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US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2017: (second edition)

US DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, Massachusetts
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NOAA Technical Memorandum NMFS-NE-245

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US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2017:

(second edition)

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CORRECTION: The first edition of this report, published in September 2018, had an incorrect value for North Atlantic right whale Potential Biological Removal (PBR). The PBR was reported as 1.4, but should have been reported as 0.9. In addition, a typographical error referencing the wrong stock name on page 149 in the Central Florida Coastal stock of bottlenose dolphins report was corrected.

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Editorial Notes

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

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

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Internet Availability: This issue of the NOAA Technical Memorandum NMFS-NE series is being as a paper and Web document in HTML (and thus searchable) and PDF formats and can be accessed at: http://www.nefsc.noaa.gov/nefsc/publications/.

Editorial Treatment: To distribute this report quickly, it has not undergone the normal technical and copy editing by the Northeast Fisheries Science Center's (NEFSC's) Editorial Office as have most other issues in the *NOAA Technical Memorandum NMFS-NE* series. Other than the four covers and first two preliminary pages, all writing and editing have been performed by – and all credit for such writing and editing rightfully belongs to – those so listed on the title page.

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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 2016 publication. Most of the changes incorporate new information into sections on population size and/or mortality estimates. A total of 21 of the Atlantic and Gulf of Mexico stock assessment reports were revised for 2017. The revised SARs include 13 strategic and 8 non-strategic stocks.

This report was prepared by staff of the Northeast Fisheries Science Center (NEFSC) and Southeast Fisheries Science Center (SEFSC). NMFS staff presented the reports at the February 2017 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 2018 date-stamp at the top right corner at the beginning of each report. Stocks not updated in 2017 are listed in Appendix VI.

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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 2011-2015. The "SAR revised" column indicates 2017 stock assessment reports that have been revised relative to the 2016 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	458	0	455	0.04[a]	0.1	0.9	5.36[a]	4.55[a]	Y	Y (a, m, p)
Humpback whale	Gulf of Maine	NEC	335	0.42	239	0.065	0.5	3.7	8.5[b]	6.7[b]	Y	Y (a, m, p, status)
Fin whale	Western North Atlantic	NEC	1,618	0.33	1,234	0.04	0.1	2.5	2.65[c]	1.05[c]	Y	Y (m)
Sei whale	Nova Scotia	NEC	357	0.52	236	0.04	0.1	0.5	0.8[d]	0[d]	Y	N (2016)
Minke whale	Canadian east coast	NEC	2,591	0.81	1,425	0.04	0.5	14	9.15[e]	7.75[e]	N	Y (m)
Blue whale	Western North Atlantic	NEC	unk	unk	440	0.04	0.1	0.9	unk	unk	Y	N (2010)
Sperm whale	North Atlantic	NEC	2,288	0.28	1,815	0.04	0.1	3.6	0.8	0.8	Y	N (2014)

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
Dwarf sperm whale	Western North Atlantic	SEC	3,785[h]	0.47[k]	2,598[h]	0.04	0.4	21	3.5	3.5 (1.0)	N	N (2016)
Pygmy sperm whale	Western North Atlantic	SEC	3,785[h]	0.47[k]	2,598[h]	0.04	0.4	21	3.5	3.5 (1.0)	N	N (2016)
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	N (2007)
False killer whale	Western North Atlantic	SEC	442	1.06	212	0.04	0.5	2.1	unk	unk	Y	N (2014)
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	6,532	0.32	5,021	0.04	0.5	50	0.4	0.2	N	N (2013)
Blainville's beaked whale	Western North Atlantic	NEC	7,092[g]	0.54	4,632[g]	0.04	0.5	46	0.2	0.2	N	N (2013)
Gervais beaked whale	Western North Atlantic	NEC	7,092[g]	0.54	4,632[g]	0.04	0.5	46	0	0	N	N (2013)
Sowerby's beaked whale	Western North Atlantic	NEC	7,092[g]	0.54	4,632[g]	0.04	0.5	46	0	0	N	N (2014)
True's beaked whale	Western North Atlantic	NEC	7,092[g]	0.54	4,632[g]	0.04	0.5	46	0	0	N	N (2013)

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
Melon-headed whale	Western North Atlantic	SEC	unk	unk	unk	0.04	0.5	unk	0	0	N	N (2007)
Risso's dolphin	Western North Atlantic	NEC	18,250	0.46	12,619	0.04	0.48	126	43.2	43 (0.27)	N	Y (m)
Pilot whale, long-finned	Western North Atlantic	NEC	5,636	0.63	3,464	0.04	0.5	35	38	38 (0.15)	Y	N (2016)
Pilot whale, short-finned	Western North Atlantic	SEC	21,515	0.37	15,913	0.04	0.5	159	192	192 (0.17)	Y	N (2016)
Atlantic white-sided dolphin	Western North Atlantic	NEC	48,819	0.61	30,403	0.04	0.5	304	57	57 (0.15)	N	Y (m)
White-beaked dolphin	Western North Atlantic	NEC	2,003	0.94	1,023	0.04	0.5	10	0	0	N	N (2007)
Common dolphin	Western North Atlantic	NEC	70,184	0.28	55,690	0.04	0.5	557	437	437 (0.10)	N	Y (m)
Atlantic spotted dolphin	Western North Atlantic	SEC	44.715	0.43	31,610	0.04	0.5	316	0	0	N	N (2013)
Pantropical spotted dolphin	Western North Atlantic	SEC	3,333	0.91	1,733	0.04	0.5	17	0	0	N	N (2013)
Striped dolphin	Western North Atlantic	NEC	54,807	0.3	42,804	0.04	0.5	428	0	0	N	N (2013)
Fraser’s dolphin	Western North Atlantic	SEC	unk	unk	unk	0.04	0.5	unk	0	0	N	N (2007)

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	Western North Atlantic	SEC	271	1.0	134	0.04	0.5	1.3	0	0	N	N (2013)
Clymene dolphin	Western North Atlantic	SEC	unk	unk	unk	0.04	0.5	undet	0	0	N	N (2013)
Spinner dolphin	Western North Atlantic	SEC	unk	unk	unk	0.04	0.5	unk	0	0	N	N (2013)
Common bottlenose dolphin	Western North Atlantic, offshore	SEC	77,532[f]	0.40	56,053[f]	0.04	0.5	561	39.4	39.4 (0.29)	N	N (2016)
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	6.1-13.2	Y	Y (a, m, p)
Common bottlenose dolphin	Western North Atlantic, southern migratory coastal	SEC	3,751	.060	2,353	0.04	0.5	23	0-14.3	0-14.3	Y	Y (a, m, p)
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	1.0-1.2	Y	Y (a, m, p)
Common bottlenose dolphin	Western North Atlantic, northern Florida coastal	SEC	877	0.49	595	0.04	0.5	6.0	0.6	0	Y	Y (a, m, p)
Common bottlenose dolphin	Western North Atlantic, central Florida coastal	SEC	1,218	0..35	913	0.04	0.5	9.1	0.4	0.4	Y	Y (a, m, p)
Common bottlenose dolphin	Northern North Carolina Estuarine System	SEC	823	0.06	782	0.04	0.5	7.8	0.8-18.2	0.2-17.6	Y	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
Common bottlenose dolphin	Southern North Carolina Estuarine System	SEC	unk	unk	unk	0.04	0.5	undet	0.4-0.6	0.4-0.6	Y	Y (m)
Common bottlenose dolphin	Northern South Carolina Estuarine System	SEC	unk	unk	unk	0.04	0.5	unk	0.2	0.2	Y	N (2015)
Common bottlenose dolphin	Charleston Estuarine System	SEC	unk	unk	unk	0.04	0.5	undet	unk	unk	Y	N (2015)
Common bottlenose dolphin	Northern Georgia/ Southern South Carolina Estuarine System	SEC	unk	unk	unk	0.04	0.5	unk	1.4	1.4	Y	N (2015)
Common bottlenose dolphin	Central Georgia Estuarine System	SEC	192	0.04	185	0.04	0.5	1.9	unk	unk	Y	N (2015)
Common bottlenose dolphin	Southern Georgia Estuarine System	SEC	194	0.05	185	0.04	0.5	1.9	unk	unk	Y	N (2015)
Common bottlenose dolphin	Jacksonville Estuarine System	SEC	unk	unk	unk	0.04	0.5	unk	1.2	1.2	Y	N (2015)
Common bottlenose dolphin	Indian River Lagoon Estuarine System	SEC	unk	unk	unk	0.04	0.5	unk	4.4	4.4	Y	N (2015)
Common bottlenose dolphin	Biscayne Bay	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2013)
Common bottlenose dolphin	Florida Bay	SEC	unk	unk	unk	0.04	0.5	undet	unk	unk	N	N (2013)
Harbor porpoise	Gulf of Maine/Bay of Fundy	NEC	79,833	0.32	61,415	0.046	0.5	706	307	307 (0.16)	N	Y (m)
Harbor seal	Western North Atlantic	NEC	75,834	0.15	66,884	0.12	0.5	2,006	368	356 (0.11)	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
Gray seal	Western North Atlantic	NEC	27,131	0.10	23,158	0.12	1.0	1,389	5,207	1,088 (0.09)	N	Y (a,m,p)
Harp seal	Western North Atlantic	NEC	unk	unk	unk	0.12	1.0	unk	215,998	43 (0.24)	N	Y (2013)
Hooded seal	Western North Atlantic	NEC	unk	unk	unk	0.12	0.75	unk	5,199	25(0.82)	N	N (2007)
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	Y (m)
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	0.6	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	1.6	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	0.4	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	0.6	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	Gulf of Mexico, bay, sound and estuary (27 stocks)	SEC	unk for all but 2 stocks	unk	unk for all but 2 stocks	0.04	0.5	undet for all but 2 stocks	unk	unk	Y for all	N (2016)

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	Barataria Bay	SEC	2,306	0.09	2,138	0.04	04	17	160	0.8	Y	Y (a, m, p)
Common bottlenose dolphin	Mississippi Sound, Lake Borgne, Bay Boudreau	SEC	3,046	0.06	2,896	0.04	0.4	23	310	1.0	Y	Y(a, m, p)
Common bottlenose dolphin	St. Joseph Bay	SEC	152	0.08	unk	0.04	0.5	undet	unk	unk	Y	N (2015)
Common bottlenose dolphin	Choctawhatchee Bay	SEC	179	0.04	unk	0.04	0.5	undet	0.4	0.4	Y	N (2015)
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)
Rough-toothed dolphin	Gulf of Mexico	SEC	624	0.99	311	0.04	0.4	3	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)

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
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[i]	Gulf of Mexico	SEC	2,415	0.66	1456	0.04	0.5	15	0.5	0.5 (1.0)	N	N (2015)
Sperm Whale	Puerto Rico and US Virgin Islands	SEC	unk	unk	unk	0.04	0.1	unk	unk	unk	Y	N (2010)
Common bottlenose dolphin	Puerto Rico and US Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)
Cuvier's beaked whale	Puerto Rico and US Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)
Pilot whale, short-finned	Puerto Rico and US Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)
Spinner dolphin	Puerto Rico and US Virgin Islands	SEC	unk	unk	unk	0.04	0.5	unk	unk	unk	Y	N (2011)
Atlantic spotted dolphin	Puerto Rico and US 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 5.36 per year. This is derived from two components: 1) non-observed fishery entanglement records at 4.55 per year, and 2) ship strike records at 0.81 per year.

b. The total estimated human-caused mortality and serious injury to the Gulf of Maine humpback whale stock is estimated as 8.25 per year. This average is derived from two components: 1) incidental fishery interaction records 6.45; 2) records of vessel collisions, 1.8.

c. The total estimated human-caused mortality and serious injury to the Western North Atlantic fin whale stock is estimated as 2.65 per year . This average is derived from two components: 1) incidental fishery interaction records 1.05; 2) records of vessel collisions, 1.6.

d. The total estimated human-caused mortality and serious injury to the Nova Scotia sei whale stock is estimated as 0.8 per year. This average is derived from two components: 1) incidental fishery interaction records 0; 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 9.0 per year. This average is derived from two components: 1) 7.6 minke whales per year (unknown CV) from U.S. and Canadian fisheries using strandings and entanglement data; and 3) 1.4 per year from vessel strikes.

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 species of *Mesoplodon* 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.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The western North Atlantic right whale population ranges primarily from calving grounds in coastal waters of the southeastern United States 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 individual female and calf pairs beyond the sole known calving and wintering ground in the waters of the southeastern United States. Whatever the case, the location of much of the population is unknown during the winter. Surveys flown in an area from 31 to 160 km from the shoreline off northeastern Florida and southeastern Georgia from 1996 to 2001 had 3 sightings in 1996, 1 in 1997, 13 in 1998, 6 in 1999, 11 in 2000, and 6 in 2001 (within each year, some were repeat sightings of previously recorded individuals). All but 1 of the sightings occurred within 90 km of the shoreline—the remaining sighting occurred ~140 km offshore (NMFS unpub. data). An offshore survey in March 2010 observed the birth of a right whale in waters 75 kms 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 United States remains unclear.

