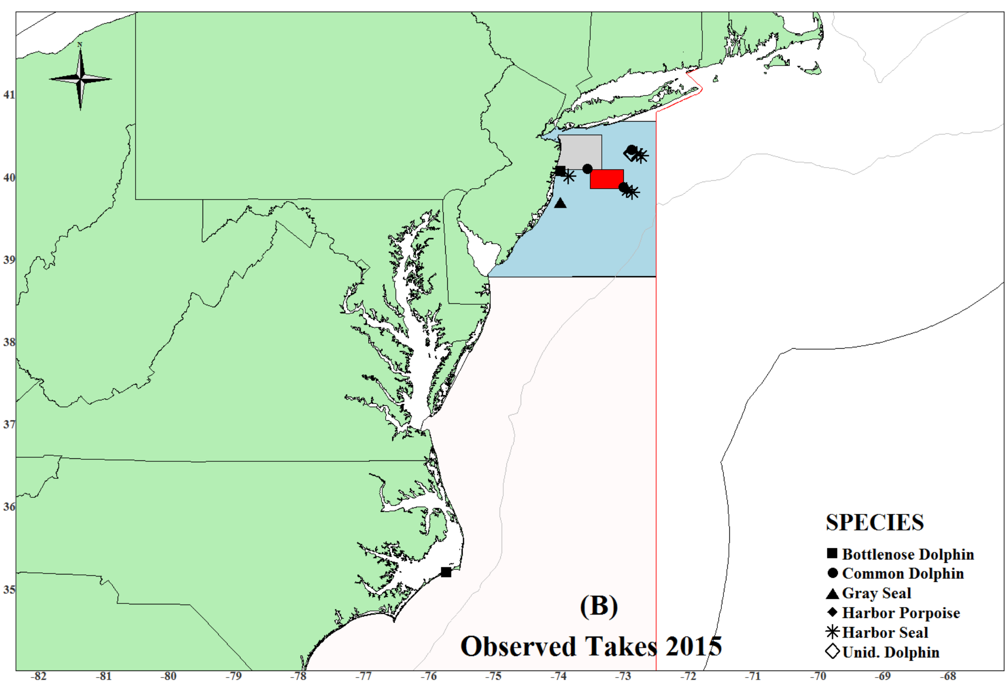
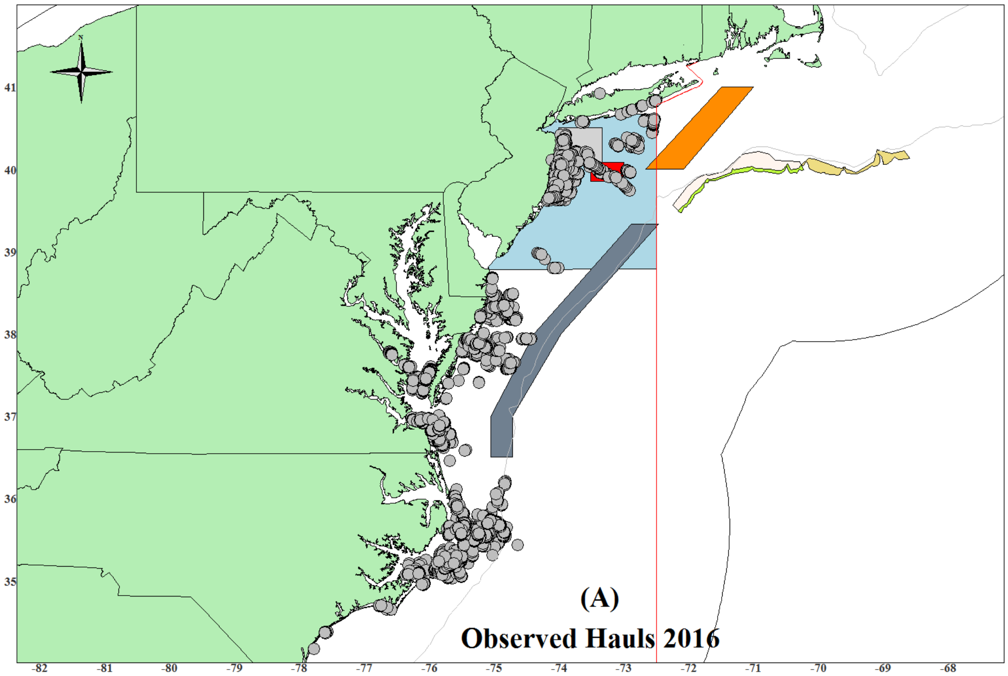


Figure 9. 2016 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters New Jersey Mudhole Mudhole South waters off New Jersey

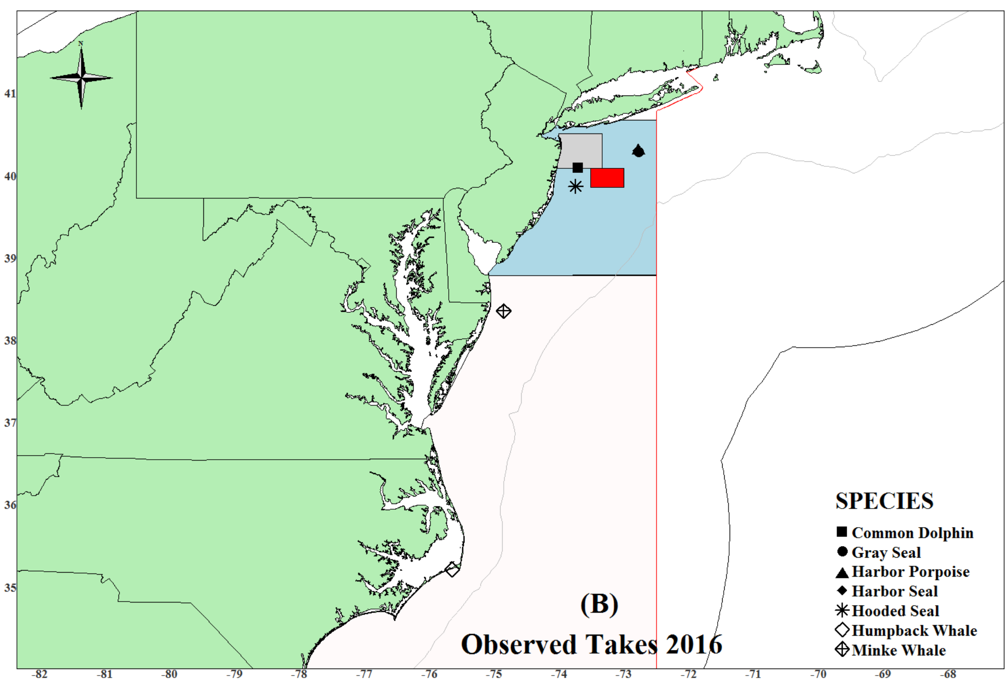
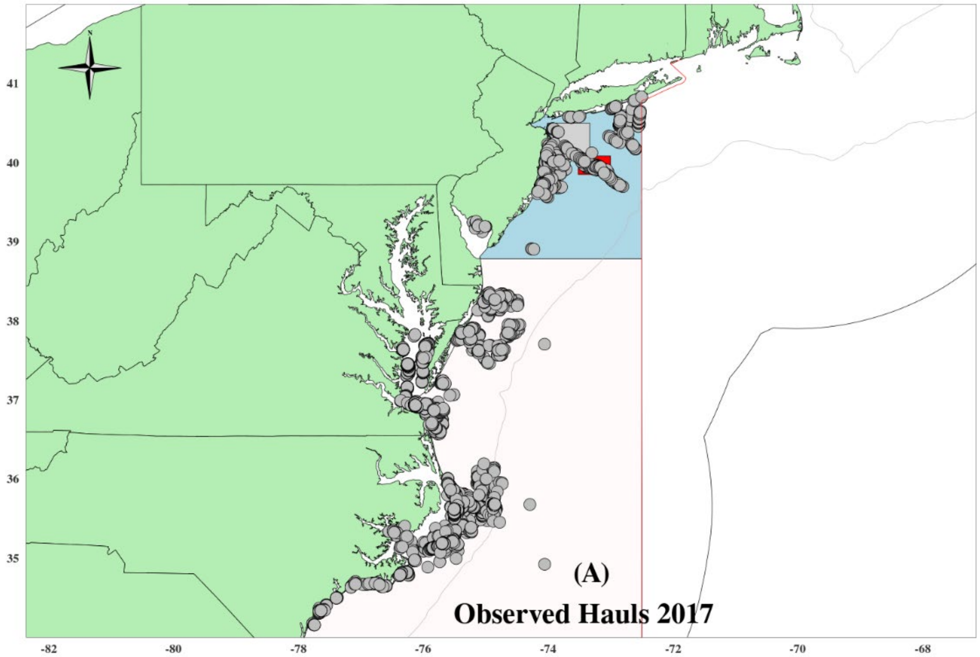


Figure 10. 2017 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:
Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

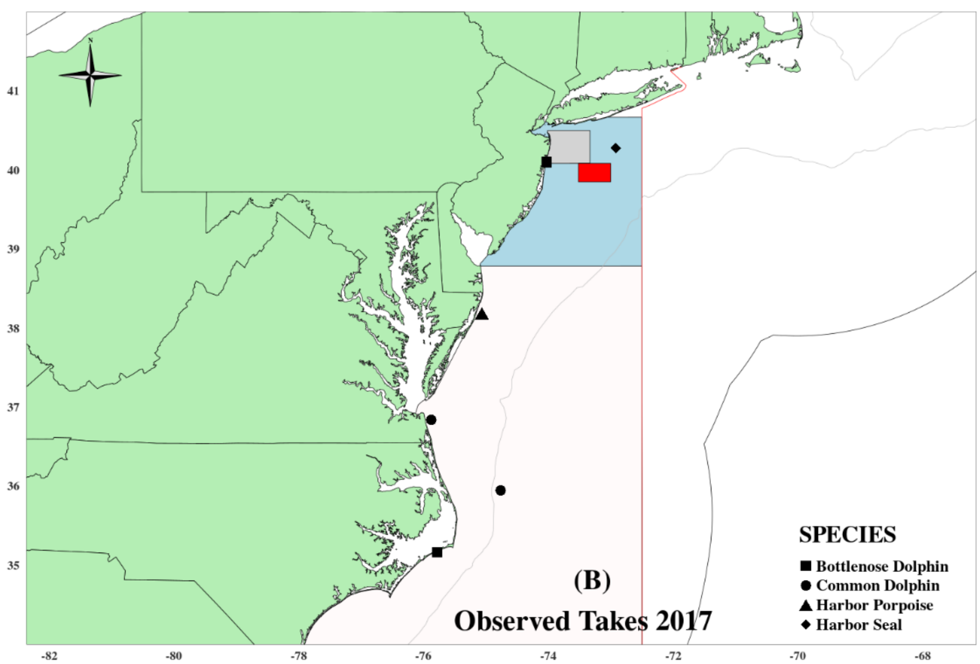


Figure 11. 2013 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

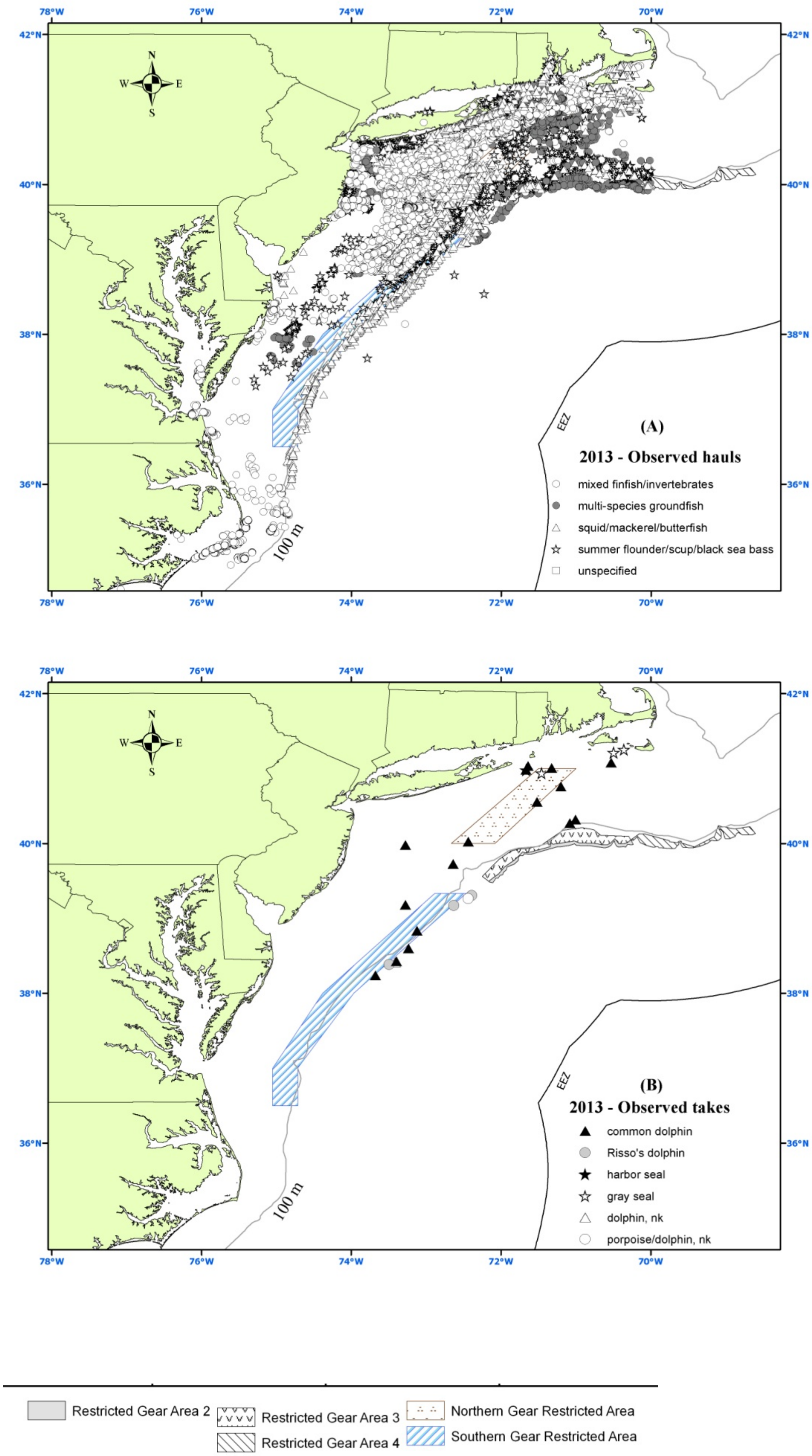


Figure 12. 2014 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

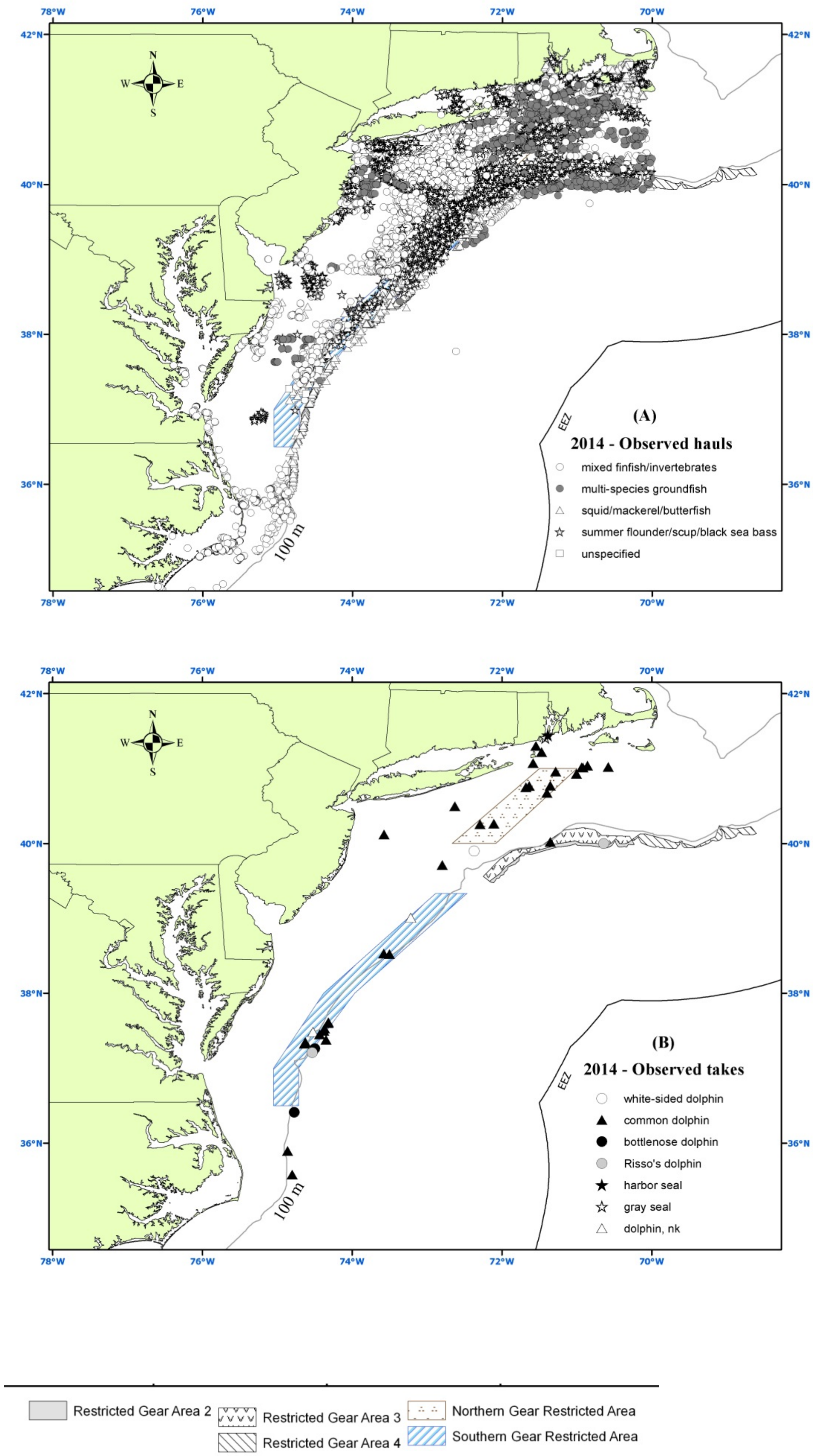
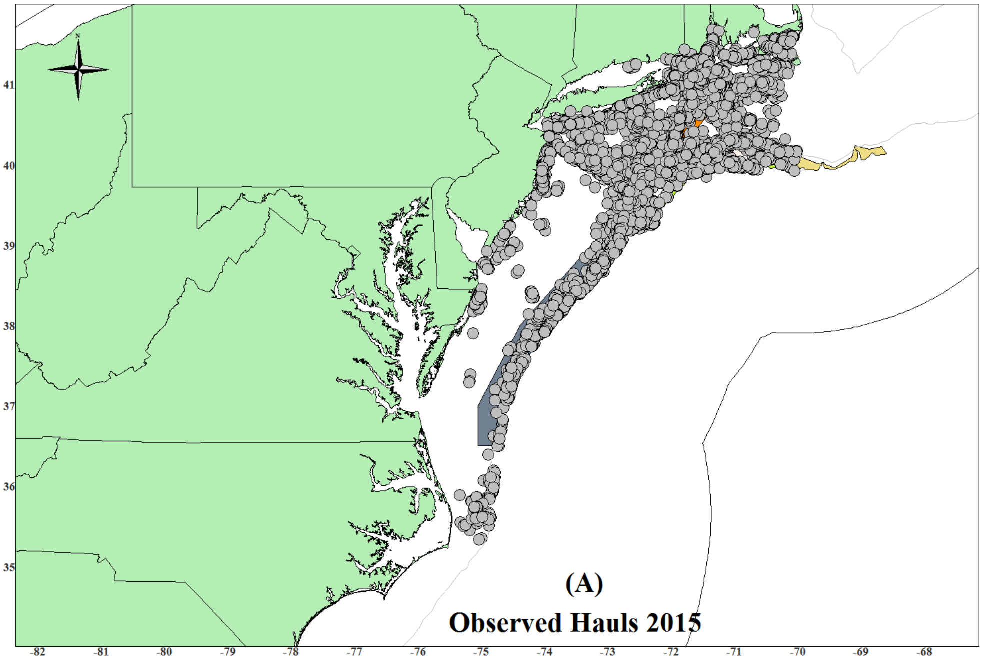


Figure 13. 2015 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).



Southern Gear Restricted Area | Northern Gear Restricted Area

Restricted Area 2 | Restricted Area 3 | Restricted Area 4

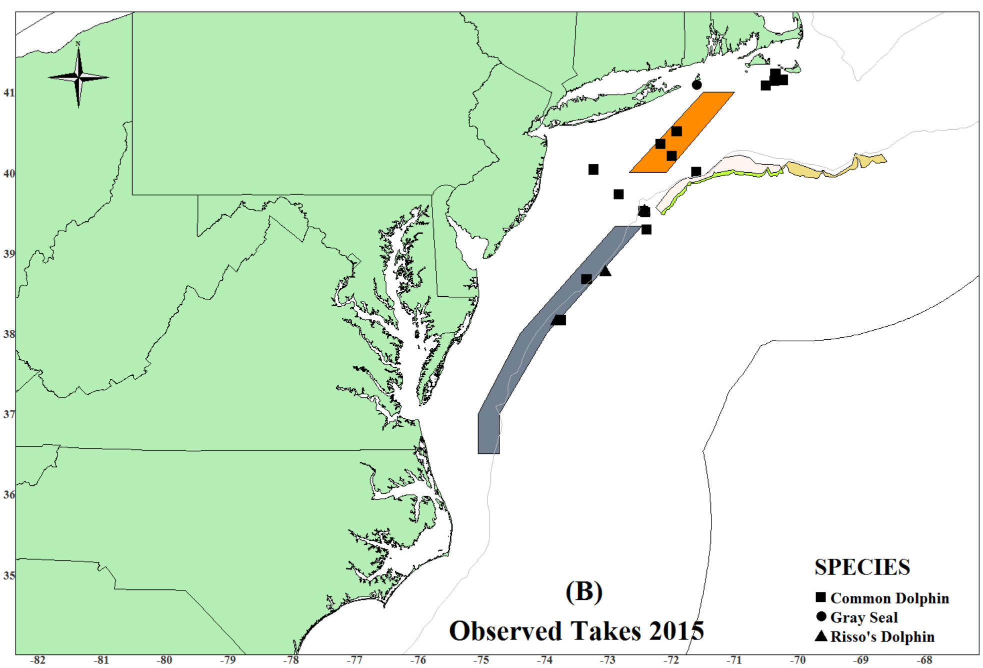


Figure 14. 2016 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

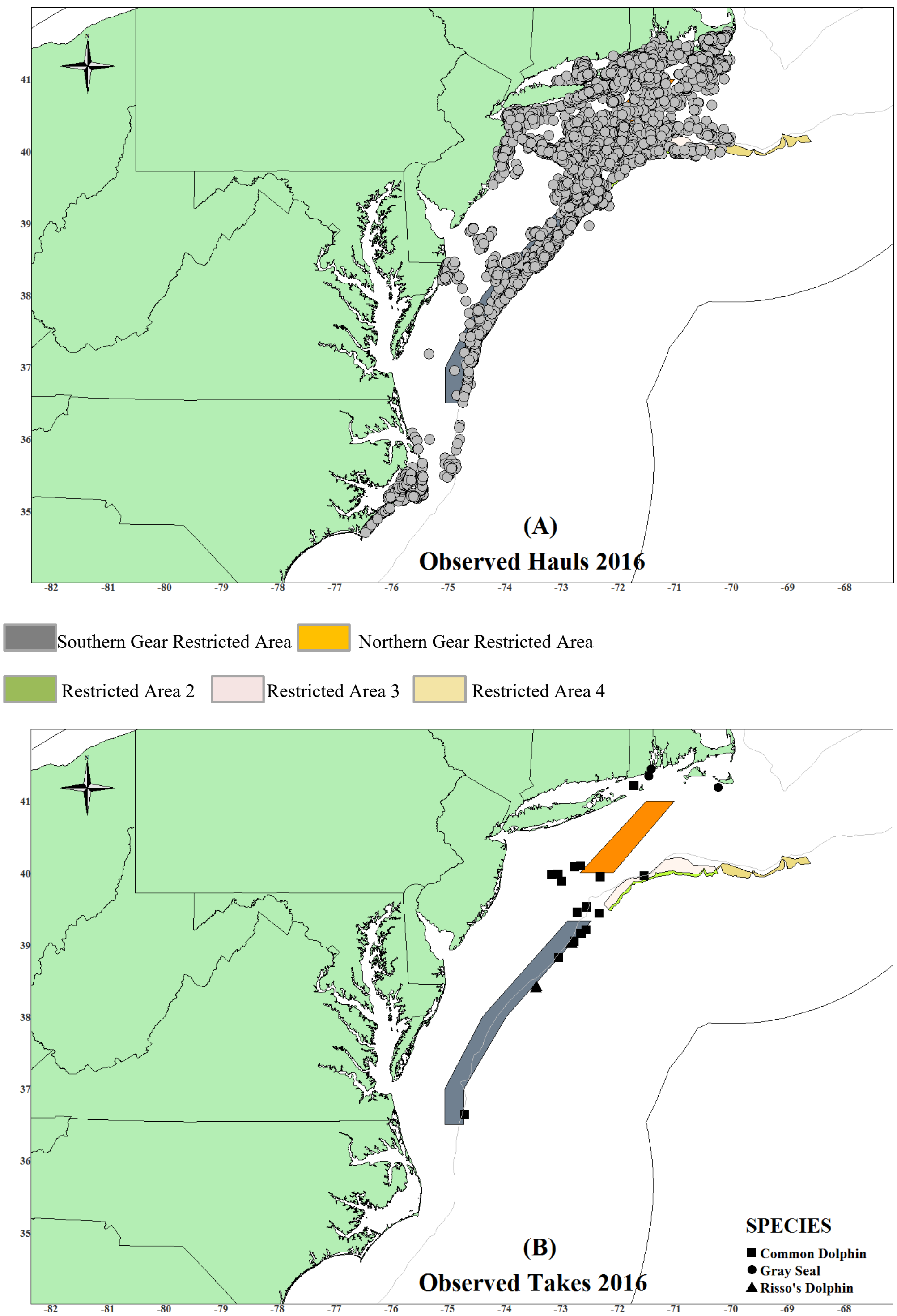


Figure 15. 2017 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

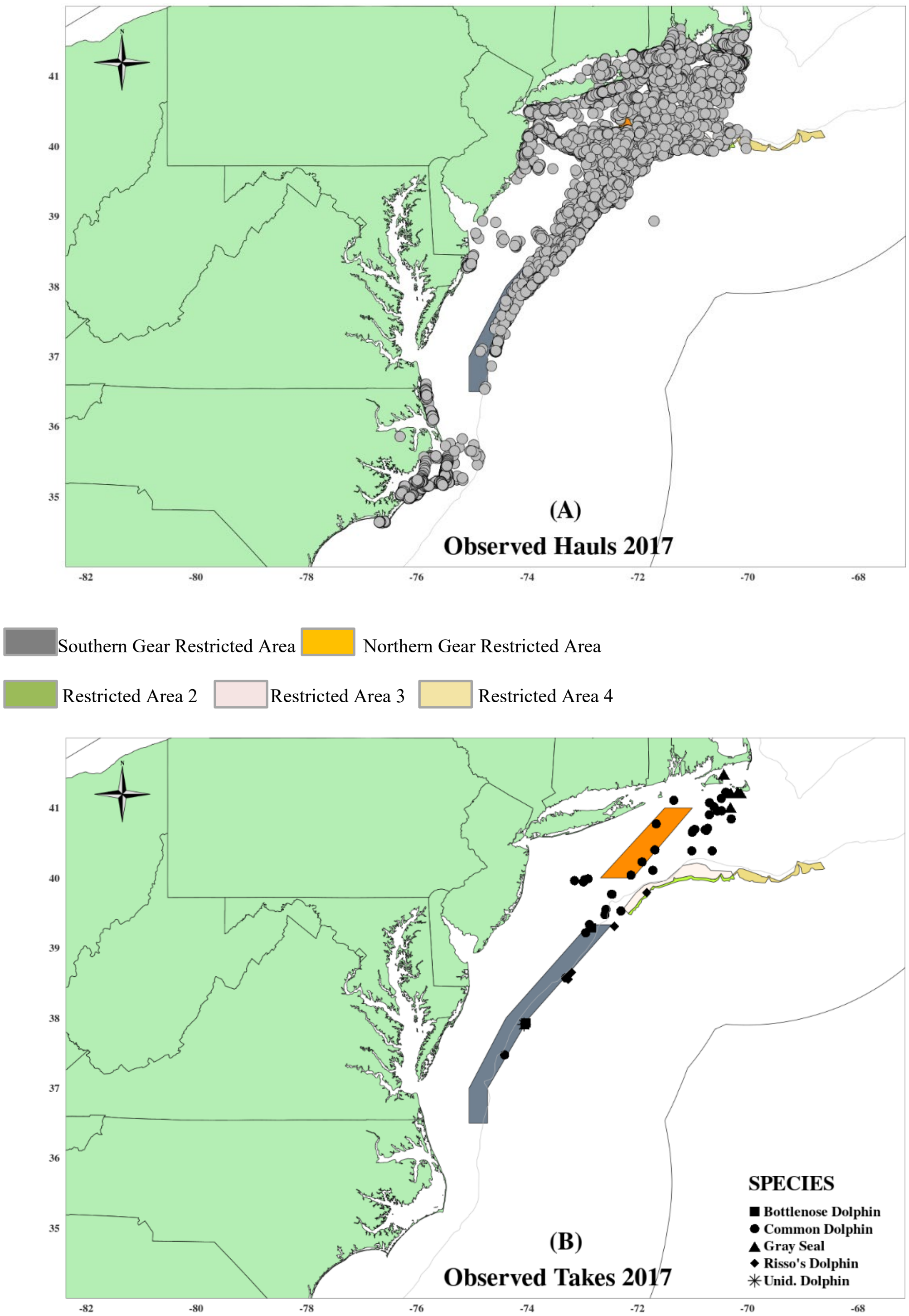


Figure 16. 2013 Northeast bottom trawl observed tows (A) and observed takes (B).

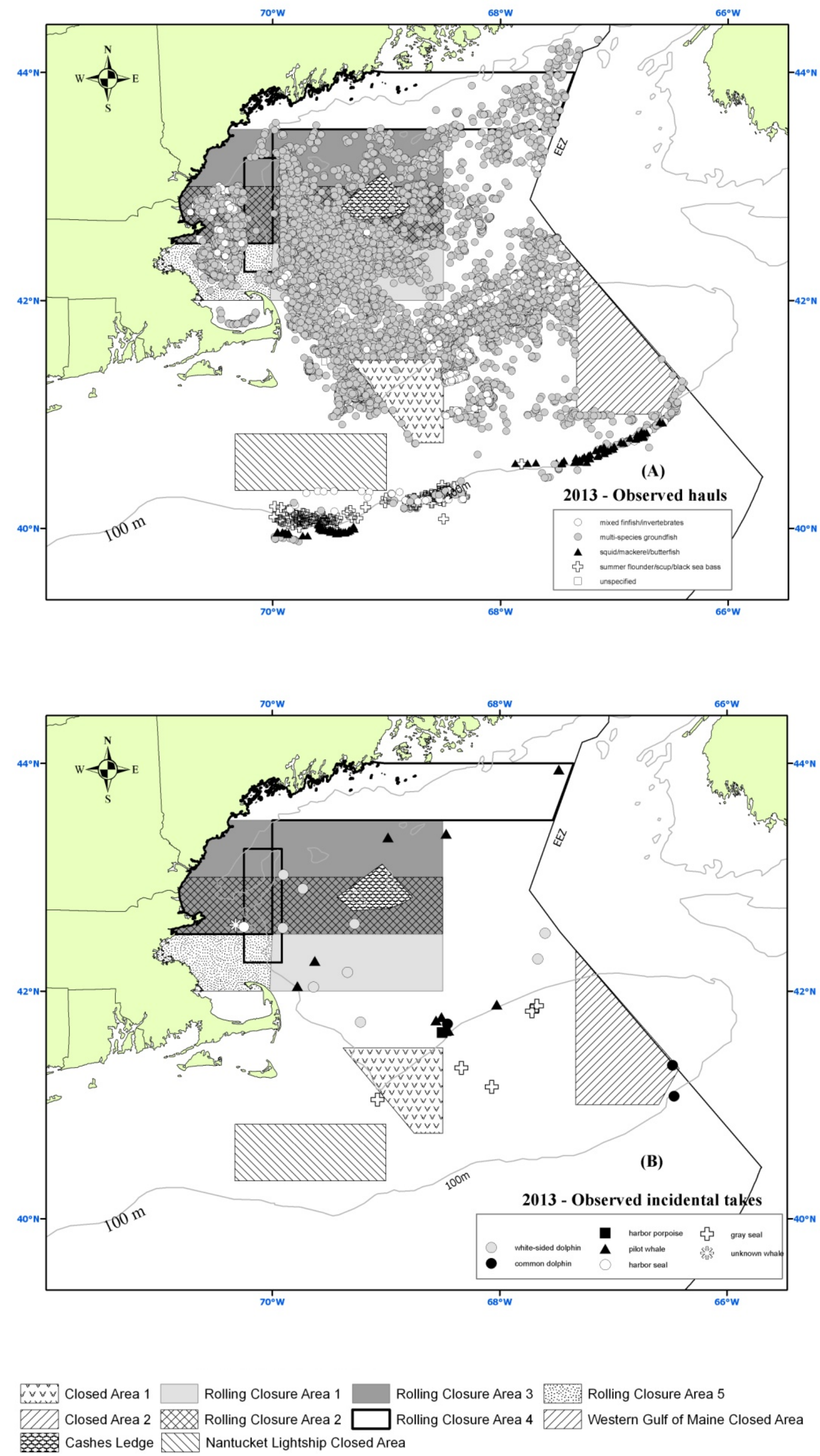


Figure 17. 2014 Northeast bottom trawl observed tows (A) and observed takes (B).

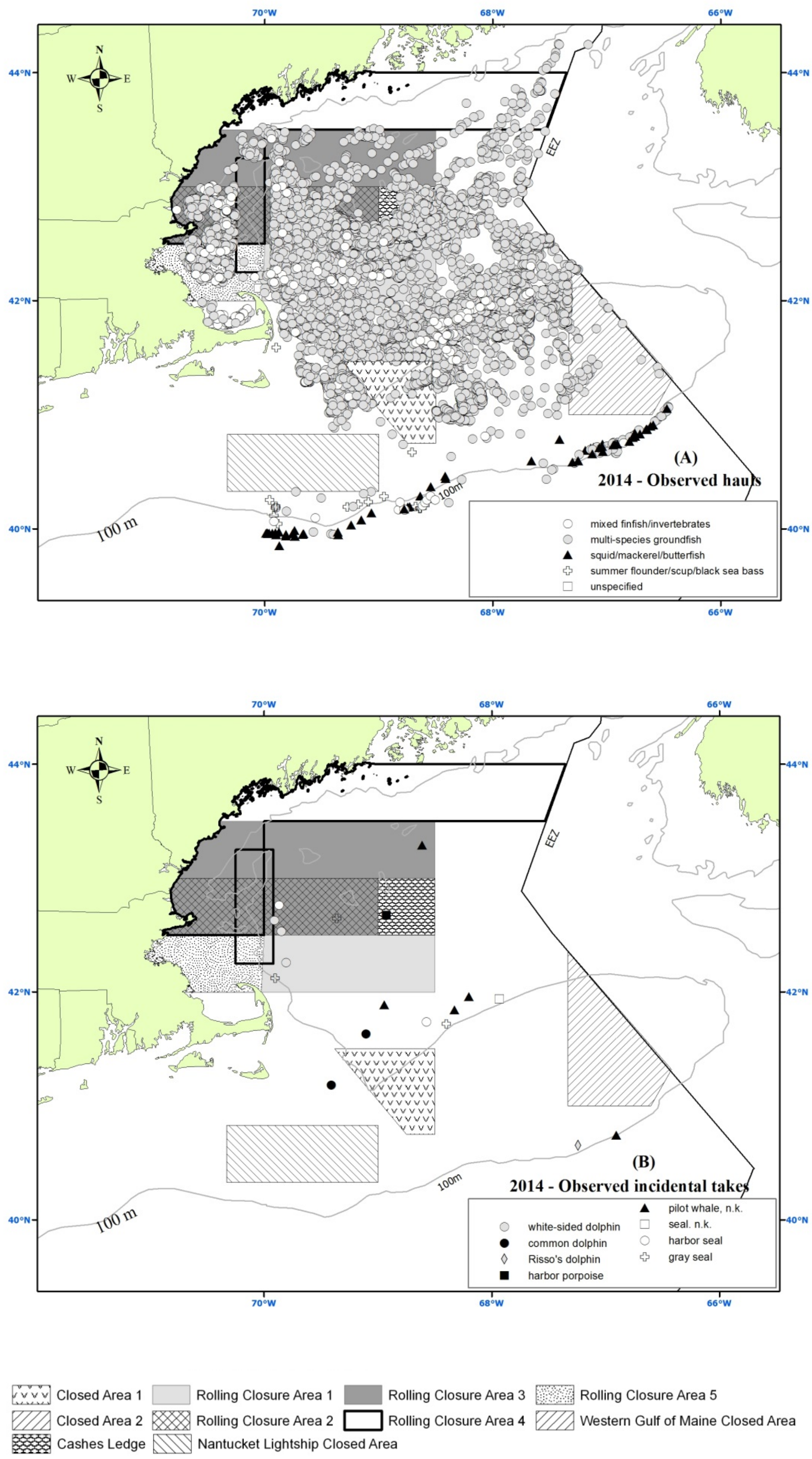


Figure 18. 2015 Northeast bottom trawl observed tows (A) and observed takes (B).

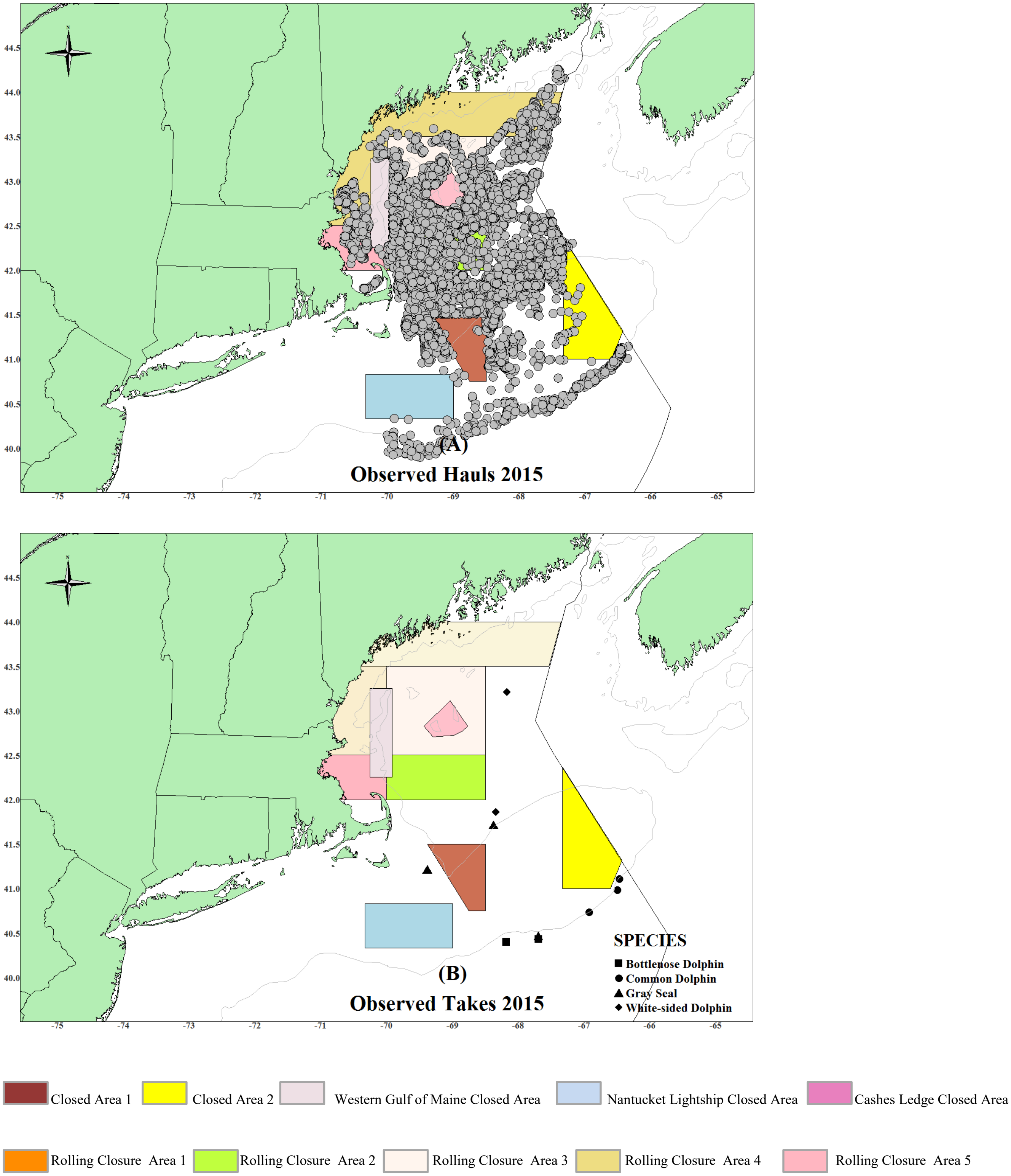


Figure 19. 2016 Northeast bottom trawl observed tows (A) and observed takes (B).

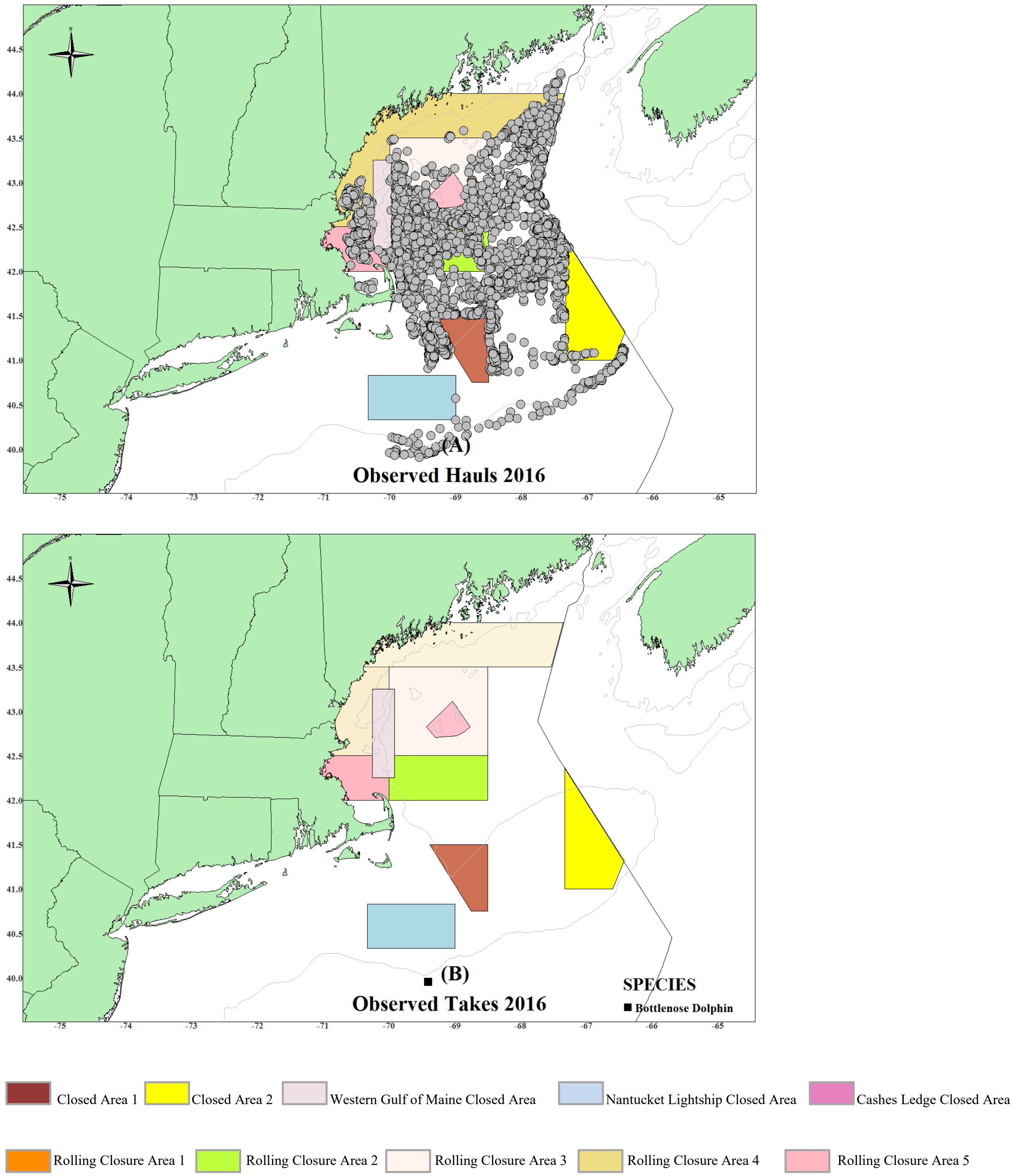


Figure 20. 2017 Northeast bottom trawl observed tows (A) and observed takes (B).