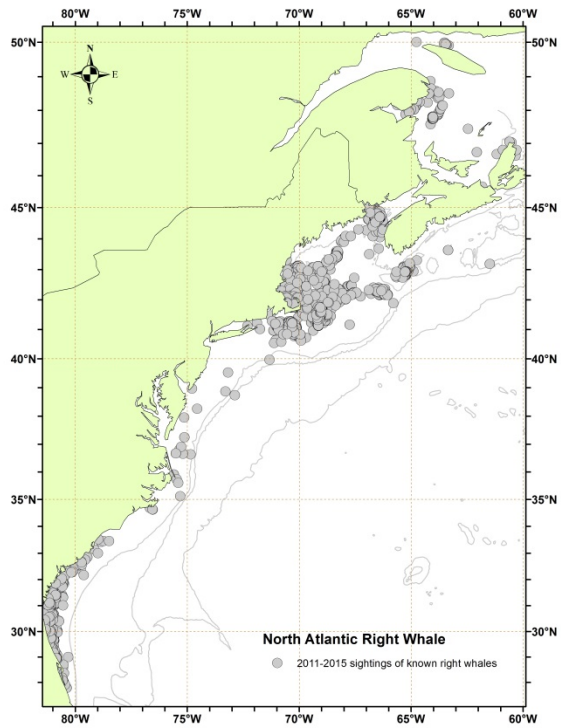

Figure 1. Distribution of sightings of known North Atlantic right whales, 2011-2015. Isobaths are the 100-m, 1000-m and 4000-m depth contours.

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 United States; the Great South Channel; Jordan Basin; Georges Basin along the northeastern edge of Georges Bank; Cape Cod and Massachusetts Bays; the Bay of Fundy; and the Roseway Basin on the Scotian Shelf (Brown *et al*. 2001; Cole *et al.* 2013). 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 and somewhat distant 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 recent case of a calf apparently being born in the Gulf of Maine (Patrician *et al*. 2009) and another newborn was detected in Cape Cod Bay in 2013 (Center for Coastal Studies, Provincetown, MA USA, unpub. data).

New England waters are important feeding habitats for right whales, where they feed primarily on copepods (largely of the genera *Calanus* and *Pseudocalanus*). Right whales must locate and exploit extremely dense patches of zooplankton to feed efficiently (Mayo and Marx 1990). These dense zooplankton patches are likely a primary characteristic of the spring, summer, and fall right whale habitats (Kenney *et al.* 1986, 1995). While feeding in the coastal waters off Massachusetts has been better studied than in other areas, right whale feeding has also been observed on the margins of Georges Bank, in the Great South Channel, in the Gulf of Maine, in the Bay of Fundy, and over the Scotian Shelf (Baumgartner *et al.* 2007). The characteristics of acceptable prey distribution in these areas are beginning to emerge (Baumgartner *et al.* 2003; Baumgartner and Mate 2003). The National Marine Fisheries Service (NMFS) and Center for Coastal Studies aerial surveys during springs of 1999–2006 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 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). 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). In the most recent years (2012–2015), surveys have detected fewer individuals in the Great South Channel and the Bay of Fundy, indicating an important shift in habitat use patterns.

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.

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 be indicative of 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 66% of all North Atlantic right whales identified through 2001. This work has improved our understanding of genetic variability, number of reproductively active individuals, reproductive fitness, parentage, and relatedness of individuals (Frasier *et al.* 2007).

One emerging result of the genetic studies is the importance of obtaining biopsy samples from calves on the calving grounds. Only 60% of all known calves are 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% are 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. However, 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). This inference of additional animals that have never been captured photographically and/or genetically suggests the existence of potentially important habitats that remain to be described. Although intriguing to consider yet undiscovered habitats, hypotheses relying on different assumptions of sampling processes and breeding strategies may provide alternative explanations for missing males. Taking all available evidence together, it is possible that there is more than one breeding stock of North Atlantic right whales.

POPULATION SIZE

The western North Atlantic minimum stock size is based on models 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 on 20 October 2016. A hierarchical, state-space Bayesian open population model of these histories produced a median abundance value of 458 individuals (95% credible intervals 444–471). North Atlantic right whales represent one of the most intensely studied populations of cetaceans in the world with effort supported by a rigorously maintained individual sightings database and considerable survey effort throughout its range. However, minor uncertainties exist in the estimation of Nmin because it is based on a probabilistic model.

Historical Abundance

An estimate of pre-exploitation population size is not available. 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). The stock of right whales may have already been substantially reduced by the time whaling was begun by colonists in Massachusetts 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 during January 1700. Reeves *et al*. (2007) calculated that a minimum of 5500 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 estimate. 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 458. The minimum population estimate is 455.

Current Population Trend

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

An increase in carcass detections in 2004 and 2005 was cause for serious concern (Kraus *et al.* 2005). Of those mortalities, six were adult females, three of which were carrying near-term fetuses. Furthermore, four of these females were just starting to bear calves, losing their complete lifetime reproduction potential. Calculations based on demographic data through 1999 (Fujiwara and Caswell 2001) indicated that this mortality rate increase would reduce population growth by approximately 10% per year (Kraus *et al.* 2005). Strong evidence for flat or negative growth exists in the time series of minimum number alive during 1998–2000, which coincided with very low calf production in 2004. However, the population continued to grow since that apparent interval of decline until the most recent year included in this analysis (Figure 2).

Examination of the abundance estimates for the years 1990–2011 (Figure 2) suggested 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 2015 when the final estimate was 458 individuals (95% credible intervals 444–471; Pace *et al.* 2017). 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. 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. Some caution is advised in interpreting the apparent downward trend in abundance in 2012, but without evidence to the contrary, it is possible that this deflection represents a true population decline.

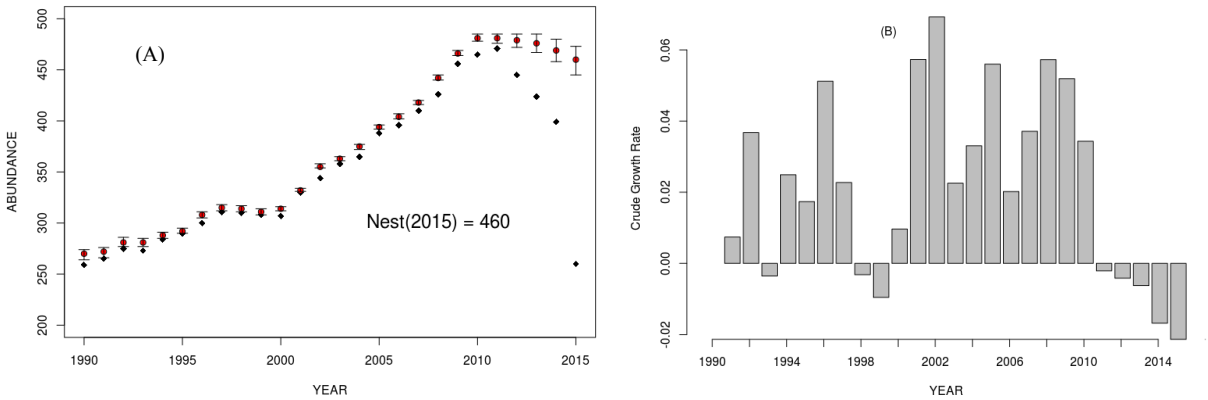

Figure 2. (A) Abundance estimates for North Atlantic right whales. Estimates are the median values of a posterior distribution from modeled capture histories. Also shown are minimum number alive (diamonds) of cataloged individuals known to be alive in any given year, which includes all whales known to be alive prior to that year and seen in that year or subsequently plus all whales newly cataloged that year. 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 1990–2014, at least 411 calves were born into the population. The number of calves born annually ranged from 1 to 39, and averaged 16.4 but was highly variable (SD=9.2). The fluctuating abundance observed from 1990 to 2014 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. 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, which might be expected from a small population, productivity in North Atlantic right whales lacks a definitive trend.

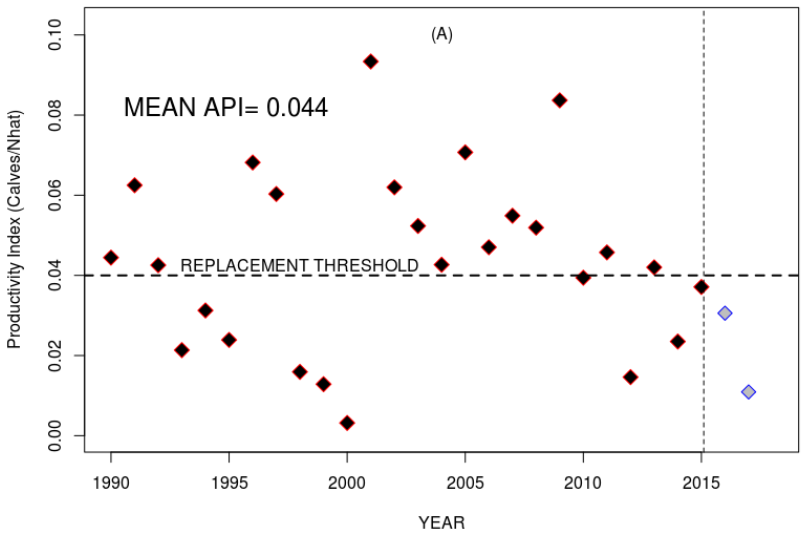

Figure 3. Productivity in the North Atlantic right whale population as characterized by calves detected/(N_{est}). The Nest values are the model-based estimates of Figure 2(A). Note that the 2016 and 2017 values assume the 2015 estimated abundance.

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 and possibly increased energy expenditures related to non-lethal entanglements (Rolland *et al.* 2016; 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 0.10 because this species is listed as endangered under the Endangered Species Act (ESA). The minimum population size is 455. 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 0.9.

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2011 through 2015, the minimum rate of annual human-caused mortality and serious injury to right whales averaged 5.36 per year. This is derived from two components: 1) incidental fishery entanglement records at 4.55 per year, and 2) vessel strike records at 0.81 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.

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 also 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.* 2017). For the purposes of this report, discussion is primarily limited to those records considered confirmed human-caused mortalities or serious injuries. Annual rates calculated from detected mortalities should not be considered an unbiased estimate of human-caused mortality, but they represent a definitive lower bound. 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 biased low. 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). 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 before being detected.

Background

The details of a particular mortality or serious injury record often require a degree of interpretation (Moore *et al.* 2005). 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 may have a greater effect relative to population growth rates than for other whales. The principal factors believed to be retarding growth and recovery of the population are vessel strikes and entanglement with fishing gear. Between 1970 and 1999, a total of 45 right whale mortalities was recorded (IWC 1999; Knowlton and Kraus 2001; Glass *et al.* 2009). Of these, 13 (28.9%) were neonates that were believed to have died from perinatal complications or other natural causes. Of the remainder, 16 (35.6%) resulted from vessel strikes, 3 (6.7%) were related to entanglement in fishing gear (in two cases lobster gear, and in one gillnet gear), and 13 (28.9%) were of unknown cause. At a minimum, therefore, 42.2% of the observed total for the period and 50% of the 32 non-calf deaths was attributable to human impacts (calves accounted for three deaths from ship strikes). Young animals, ages 0–4 years, are apparently the most impacted portion of the population (Kraus 1990).

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*. 2017).

Fishery-Related Mortality and Serious Injury

Reports of mortality and serious injury relative to PBR as well as total human impacts are contained in records maintained by the New England Aquarium and the NMFS Northeast and Southeast Regional Offices (Table 1). From 2011 through 2015, 24 records of mortality or serious injury (including records from both U.S. and Canadian waters, pro-rated to 22.75 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 4.55 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. Five serious injuries were prevented by intervention during 2011–2015 (Henry *et al.* 2017). 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.

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 1032 definite, unique entanglement events on the 626 individual whales identified (Knowlton *et al*. 2012). Most individual whales (83%) were entangled at least once, and almost half of them (306 of 626) 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.

Incidents of entanglements in waters of Atlantic Canada and the U.S. east coast were summarized by Read (1994) and Johnson *et al.* (2005). In six records of right whales that were entangled in groundfish gillnet gear in the Bay of Fundy and Gulf of Maine between 1975 and 1990, the whales were either released or escaped on their own, although several whales were observed carrying net or line fragments. A right whale mother and calf were released alive from a herring weir in the Bay of Fundy in 1976. Gillnet gear entanglements in the U.S. can also be fatal. A calf died in 2006, apparently victim of a gillnet entanglement, and other whales initially detected in gillnet gear have subsequently not been seen alive (NMFS unpub. data).

For all areas, specific details of right whale entanglement in fishing gear are often lacking. When direct or indirect mortality occurs, some carcasses come ashore and are subsequently examined, or are reported as "floaters"

at sea. The number of unreported and unexamined carcasses is unknown, but may be significant in the case of floaters. More information is needed about fisheries interactions and where they occur.