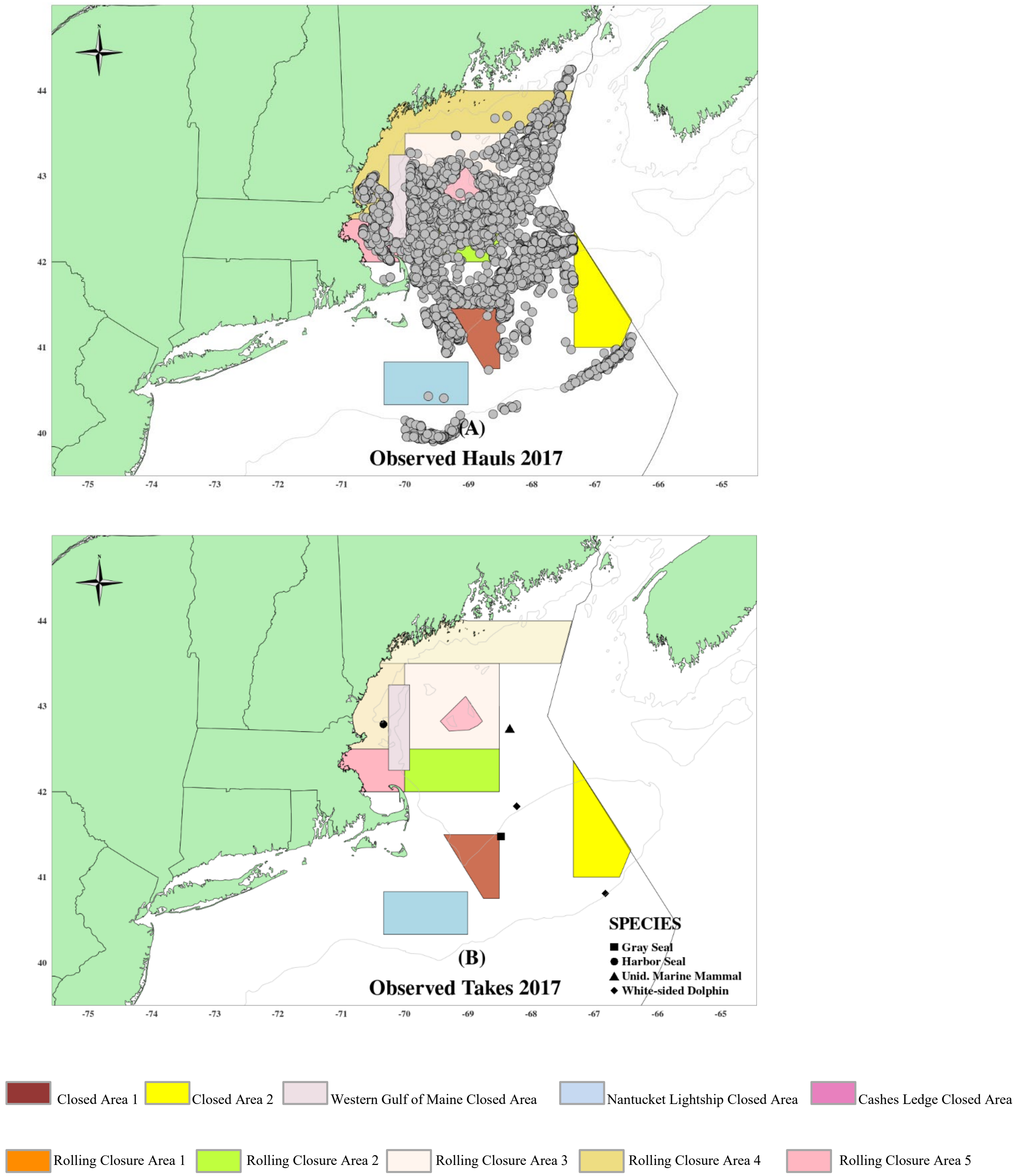
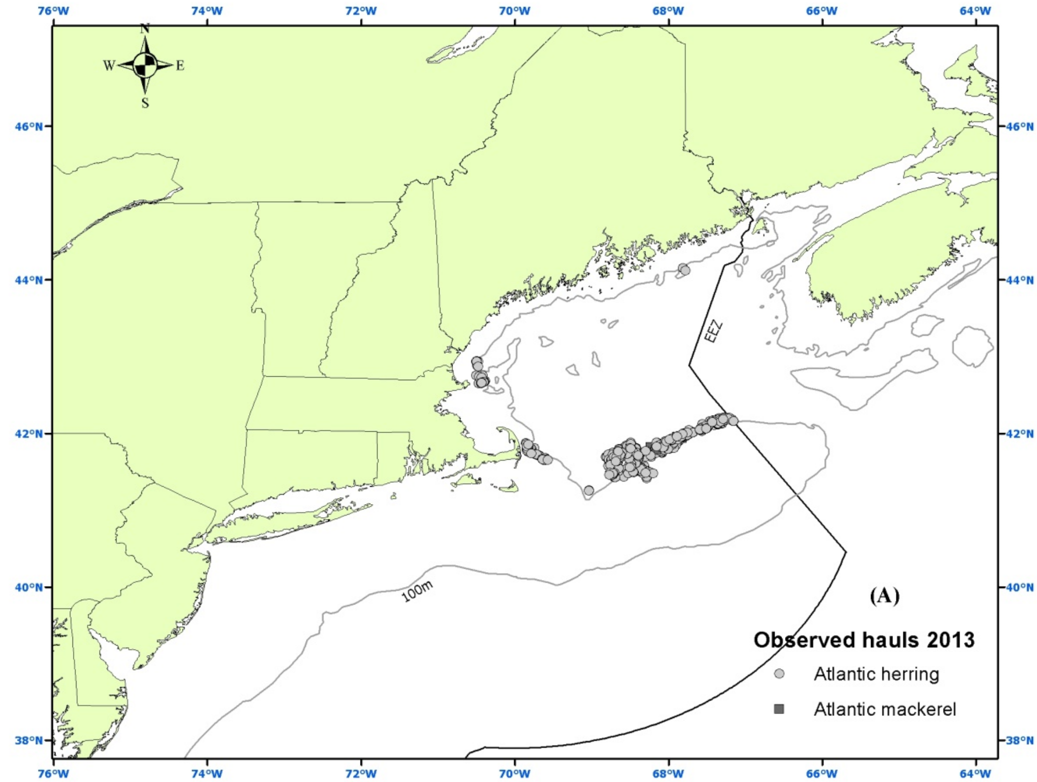


Figure 21. 2013 Northeast mid-water trawl observed tows (A) and observed takes (B).



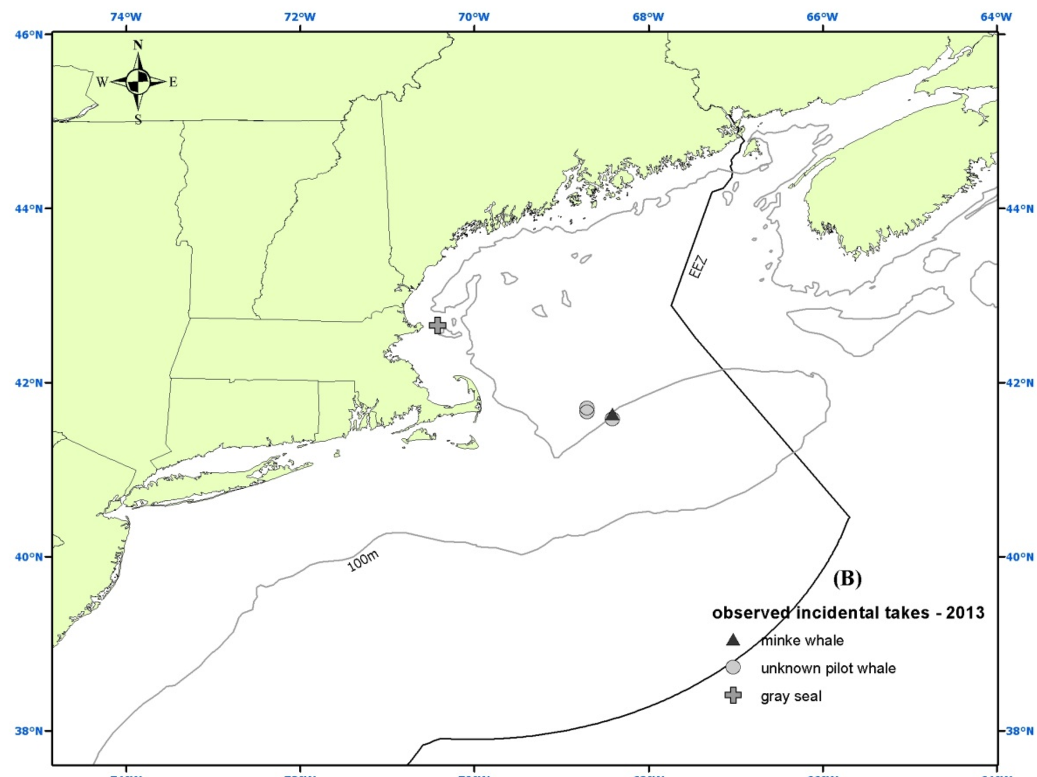
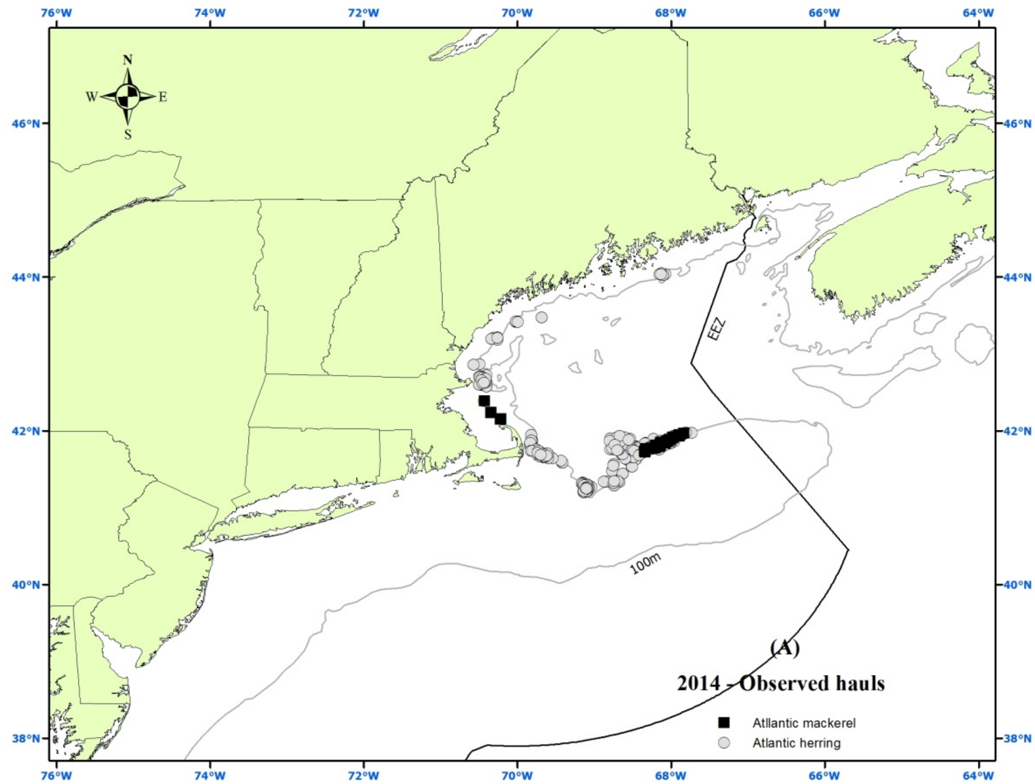


Figure 22. 2014 Northeast mid-water trawl observed tows (A) and observed takes (B).



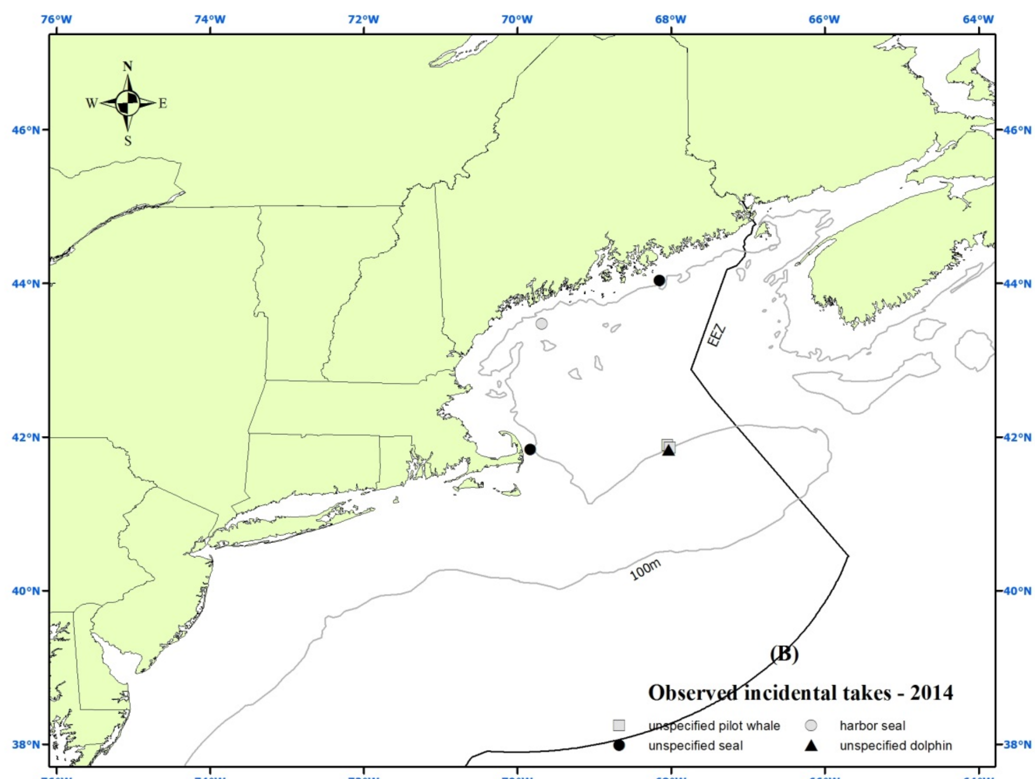


Figure 23. 2015 Northeast mid-water trawl observed tows (A) and observed takes (B).

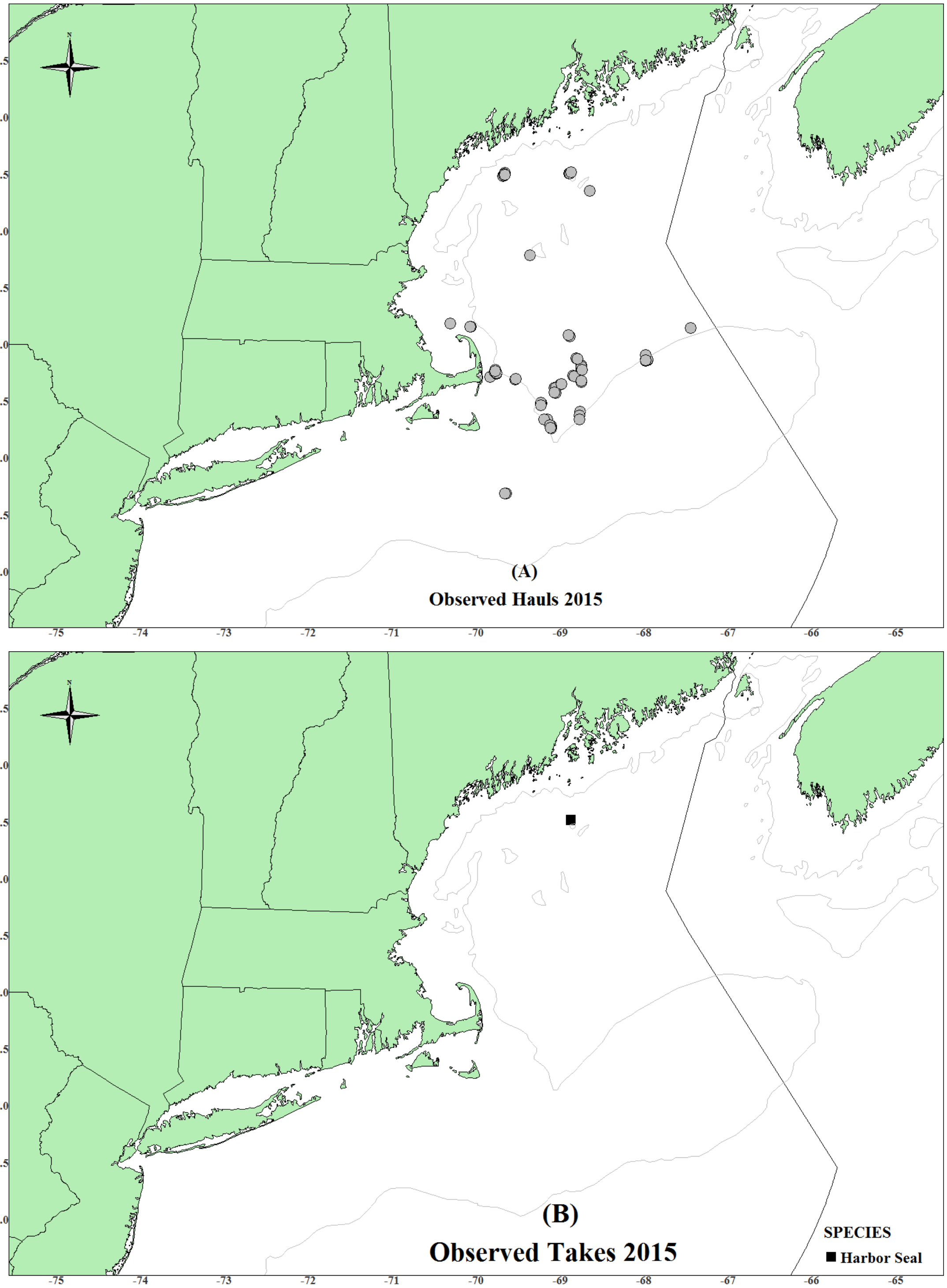
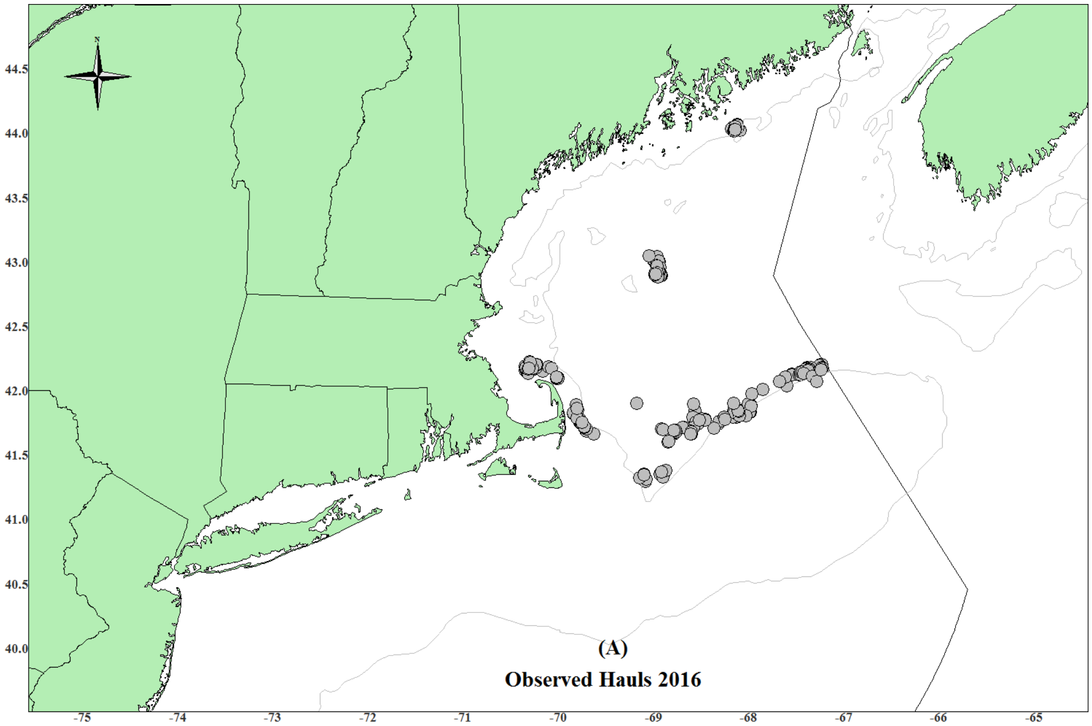


Figure 24. 2016 Northeast mid-water trawl observed tows (A) and observed takes (B).



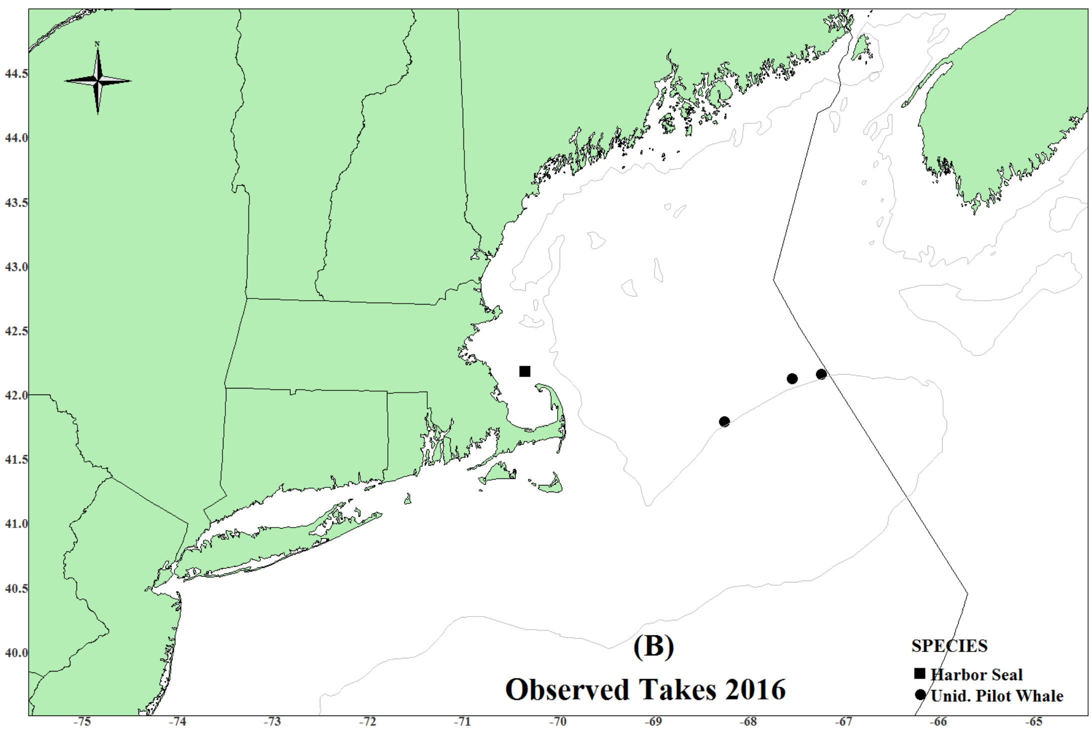
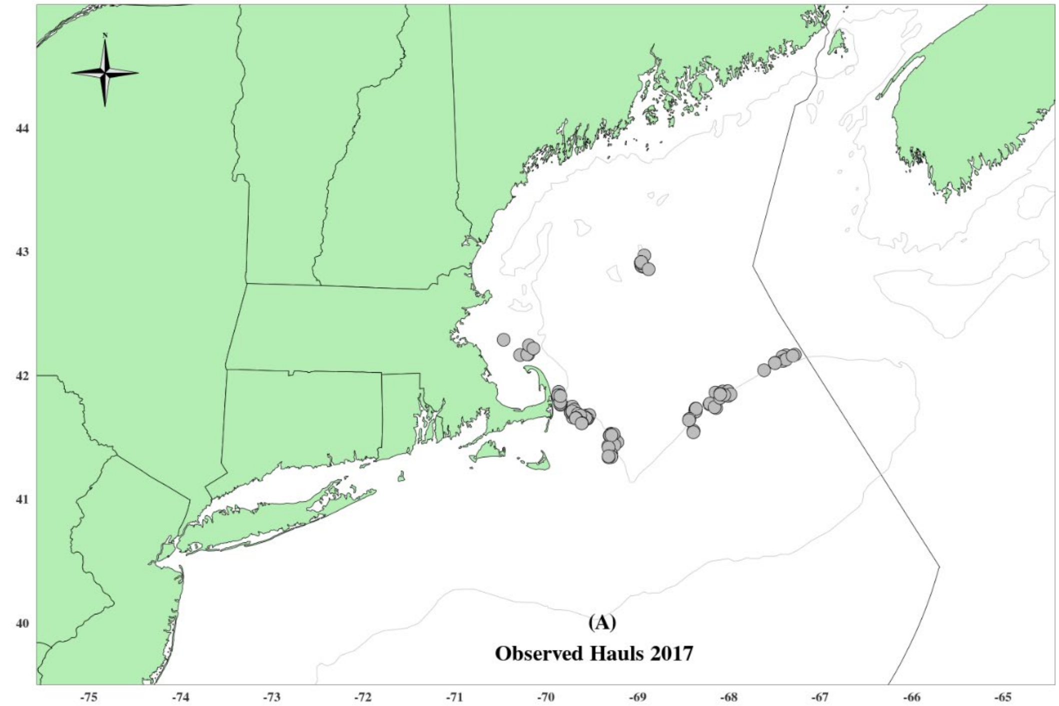


Figure 25. 2017 Northeast mid-water trawl observed tows (A) and observed takes (B).



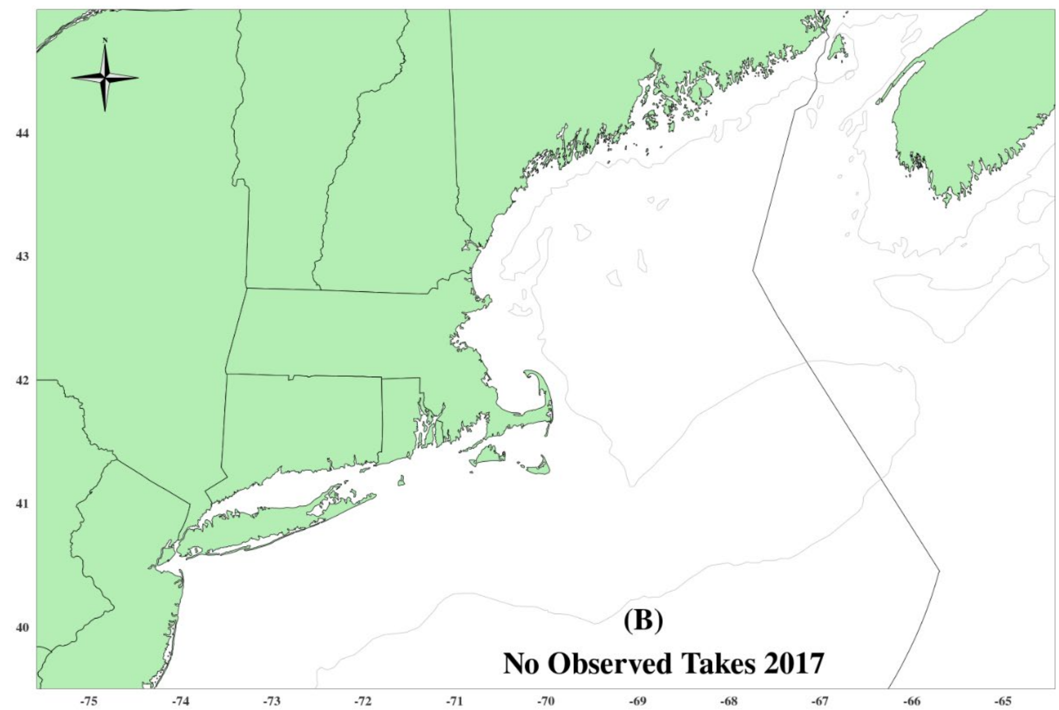
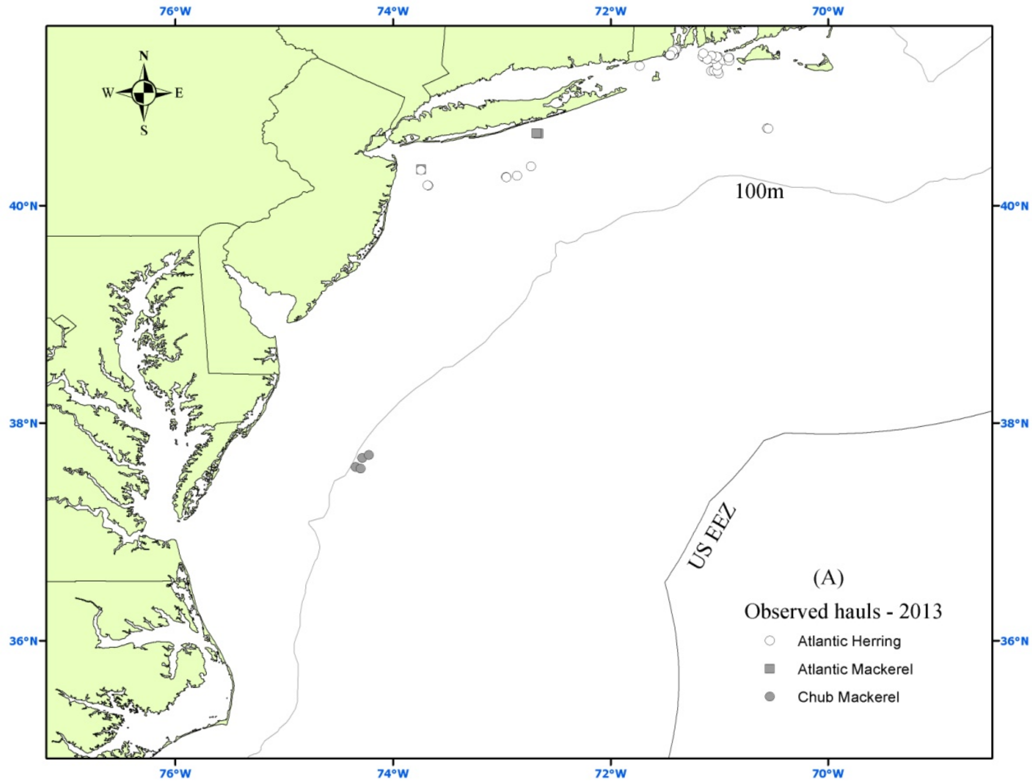


Figure 26. 2013 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



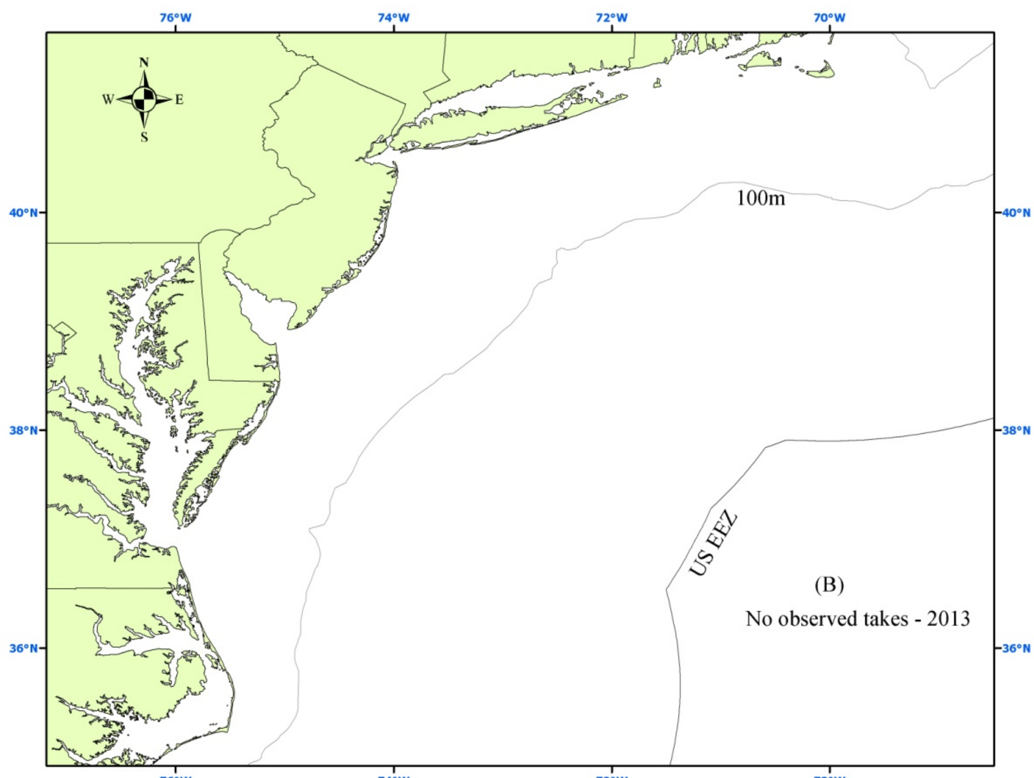
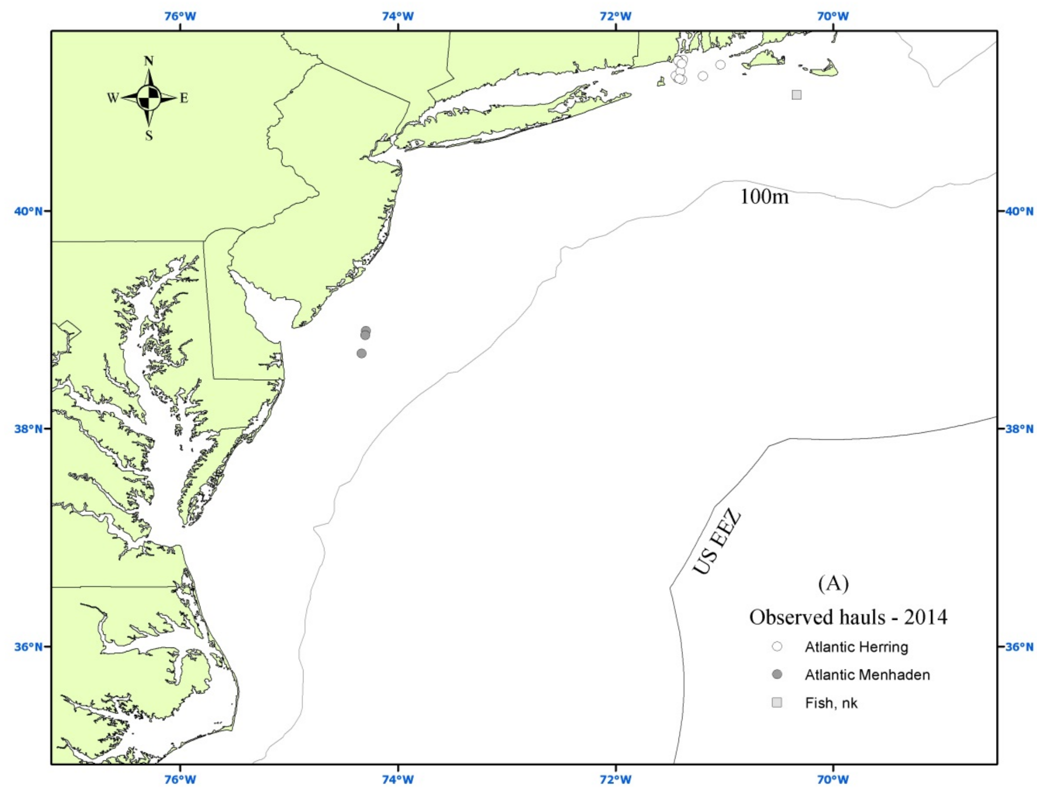


Figure 27. 2014 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



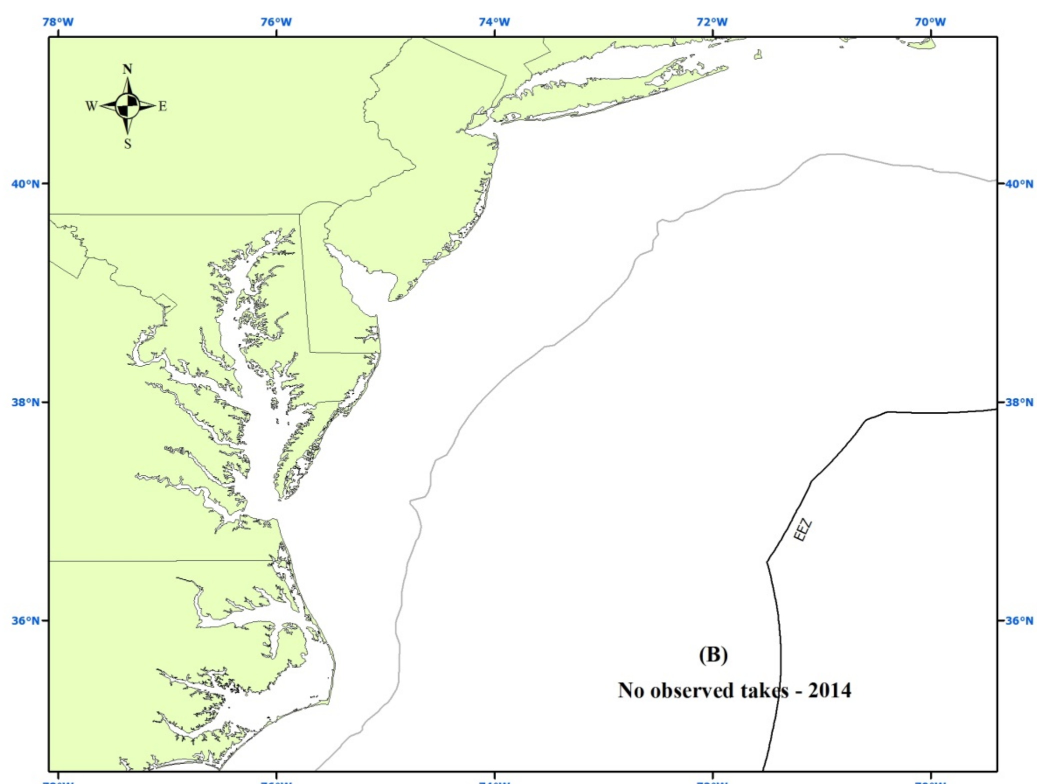
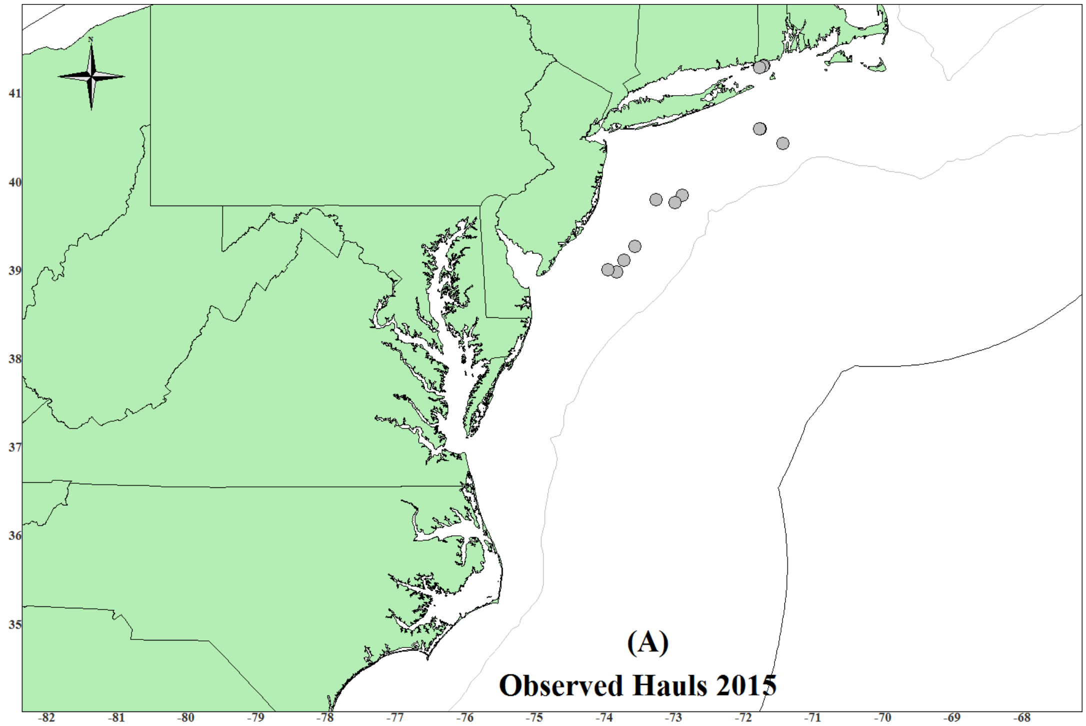


Figure 28. 2015 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



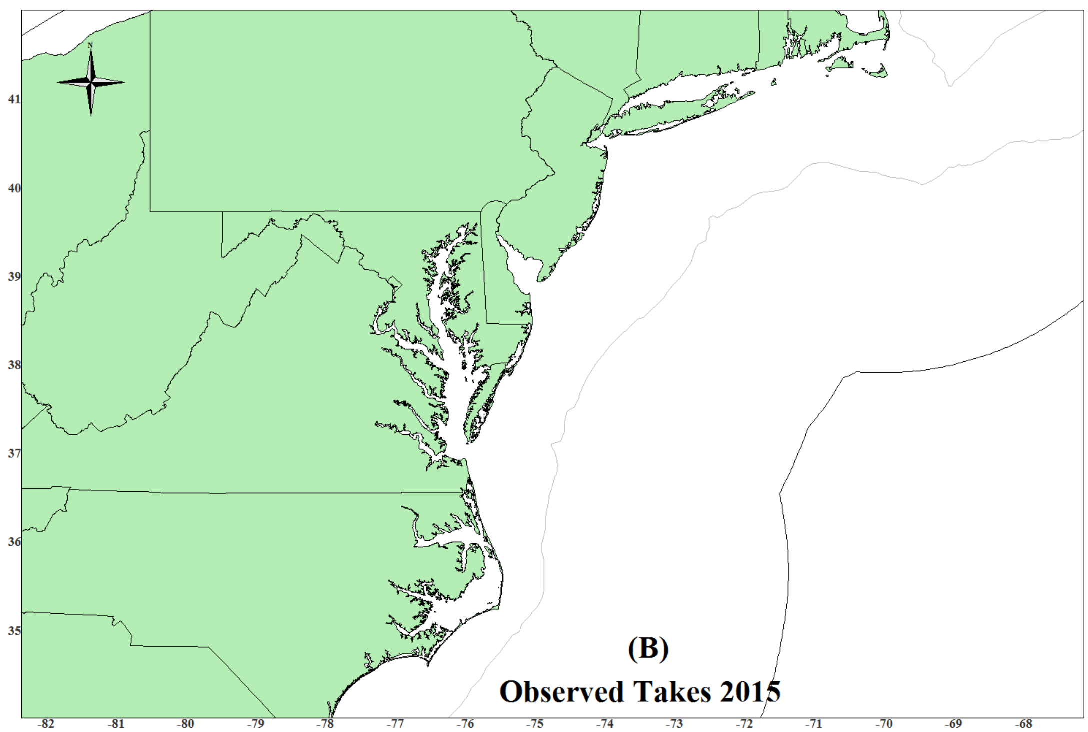
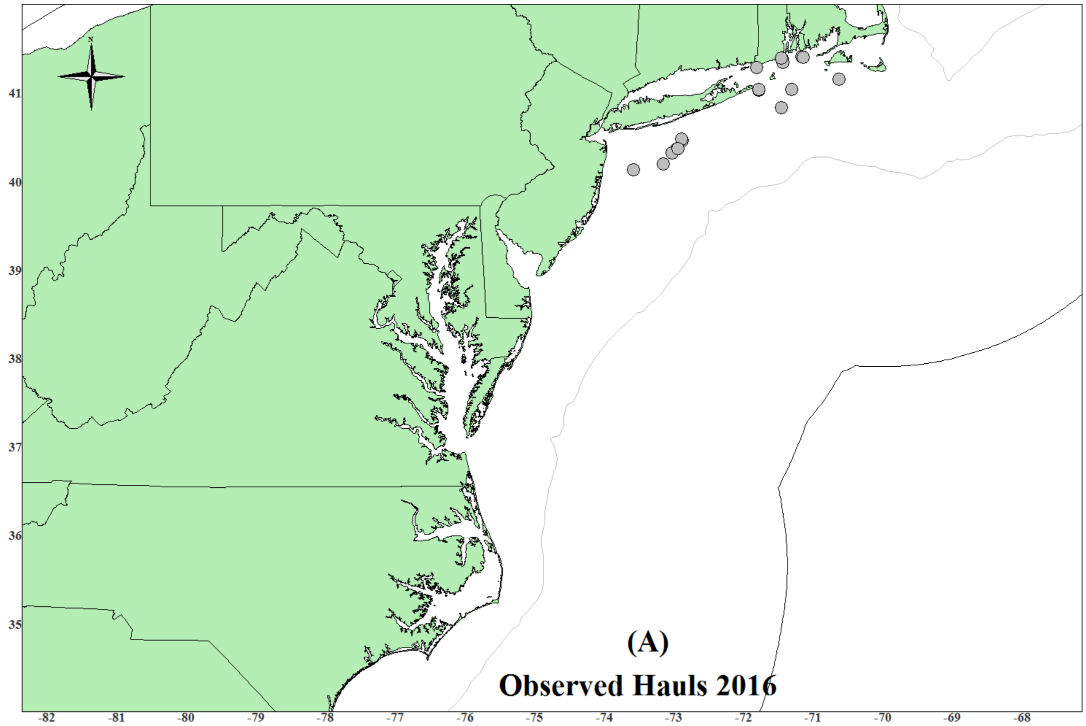


Figure 29. 2016 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



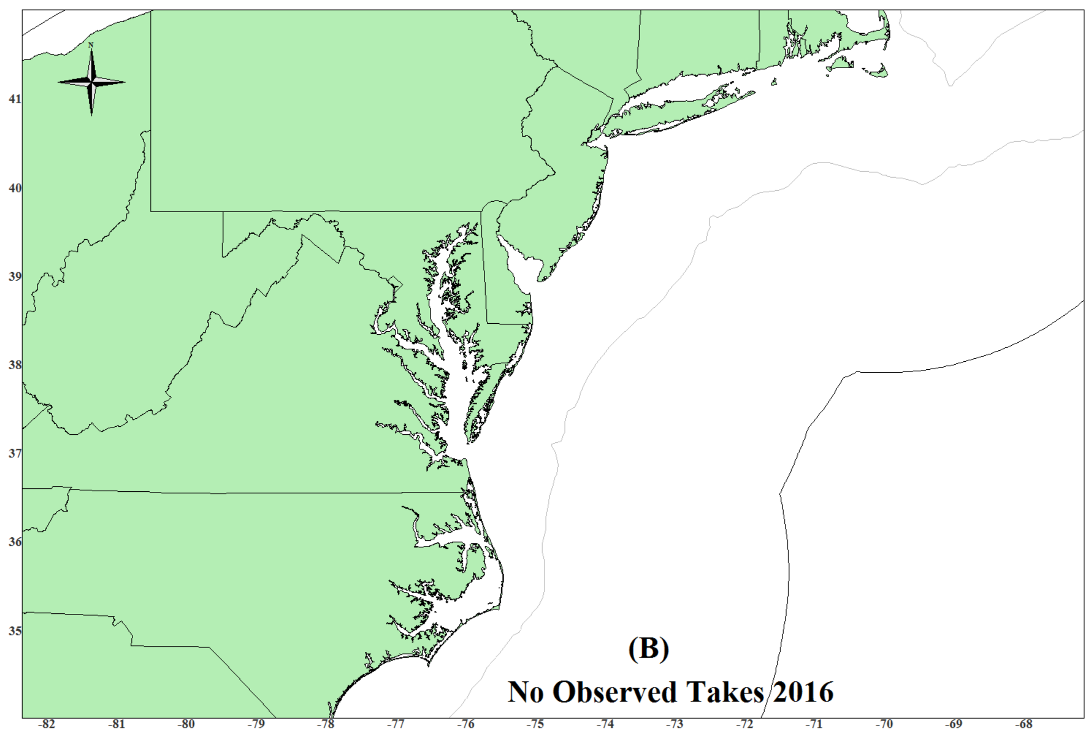
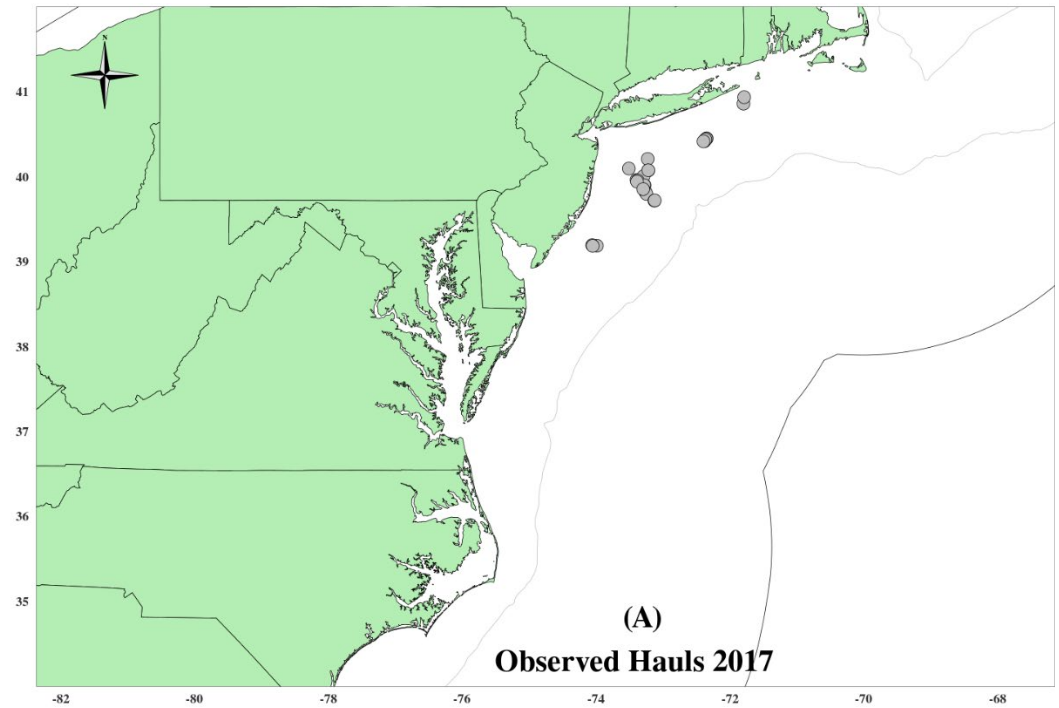