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 2011 through 2015 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 0.81 whales per year.

Table 2. Confirmed human-caused mortality and serious injury records of right whales: 2011–2015 [a]

Date[b]	Fate	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
01/20/2011	Serious Injury	3853	off Edisto Beach, SC	VS	1	US	-	16 deep lacerations across back, potentially penetrating body cavity. Vessel determined to be > 65 ftb. No resights.
02/13/2011	Serious Injury	3993	off Tybee Island, GA	EN	1	XU	NR	Line originates from front of whale. Attachment point(s) unknown. Right flipper very light in color. Overall body color light and body condition thin. No resights.
03/16/2011	Mortality		Cape Island SC	EN	1	XU	GU	Multiple wraps embedded in right pectoral bones.
03/27/2011	Serious Injury		Nags Head, NC	VS	1	US	-	Dependent/newborn calf of 1308. 1308 was found dead on 27Mar2011--COD[c] = ship strike. Calf last seen (with 1308) in January 2011.
03/27/2011	Mortality	1308	Nags Head, NC	VS	1	US	-	Fractured right skull.
04/22/2011	Serious Injury	3302	off Martha's Vineyard, MA	EN	1	XU	NR	Single, dark line just behind nares, likely entering either side of the mouth. Line appears tight. No trailing line seen. Last sighted November 2011 with no change in configuration.

09/03/2011	Serious Injury	2660	off Cap-de-Rosiers, Gaspe Bay, QC	EN	1	XC	NP	No gear present, but evidence of extensive, constricting gear with major ent. wounds and in poor condition. No longer with calf. Only blowing out of left blowhole. No resights post Sep. 2011.
09/18/2011	Prorated Injury	4090	off Gloucester, MA	EN	.75	XU	GN	Juvenile whale with gillnet mesh and floatline w/ floats wrapped around whale's head and tail. Whale thrashing. Full configuration unknown. No resights.
09/27/2011	Prorated Injury	3111	off Grand Manan, NB	EN	.75	XC	NR	Attachment point at left mouthline and/or flipper with multiple lines draping over peduncle. Lt. to mod. cyamid load and sloughing skin.
02/15/2012	Serious Injury		off Provincetown, MA	EN	1	XU	NR	Constricting gear across head and health decline.
07/19/2012	Mortality		Clam Bay, NS	EN	1	XC	GU	Multiple constricting wraps on peduncle; COD - peracute underwater entrapment.
09/24/2012	Serious Injury		Bay of Fundy, NS	EN	1	XC	NP	New significant raw & healing entanglement wounds on head, dorsal & ventral peduncle, and leading fluke edges. Health decline: moderate cyamid load, thin
12/07/2012	Prorated Injury		off Ossabaw Island, GA	VS	.52	US	-	46' vessel, 12-13 kts struck whale. Animal not resighted but large expanding pool of blood at surface.

12/18/2012	Mortality		off Palm Coast, FL	EN	1	US	PT	Constricting & embedded wraps w/ associated hemorrhaging at peduncle, mouthline, tongue, oral rete, rostrum & pectoral; malnourished.
07/12/2013	Prorated Injury	3123	off Virginia Beach, VA	EN	.75	XU	NR	Constricting gear cutting into mouthline; Partially disentangled; final configuration unknown
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	Entanglement discovered during photo processing just after the sighting. Constricting rostrum wrap with line trailing to at least mid-body. No resights.
04/02/2014	Serious Injury	3390	Cape Cod Bay, MA	EN	1	XU	NP	No gear present but new, healing ent. injuries. Evidence of a rostrum wrap, body wrap, and damage to right pectoral, peduncle, and leading fluke edges. Resights indicate health decline.
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 Provincetown, MA	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 October 2014.
09/04/2014	Mortality		Far south of St. Pierre & Miquelon, off the south coast ofNL	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.
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.
Five-year totals		Shipstrike (US/CN/XU/XC)			0.81 (0.81/ 0.00/ 0.00/ 0.00)			
		Entanglement (US/CN/XU/XC)			4.55 (0.40/ 0.00/ 2.45/ 1.70)			

a. For more details on events please see Henry *et al.* 2017.

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 is 99.99% likely to be declining (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). A status review by the National Marine Fisheries Service affirms endangered status (NMFS 2012). 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 5.36 right whales per year from 2011 through 2015. Given that PBR has been calculated as 0.9, human-caused mortality or serious injury for this stock can 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).

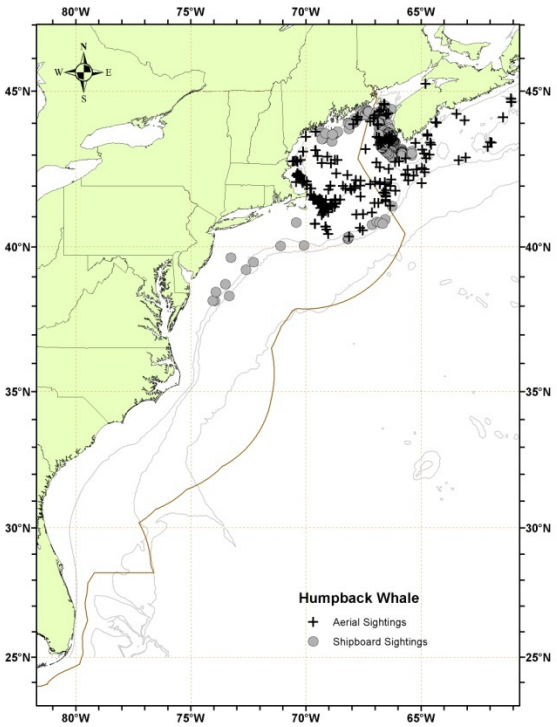

Figure 1. Distribution of humpback whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010 and 2011. Isobaths are the 100-m, 1000-m and 4000-m depth contours.

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.

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). 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 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. In collaboration with colleagues in the French islands of St. Pierre and Miquelon, a new photographic catalogue and concurrent matching effort is being undertaken for this region (J. Lawson, DFO, pers. comm.).

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 areas were becoming an increasingly important habitat for juvenile humpback whales and that anthropogenic factors may negatively impact whales in this area. For the period 2011–2015, there are records of 43 humpback whale strandings between New York and Florida in the Marine Mammal Health and Stranding Response database (accessed 17 May 2017). There have also been a number of wintertime humpback sightings in coastal waters of the southeastern U.S. 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. Other sightings of note include multiple humpbacks 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 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).

A key question with regard to humpback whales off the southeastern and mid-Atlantic states is their stock identity. This topic 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.

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 (Wienrich *et al.* 1997). Diel patterns in humpback foraging behavior have been shown to correlate with diel patterns in sand lance behavior (Friedlaender *et al.* 2009).

The key 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. 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.

POPULATION SIZE

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

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.

Minimum Population Estimate

For statistically-based estimates, 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 most recent line-transect survey, which did not include the Scotian Shelf portion of the stock, produced an estimate of abundance for Gulf of Maine humpback whales of 335 animals (CV=0.42), with a resultant minimum population estimate for this stock of 239 animals. In addition to the uncertainty associated with any statistical model and those uncertainties particular to distance sampling for which the magnitude is ostensibly accounted for in the CV, a key uncertainty with this minimum population estimate is its age. The line-transect estimate is negatively biased due to not being corrected for availability.

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

Current Population Trend

As detailed below, previous analyses concluded that the Gulf of Maine humpback whale stock is characterized by a positive trend in size. 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), but results may have been confounded by distribution shifts. Whether the reported positive trends continued into the current evaluation period is uncertain. As stated above, the current population estimate cannot be compared with the previous one due to differences in methodology. There is no observed increase in detected mortalities, so the reported drop in abundance from previous years could be an artifact of this methodology shift.

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 239 whales. The maximum productivity rate is 0.065. 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. 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 3.7 whales.

ANNUAL HUMAN-CAUSED SERIOUS INJURY AND MORTALITY

For the period 2011 through 2015, the minimum annual rate of human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 8.25 animals per year. This value includes incidental fishery interaction records, 6.45; and records of vessel collisions, 1.8 (Table 2; Henry *et al.* 2017).

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, discussion is primarily limited to those records considered to be confirmed human-caused mortalities or serious injuries.

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. 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). 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.

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 1762 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.

Robbins and Mattila (2001) reported that males were more likely to be entangled than females. Annually updated inferences made from scar prevalence and multistate models of GOM humpback whales that (1) younger animals are more likely to become entangled than adults, (2) juvenile scarring rates may be trending up, (3) maybe less than 10% of humpback entanglements are ever reported, and (4) 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 in Newfoundland and Labrador, cod traps caused the most entanglements and entanglement mortalities (21%) of humpbacks between 1979 and 1992. They also reported that gillnets were the primary cause of entanglements and entanglement mortalities (20%) of humpbacks in the Gulf of Maine between 1975 and 1990. In more recent times, following the collapse of the cod fishery, groundfish gillnets for other fish species and crab pot lines have been the most common sources of humpback entanglement in Newfoundland. Since the crab pot fishery is primarily an offshore activity on the Grand Banks, these entanglements are hard to respond to and are likely underreported. One humpback whale was reported released alive (status unknown) from a herring weir off Grand Manan in 2009 (H. Koopman, UNC Wilmington, pers. comm.). In U.S. waters, 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.

Wiley *et al.* (1995) reported that serious injuries attributable to ship strikes are 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 are extremely common (see Robbins 2010, 2011, 2012) and 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 2010 through 2014, there are only 9 reports of serious injuries and mortalities as a result of collision with a vessel and 40 records of injuries (prorated or serious) and mortalities attributed to entanglement. 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 this writing (Henry *et al.* 2016).

Fishery-Related Serious Injuries and Mortalities

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

In 2011 a humpback was caught on an observed gillnet trip (disentangled and released free of gear; Henry *et al.* 2016), and in 2012 there was an observed interaction with a humpback whale in mid-Atlantic gillnet gear (non-serious injury). 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.

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*. *2017*) 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, which is nearly 7 times PBR.

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): 2011–2015 [a]

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
07-Jan-11	Serious Injury	-	off Oregon Inlet, NC	EN	1	US	GN	Extensive entanglement w/ netting covering majority of body including head, blowholes, & flukes. Immobile & drifting.
01-Feb-11	Serious Injury	EKG	off Bar Harbor, ME	EN	1	US	NR	Anchored. Cuts were made to gear but whale remained anchored.
07-Mar-11	Mortality	-	Thorofare Bay, NC	VS	1	US	-	Live stranded w/ 8 deep lacerations across back. Euthanized.
11-Apr-11	Prorated Injury	-	off Rockport, MA	EN	0.75	XU	NR	Full configuration unknown.
5-May-11	Mortality	-	Little Compton, RI	VS	1	US	-	Hemorrhaging at left jaw associated w/ blunt trauma.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
27-May-11	Mortality	-	Island Beach State Park, NJ	VS	1	US	-	5 broken vertebral processes along left side w/ associated hemorrhaging.
30-May-11	Prorated Injury	-	off Orleans, MA	EN	0.75	XU	NR	Full configuration unknown.
2-Jul-11	Serious Injury	-	off Provincetown, MA	EN	1	XU	NP	Young whale. Missing flukes attributed to chronic entanglement. Laceration due to VS appears minor. Significant health decline: emaciated, swimming by use of pectorals only
9-Jul-11	Prorated Injury	-	off Monomoy Island, MA	EN	0.75	XU	NR	Full configuration unknown.
10-Jul-11	Prorated Injury	-	off Monomoy Island, MA	EN	0.75	XU	NR	Report of two entangled whales but could not confirm that both were entangled. Full configuration unknown.
21-Jul-11	Prorated Injury	-	off Oregon Inlet, NC	EN	0.75	XU	NR	Full configuration unknown.
10-Oct-11	Serious Injury	Clutter	off Grand Manan Island, New Brunswick	EN	1	XC	NR	Embedded wraps at fluke insertion.
29-Apr-12	Serious Injury	-	off Chatham, MA	EN	1	US	NR	SI based on description of body position, which indicates anchored
29-Jul-12	Serious Injury	-	off Gloucester, MA	EN	1	XU	NR	Calf w/ line cutting into peduncle

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
4-Aug-12	Serious Injury	Aphid	off Provincetown, MA	EN	1	XU	NR	Line exiting both sides of mouth, under flippers, twisting together aft of the dorsal fin & trailing 75 ft past flukes; no wraps. Health decline: thin w/ graying skin.
21-Aug-12	Prorated Injury	2011 Calf of Wizard	off Provincetown, MA	EN	0.75	XU	MF	Full configuration unknown
24-Aug-12	Serious Injury	Forceps	off Provincetown, MA	EN	1	US	NR	Closed, possibly weighted, bridle w/ large tangle of line just above left eye. SI due to odd behavior & apparent difficulty staying at the surface.
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	GU	Embedded line in mouth w/ associated hemorrhaging & necrosis; evidence of constriction at pectorals, peduncle & fluke w/ associated hemorrhaging; emaciated