Figure 30. 2017 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



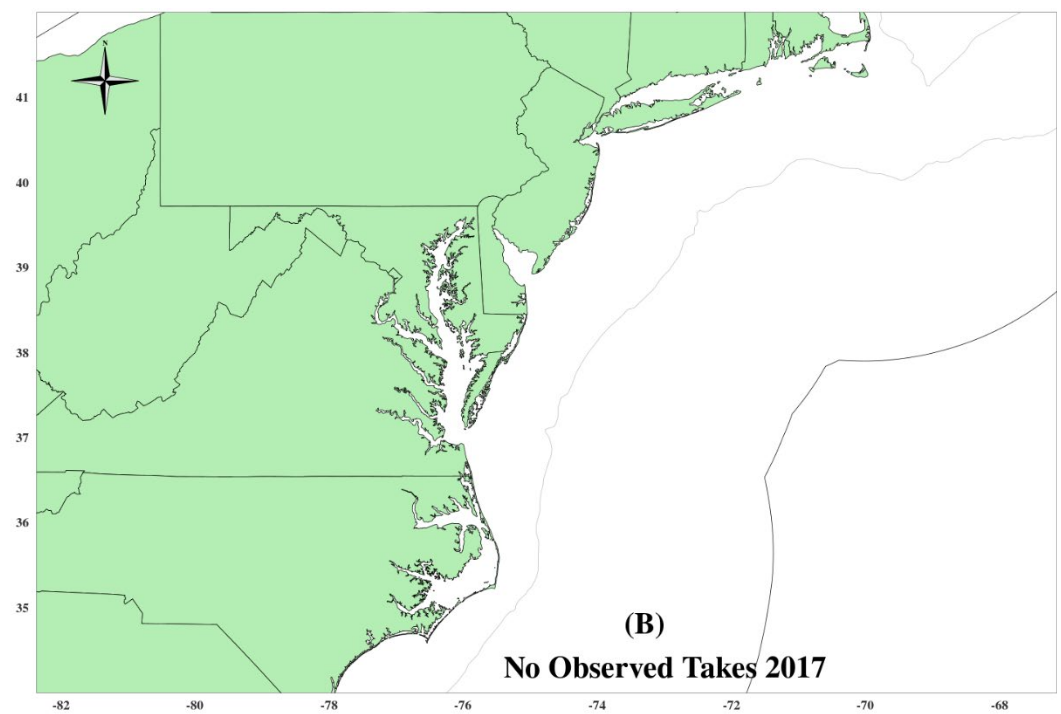
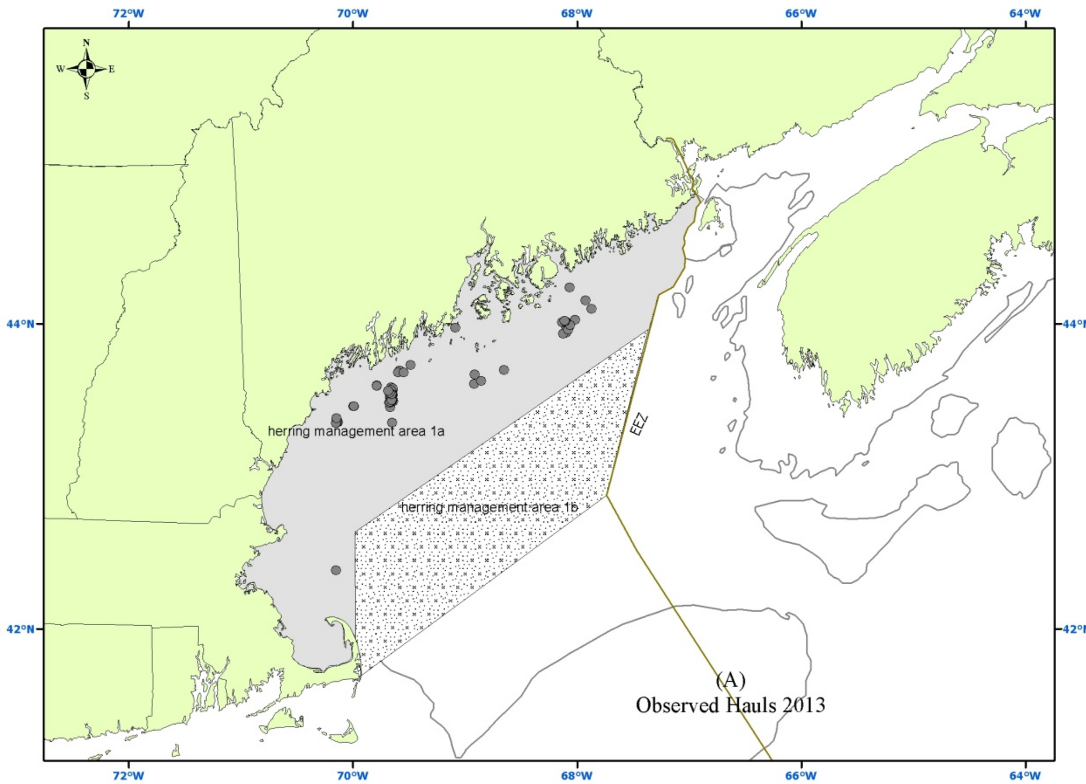


Figure 31. 2013 Herring Purse Seine observed hauls (A) and observed takes (B).



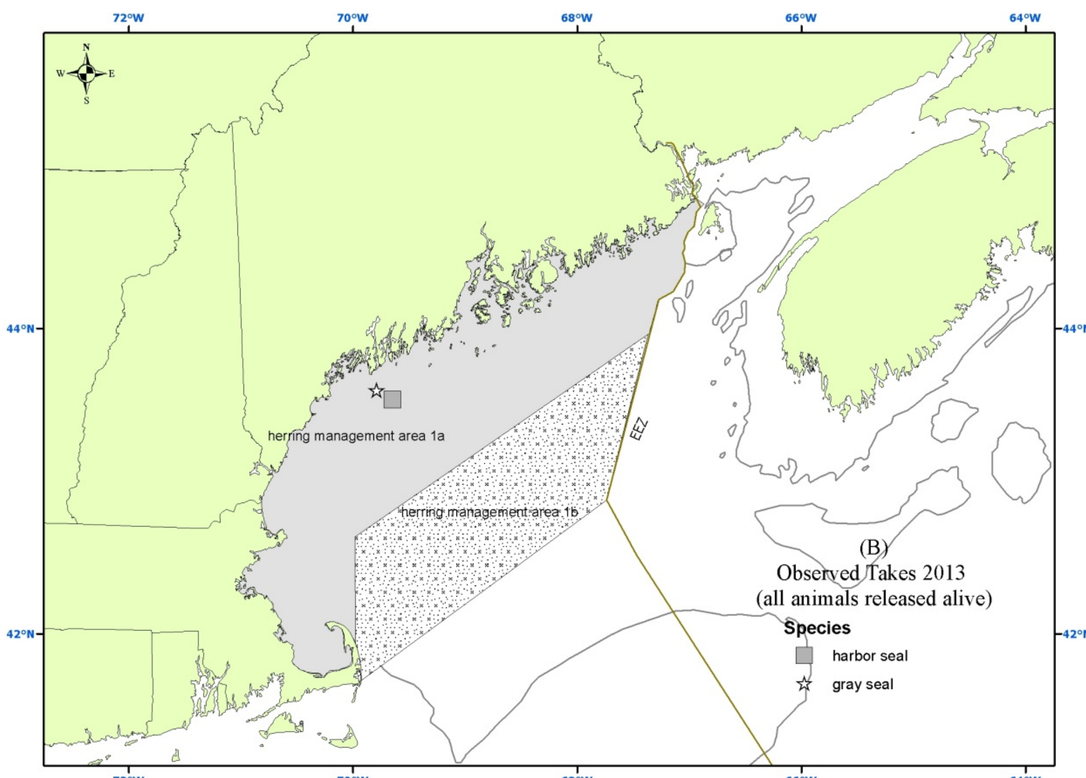
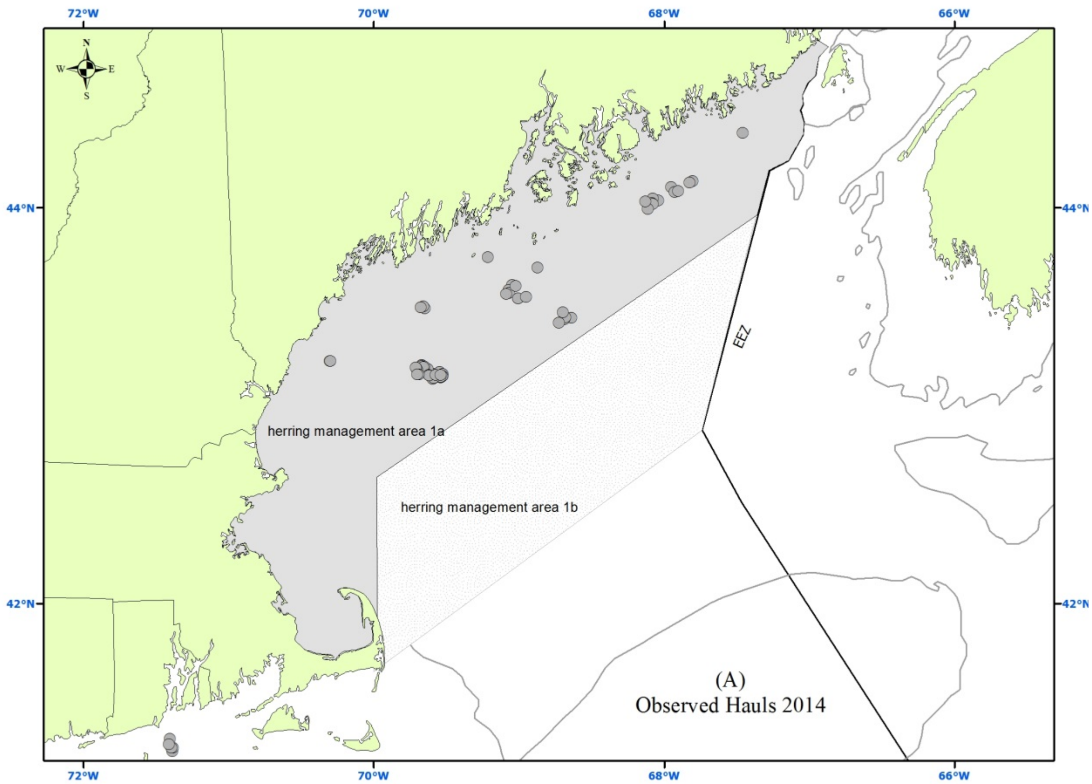


Figure 32. 2014 Herring Purse Seine observed hauls (A) and observed takes (B).



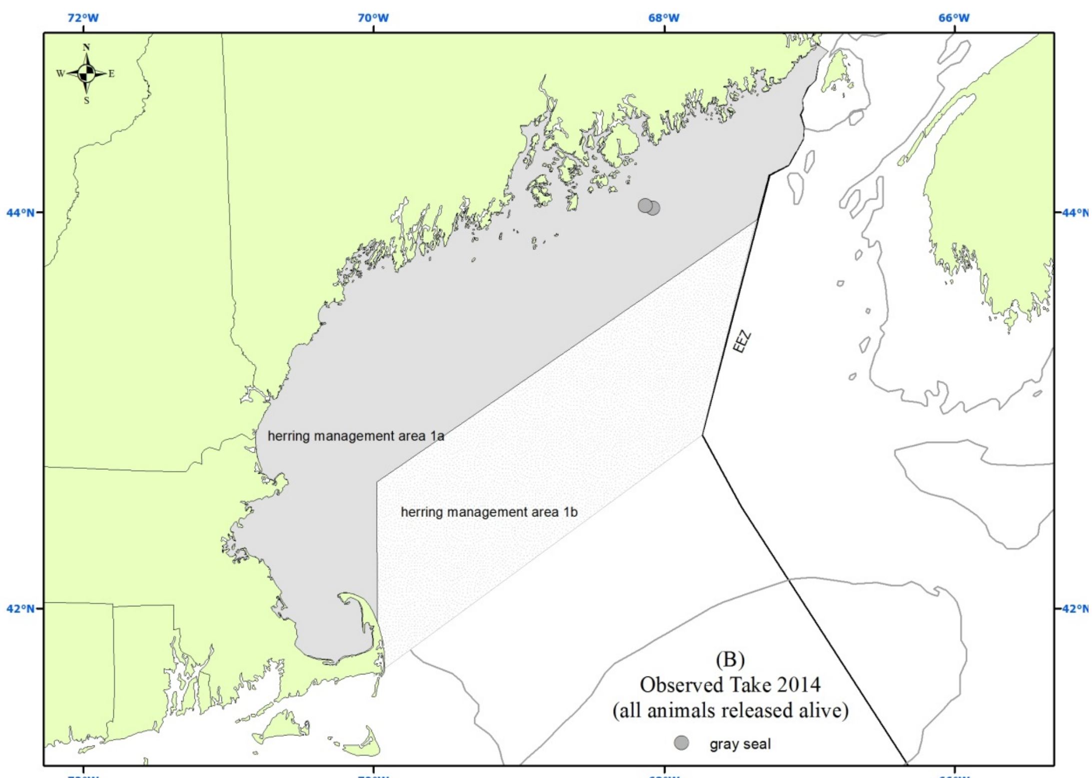


Figure 33. 2015 Herring Purse Seine observed hauls (A) and observed takes (B).

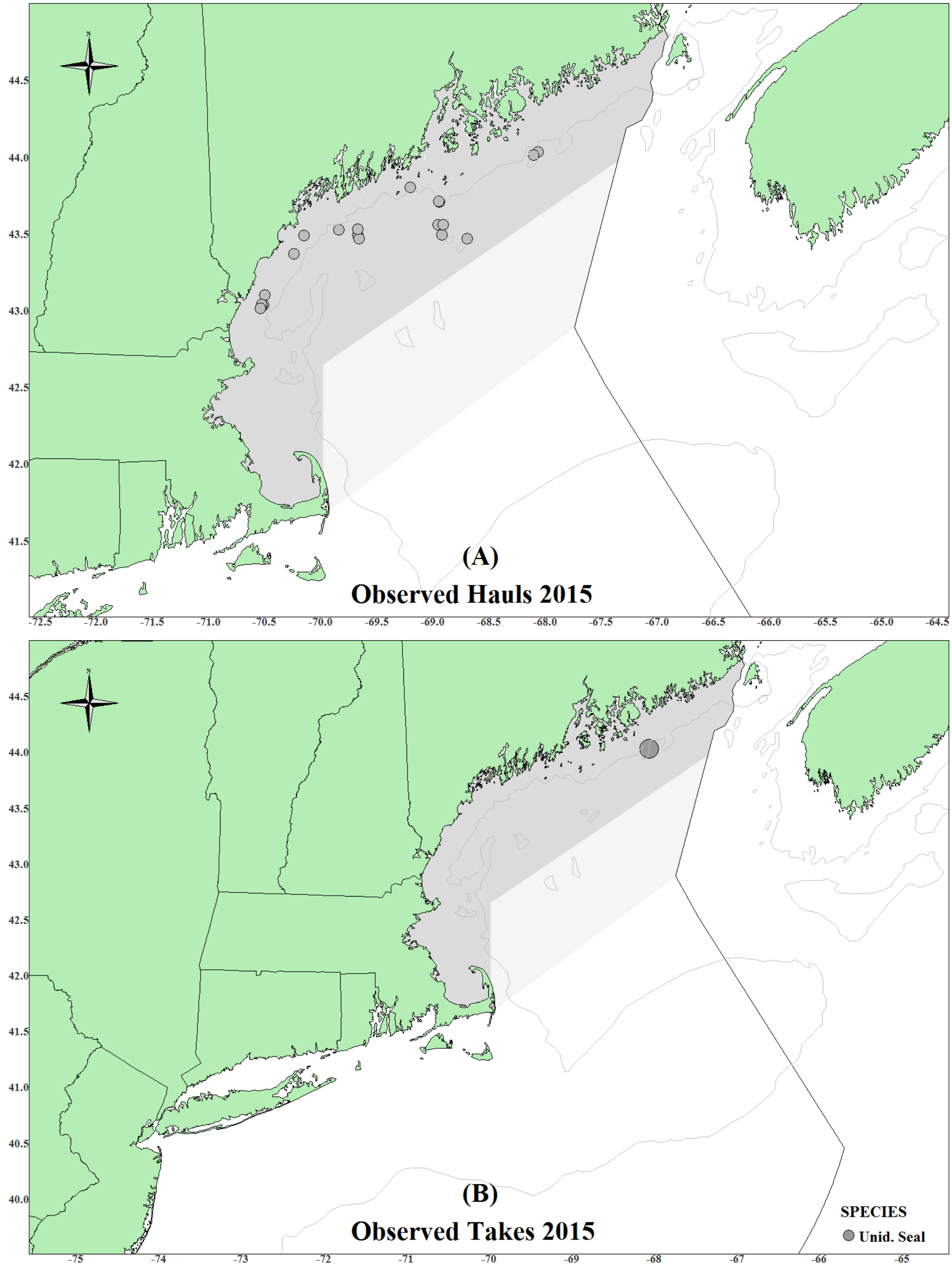
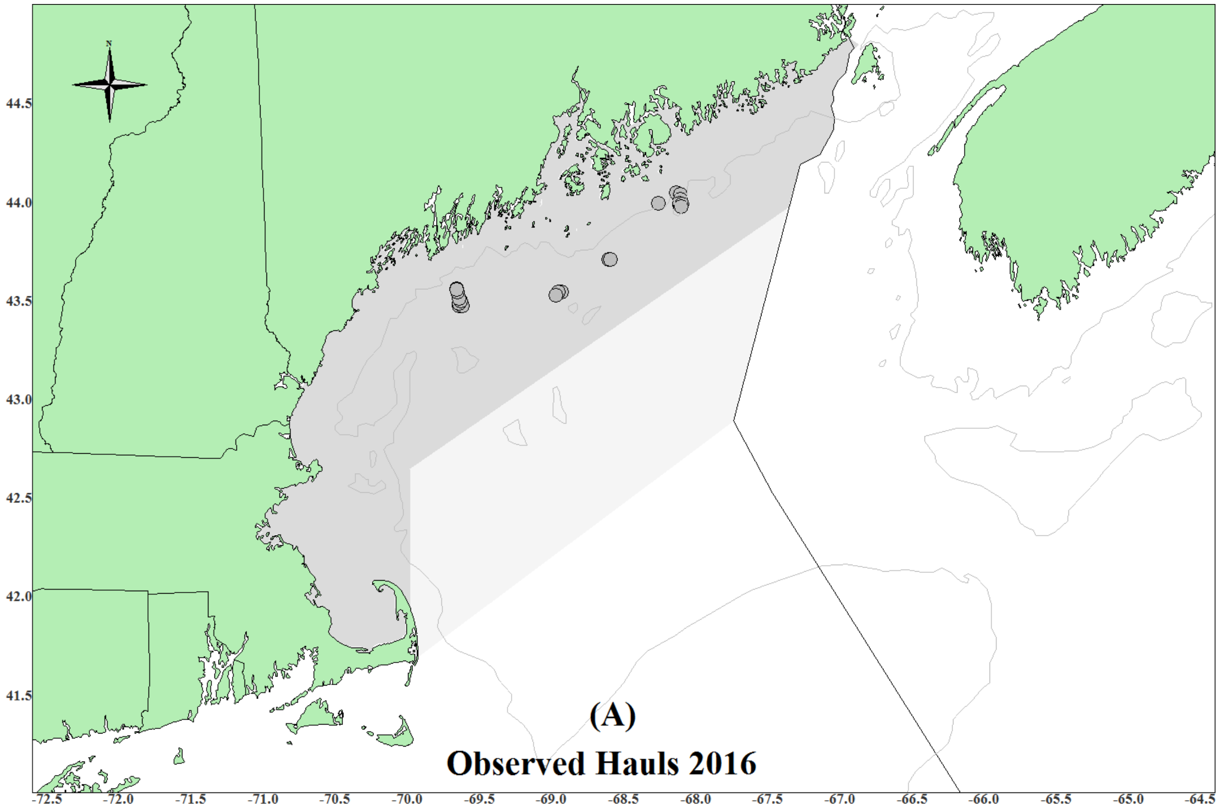


Figure 34. 2016 Herring Purse Seine observed hauls (A) and observed takes (B).



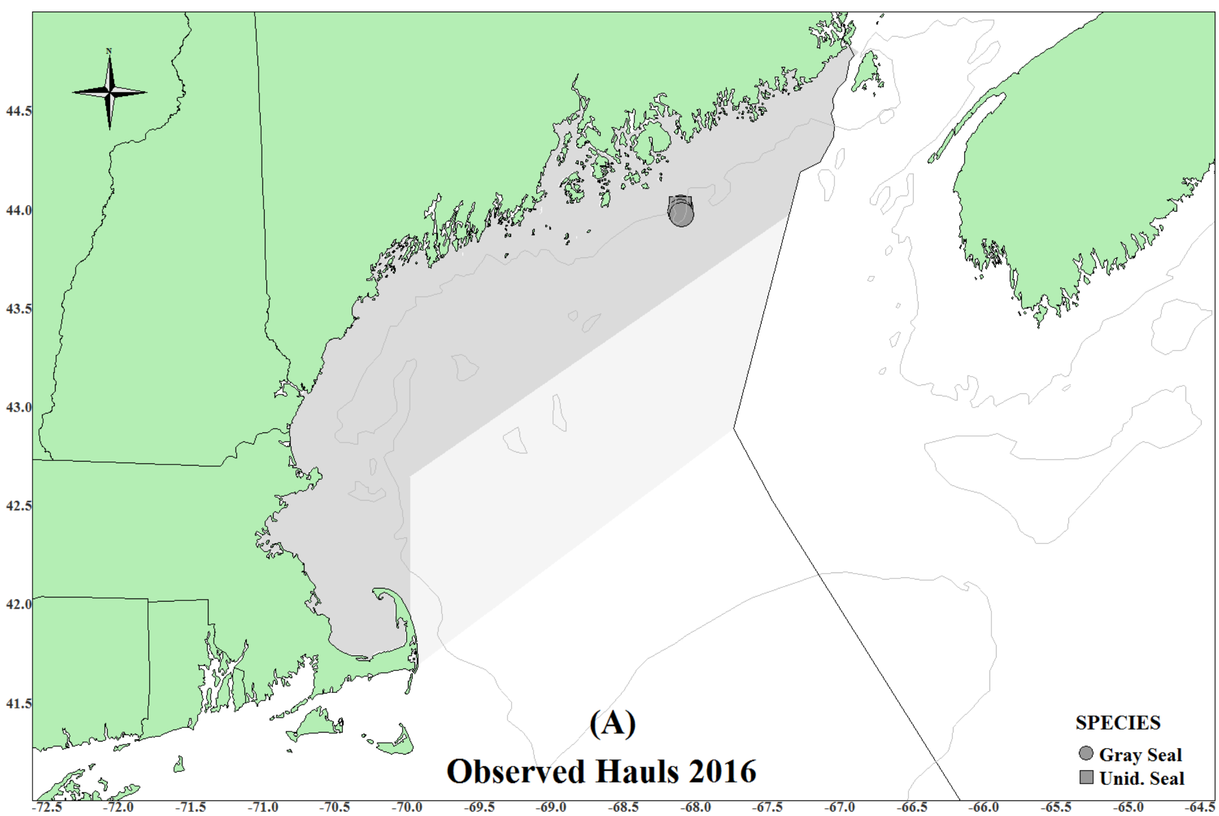
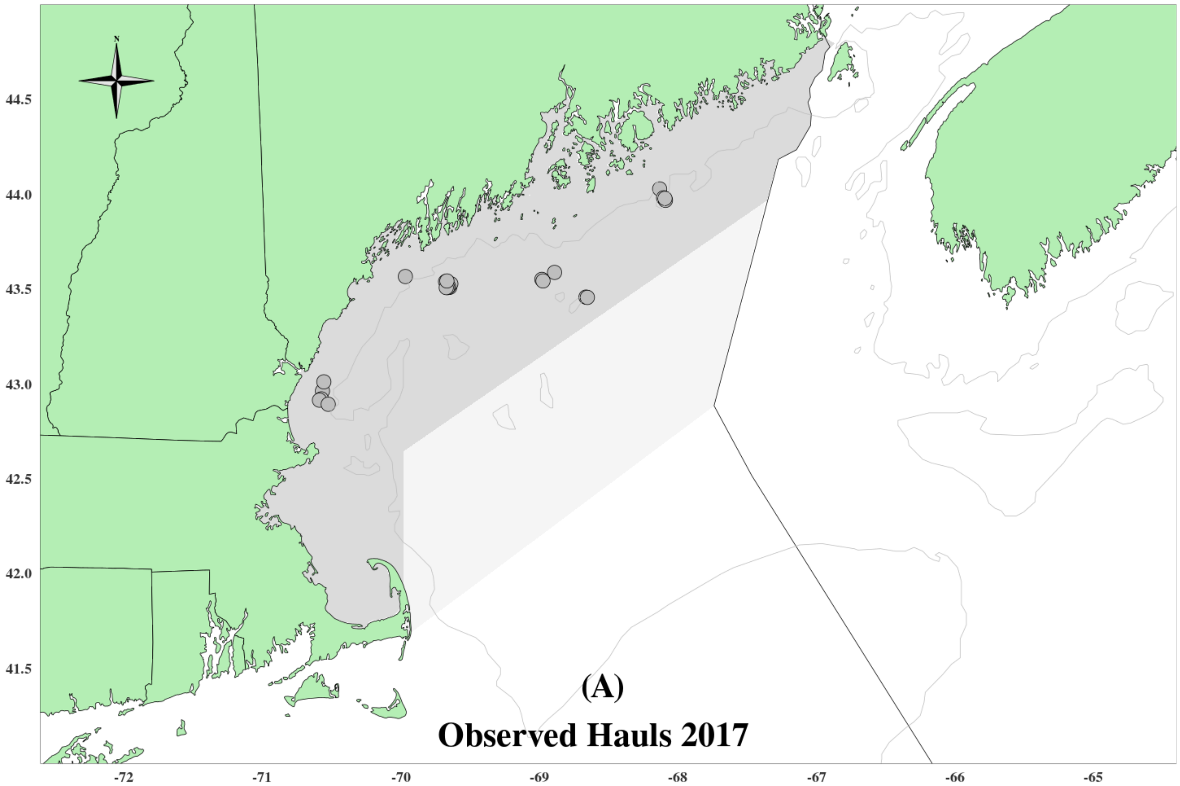


Figure 35. 2017 Herring Purse Seine observed hauls (A) and observed takes (B).



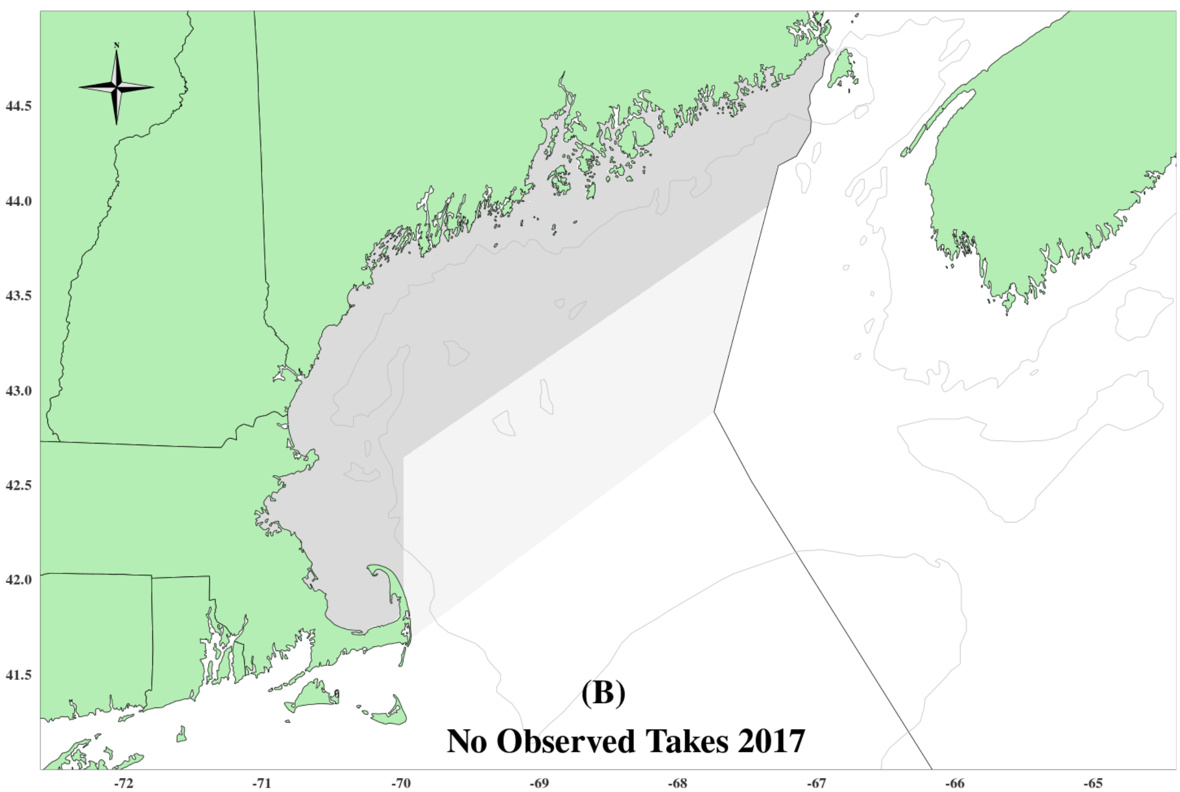


Figure 36. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2013. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

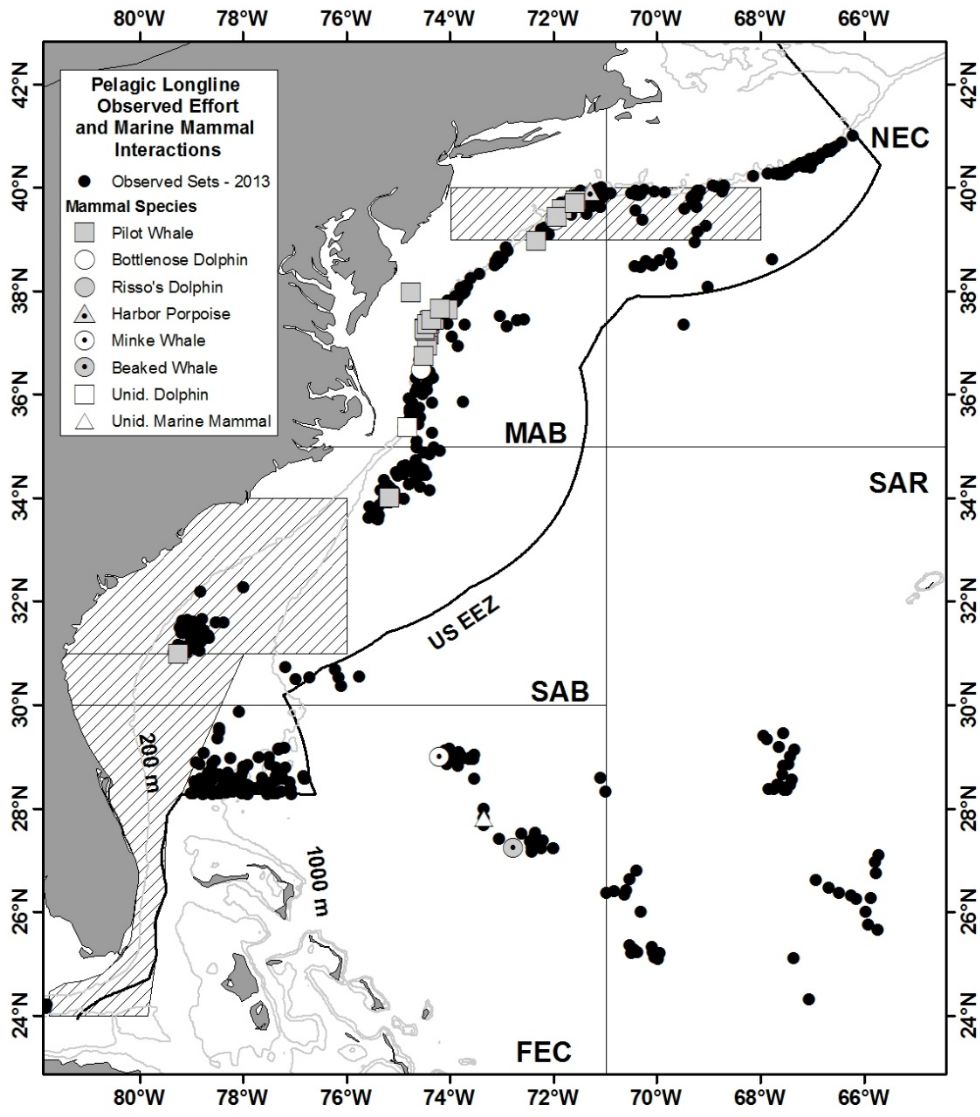


Figure 37. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2014. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

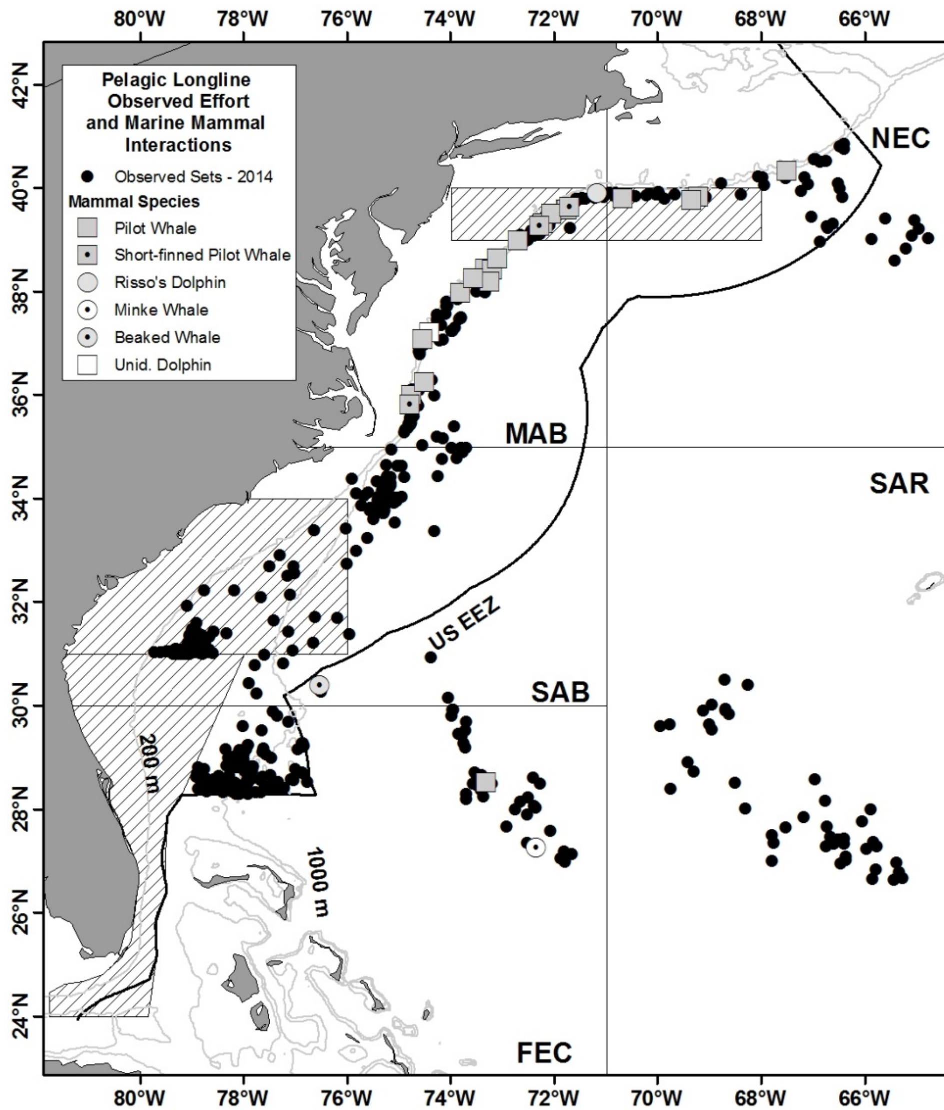


Figure 38. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2015. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

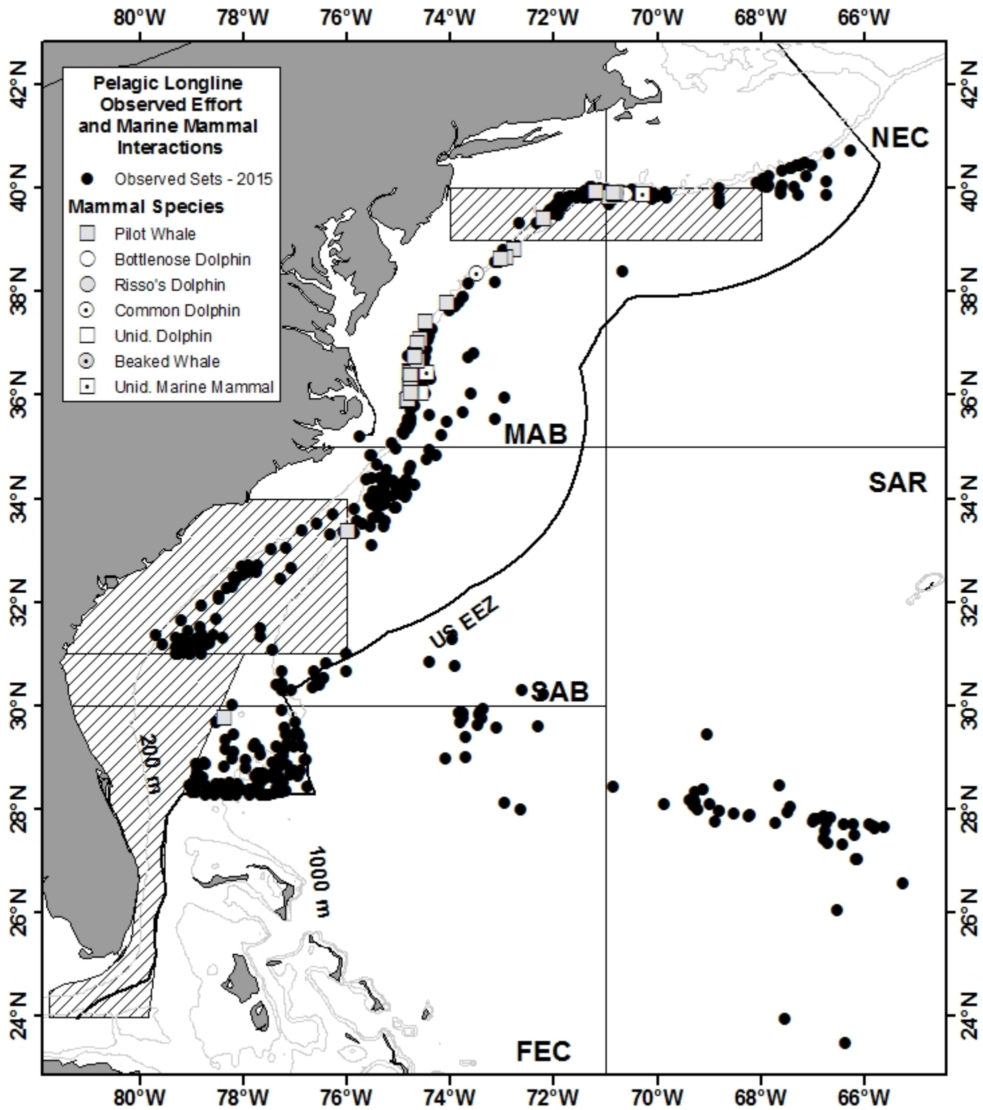


Figure 39. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2016. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

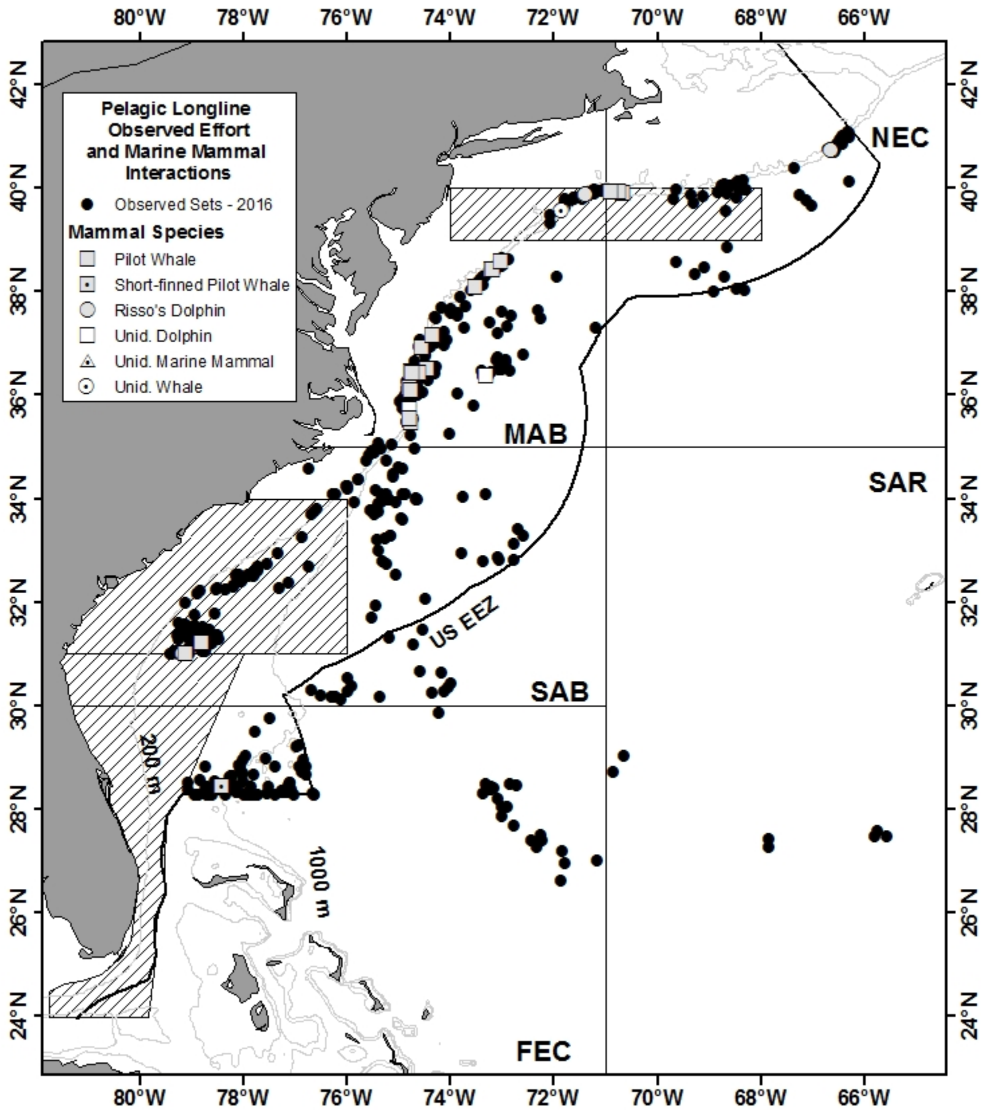


Figure 40. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2017. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

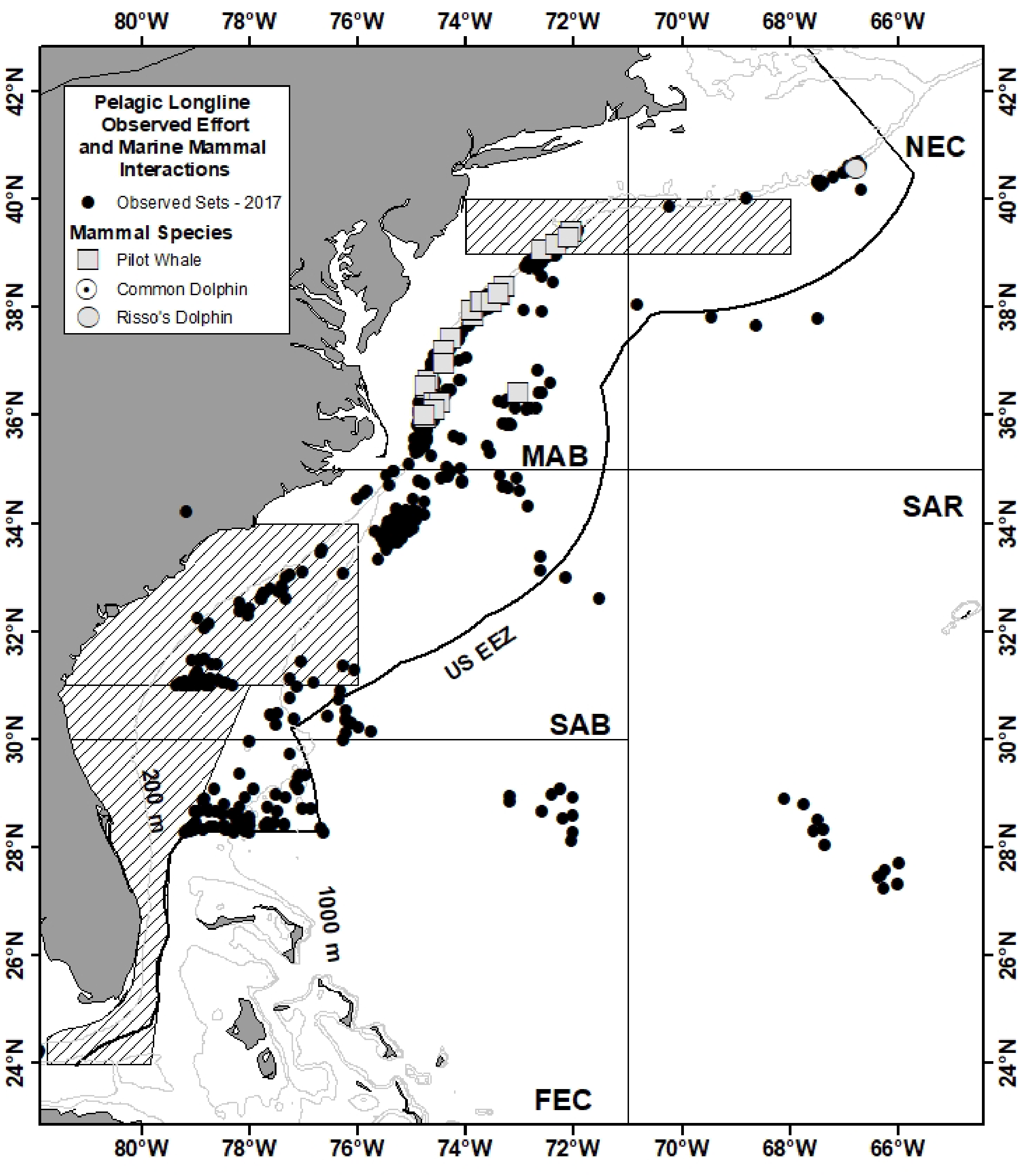


Figure 41. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2013. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.