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
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		Free-swimming with buoy and highflier trailing 100ft aft of flukes. Attachment point(s) unknown. Unable to confirm if resighted on 21Jun2014.
21-Jun-14	Prorated Injury		5 mi E of Gloucester, MA	EN	0.75	XU		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.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
18-Jul-14	Serious Injury		Provincetown Harbor, MA	EN	1	XU		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

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		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		NYC0010	off Rockaway Beach, Long Island, NY	EN	.75	US		Free-swimming with netting and rope with floats wrapping flukes. Entanglement noticed during photo processing. Full 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-14			15 mi E of Metompkin Inlet, VA	EN	.75	US		Free-swimming whale with line & netting on left fluke blade. Gear appeared heavy. Full configuration unknown. No resights.
15-Dec-14	Prorated Injury		8.5 nm S of Grand Manan, NB	EN	.75	XC	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.
25-Dec-14	Mortality	Triomphe	Little Cranberry Island, ME	EN	1	XU		Fresh carcass with evidence of extensive constricting entanglement. No necropsy, but robust body condition and histopathology results of samples support EN as COD.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
01-Feb-15	Serious Injury		off Beaufort, NC	EN	1	XU		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.
03-Feb-15	Mortality		Corolla, NC	EN	1	US		Fresh carcass with injuries consistent with constricting gear. No gear present. Full stomach indicating fed recently. COD likely peracute under water entrapment.
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.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
09-Jul-15	Prorated Injury		off Sandy Hook, NJ	EN	0.75	XU		High flier trailing 30 ft aft of flukes. Attachment point(s) and configuration unknown. No resights.
02-Aug-15	Serious Injury		off Race Point, Provincetown, MA	EN	1	XU		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		Calf with line around tail leading to buoys 4 ft aft of flukes. Full configuration unknown. No resights post 22Aug2015.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
07-Sep-15	Prorated Injury		off Race Point, Provincetown, MA	EN	0.75	XU		Monofilament line trailing from flukes. Attachment point(s) and configuration unknown. No resights.
24-Sep-15	Prorated Injury		off Hampton, NH	EN	0.75	US		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		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.

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
04-Dec-15	Prorated Injury		off Brier Island, NS	EN	0.75	CN		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.
15-Dec-15	Prorated Injury		off North East Harbour, NS	EN	0.75	CN		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.
Five-year averages		Shipstrike (US/CN/XU/XC)			1. 80 (1.80/ 0.00/ 0.00/ 0.00)			
		Entanglement (US/CN/XU/XC)			6.45 (1.65/ 0.30/ 4.15/ 0.35)			
a. For more details on events please see Henry *et al.* 2017.								
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

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. Causes of these UME events have not been determined.

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 Dopulation 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, which is likely biased low, exceeds the calculated PBR and, therefore, this is a strategic stock. However, because the abundance estimate is fairly imprecise, as well as negatively biased due to the survey being incomplete in geographic coverage, 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, current 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).

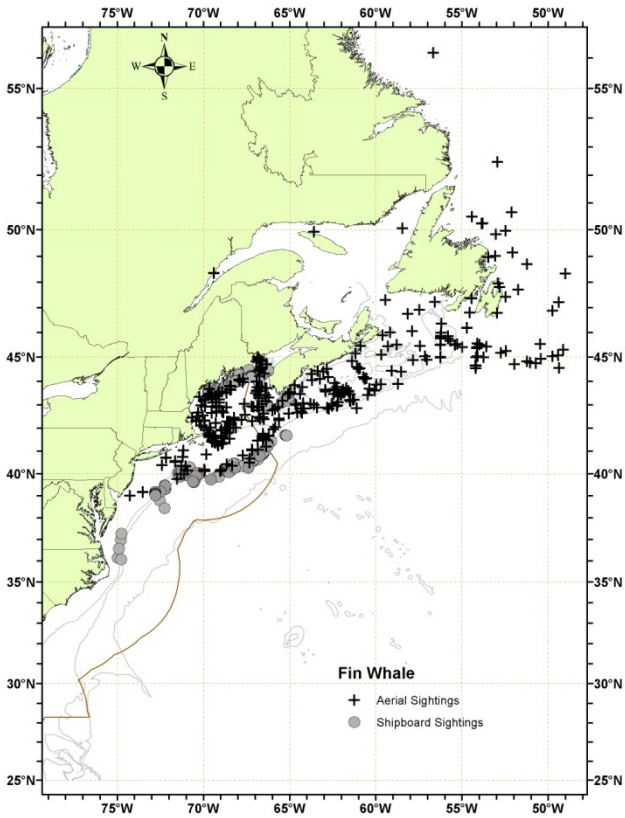

Figure 1. Distribution of fin whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010 and 2011 and DFO's 2007 TNASS survey. Isobaths are the 100-m, 100-m and 4000-m depth contours.

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; 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 deleting 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 1,618 (CV=0.33). This is the estimate derived from the 2011 NOAA shipboard and aerial surveys and is considered best because it represents the only current data in spite of the survey not including all of the stock's range.

A key uncertainty in the current abundance estimate is the number of animals in Canadian waters. The northern part of the stock's range was not surveyed in the 2011 shipboard survey (Palka 2012). This new abundance estimate largely represents only the U.S. portion of this stock, and a small portion in Canadian waters. Additionally, the current abundance estimate does not account for availability bias due to submerged animals. Without a correction for this bias, the abundance estimate is likely biased low. Finally, since the most current estimate dates from a survey done in 2011, the ability for that estimate to accurately represent the present population size has become increasingly uncertain.

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 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 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) 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).

Table 1. Summary of recent abundance estimates for western North Atlantic fin whales with month, year, and area covered during each abundance survey, and resulting abundance estimate (N_{best}) 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

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 1,618 (CV=0.33). The minimum population estimate for the western North Atlantic fin whale is 1,234.

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).

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 1,234. 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 2.5. Because uncertainties exist in stock definition and because the current N_{min} used to calculate PBR is not derived from the full range of the stock as currently defined, considerable uncertainties exist in this calculated PBR.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

For the period 2011 through 2015, the minimum annual rate of human-caused mortality and serious injury to fin whales was 2.65 per year. This value includes incidental fishery interaction records, 1.05 (0.2 U.S./0.85 unknown but first reported in U.S. waters); and records of vessel collisions, 1.6 (all U.S.) (Table 2; Henry *et al.* 2017). Human-caused serious injury and mortality records from Canadian waters are reported in Table 2b but not included

in the summary calculation as they occurred outside the area covered by the abundance estimate. 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

US

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 2011 through 2015 on file at NMFS found 2 records with substantial evidence of fishery interactions causing mortality (Table 2a; Henry *et al.* 2017). Serious injury determinations from non-fatal fishery interaction records yielded a value of 3.25 over five years, for an annual average of 0.65 (Table 2a; Henry *et al.* 2017). The resultant estimated minimum annual rate of serious injury and mortality from fishery interactions for this fin whale stock is 1.05. 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 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. Two records with substantial evidence of fishery interactions causing mortality and 2 that were classified as serious injuries were reported for the 2011–2015 period, resulting in a 5-year annual average of 0.8 animals. All of these interactions occurred (or were discovered in) waters outside the area covered by the abundance estimate, and so were not included in the totals.

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): 2011–2015[a]

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
1/1/2011	Mortality	-	off Portland, ME	EN	1	XU	NP	Fresh carcass w/ evidence of constricting gear.
6/5/2011	Mortality	-	off Long Branch, NJ	VS	1	US	-	Extensive hemorrhage & soft tissue damage to the dorsal & right lateral thoracic region.
9/21/2011	Mortality	-	off Atlantic City, NJ	EN	1	US	NP	Fresh carcass w/ evidence of extensive entanglement.

1/23/2012	Mortality	-	Ocean City, NJ	VS	1	US	-	Hemorrhaging along right, midlateral surface.
2/19/2012	Mortality	-	Norfolk, VA	VS	1	US	-	Deep laceration on head. Skeletal fractures of rostrum and vertebrae. Extensive hemorrhaging.
7/16/2012	Prorated Injury	-	off Portland, ME	EN	0.75	XU	NR	Full configuration unknown.
8/10/2012	Mortality	-	Hampton Bays, NY	VS	1	US	-	Extensive bruising along right lateral and ventral aspects.
10/7/2012	Mortality	-	Boston Harbor, MA	VS	1	US	-	Deep mid-line impression with associated hemorrhaging consistent with being folded across bow of ship.
1/13/2013	Mortality	-	East Hampton, NJ	VS	1	US	-	Fracturing of left cranium with associated hematoma
4/12/2014	Mortality	-	Port Elizabeth, NJ	VS	1	US	-	Fresh carcass on bow of vessel. Large external abrasions w/ associated hemorrhage and skeletal fractures along right side.
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.
10/5/2014	Mortality	-	off Manasquan, NJ	VS	1	US	-	Large area of hemorrhage along dorsal, ventral, and right lateral surfaces consistent with blunt force trauma.
06/06/2015	Serious Injury		off Bar Harbor, ME	EN	1	XU	NR	Free-swimming with 2 buoys and 80 ft of line trailing from fluke. Line cutting deeply into right fluke blade. Emaciated. No resights.
Five-year averages		Shipstrike (US/ XU)			1.6 (1.6/ 0.0)			
		Entanglement (US/ XU)			1.05 (0.2/ 0.85)			

a. For more details on events please see Henry *et al.* 2017.

b. The date sighted and location provided in the table are not necessarily when or where the serious injury or mortality occurred; rather, this information indicates when and where the whale was first reported beached, entangled, or injured.

c. Mortality events are counted as 1 against PBR. Serious injury events have been evaluated using NMFS guidelines (NOAA 2012)

d. US=United States, XU=Unassigned 1st sight in U.S.

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): 2011–2015[a]

Date[b]	Injury Determination	ID	Location[b]	Assigned Cause	Value against PBR[c]	Country[d]	Gear Type[e]	Description
7/2/2011	Serious Injury	F100	Gulf of St. Lawrence	EN	1	CN	PT	Deep lacerations at peduncle. Unconfirmed if gear free.
7/24/2011	Mortality	-	Cheticamp, Nova Scotia	EN	1	CN	NP	Fresh carcass w/ evidence of extensive entanglement.
6/6/2013	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
5/13/2014	Mortality	-	Rocky Harbour, NL	EN	1	CN	PT	Fresh carcass hog-tied in gear.
Five-year averages		Shipstrike (CN/XC)			0			
		Entanglement (CN/XC)			0.8 (0.8/ 0.0)			

a. For more details on events please see Henry *et al.* 2017.

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 2011 through 2015, 8 were found that had sufficient information to confirm the cause of death as collisions with vessels (Table 2; Henry *et al.* 2017). These records constitute an annual rate of serious injury or mortality of 1.6 fin whales from vessel collisions in U.S. waters.

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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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 occurance in deep-ocean waters throughout most of the Western North Atlantic (Clark and Gagnon 2002; *Risch et al.* 2014). During the spring-fall period the whales are most abundant in New England waters. 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).

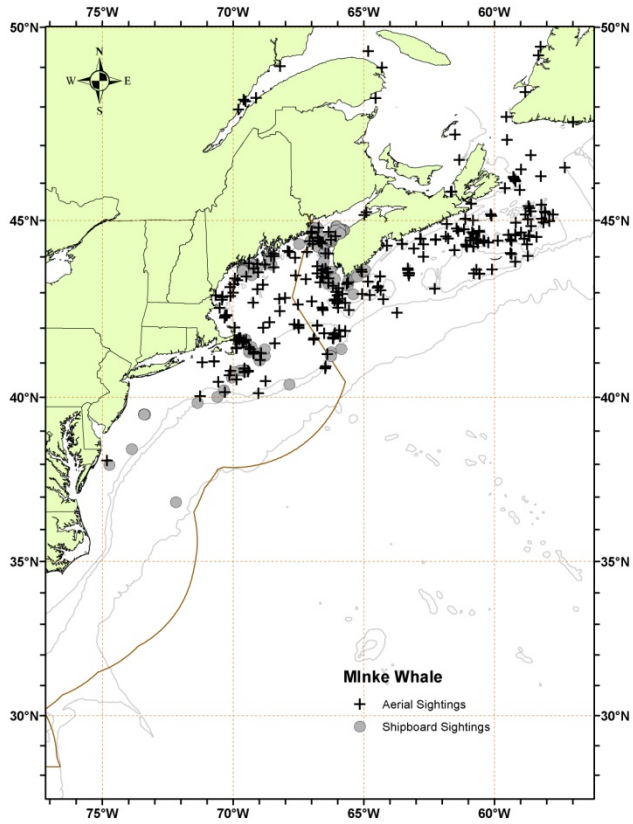

***Figure 1**. Distribution of minke whale sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, and 2011and DFO's 2007 TNASS survey. Isobaths are the 100-m, 1000-m and 4000-m countours.*

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 abundance estimate for this stock is 2,591 (CV=0.81; Palka 2012) common minke whales. This estimate,

derived from 2011 shipboard and aerial surveys, is the only current estimate available. This estimate is substantially lower than the estimate from the 2015 SAR (20,741, CV=.30). This is because the previous estimate included data from the 2007 TNASS surveys of Canadian waters. For the purposes of this SAR, as recommended in the 2016 guidelines for preparing stock assessment reports (NMFS 2016), estimates older than eight years are deemed unreliable, so this new estimate must not include data from the 2007 TNASS survey. This new estimate should not be interpreted as a decline in abundance of this stock, as previous estimates are not directly comparable.

A key uncertainty in the current abundance estimate is the number of animals in Canadian waters. The northern part of the stock's range was not surveyed in the 2011 shipboard survey (Palka 2012). This new abundance estimate largely represents only the U.S. portion of this stock, and a small portion in Canadian waters. Additionally, the current abundance estimate does not account for availability bias due to submerged animals. Without a correction for this bias, the abundance estimate is likely biased low. Finally, since the most current estimate dates from a survey done in 2011, it is not certain that the 2011 estimate accurately represents the present population size.

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 (MCDS) option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

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 coefficient of variation. (CV).

Month/Year	Area	N_{best}	CV
Jul-Aug 2011	Central Virginia to lower Bay of Fundy	2,591	0.81

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 2,591animals (CV=0.81). The minimum population estimate is 1,425 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).

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 1,425. 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 14.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

During 2011 to 2015, the average annual minimum detected human-caused mortality and serious injury was 9.15 common minke whales per year, which is the sum of 7.75 (1.9 U.S/3.25 Canada/2.6 unassigned but first reported in the U.S.) minke whales per year (unknown CV) from U.S. and Canadian fisheries using strandings and entanglement data, and 1.4 (1.2 U.S./0.2 Canada) per year from vessel strikes.

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.

Northeast Mid-water Trawl Fishery (Including Pair Trawl)

In July 2013, one minke whale was observed dead in the mid-water otter trawl fishery on Georges Bank. This animal was too decomposed to have been taken in a haul that was only 3 hours long. Annual average estimated minke whale mortality and serious injury from the mid-Atlantic mid-water trawl (including pair trawl) during 2011 to 2015 was 0.

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. During 2011 to 2015, 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 4.35 common minke whales per year in U.S. fisheries (Table 2a). 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.

Herring Weirs

Weir interactions that may have resulted in serious injury to minke whales are reported in Table 2b.

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 2011 to 2015, 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 3.25 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.: 2011–2015[a]

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
5/6/2011	Mortality	-	off Martha's Vineyard, MA	EN	1	US	PT	Anchored in gear. Embedded line at fluke. Evidence of entanglement w/ associated hemorrhaging at mouth corners & insertion of pectorals
7/17/2011	Prorated Injury	-	off Nahant, MA	EN	0.75	XU	NR	Full configuration unknown.
7/24/2011	Prorated Injury	-	off North Truro, MA	EN	0.75	XU	NR	Full configuration unknown
8/4/2011	Mortality	-	Sandy Hook Bay, NJ	VS	1	US	-	4 propeller lacerations across dorsal surface. Fractured ribs w/associated hemorrhaging
8/26/2011	Mortality	-	Horseshoe Cove, NJ	EN	1	US	NP	Fresh carcass w/ evidence of extensive entanglement
8/29/2011	Mortality	-	Moriches Bay, NY	VS	1	US	-	Extensive hemorrhage & edema along dorsal & both lateral surfaces
10/6/2011	Mortality	-	off Matinicus Island, ME	EN	1	US	PT	Fresh carcass anchored in gear
12/7/2011	Mortality	-	Carolina Beach, NC	VS	1	US	-	Healed deep & superficial propeller lacerations; internal lesions associated w/ deep lacerations indicative of peritonitis & infection

2/4/2012	Prorated Injury	-	off Virginia Beach, VA	EN	0.75	XU	CE	Reported with hook/monofilament gear. Attachment point unknown.
3/16/2012	Mortality	-	Ipswich, MA	EN	1	US	NP	Evidence of extensive, constricting gear w/ associated hemorrhaging
6/21/2012	Serious Injury	-	off Frenchboro, ME	EN	1	XU	NR	Constricting body wrap, flipper pinned, embedded in mouthline; emaciated
6/23/2012	Mortality	-	Newark, NJ	VS	1	US	-	Fresh carcass on bow of ship. Deep laceration across ventral surface; Cause of death - disembowelment & hypovolemic shock
7/1/2012	Prorated Injury	-	off Portsmouth, NH	EN	0.75	XU	NR	Full configuration unknown
7/13/2012	Prorated Injury	-	off Jonesport, ME	EN	0.75	US	NR	Anchored. Partial disentanglement; Final configuration unknown
7/17/2012	Serious Injury	-	off Chatham, MA	EN	1	XU	NR	Tight wrap across back; health decline: emaciated
8/2/2012	Prorated Injury	-	off Provincetown, MA	EN	0.75	XU	NR	Full configuration unknown
8/5/2012	Mortality	-	Chatham, MA	EN	1	US	NR	Multiple constricting wraps through & around mouth and on fluke blades; COD - acute underwater entrapment
10/4/2012	Mortality	-	Cliff Island, ME	EN	1	US	NR	Evidence of constricting gear at mouthline, across ventral pleats, & at peduncle
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
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. Proximate COD - entanglement. Ultimate COD - euthanasia.
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 hemorrhaging. No gear present.
Five-year averages		Vessel strike (US/ XU)			1.2 (1.2/ 0.0)			
		Entanglement (US/ XU)			4.5 (1.9/ 2.6)			

a. For more details on events please see Henry *et al.* 2017.

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, MT=midwater trawl, 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 common minke whales (*Balaenoptera acutorostrata acutorostrata*) first reported in Canadian waters or attributed to Canada: 2011–2015[a]

Date[b]	Injury determination	ID	Location[b]	Assigned Cause[f]	Value against PBR[c]	Country[d]	Gear Type[e]	Description
6/3/2011	Serious Injury	-	Tadoussac, Quebec	EN	1	CN	NR	Tight rostrum wrap.
9/7/2011	Prorated Injury	-	Greenspond, NL	EN	0.75	CN	GN	Partially disentangled from anchoring gear. Final configuration unknown.
9/19/2011	Prorated Injury	-	Northumberland Strait, PEI	EN	0.75	CN	NR	Partially disentangled from anchoring gear. Final configuration unknown.
12/19/2011	Mortality	-	off Grand Manan Island, NB	EN	1	CN	PT	Live entanglement; recovered dead in gear the following day. Constricting peduncle wraps
5/15/2012	Serious Injury	-	Sable Island Bank, Canada	EN	1	CN	PT	Disentangled from gear embedded down to bone of peduncle.
6/26/2012	Mortality	-	Renews Rock, NL	EN	1	CN	PT	Fresh carcass w/ constricting gear around peduncle
6/30/2012	Mortality	-	off Naufrage, PEI	EN	1	CN	PT	Fresh carcass anchored in gear
7/1/2012	Mortality	-	Northern Lake Harbor, PEI	EN	1	CN	PT	Constricting gear w/ associated hemorrhaging; COD - drowning
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.

7/10/2014	Mortality	-	Cape George, Antigonish, NS	VS	1	CN	-	Fresh carcass with jaw fractures.
7/29/2014	Mortality	-	5 nm E of Herring Cove, NS	EN	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 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 inbetween 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.
Five-year averages		Vessel strike (CN/ XC)			0.20 (0.20/ 0.00)			

	Entanglement (CN/ XC)	3.25 (3.25/ 0.00)
a. For more details on events please see Henry *et al.* 2017.		
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, MT=midwater trawl, NP=none present, NR=none recovered/received, PT=pot/trap, WE=weir		
f. Assigned cause: EN=entanglement, VS=vessel strike, ET=entrapment (summed with entanglement).		

Other Mortality

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

U.S.

Common minke whales inhabit coastal waters during much of the year and are thus susceptible to collision with vessels. In 2011, three juvenile common minkes were confirmed dead due to vessel strikes: a female off Sandy Hook, New Jersey, a female off Moriches, New York, and a male off Carolina Beach, North Carolina. In 2012, a confirmed vessel strike resulted in a mortality off Newark, New Jersey. 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. Thus, during 2011–2015, as determined from stranding and entanglement records, the minimum detected annual average was 1.2 common minke whales per year struck by vessels in U.S. waters or first seen in U.S. waters (Table 2a; Henry *et al.* 2017).

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 2011, 3 in 2012, and 0 in 2013 and 1 in 2014, and 2 in 2015. Those that have been determined to be human-caused serious injury or mortality are included in Table 2b (Ledwell and Huntington 2011, 2012a, 2012b, 2013, 2014, 2015).

During 2011–2015, 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.* 2017).

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 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 minke whales relative to OSP in the U.S. Atlantic EEZ is unknown.

It is expected that the uncertainties described above will have little effect on the designation of the status of the entire stock. Even though the estimate of human-caused mortality and serious injury in this assessment is negatively biased due to using strandings and entanglement data as the primary source, the abundance estimate is a very negatively-biased estimate for the entire stock as it only includes the U.S. portion of the Canadian East Coast common minke whale stock's habitat and the habitat in Canadian waters. If the current PBR representing only the U.S. portion of the stock (9.4) is compared to only the negatively-biased U.S. mortalities and serious injuries (5.7), the stock would still be designated as not strategic. However, this designation may be reversed if the negative bias in the mortality estimate is large. Thus, key uncertainties that need to be resolved include the stock structure, particularly as it is influenced by movement patterns between US and Canadian waters, and the estimated human-caused mortalities and serious injuries.

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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).

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.

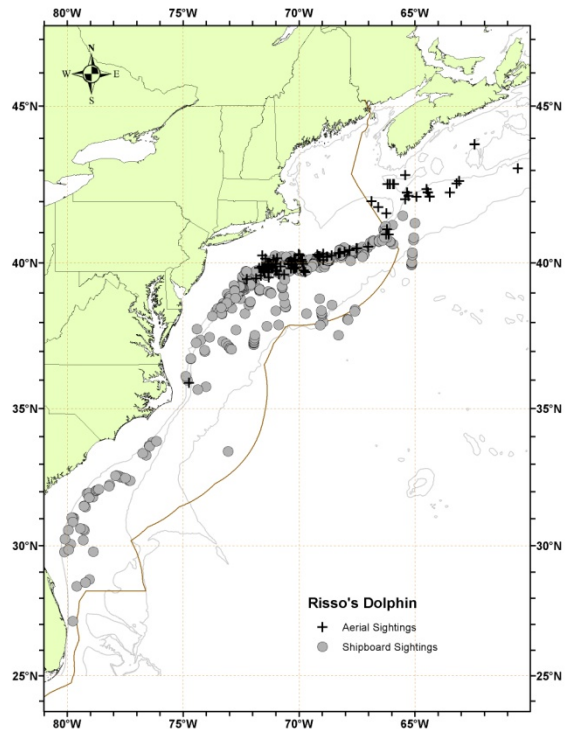

Figure 1. Distribution of Risso's dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008 2010 and 2011. Isobaths are the 100-m, 1,000-m, and 4,000-m depth contours.

POPULATION SIZE

The best abundance estimate for Risso's dolphins is the sum of the estimates from the 2011 surveys—18,250 (CV = 0.46: Table 1). The current abundance estimate did not account for availability bias due to submergence of animals. Without a correction for availability bias, the abundance estimate is expected to be biased low. Additionally, since the most current estimate dates from a survey done in 2011, the ability for that estimate to accurately represent the present population size has become increasingly uncertain.

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 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. 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).

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 (N_{best}) 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

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 18,250 (CV = 0.46), obtained from the 2011 surveys. The minimum population estimate for the western North Atlantic Risso's dolphin is 12,619.

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).

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 12,619. 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 126.

ANNUAL HUMAN-CAUSED MORTALITY

Total annual estimated average fishery-related mortality or serious injury to this stock during 2011–2015 was 43.2 Risso's dolphins, derived from 2 components: 1) 43 estimated mortalities in observed fisheries (CV = 0.27; Table 2) and 2) 0.2 from average 2011–2015 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 2011–2015 are documented in Garrison and Stokes (2012, 2013, 2014, 2016, 2017). 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 (Table 2). Annual Risso's dolphin mortalities were estimated using annual stratified ratio-estimator methods (Chavez-Rosales *et al.* 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.