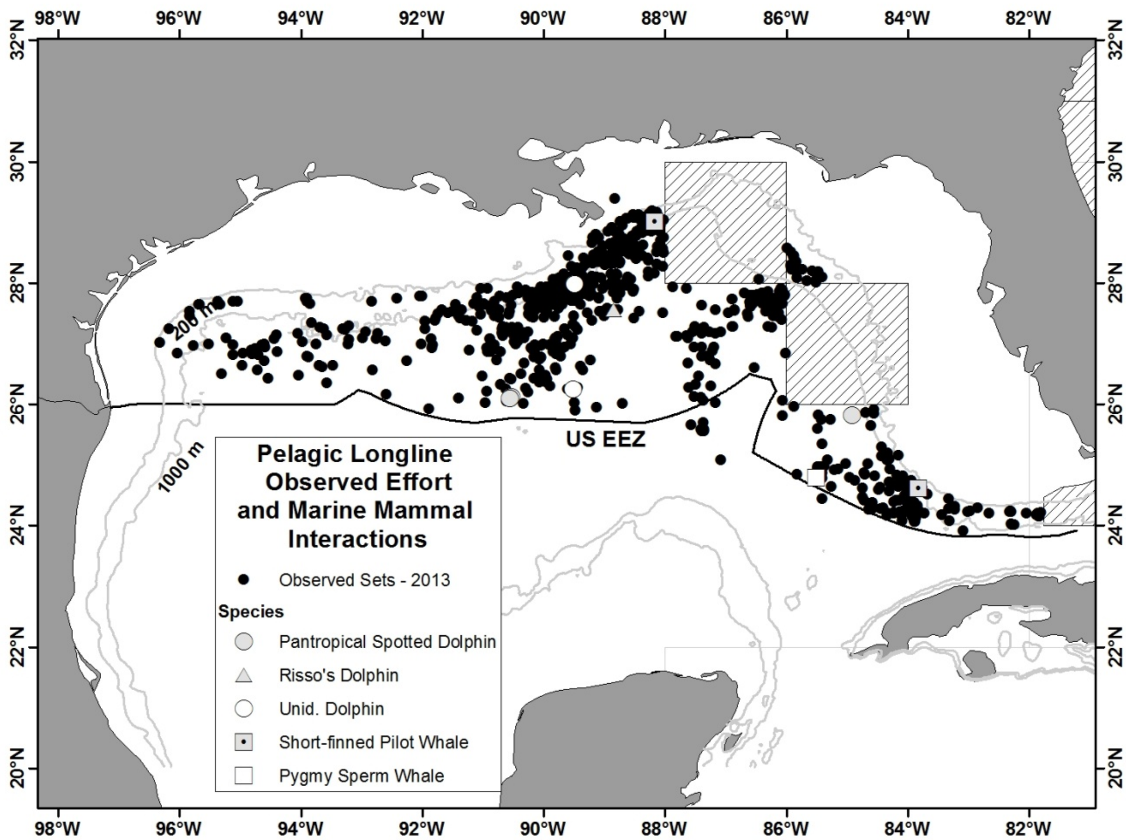


Figure 42. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2014. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.

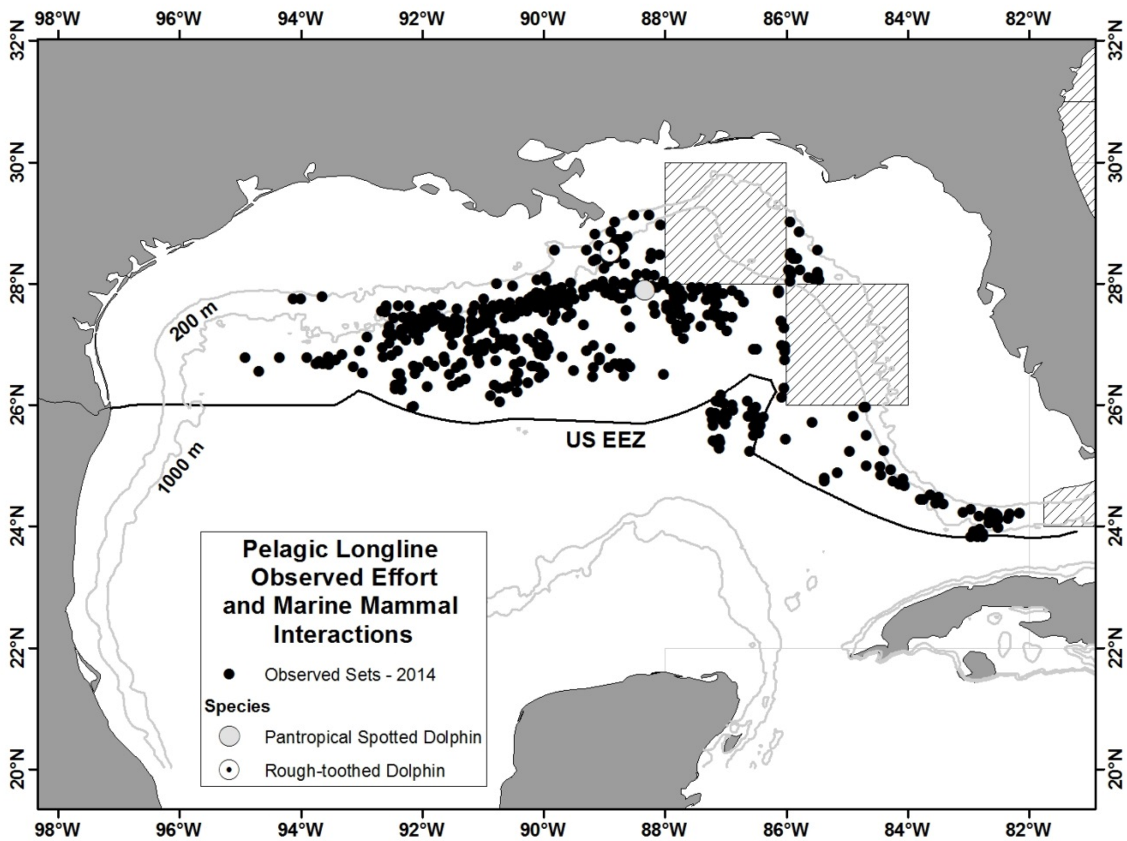


Figure 43. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2015. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.

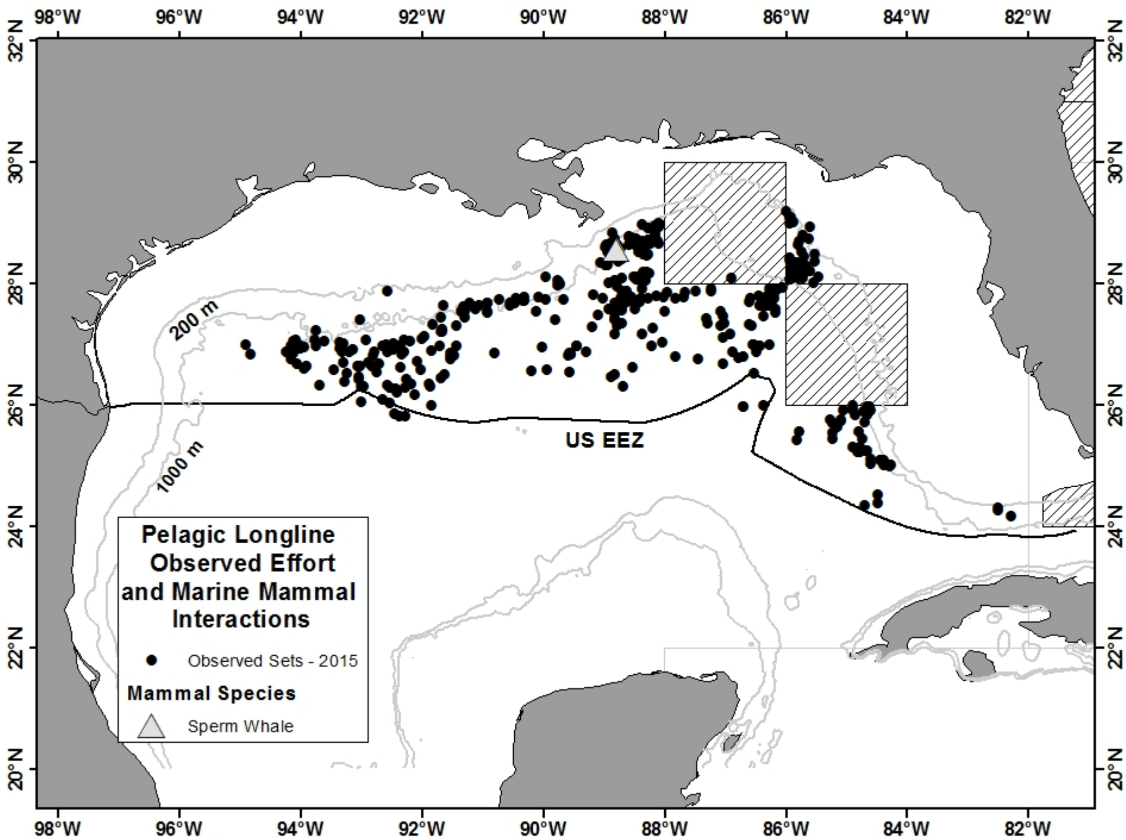


Figure 44. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2016. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.

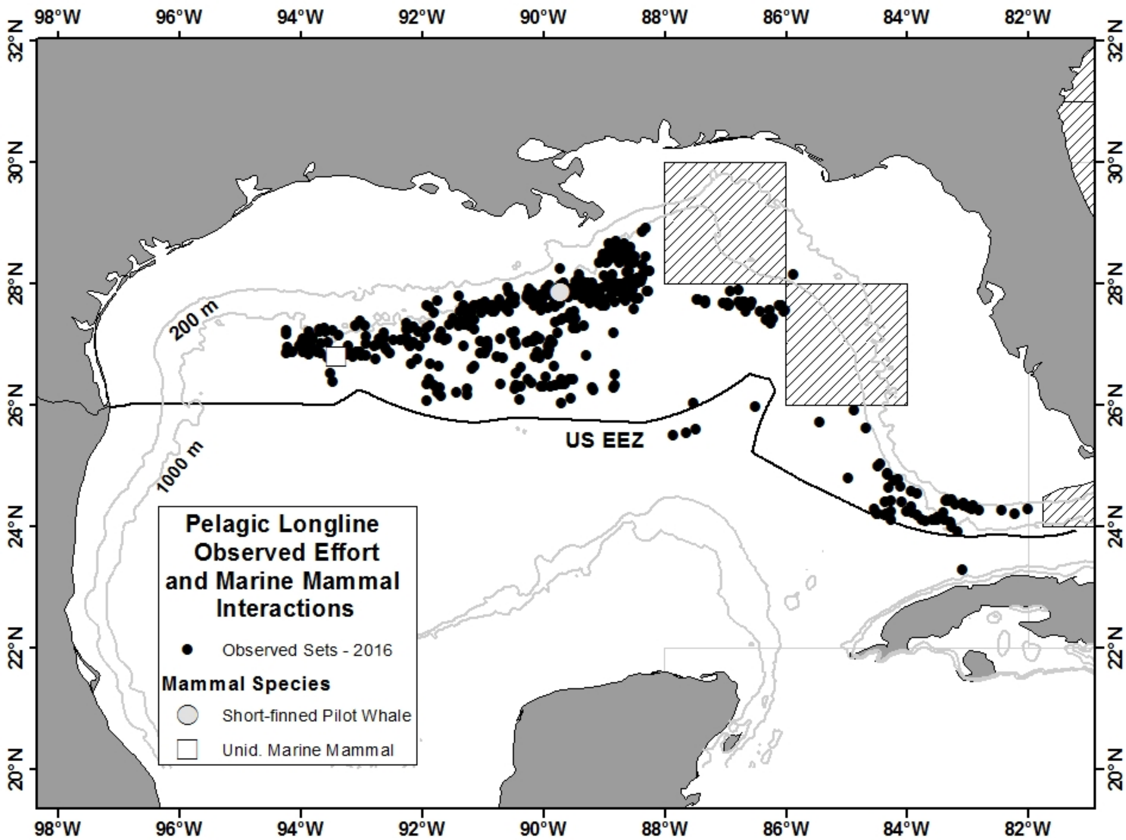
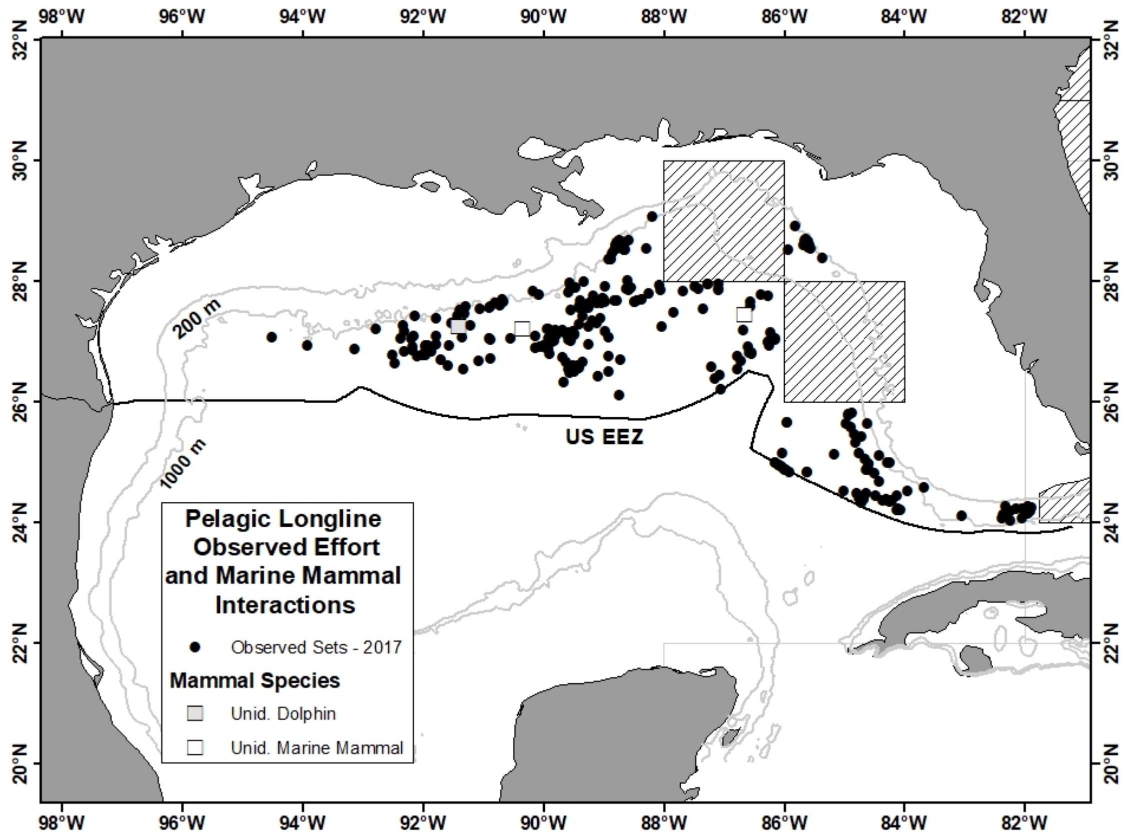


Figure 45. Observed sets and marine mammal interactions in the Pelagic longline fishery in the Gulf of Mexico during 2017. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.



Appendix IV: Table A. Surveys

Survey Number	Year	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
1	1982	year-round	plane	211,585	Cape Hatteras, NC to Nova Scotia, continental shelf and 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 the 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 and 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 and 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 and 2,000m isobaths	NEC	One team data analyzed by DISTANCE.	N	Waring et al. 1992; Waring 1998

Survey Number	Year	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
8	1992	Jul-Sep	ship (Abel-J)	3,710	N. Gulf of Maine and 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	shelf edge and slope waters of Georges Bank	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 procedure that modeled the left truncated sighting curve	N	Kingsley and Reeves 1998
12	1995	Jul-Sep	2 ships (Abel-J and Pelican) and 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 procedure on line transect methods that modeled the left	N	Kingsley and Reeves 1998

Survey Number	Year	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
							truncated sighting curve		
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 (1995 and 1998)		Gulf of St. Lawrence	DFO			Kingsley and Reeves 1998
16	1998	Jul-Sep	ship (Abel-J) and plane (Twin Otter)	15,900	north of Maryland	NEC	Ship: two independent team data analyzed with the modifed 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) and 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	Y	

Survey Number	Year	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
							abundance estimates for all years except 2008.		
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 - 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) and 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
24	2007	Aug	ship (Bigelow) and 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

Survey Number	Year	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
25	2007	Jul-Aug	plane	46,804	Canadian waters from Nova Scotia to Newfoundland	DFO	uncorrected counts	N	Lawson and Gosselin 2009
26	2008	Aug	plane (Twin Otter)	6,267	NY to Maine in US waters	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 and sounds, coastal waters from shoreline to 18-m isobath, and OCS waters from 18-m isobath to 9.3 km past the 18-m 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 200 m 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 and sounds, coastal waters from shoreline to 18-m isobath, and OCS waters from 18-m isobath to 9.3 km past the 18-m isobath	GOMEX92, GOMEX93	One team data analyzed by DISTANCE	N	Blaylock and Hoggard 1994

Survey Number	Year	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
33	1996-1997,1999-2001	Apr-Jun	ship (Oregon II and Gunter)	12,162	northern Gulf of Mexico from 200 m to U.S. EEZ	SEC	One team data analyzed by DISTANCE	N	Mullin and Fulling 2004
34	1998-2001	end Aug-early Oct	ship (Gunter and Oregon II)	2,196	northern Gulf of Mexico outer continental shelf (OCS, 20-200 m)	SEC	One team data analyzed by DISTANCE	N	Fulling et al. 2003
36	2004	12-13 Jan	helicopter		Sable Island	DFO	Pup count	na	Bowen et al. 2007
37	2004		plane		Gulf of St Lawrence and Nova Scotia Eastern Shore	DFO	Pup count	na	Hammill 2005
38	2009	10 Jun-13 Aug	ship	4,600	northern Gulf of Mexico from 200m to U.S. EEZ	SEC	One team data analyzed by DISTANCE		
39	2007	17 Jul-8 Aug	plane		northern Gulf of Mexico from shore to 200m(majority of effort 0- 20m)	SEC	One team data analyzed by DISTANCE		
40	2011	4 Jun-1 Aug	ship (Bigelow)	3,107	Virginia to Massachusetts (waters that were deeper than the 100-m 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	7-26 Aug	Plane (Twin Otter)	5,313	Massachusetts to New Brunswick, Canada (waters north of New Jersey and	NEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the	Y	Palka 2012

Survey Number	Year	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
					shallower than the 100-m depth contour, through the US and Canadian Gulf of Maine and up to and including the lower Bay of Fundy)		independent observer option assuming point independence		
42	2011	19 Jun- 1 Aug	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	24 July–14 Aug	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		

Survey Number	Year	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
46	2011	6–29 July	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	27 Jun–25 Aug	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	Season	Platform	Track line length (km)	Area	Agency/ Program	Analysis	Corrected for g(0)	Reference
48	2016	30 June–19 Aug	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 and Sep	plane	50,160	Gulf of St. Lawrence, Bay of Fundy, Scotian Shelf, Newfoundland, Labrador	DFO	NAISS		Lawson and Gosselin 2018

APPENDIX IV: Table B. Abundance estimates – "Survey Number" refers to surveys described in Table A. "Best" estimate for each species in bold font .

Species	Stock	Year	Nbest	CV	Survey Number	Notes
Humpback Whale	Gulf of Maine	1992	501			minimum pop'n size estimated from photo-ID data
		1993	652	0.29		YONAH sampling (Clapham *et al.* 2003)
		1997	497			minimum pop'n 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 pop'n 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

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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	.40	47+48	Garrison 2020; Palka 2020
		2016	2,235	.41	49	Lawson and Gosselin 2018 (Bay of Fundy/Scotian Shelf)
		2016	2,177	.47	49	Lawson and Gosselin 2018 (Newfoundland/Labrador)
		2016	**7,418**	**.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 generated by Roberts *et al*. (2016)

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		1995─2013	717	0.30		Summer habitat-based density estimates generated by Roberts *et al*. (2016)
		2016	28	0.55	47	Palka 2016
Minke Whale	Canadian East Coast	1982	320	0.23	1	CETAP 1982
		1992	2,650	0.31	3+8	
		1993	330	0.66	9	
		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**	Lawson and Gosselin 2018 (Bay of Fundy/Scotian Shelf)
		2016	13,008	0.46	**49**	Lawson and Gosselin 2018 (Newfoundland/Labrador)
		2016	**24,202**	**0.30**	**47+49**	
Sperm Whale	North Atlantic	1982	219	0.36	1	CETAP 1982

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		1990	338	0.31	2	
		1991	736	0.33	7	Waring *et al.*1992:1998
		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.		1998	115	0.61	16	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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
	Western North Atlantic	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	
		1991	262	0.99	7	Waring *et al.*1992:1998
		1991	370	0.65	6	
Beaked Whales	Western North Atlantic	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	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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.*1992:1998
		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
		2004	20,479	0.59	21+22	Estimate summed from north and south surveys

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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	0.445	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.*1992:1998
		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	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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	Lawson and Gosselin 2009; long-finned pilot whales
		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

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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**
Atlantic white-sided Dolphin	Western North Atlantic	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
		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	Lawson and Gosselin 2018 (Canadian part of Gulf of Maine and all of Gulf of St. Lawrence population)

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		2016	**93,233**	**0.710**	**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.*1992:1998
		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

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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	Lawson and Gosselin 2018 (Newfoundland/Labrador)
		2016	43,124	0.28	49	Lawson and Gosselin 2018 (Bay of Fundy/Scotian Shelf)
		2016	**172,825**	**0.21**	**47+48+49**	**Estimate summed from north, south and Canadian surveys**
Atlantic Spotted Dolphin	Western North Atlantic	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
		2004	50,978	0.42	21+22	Estimate summed from north and south surveys
		2011	26,798	0.66	40+41	Palka 2012

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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**

Species	Stock	Year	Nbest	CV	Survey Number	Notes
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	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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	
		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	Palka 2006; survey discovered portions of the range not previously surveyed
		2002	64,047	0.48	21	Palka 2006
		2004	51,520	0.65	23	Palka 2006
		2006	89,054	0.47	24	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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		DFO 2013 (Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island)
		2014	505,000	95% CI= 329,000–682,000		DFO 2014 (Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island)
		2016	424,300	95% CI= 263,600–578,300		DFO 2017 (Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island)
		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	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		2009	**33**	**1.07**	**38**	
Sperm Whale	Northern Gulf of Mexico	1991-1994	530	0.31	30	Hansen *et al.* 1995
		1996-2001	1,349	0.23	33	Mullin and Fulling 2004
		2003-2004	1,665	0.2	35	
		2009	**763**	**0.38**	**38**	
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**	
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**	
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**	
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

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		2003-2004	49	0.77	35	
		2009	**28**	**1.02**	**38**	
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	
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**	
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**	
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**	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
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**	
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**	
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**	
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**	
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	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		2009	**129**	**1**	**38**	
Atlantic Spotted Dolphin	Northern Gulf of Mexico	1991-1994 oceanic	3,213	0.44	30	Hansen *et al.* 1995
		1996-2001 oceanic	175	0.84	33	Mullin and Fulling 2004
		1998-2001 OCS	37,611	0.28	34	This 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.
		2003-2004 oceanic	0	-	35	
		2009	2968	0.67	38	
Fraser's Dolphin	Northern Gulf of Mexico	1991-1994	127	0.9	30	Hansen *et al.* 1995
		1996-2001	726	0.7	33	
		2003-2004	0	-	35	
		2009	0	-	38	Current best population size estimate is unknown.
Rough-toothed Dolphin	Northern Gulf of Mexico	1991-1994 oceanic	852	0.31	30	
		1996-2001 oceanic	985	0.44	33	Mullin and Fulling 2004
		1998-2001 OCS	1,145	0.83	34	This 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.
		2003-2004 oceanic	1,508	0.39	35	

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		2009	624	0.99	0.05	
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**	
Bottlenose Dolphin	Northern Gulf of Mexico Continental Shelf	1998-2001	17,777	0.32	34	This 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	Blaylock and Hoggard 1994; Current best population size estimate for this stock is unknown because data are more than 8 years old.
		Northern 2007	**2,473**	**0.25**	**39**	
		Western 1992	3,499	0.21	31	Blaylock and Hoggard 1994; Current best population size estimate for this stock is unknown because data are more than 8 years old.
Bottlenose Dolphin	Northern Gulf of Mexico Bay, Sound and Estuarine (33 stocks)	Choctawhatchee Bay, 2007	179	0.04		Conn *et al.* 2011
		St. Joseph Bay, 2005-2007	146	0.18		Balmer *et al.* 2008

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		St. Vincent Sound, Apalachicola Bay, St. George Sound, 2008	439	0.14		Tyson *et al.* 2011
		Sarasota Bay, Little Sarasota Bay, 2007	160	-		Direct count; Wells 2009.
		Mississippi River Delta, 2011-12	332	.93		
		Mississippi Sound/ Lake Borgne, Bay Boudreau	901	0.63		
		Mississippi Sound/ Lake Borgne, Bay Boudreau	**3,046**	**0.06**		**Mullin 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		

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		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		
		Galveston Bay/East Bay/Trinity Bay[j]	152	0.43		
		Terrebonne Bay/Timbalier Bay	3,870	0.15		

Species	Stock	Year	Nbest	CV	Survey Number	Notes
		Mobile Bay/Bonsecour Bay	122	0.34		
		Pensacola Bay/East Bay	33	0.80		
		St. Andrew Bay	199	0.09		
		Apalachee Bay	491	0.39		
		Remaining 27 stocks	unknown	undetermined	31	Blaylock and Hoggard 1994; Current best population size estimate for each of these 27 stocks is unknown because data are more than 8 years old.