Mid-Atlantic Bottom Trawl

Risso's dolphins have been observed taken in mid-Atlantic bottom trawl fisheries (Table 2). It was discovered in 2010 that a small segment of the mid-Atlantic bottom trawl fleet was equipping fishing nets with acoustic deterrent devices (i.e., pingers). To the extent possible, the use of pingers on bottom trawl gear has been taken into account when estimating bycatch mortality of Risso's dolphins (methodology is detailed in Chavez-Rosales *et al.* 2017). Annual Risso's dolphin mortalities were estimated using annual stratified ratio-estimator methods (Chavez-Rosales *et al.* 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 Sink Gillnet

In the northeast sink gillnet fishery, Risso's dolphin interactions have historically been rare, but in 2012 and 2013 one animal was observed each year in the waters south of Massachusetts (Hatch and Orphanides 2014, 2015, 2016). 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[e]	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Combined Annual Mortality

Pelagic Longline	11-15	Obs. Data Logbook	.09, .07, .09,	2, 1, 1, 1, 2	0, 0, 0, 0, 0	12, 15, 1.9, 7.7, 8.4	0, 0, 0, 0. 0	12, 15, 1.9, 7.7, 8.4	.63, 1.0, 1.0, 1.0, .71	8.98 (0.44)
Northeast Sink Gillnet	11-15	Obs. Data, Trip Logbook, Allocated Dealer Data	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	0, 1, 1, 0	0, 0, 0, 0, 0	0, 6, 23, 0, 0	0, 6, 23, 0, 0	0, .87, 1, 0, 0	5.8 (0.79)
Northeast Bottom Trawl	11-15	Obs. Data Dealer Data VTR Data	.26, .17, .15, .17, .19	0, 0, 0, 0, 0	0, 0, 0, 1, 0	0, 0, 0, 0, 0	0, 0, 0, 4.2, 0	0, 0, 0, 4.2, 0	0, 0, 0, .91, 0	0.8 (0..91)
Mid-Atlantic Bottom Trawl	11-15	Obs. Data Dealer Data	.08, .05, .06, .08, .09	0, 0, 0, 0, 2	2, 1, 4, 2, 1	0, 0, 0, 0, 27	27, 7.6, 42, 21, 13	27, 7.6, 42, 21, 40	.68, 1.0, .71, .93, .63	28 (.35)
TOTAL										43 (0.27)

[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 2011–2015 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2017).

Other Mortality

From 2011 to 2015, 26 Risso's dolphin strandings were recorded along the U.S. Atlantic coast (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 September 2016). Three animals had confirmed indications of human interaction, 2 of which were 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, 2011-2015.

STATE	2011	2012	2013	2014	2015	TOTALS
Massachusetts[a]	0	0	3	2	1	6

New York	1	0	2	0	2	5
Maryland	0	0	1	0	0	1
Virginia[b]	1	0	0	1	1	6
North Carolina[c]	1	2	1	1	0	5
Florida	2	2	2	0	0	6
Puerto Rico	1	0	0	0	0	1
TOTAL	**6**	**4**	**9**	**4**	**4**	**26**
a. One animal in 2014 was classified as CBD for human interaction due to signs of ear trauma.						
b. One animal in 2014 classified as HI due to plastic ingestion.						
c. Two animals in 2012 showed signs of fishery interaction.						

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.

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 2011–2015 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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ATLANTIC WHITE-SIDED DOLPHIN (*Lagenorhynchus acutus*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

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. No comparative genetic analysis of samples from U.S. waters and the Gulf of St. Lawrence and/or Newfoundland have been made.

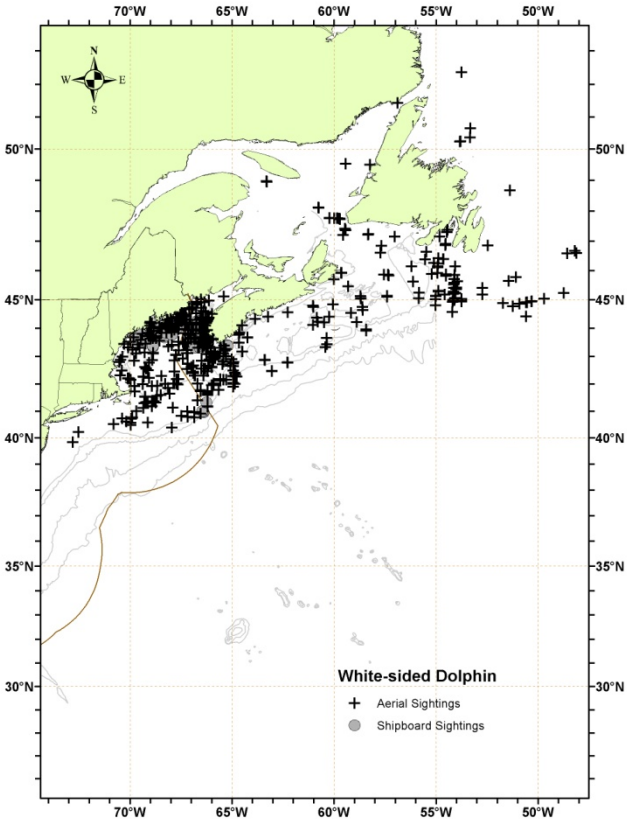

Figure 1. *Distribution of white-sided dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, and 2011, and DFO's 2007 TNASS survey. Isobaths are the 100-m, 1000-m and 4000-m depth contours.*

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 distributed 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 48,819 (CV= 0.61), resulting from a June–August 2011survey. However, this estimate actually only covers the Gulf of Maine population, not the entire western North Atlantic stock. A current abundance survey that accounts for availability bias and covers at least the Atlantic U.S. and Canadian waters is needed to estimate the abundance of the entire, or at least most of, the western North Atlantic stock. Additionally, since the most current estimate dates from a survey done in 2011, the ability for that estimate to accurately represent the present population size has become increasingly uncertain.

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 48,819 (CV=0.61) white-sided 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 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 MRDS option in the computer program Distance (version 6.0, release 2, Thomas *et al.* 2009).

No white-sided dolphins were detected in the aerial and ship abundance surveys that were 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.

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

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 48,819 (CV=0.61). The minimum population estimate for these white-sided dolphins is 30,403.

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).

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 30,403. 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 304.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total annual estimated average fishery-related mortality or serious injury to this stock during 2011–2015 was 56 (CV=0.15) white-sided dolphins (Table 2).

Key uncertainties include the potential that the observer coverage in the Mid-Atlantic gillnet was not representative of the fishery during all times and places, since the observer coverage was relatively low (0.02 – 0.06). The effect of this is unknown.

There are no major known sources of unquantifiable human-caused mortality or serious injury for 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

Annual white-sided dolphin mortalities were estimated using annual stratified ratio-estimator methods (Table 2; Orphanides 2013; Hatch and Orphanides 2014, 2015, 2016, Orphanides and Hatch 2017). 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

Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos 2015; Chavez-Rosales *et al.* 2017). 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

Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Lyssikatos 2015; Chavez-Rosales *et al.* 2017). 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 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	11-15	Obs. Data Weighout Trip Logbook	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	5, 1, 1, 2, 0	0, 0, 0, 0, 0	18, 9, 4, 10, 0	18, 9, 4, 10, 0	.43, .92, 1.03, .66, 0	8.2 (0.34)
Northeast Bottom Trawl	11-15	Obs. Data Trip Logbook	.26, 0.17, .15, 17, .19	2, 0, 0, 0, 0	47, 9, 8, 3, 3	3, 0, 0, 0, 0	138, 27, 33, 16, 15	140, 27, 33, 16, 15	.24, .47, .31, .5, .52	46 (0.17)
Mid-Atlantic Bottom Trawl	11-15	Obs. Data Trip Logbook	.08, .05, .06, .08, .09	0, 0, 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	1.9 (0.94)
Total										56 (0.15)

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 and 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 2011–2015 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2017).

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 the 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 which 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.

Herring Weirs

Previously only one white-sided dolphin was released alive and unharmed from a herring weir in the Bay of Fundy (A. Westgate, pers. comm.). Due to the formation of a cooperative program between Canadian fishermen and biologists, it is expected that most dolphins and whales will be able to be released alive. Fishery information is available in Appendix III.

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 13 September 2016). Eight of these animals were released alive. Human interaction was indicated in three 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

Small numbers of white-sided dolphins have been hunted off southwestern Greenland. (Reeves *et al.* 1999). 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 2010, 2011, 2012a, 2012b, 2013, 2014, 2015).

Table 3. White-sided dolphin (*Lagenorhynchus acutus*) reported strandings along the U.S. and Canadian Atlantic coast, 2010-2014.

Area	**Year**					**Total**
	2011	**2012**	**2013**	**2014**	**2015**	
Maine	2	1	1	2	1	7
New Hampshire	0	2	0	0	0	2
Massachusetts[a,b]	42	3	10	4	3	62
Rhode Island	1	1	1	0	0	3
New York	0	3	2	0	0	5
New Jersey	1	0	0	0	0	1
Delaware	1	0	0	0	0	1
Maryland	1	0	0	0	0	1
North Carolina	1	0	0	0	0	1
Georgia	1	0	0	0	0	1

TOTAL US	**50**	**10**	**14**	**6**	**4**	**84**
Nova Scotia[c]	6	5	7	12	11	41
Newfoundland and Labrador[d]	0	3	0	5	0	8
GRAND TOTAL	**56**	**18**	**21**	**23**	**15**	**133**

[a] Records of mass strandings in Massachusetts during this period are: March 2011 - 4 animals (2 released alive), 2 animals (released alive); April 2013 - 2 animals (one released alive); December 2013 - 3 animals (all released alive).

In 2011, 1 animal in Massachusetts was classified as human interaction due to post-mortem mutilation. 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.

[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 2011, 2012a, 2012b, 2013, 2014, 2015).

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 but is not less than 10% of the calculated PBR; therefore, it cannot be 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.

Based on the levels of uncertainties regarding the Gulf of Maine population 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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COMMON DOLPHIN (*Delphinus delphis delphis*): Western North Atlantic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

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 have 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˚ (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).

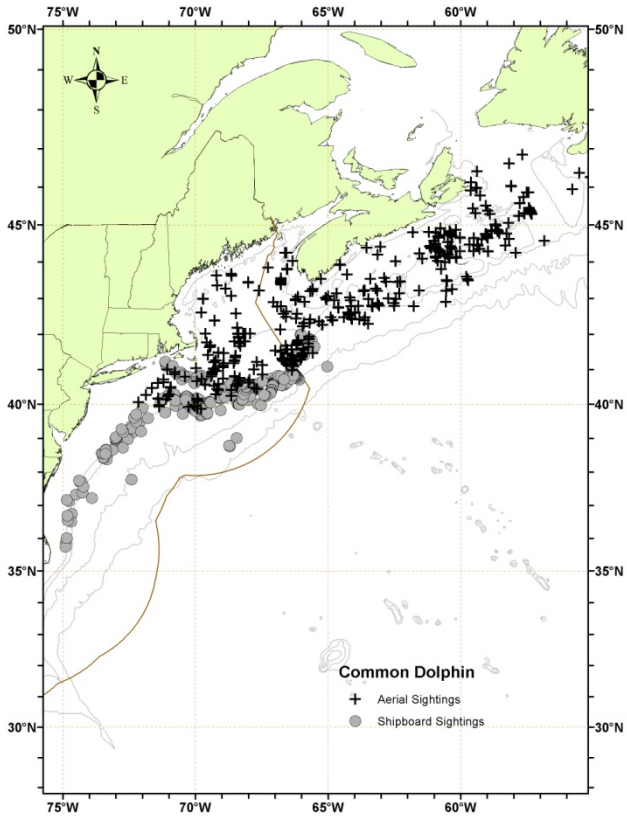

Figure 1. *1Distribution of common dolphin sightings from NEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008 2010 and 2011. Isobaths are the 100-m, 1,000-m, and 4,000-m depth contours.*

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 common dolphins off the U.S. Atlantic coast is 70,184 (CV=0.28). This estimate, derived from 2011 shipboard and aerial surveys, is the only current estimate available. This estimate is substantially lower than the estimate from the 2015 SAR (173,486, CV=0.55). This is because the previous estimate included data from the 2007 TNASS surveys of Canadian waters. For the purposes of this SAR, as recommended in the guidelines for preparing Stock Assessment Reports (NMFS 2016), estimates older than eight years are deemed unreliable, so this new estimate must not include data from the 2007 TNASS survey. This new estimate should not be interpreted as a decline in abundance of this stock, as previous estimates are not directly comparable.

A key uncertainty in the current abundance estimate is the number of animals in Canadian waters. The northern part of the stock's range was not surveyed in the 2011 shipboard survey (Palka 2012). This new abundance estimate largely represents only the US portion of this stock, and a small portion in Canadian waters. Additionally, the current abundance estimate does not account for availability bias due to submerged animals. Without a correction for this bias, the abundance estimate is likely biased low. Finally, since the most current estimate dates from a survey done in 2011, the ability for that estimate to accurately represent the present population size has become increasingly uncertain.

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. 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

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 70,184 animals (CV=0.28), derived from the 2011 aerial and shipboard surveys. The minimum population estimate for the western North Atlantic common dolphin is 55,690.

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).

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 55,690 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 557.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Average annual estimated fishery-related mortality or serious injury to this stock during 2011–2015 was 437 (CV=0.10) common dolphins.