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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	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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

	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	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
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	0.82	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

	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	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
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		
2017	136	0.28	8	0.92	0	0	133	0.28	0	0	0	0	298	0.18	930	0.16		

Note: this table only includes observed bycatch. For a complete list of marine mamal 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

[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.

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		Bottlenose Dolphin, Northern Migratory Coastal Stock		Bottlenose Dolphin, Southern Migratory Coastal Stock		Bottlenose Dolphin, Northern NC Estuarine Stock		Bottlenose Dolphin, Southern NC Estuarine Stock		White-Sided Dolphin		Common Dolphin		Risso's Dolphin		Pilot Whale, Unid.		Harbor Seal		Gray Seal		Harp Seal	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
1994	0	0	0	0	na	na	na	na	na	na	na	na	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1995	103	0.57	56	1.66	na	na	na	na	na	na	na	na	0	0	7.4	0.69	0	0	0	0	0	0	0	0	0	0
1996	311	0.31	64	0.83	na	na	na	na	na	na	na	na	0	0	43	0.79	0	0	0	0	0	0	0	0	0	0
1997	572	0.35	0	0	na	na	na	na	na	na	na	na	45	0.82	0	0	0	0	0	0	0	0	0	0	0	0
1998	446	0.36	63	0.94	na	na	na	na	na	na	na	na	0	0	0	0	0	0	7	0	11	0.77	0	0	17	1.02
1999	53	0.49	0	0	na	na	na	na	na	na	na	na	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2000	21	0.76	0	0	na	na	na	na	na	na	na	na	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2001	26	0.95	na	na	na	na	na	na	na	na	na	na	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2002	unk	na	0	0	8.25-9.29	0.34-0.33	11.96-30.68	0.79-0.52	5.21-24.38	0.63-0.53	0.59-1.45	0.35-0.30	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2003	76	1.13	0	0	3.92-6.66	0.36-0.30	15.71-41.55	0.51-0.62	3.68-27.17	0.58-0.59	1.04-1.57	0.42-0.34	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		Bottlenose Dolphin, Northern Migratory Coastal Stock		Bottlenose Dolphin, Southern Migratory Coastal Stock		Bottlenose Dolphin, Northern NC Estuarine Stock		Bottlenose Dolphin, Southern NC Estuarine Stock		White-Sided Dolphin		Common Dolphin		Risso's Dolphin		Pilot Whale, Unid.		Harbor Seal		Gray Seal		Harp Seal	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
2004	137	0.91	0	0	4.86-7.28	0.35-0.33	33.50-40.10	0.79-0.51	4.03-18.96	0.62-0.49	0.92-2.17	0.43-0.36	0	0	0	0	0	0	0	0	15	0.86	69	0.92	0	0
2005	470	0.51	1[a]	na	4.89-6.52	0.39-0.32	69.40-80.30	0.60-0.64	3.95-15.20	0.60-0.49	0.48-0.78	0.41-0.30	0	0	0	0	0	0	0	0	63	0.67	0	0	0	0
2006	511	0.32	0	0	4.64-5.19	0.33-0.33	4.00-79.50	0.48-0.53	2.16-35.55	0.35-0.49	0.75-1.05	0.51-0.37	0	0	0	0	0	0	0	0	26	0.98	0	0	0	0
2007	58	1.03	0	0	0.00-3.18	0.00-1.08	0.00-6.00	0.00-0.97	0.00-9.69	0.00-0.95	0.00-0.00	0.00-0.00	0	0	0	0	34	0.73	0	0	0	0	0	0	38	0.9
2008	350	0.75	0	0	0.00-3.05	0.00-1.08	0.00-5.27	0.00-0.97	0.00-8.08	0.00-0.95	0.00-0.00	0.00-0.00	0	0	0	0	0	0	0	0	88	0.74	0	0	176	0.74
2009	201	0.55	0	0	0.00-23.86	0.00-0.83	0.00-37.61	0.00-0.86	0.00-46.79	0.00-0.82	0.00-0.00	0.00-0.00	0	0	0	0	0	0	0	0	47	0.68	0	0	0	0
2010	259	0.88	0	0	0.00-2.62	0.00-1.08	0.00-4.11	0.00-0.97	0.00-6.96	0.00-0.95	0.00-0.00	0.00-0.00	0	0	30	0.48	0	0	0	0	89	0.39	267	0.75	0	0

Year	Harbor Porpoise		Bottlenose Dolphin, Atlantic Offshore Stock		Bottlenose Dolphin, Northern Migratory Coastal Stock		Bottlenose Dolphin, Southern Migratory Coastal Stock		Bottlenose Dolphin, Northern NC Estuarine Stock		Bottlenose Dolphin, Southern NC Estuarine Stock		White-Sided Dolphin		Common Dolphin		Risso's Dolphin		Pilot Whale, Unid.		Harbor Seal		Gray Seal		Harp Seal	
	SI&M_est	CV	SI&M_est	CV	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est (min-max)[b]	CV[b]	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
2011	123	0.41	0	0	0.00-2.98	0.00-1.08	0.00-4.33	0.00-0.97	0.00-8.38	0.00-0.95	0.00-0.00	0.00-0.00	0	0	29	0.53	0	0	0	0	21	0.67	19	0.6	0	0
2012	63.41	0.83	0	0	tbd	tbd	tbd	tbd	tbd	tbd	tbd	tbd	0	0	15	0.93	0	0	0	0	0	0	14	0.98	0	0
2013	19	1.06	26	0.95	tbd	tbd	tbd	tbd	tbd	tbd	tbd	tbd	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			6.1-13.2	0.32-0.22	0-14.3	0.32	0.8-18.2	0.23			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	22	0.71	0	0	0	0	3	0.62	0	0	0	0

Note: this table only includes observed bycatch. For a complete list of marine mamal species interactions with this fishery please seehttps://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-gillnet-fishery-mmpa-list-fisheries

[a] Unextrapolated mortalities

[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.

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	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	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

	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	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
2004	0	0	0	0	200	0.3	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	0.3	0	0	0	0	unk	unk	unk	unk	0	0
2006	6.5	0.49	0	0	40	0.5	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	0.6	3	0.53	0	0	13	0.7	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	18.6	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

	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	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
2017	0	0	0	0	14.8	0.64	0	0	0	0	0	0	0	0	0	0	16	0.24	0	0	0	0

Note: this table only includes observed bycatch. For a complete list of marine mamal species interactions with this fishery please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/northeast-bottom-trawl-fishery-mmpa-list-fisheries[a] Unextrapolated mortalities

[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.

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

	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	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
1997	0	0	0	0	161	1.58	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
1999	0	0	0	0	0	0	0	0	0	0	228	1.03	0	0	0	0	0	0
2000	0	0	0	0	27	0.17	0	0	0	0	0	0	0	0	0	0	0	0
2001	0	0	0	0	27	0.19	103	0.27	0	0	39	0.3	0	0	0	0	0	0
2002	0	0	0	0	25	0.17	87	0.27	0	0	38	0.36	0	0	0	0	0	0
2003	0	0	0	0	31	0.25	99	0.28	0	0	31	0.31	0	0	0	0	0	0
2004	0	0	0	0	26	0.2	159	0.3	0	0	35	0.33	0	0	0	0	0	0
2005	0	0	0	0	38	0.29	141	0.29	0	0	31	0.31	0	0	0	0	0	0
2006	0	0	0	0	3	0.53	131	0.28	0	0	37	0.34	0	0	0	0	0	0
2007	0	0	11	0.42	2	1.03	66	0.27	33	0.34	0	0	0	0	0	0	0	0
2008	0	0	16	0.36	0	0	23	1	39	0.69	0	0	0	0	0	0	0	0
2009	0	0	21	0.45	0	0	167	0.46	23	0.5	0	0	0	0	24	0.92	38	0.7
2010	0	0	20	0.34	0	0	21	0.96	54	0.74	0	0	0	0	11	1.1	0	0
2011	0	0	34	0.31	0	0	271	0.25	62	0.56	0	0	0	0	0	0	25	0.57
2012	0	0	16	1.00	0	0	323	0.26	8	1	0	0	0	0	23	1	30	1.1
2013	0	0	0	0	0	0	269	0.29	42	0.71	0	0	0	0	11	0.96	29	0.67
2014	0	0	25	0.66	9.7	0.94	329	0.29	21	0.93	0	0	0	0	10	0.95	7	0.96
2015	0	0	0	0	0	0	250	0.32	40	0.63	0	0	0	0	7.4	1.0	0	0
2016	0	0	7.3	0.93	0	0	177	0.33	39	0.56	0	0	0	0	0	0	26	0.57

	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	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
2017	0	0	22.1	0.66	0	0	380	0.23	31	0.51	0	0	0	0	0	0	26	-.40

Note: this table only includes observed bycatch. For a complete list of marine mamal species interactions with this fishery please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/mid-atlantic-bottom-trawl-fishery-mmpa-list-fisheries

[a] Unextrapolated mortalities

[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.

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

	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	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
1999	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	0	0	0	0	0	0	4.6	0.74	0	0	0	0	0	0
2001	0	0	0	0	unk	na	0	0	0	0	11	0.74	0	0	0	0	0	0
2002	0	0	0	0	unk	na	0	0	0	0	8.9	0.74	0	0	0	0	0	0
2003	0	0	0	0	22	0.97	0	0	0	0	14	0.56	0	0	0	0	0	0
2004	0	0	0	0	0	0	0	0	0	0	5.8	0.58	0	0	0	0	0	0
2005	0	0	0	0	9.4	1.03	0	0	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	0	0	0	0	0	0
2007	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2008	0	0	0	0	0	0	0	0	0	0	0	0	16	0.61	0	0	0	0
2009	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1.3	0.81	0	0
2010	0	0	0	0	0	0	1[a]	na	0	0	0	0	0	0	2[a]	na	0	0
2011	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
2012	0	0	0	0	0	0	1[a]	na	0	0	0	0	1	0	1[a]	na	1[a]	na
2013	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	1[a]	na
2014	0	0	0	0	0	0	0	0	0	0	0	0	4	na	1[a]	na	0	0
2015	0	0	0	0	0	0	0	0	0	0	0	0	0	na	2[a]	na	0	0
2016	0	0	0	0	0	0	0	0	0	0	0	0	3	na	1[a]	na	0	0
2017	0	0	0	0	0	0	0	0	0	0	0	0	0	na	0	na	0	0

Note: this table only includes observed bycatch. For a complete list of marine mamal 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
[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.
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		Pilot Whale, Unidentified		Long-finned Pilot Whale		Harbor Seal		Gray Seal	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
1999	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2000	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2001	unk	na	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
2003	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2004	22	0.99	0	0	0	0	0	0	0	0	0	0	0	0
2005	58	1.02	0	0	0	0	0	0	0	0	0	0	0	0
2006	29	0.74	0	0	0	0	0	0	0	0	0	0	0	0
2007	12	0.98	3.2	0.7	0	0	0	0	0	0	0	0	0	0
2008	15	0.73	0	0	1[a]	na	0	0	0	0	0	0	0	0
2009	4	0.92	0	0	0	0	0	0	0	0	0	0	0	0
2010	0	0	0	0	0	0	0	0	0	0	1[a]	na	1[a]	na
2011	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2012	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2013	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2014	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2015	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2016	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	White-Sided Dolphin		Common Dolphin		Risso's Dolphin-Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Harbor Seal		Gray Seal	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
2017	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Note: this table only includes observed bycatch. For a complete list of marine mamal 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

[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.

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 - Atl.		Short-finned Pilot Whale - Atlantic		Beaked whale, Unidentified	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	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	0.732	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	11.8	0.699	1.5	1	0	0	305	0.29	0	0

	Pantropical Spotted dolphin - GMex		Bottlenose Dolphin, Atlantic Offshore Stock		Common Dolphin		Risso's Dolphin - Atlantic		Risso's Dolphin - Gmex		Pilot Whale, Unidentified/long-finned - Atl.		Short-finned Pilot Whale - Atlantic		Beaked whale, Unidentified	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
2012	0	0	61.8	0.68	0	0	15.1	1	29.8	1	0	0	170.1	0.33	0	0
2013	2.1	1	0	0	0	0	1.9	1	15.2	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	16.1	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

Note: this table only includes observed bycatch. For a complete list of marine mamal species interactions with this fishery please see https://www.fisheries.noaa.gov/national/marine-mammal-protection/atlantic-ocean-caribbean-gulf-mexico-large-pelagics-longline

[a] Unextrapolated mortalities

[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.

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	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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.
[a] Unextrapolated mortalities
[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks

Pelagic Pair Trawl.

	White-Sided Dolphin		Common Dolphin		Risso's Dolphin-Atlantic		Pilot Whale, Unidentified		Long-finned Pilot Whale		Bottlenose dolphin- Atlantic offshore	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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.
[a] Unextrapolated mortalities
[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.
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	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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.

[a] Unextrapolated mortalities

[b] Due to uncertainty in stock identification both minimum and maximum estimates are provide with associated CV's. As a result of uncertainty in stock identification, minimum and maximum mortality estimates are not additive across the Atlantic coastal and estuarine bottlenose dolphin stocks.

na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Part B: by Species

Harbor Porpoise

	Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gilllnet		Pelagic Drift Gillnet	
Year	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_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		

	Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gilllnet		Pelagic Drift Gillnet	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
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		

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

Common Bottlenose Dolphin, Atlantic Offshore Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gilllnet		Pelagic Drift Gillnet		Pelagic Longline	
Year	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_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	.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

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gilllnet		Pelagic Drift Gillnet		Pelagic Longline	
Year	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**
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

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 Gilllnet		Northeast Midwater Trawl		Pelagic Drift Gillnet	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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		

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		North Atlantic Bottom Trawl		NE Sink Gilllnet		Northeast Midwater Trawl		Pelagic Drift Gillnet	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
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	14.8	0.64	0	0	0	0		

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

Risso's Dolphin, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gilllnet		Pelagic Longline	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gilllnet		Pelagic Longline	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV
2017	31	0.51	0	0	0	0	0	0	0.2	1

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

Long-finned Pilot Whale, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Midwater Trawl		North Atlantic Bottom Trawl		NE Sink Gilllnet		Northeast Midwater Trawl		Pelagic Longline	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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

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

	PLL	
Year	**SI&M_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

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

Common Dolphin, Western North Atlantic Stock

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gilllnet		Northeast Midwater Trawl		Pelagic Drift Gillnet		Pelagic Longline	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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

	Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		North Atlantic Bottom Trawl		NE Sink Gilllnet		Northeast Midwater Trawl		Pelagic Drift Gillnet		Pelagic Longline	
Year	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**
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

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

Harbor Seal

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gilllnet		Northeast Midwater Trawl	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_est	CV	SI&M_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

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gilllnet		Northeast Midwater Trawl	
Year	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**
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

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

Gray Seal

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gilllnet		Northeast Midwater Trawl	
Year	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**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

	Herring Purse Seine		Mid-Atlantic Bottom Trawl		Mid-Atlantic Gillnet		Mid-Atlantic Midwater Trawl		Northeast Bottom Trawl		NE Sink Gilllnet		Northeast Midwater Trawl	
Year	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**
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

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 Gilllnet	
Year	SI&M_est	CV	SI&M_est	CV	SI&M_est	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

	Mid-Atlantic Gillnet		Northeast Bottom Trawl		NE Sink Gilllnet	
Year	**SI&M_est**	**CV**	**SI&M_est**	**CV**	**SI&M_est**	**CV**
2015	0	0	0	0	119	0.34
2016						
2017						

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

APPENDIX VI: Reports not updated in 2019
(all reports may be accessed at https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessment-reports-region)

Species	Stock	Updated
Killer whale	Western North Atlantic	2014
Northern bottlenose whale	Western North Atlantic	2014
Rough-toothed dolphin	Western North Atlantic	2018
Common bottlenose dolphin	Western North Atlantic Northern Migratory Coastal	2017
Common bottlenose dolphin	Western North Atlantic Southern Migratory Coastal	2017
Common bottlenose dolphin	Western North Atlantic South Carolina/Georgia Coastal	2017
Common bottlenose dolphin	Western North Atlantic Northern Florida Coastal	2017
Common bottlenose dolphin	Western North Atlantic Central Florida Coastal	2017
Common bottlenose dolphin	Northern North Carolina Estuarine System	2017
Common bottlenose dolphin	Southern North Carolina Estuarine System	2017
Common bottlenose dolphin	Northern South Carolina Estuarine System	2017
Common bottlenose dolphin	Charleston Estuarine System	2015
Common bottlenose dolphin	Northern Georgia/Southern South Carolina Estuarine System	2015
Common bottlenose dolphin	Central Georgia Estuarine System	2015
Common bottlenose dolphin	Southern Georgia Estuarine System	2015
Common bottlenose dolphin	Jacksonville Estuarine System	2015
Common bottlenose dolphin	Indian River Lagoon Estuarine System	2015
Common bottlenose dolphin	Biscayne Bay	2013
Common bottlenose dolphin	Florida Bay	2013
Hooded seal	Western North Atlantic	2018
Bryde’s whale	Northern Gulf of Mexico	2017
Cuvier's beaked whale	Northern Gulf of Mexico	2012
Blainville's beaked whale	Northern Gulf of Mexico	2012

Species	Stock	Updated
Gervais' beaked whale	Northern Gulf of Mexico	2012
Common bottlenose dolphin	Northern Gulf of Mexico	2014
Common bottlenose dolphin	Northern Gulf of Mexico, Continental shelf	2015
Common bottlenose dolphin	Northern Gulf of Mexico, Eastern coastal	2015
Common bottlenose dolphin	Northern Gulf of Mexico, Northern coastal	2015
Common bottlenose dolphin	Northern Gulf of Mexico, Western coastal	2015
Common bottlenose dolphin	Northern Gulf of Mexico, Oceanic	2015
Common bottlenose dolphin	Laguna Madre	2018
Common bottlenose dolphin	Neuces Bay/Corpus Christi Bay	2018
Common bottlenose dolphin	Copano Bay/Aransas Bay/San Antonio Bay/Redfish Bay/Espiritu Santo Bay	2018
Common bottlenose dolphin	Matagorda Bay/Tres Palacios Bay/Lavaca Bay	2018
Common bottlenose dolphin	Galveston Bay/East Bay/Trinity Bay	2018
Common bottlenose dolphin	Sabine Lake	2018
Common bottlenose dolphin	Calcasieu Lake	2018
Common bottlenose dolphin	Vermilion Bay/West Cote Blanche Bay/Atchafalaya Bay	2018
Common bottlenose dolphin	Terrebonne Bay/Timbalier Bay Estuarine System	2018
Common bottlenose dolphin	Mississippi River Delta	2018
Common bottlenose dolphin	Mobile Bay/Bonsecour Bay	2018
Common bottlenose dolphin	Perdido Bay	2018
Common bottlenose dolphin	Pensacola Bay/East Bay	2018
Common bottlenose dolphin	Choctawhatchee Bay	2015
Common bottlenose dolphin	Barataria Bay Estuarine System	2017
Common bottlenose dolphin	Mississippi Sound/Lake Borgne/Bay Boudreau	2017
Common bottlenose dolphin	St. Vincent Sound/Apalachicola Bay/St. George Sound	2018
Common bottlenose dolphin	Apalachee Bay	2018
Common bottlenose dolphin	Waccasassa Bay/Withlacoochee Bay/Crystal Bay	2018

Species	Stock	Updated
Common bottlenose dolphin	St. Joseph Sound/Clearwater Harbor	2018
Common bottlenose dolphin	Tampa Bay	2018
Common bottlenose dolphin	Sarasota Bay/Little Sarasota Bay	2018
Common bottlenose dolphin	Pine Island Sound/Charlotte Harbor/Gasparilla Sound/Lemon Bay	2018
Common bottlenose dolphin	Caloosahatchee River	2018
Common bottlenose dolphin	Estero Bay	2018
Common bottlenose dolphin	Chokoloskee Bay/Ten Thousand Islands/Gullivan Bay	2018
Common bottlenose dolphin	Whitewater Bay	2018
Common bottlenose dolphin	Florida Keys (Southwest Marathon Key to Marquesas Keys)	2018
Atlantic spotted dolphin	Northern Gulf of Mexico	2015
Pantropical spotted dolphin	Northern Gulf of Mexico	2015
Striped dolphin	Northern Gulf of Mexico	2012
Spinner dolphin	Northern Gulf of Mexico	2012
Rough-toothed dolphin	Northern Gulf of Mexico (Outer continental shelf and Oceanic)	2016
Clymene dolphin	Northern Gulf of Mexico	2012
Fraser's dolphin	Northern Gulf of Mexico	2012
Killer whale	Northern Gulf of Mexico	2012
False killer whale	Northern Gulf of Mexico	2012
Pygmy killer whale	Northern Gulf of Mexico	2012
Dwarf sperm whale	Northern Gulf of Mexico	2012
Pygmy sperm whale	Northern Gulf of Mexico	2012
Melon-headed whale	Northern Gulf of Mexico	2012
Risso’s dolphin	Northern Gulf of Mexico	2015
Pilot whale, short-finned	Northern Gulf of Mexico	2015
Sperm whale	Northern Gulf of Mexico	2015

Species	Stock	Updated
Sperm whale	Puerto Rico and US Virgin Islands stock	2010
Common bottlenose dolphin	Puerto Rico and US Virgin Islands stock	2011
Cuvier's beaked whale	Puerto Rico and US Virgin Islands stock	2011
Pilot whale, short-finned	Puerto Rico and US Virgin Islands stock	2011
Spinner dolphin	Puerto Rico and US Virgin Islands stock	2011
Atlantic spotted dolphin	Puerto Rico and US Virgin Islands stock	2011

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