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, US 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 (Orphanides 2013; Hatch and Orphanides 2014, 2015, 2016; Orphanides and Hatch 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.

Mid-Atlantic Gillnet

Common dolphins were taken in observed trips during most years. Annual common dolphin mortalities were estimated using annual ratio-estimator methods (Orphanides 2013; Hatch and Orphanides 2014, 2015, 2016; Orphanides and Hatch 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

This fishery is active in New England waters in all seasons. Annual common dolphin mortalities were estimated using annual stratified ratio-estimator methods (Chavez-Rosales *et al.* 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.

Mid-Atlantic Bottom Trawl

Annual common dolphin mortalities were estimated using annual stratified ratio-estimator methods (Chavez-Rosales *et al.* 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 Mid-water Trawl Fishery (Including Pair Trawl)

A common dolphin mortality was observed in this fishery in 2012 (Table 2). An expanded bycatch estimate has not been calculated so the minimum raw count is reported.

Pelagic Longline

Pelagic longline bycatch estimates of common dolphins for 2011–2015 were documented in Garrison and Stokes (2012, 2013, 2014, 2016, 2017). 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.

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	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Annual Combined Mortality

Northeast Sink Gillnet[d]	11-15	Obs. Data, Trip Logbook, Allocated Dealer Data	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	6, 6, 5, 11, 3	0, 0, 0, 0, 0	49, 95, 104, 111, 55	49, 95, 104, 111, 55	.71, .40, .46, .47, .54	86 (.23)
Mid-Atlantic Gillnet	11-15	Obs. Data, Trip Logbook, Allocated Dealer Data	.02, .02, .03, .05, .06	0, 0, 0, 0, 0	3, 1, 2, 1, 3	0, 0, 0, 0, 0	29, 15, 62, 17, 30	29, 15, 62, 17, 30	.53, .93, .67, .86, .55	31(.34)
Northeast Mid-water Trawl - Including Pair Trawl	11-15	Obs. Data Trip Logbook	.41, .45, .37, .42, .08	0, 0, 0, 0, 0	0, 1, 0, 0, 0	0, 0, 0, 0, 0	0, na, 0, 0, 0	0, 1, 0, 0, 0	0, 1, 0, 0, 0	0.2
Northeast Bottom Trawl[c]	11-15	Obs. Data Trip Logbook	.26, .17, .15, .17, .19	0, 0, 0, 0, 0	22, 10, 4, 3, 4	3, 2, 0, 0, 0	73, 42, 17, 17, 22	73, 42, 17, 17, 22	32, .47, .54, .53, .45	34 (.22)
Mid-Atlantic Bottom Trawl[c]	11-15	Obs. Data Trip Logbook	.08, .05, .06, .08, .09	1, 0, 0, 3, 0	29, 32, 24, 35, 26	8, 7, 0, 24, 0	263, 316, 269, 305, 250	271, 323, 269, 329, 250	.25, .26, .29, .29, .32	285(.12)
Pelagic Longline	11-15	Obs. Data Logbook	.09, .07, .09, .10, .12	0, 0, 0, 0, 1	0, 0, 0, 0, 0	0, 0, 0, 0, 9.05	0, 0, 0, 0. 0	0, 0, 0, 0, 9.05	0, 0, 0, 0, 1.0	1.81 (1.0)
TOTAL										437 (.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 2011-2015 were estimated using an annual stratified ratio-estimator (Chavez-Rosales *et al.* 2017).

d. Serious injuries were evaluated for the 2011–2015 period and include both at-sea monitor and traditional observer data (Josephson *et al. 2017*)

CANADA

One common dolphin was reported as a bycatch mortality in Canadian bottom otter trawl fishing on Georges Bank in 2012 (pers. comm. Marine Animal Response Society, Nova Scotia). Canadian mortalities are not added to the U.S. estimates for this SAR, as the abundance estimate and PBR apply mainly to U.S. waters.

Other Mortality

From 2011 to 2015, 654 common dolphins were reported stranded between Maine and Florida (Table 3; (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 September 2016). The total includes mass-stranded common dolphins in Massachusetts during 2011 (a total of 30 animals in 5 events), 2012 (a total of 192 animals in 23 group stranding events), 2013 (a total of 9 in 3 events), 2014 (a total of 14 in 4 events), and 2015 (a total of 37 in 13 events); one mass stranding in North Carolina in 2011 (4 animals); and 2 mass strandings in Virginia in 2013 (a total of 6 in 2 events). Fifteen animals in 2011, 71 animals in 2012, 13 in 2013, and 12 in 2014 were released or last sighted alive. In 2011, 3 animals were classified as having human interactions, 2 of which were fishery interactions (one of these was satellite-tagged and released). Twelve human interaction cases were reported in 2012 (7 in Massachusetts, 3 in New York, and 2 in New Jersey), 6 of which (2 in Massachusetts, 2 in New York, and 1 in New Jersey) were classified as fisheries interactions. 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. 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 2 common dolphins (one a fisheries interaction) stranded in 2011, 0 in 2012 and 2013, 3 in 2014 and 2 in 2015 (Tonya Wimmer/Andrew Reid, pers. comm.).

Table 3. Common dolphin (*Delphinus delphis delphis*) reported strandings along the U.S. Atlantic coast, 2011-2015.

STATE	2011	2012	2013	2014	2015	TOTALS
Maine	0	2	0	0	0	**2**
Massachusetts[a]	64	221	48	38	40	**411**
Rhode Island[c]	5	6	6	6	7	**30**
Connecticut	0	0	0	0	2	**2**
New Hampshire	0	0	0	0	1	**1**
New York [c]	17	13	24	7	3	**68**
New Jersey[a, c]	9	14	19	8	3	**53**
Delaware[c]	1	1	3	0	2	**7**
Maryland	1	1	3	0	1	**6**
Virginia[a,c]	9	4	13	9	2	**37**
North Carolina[a,c]	18	0	9	6	4	**37**
TOTALS	**124**	**262**	**125**	**74**	**65**	**654**

a. Massachusetts mass strandings (2011-3,3,4,7,13; 2012 - 23 group events ranging from 2 to 22 animals each, 2013 - 4, 3 2, 2014 – 2, 2, 5, 5, 2015-2, 2, 2, 2, 2, 2, 2, 3, 3, 3, 4, 4, 4, 4). North Carolina mass stranding of 4 animals in 2011. Two mass strandings in Virginia in April 2013 - a group of 4 and a group of 2. Three animals (one released alive) involved in mass stranding in NJ in 2012.

b. Three HI cases in 2011, all in Massachusetts, 2 of which were classified as fishery interactions (but one of those fishery interaction animals was released alive). Twelve HI cases in 2012 (7 in Massachusetts, 3 in New York and 2 in New Jersey), 6 of which (2 in Massachusetts, 2 in New York and 1 in New Jersey) were classified as fisheries interactions. 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.

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.

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 2011–2015 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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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Western North Atlantic Northern Migratory Coastal Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

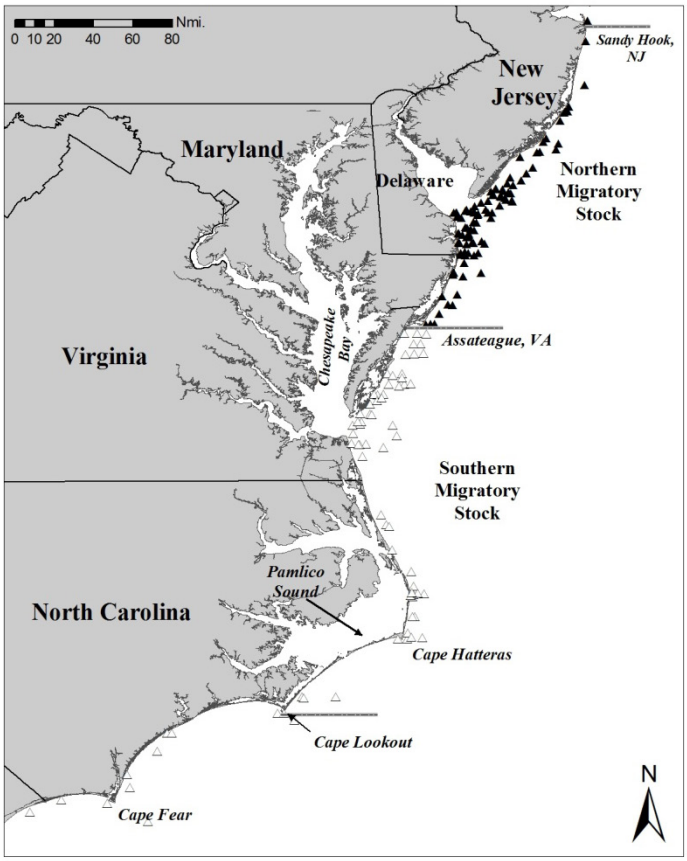

Figure 1. *The distribution of common bottlenose dolphin sightings in coastal waters from North Carolina to New Jersey during July-August 2002, 2004, 2010, 2011, and 2016. Sighting locations from aerial surveys are plotted as triangle symbols. Sightings ascribed to the Northern Migratory Coastal Stock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

This stock exhibits spatiotemporal overlap with multiple common bottlenose stocks in the wNA. The stock is best defined by its distribution during warm water months (best described by July and August) when it overlaps with the fewest stocks. During warm water months, this stock occupies coastal waters from the shoreline to approximately the 20-m isobath between Assateague, Virginia, and Long Island, New York (Figure 1) (Garrison *et al.* 2017b). The stock migrates in late summer and fall and, during cold water months (best described by January and February), occupies coastal waters from approximately Cape Lookout, North Carolina, to the North Carolina/Virginia border (Garrison *et al.* 2017b). Four common bottlenose dolphins tagged during 2003 and 2004 off the coast of New Jersey in late summer moved south to North Carolina and

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

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

POPULATION SIZE

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

Background

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

The definition of the southern summer boundary and inter-annual variation in stock distribution are significant unquantified sources of uncertainty.

Earlier abundance estimates (>8 years old)

Aerial surveys were conducted during the summers of 2002 and 2004. Survey tracklines for the 2002 and 2004 surveys were set perpendicular to the shoreline and effort was stratified into 0–20 m and 20–40 m strata with the majority of effort in the shallow depth stratum (Garrison *et al.* 2017a). The 2002 surveys employed two observer teams operating independently on the same aircraft to estimate the probability of detection on the trackline. This estimate was also applied to the 2004 survey to reduce bias in the resulting abundance estimate. The resulting abundance estimates from the 2002 and 2004 summer aerial surveys for the Northern Migratory Coastal Stock were 20,199 (CV=0.58) and 5,823 (CV=0.48), respectively (Garrison *et al.* 2017a). There were strong differences in spatial distribution between these two survey years, suggesting that the large difference in estimates was related to changes in distribution rather than population size of the stock (Garrison *et al.* 2017a). As recommended in the GAMMS II Workshop Report (Wade and Angliss 1997), these estimates are greater than eight years old and deemed unreliable and should not be used for PBR determinations. However, these estimates are included below in the assessment of trends for this stock.

Recent surveys and abundance estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters along the U.S. East Coast from southeastern Florida (26.9°N) to Sandy Hook, New Jersey (40.3°N), during the summers of 2010, 2011, and 2016 (see Garrison *et al.* 2017a for survey design). The surveys were conducted along tracklines oriented perpendicular to the shoreline and spaced latitudinally at 20-km intervals, and covered waters from the shoreline to the continental shelf break (Garrison *et al.* 2017a). The 2011 and 2016 surveys also included more closely spaced "fine-scale" tracklines in waters offshore of New Jersey and Virginia within areas being evaluated for the placement of offshore energy installations (Garrison *et al.* 2017a).

The recent surveys were conducted using a two-team approach to develop estimates of detection probabilities using the independent observer approach with Distance analysis (Laake and Borchers 2004). The detection functions from each survey indicated a decreased probability of detection near the trackline. The sighting data were therefore "left-truncated" by analyzing only sightings occurring greater than 80 m from the trackline during the 2010 survey, 70 m during the 2011 survey, and 100 m from the trackline during the 2016 survey (see Buckland *et al.* 2001 for left-truncation methodology). The independent observer method assuming point independence was used to estimate detection probability on the trackline. This estimate accounts for the probability of detecting a marine mammal group conditional on it being available to both survey teams. Covariates that may influence detection probabilities (e.g., sea state, glare, cloud cover, visibility) were incorporated into both the mark-recapture and distance function components of the detection models (Laake and Borchers 2004; Garrison *et al.* 2017a). The resulting abundance estimates are negatively biased due to the effects of animals spending some time underwater where they are not available to the survey teams. However, due to the relatively short dive times of bottlenose dolphins (Klatsky *et al.* 2007) and the large group sizes, it is likely that this bias is small (Garrison *et al.* 2017a).

The abundance estimates for 2010, 2011, and 2016 summer aerial surveys were 14,314 (CV=0.74), 15,630 (CV=0.29) and 6,639 (CV=0.41), respectively (Garrison *et al.* 2017a). The 2016 estimate was used as the best estimate of the current population size for the stock due to possible effects from the 2013–2015 unusual mortality event. Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution.

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

Month/Year	Area	N_{best}	CV
July–August 2002	Assateague, Virginia (37.9°N) to Sandy Hook, New Jersey (40.3°N)	20,199	0.58
July–August 2004	Assateague, Virginia (37.9°N) to Sandy Hook, New Jersey (40.3°N)	5,823	0.48

July–August 2010	Assateague, Virginia (37.9°N) to Sandy Hook, New Jersey (40.3°N)	14,314	0.74
July–August 2011	Assateague, Virginia (37.9°N) to Sandy Hook, New Jersey (40.3°N)	15,630	0.29
July–August 2016	Assateague, Virginia (37.9°N) to Sandy Hook, New Jersey (40.3°N)	6,639	0.41

Minimum Population Estimate

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Northern Migratory Coastal Stock during 2011–2015 is unknown. The mean annual fishery-related mortality and serious injury for observed fisheries and strandings identified as fishery-related ranged between 6.1 (CV=0.32) and 13.2 (CV=0.22). 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 2011–2015 ranged between 6.1 and 13.2 (Tables 2a, 2b and 2c). This range reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected and recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between this stock and other common bottlenose

dolphin stocks in North Carolina and Virginia introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities and serious injuries were ascribed to a stock or stocks by comparing the season and geographic location of the take/stranding to the stock boundaries and geographic range delimited for each stock.

Fishery Information

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

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the Northern Migratory Coastal Stock throughout its range. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001; Lyssikatos and Garrison 2018). This fishery is observed by the Northeast and Southeast Fisheries Observer Programs. The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006, available at http://www.nmfs.noaa.gov/pr/pdfs/fr/fr71-24776.pdf). Mortality estimates for the period 2002–2006 (immediately prior to implementation of the BDTRP) and 2007–2011 are available in the 2015 stock assessment report for the Northern Migratory Coastal Stock (Waring *et al.* 2015). The current report covers the most recent available five-year estimate (NMFS 2016) for 2011–2015.

Mortality estimation for this stock is difficult because 1) observed takes are rare events, 2) the Northern Migratory Coastal, Southern Migratory Coastal, NNCES, and Southern North Carolina Estuarine System stocks of common bottlenose dolphin overlap in coastal waters of North Carolina and Virginia at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect rare bycatch events. To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2011–2015: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data, and 2) a pooled CPUE approach (where all observer data from the most recent five years were combined into one sample to estimate CPUE) (Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018).

During the most recent five-year time period, 2011–2015, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 2.67% in state waters (0–3 miles from shore) and 5.36% in federal waters (3–200 miles from shore) (Lyssikatos and Garrison 2018). During these trips, observers documented three entangled dolphins, all off of North Carolina, that may have been from the Northern Migratory Coastal Stock. In August 2015, the NEFOP observed one mortality in gillnet gear that was ascribed solely to the Northern Migratory Coastal Stock. In January 2015, one mortality was observed by the NEFOP off Hatteras, North Carolina, entangled in a medium-mesh gillnet within 0.23 km of shore. This dolphin was ascribed to the Northern Migratory Coastal and NNCES stocks (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). In February 2013, the NEFOP documented a dolphin entangled in small-mesh gillnet gear that was released alive without serious injury; it was ascribed to the NNCES and Northern Migratory Coastal stocks (Wenzel *et al.* 2015; Lyssikatos and Garrison 2018). Because the animal was released without serious injury, it was not included in the bycatch estimation. This take was also reported by the fisherman through the Marine Mammal Authorization Program. The resultant five-year mean minimum and maximum mortality estimates (2011–2015) were 6.1 (CV=0.32) and 12.2 (CV=0.22) animals per year, respectively (Table 2a; Lyssikatos and Garrison 2018).

Historical stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear

attached (Byrd *et al.* 2014; Waring *et al.* 2015; Lyssikatos and Garrison 2018); however, none were documented over the current five-year period. From 2011 to 2015, six dead, stranded dolphins were recovered with markings indicative of interaction with gillnet gear, but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. Four of the six cases were ascribed to the Northern Migratory Coastal Stock alone, and two were ascribed to the Northern and Southern Migratory Coastal and NNCES stocks.

Chesapeake Bay Inshore Gillnet

During 2011–2015, there were two documented strandings of common bottlenose dolphins entangled in inshore gillnet gear in Chesapeake Bay. In 2013, in Maryland, a dead dolphin was recovered entangled in large-mesh gillnet gear. In 2015, in Virginia, a stranded animal was found entangled in gillnet gear (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). Both of these animals were ascribed to the Northern and Southern Migratory Coastal stocks, and both are included in the annual human-caused mortality and serious injury total for this stock (Table 2b) as well as in the stranding database and stranding totals presented in Table 3 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). There is no observer coverage of this fishery within Maryland waters of Chesapeake Bay; within Virginia waters of Chesapeake Bay, there is a low level of observer coverage (<1%). No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Virginia Pound Net

During 2011–2015, one common bottlenose dolphin stranding (mortality) that was ascribed to the Northern and Southern Migratory Coastal Stocks was found entangled in pound net gear in Virginia (in 2011; Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016), and was included in the annual human-caused mortality and serious injury total for this stock (Table 2b). An additional four dead, stranded dolphins were recovered with twisted twine markings indicative of interactions with pound net gear, but it is unknown whether the interactions with the gear contributed to the death of these animals and these cases are not included in the annual human-caused mortality and serious injury total for this stock. Of these four, one was ascribed solely to the Northern Migratory Coastal Stock, and the remaining three were ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. All of the strandings discussed here occurred inside estuarine waters near the mouth of the Chesapeake Bay in April, May, September, or October. Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Mid-Atlantic Menhaden Purse Seine

During 2011–2015, there were no documented mortalities or serious injuries in mid-Atlantic menhaden purse seine gear of common bottlenose dolphins that could be ascribed to the Northern Migratory Coastal Stock. The mid-Atlantic menhaden purse seine fishery historically reported an annual incidental take of one to five common bottlenose dolphins (NMFS 1991, pp. 5–73). There has been very limited federal observer coverage since 2008. No observer coverage was allocated to this fishery during 2011 or during 2013–2015, and for 2012 only three trips were observed. Because there is no systematic observer program for this fishery, no estimate of bycatch mortality is available.

Atlantic Blue Crab Trap/Pot

During 2011–2015, there were no documented mortalities or serious injuries in commercial blue crab trap/pot gear of common bottlenose dolphins that could be ascribed to the Northern Migratory Coastal Stock. Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots. However, stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

Mid-Atlantic Haul/Beach Seine

During 2011–2015, one serious injury of a common bottlenose dolphin occurred associated with the mid-Atlantic haul/beach seine fishery that could be ascribed to the Northern Migratory Coastal Stock. During 2014, a common bottlenose dolphin was found within a haul seine net in Virginia and released alive seriously injured (Maze-Foley and Garrison 2017). The animal was ascribed to the Northern and Southern Migratory Coastal and

NNCES stocks. This case was included in the stranding database and in the stranding totals presented in Table 3 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 9 June 2015) as well as in the annual human-caused mortality and serious injury total for this stock (Table 2b). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2012–2015. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Hook and Line (Rod and Reel)

During 2011–2015, stranding data identified two mortalities of common bottlenose dolphin that could be ascribed to the Northern Migratory Coastal Stock for which hook and line gear entanglement or ingestion were documented. For one mortality, available evidence suggested the hook and line gear interaction contributed to the cause of death (2012, Virginia). This animal was ascribed to the Northern and Southern Migratory Coastal Stocks. The second dolphin was in a state of advanced decomposition and it could not be determined whether the hook and line gear interaction contributed to cause of death (2014, Virginia). This mortality was ascribed to the Northern Migratory Coastal Stock alone. The two mortalities were included in the stranding database and in the stranding totals presented in Table 3 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). The 2012 mortality for which evidence suggested the gear contributed to the cause of death is included in the annual human-caused mortality and serious injury total for this stock (Table 2b).

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

Other Mortality

Historically, there have been occasional mortalities of common bottlenose dolphins during activities including during directed live capture-release studies, turtle relocation trawls, and fisheries surveys (Waring *et al.* 2016); however, none were documented during 2011–2015 that could be ascribed to the Northern Migratory Coastal Stock. All mortalities and serious injuries from known human-caused sources for the Northern Migratory Coastal Stock are summarized in Tables 2a, 2b and 2c.

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

Fishery	Years	Data Type	Observer Coverage	Observed Serious Injury	Observed Mortality	Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data
Mid-Atlantic Gillnet	2011–2015	Obs. Data Logbook	2.0, 2.6, 3.1, 3.6, 5.6	0, 0, 0, 0, 0	0, 0, 0, 0, 2	Min=6.1 (0.32) Max=12.2 (0.22)
Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015)						**Min=6.1 (0.32) Max=12.2 (0.22)**

Table 2b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Migratory Coastal Stock during 2011–2015 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum

and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.			
Fishery	Years	Data Type	5-year Count Based on Stranding Data
Chesapeake Bay Inshore Gillnet	2011–2015	Limited Observer and Stranding Data	Min=0 Max=2
Virginia Pound Net[a]	2011–2015	Stranding Data	Min=0 Max=1
Mid-Atlantic Menhaden Purse Seine	2011–2015	Limited Observer and Stranding Data	0
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	0
Mid-Atlantic Haul/Beach Seine	2011–2015	Limited Observer and Stranding Data	Min=0 Max=1
Hook and Line[b]	2011–2015	Stranding Data	Min=0 Max=1
Mean Annual Mortality due to unobserved commercial fisheries (2011–2015)			**Min=0 Max=1.0**

[a] Pound net interactions are included if the animal was found entangled in pound net gear. Strandings with twisted twine markings indicative of interactions with pound net gear are not included within the table. See "Virginia Pound Net" text for more details.
[b] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.

Table 2c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Migratory Coastal Stock during 2011–2015 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015) (Table 2a)	**Min=6.1 (0.32) Max=12.2 (0.22)**
Mean Annual Mortality due to unobserved commercial fisheries (2011–2015) (Table 2b)	**Min=0 Max=1.0**
Research Takes (5-year Min/Max Count)	0
Other takes (5-year Min/Max Count)	0
Mean Annual Mortality due to research and other takes (2011–2015)	**0**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)	**Min=6.1 Max=13.2**

Strandings

Between 2011 and 2015, 1,111 common bottlenose dolphins that were ascribed to the Northern Migratory Coastal Stock stranded along the Atlantic coast between North Carolina and New York (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 18 May 2016 (SER) and 6 June 2016 (NER)). It could not be determined if there was evidence of human interaction (HI) for 775 of these strandings, and for 247 it was determined there was no evidence of HI. The remaining 89 showed evidence of HI, of which 57 (64%) were fisheries interactions and 10 (11%) showed evidence of a boat strike (Table 3). It should be

recognized that evidence of HI does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions due to spatial and temporal overlap among several common bottlenose dolphin stocks. Of the 1,111 strandings ascribed to the Northern Migratory Coastal Stock, 634 were ascribed solely to this stock. Therefore, the counts in Table 3 likely include some animals from the Southern Migratory Coastal and NNCES stocks and, therefore, overestimate the number of strandings for the Northern Migratory Coastal Stock; those strandings that could not be ascribed to the Northern Migratory Coastal Stock alone are also included in the counts for these other stocks as appropriate. In addition, stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin, therefore, it is possible that some of the reported strandings were of the offshore form though that number is likely to be low (Byrd *et al.* 2014).

This stock has also been impacted by two large unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994; Morris *et al.* 2015). Both UMEs included deaths of dolphins north of Assateague, Virginia, in summer, corresponding solely to the Northern Migratory Coastal Stock area. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988), so impacts to the Northern Migratory Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988; Eguchi 2002). For the 2013–2015 UME, a total of 1,872 stranded common bottlenose dolphins were recovered in the UME area which stretched from New York to Brevard County, Florida. Of these, 381 stranded dolphins were recovered from the states of New York, New Jersey, Delaware, and Maryland (2013-2015 Bottlenose Dolphin Unusual Mortality Event in the Mid-Atlantic, accessed October 3, 2016). While some of these deaths may be attributable to the Offshore Stock, the majority likely came from the Northern Migratory Coastal Stock given their geographic location. This number is likely an underestimate of the total number of deaths for this stock, however, because it does not include animals that stranded in Virginia and North Carolina in cold water months that might have come from this stock, and not all dolphins that died during the UME would have been recovered. An analysis of trends in abundance for common bottlenose dolphins coast-wide (New Jersey to Florida) indicated a statistically significant decline in population size between 2011 and 2016 (Garrison *et al*. 2017a). A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model included a linear term for survey year and an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant ($p=0.0308$) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse-variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison *et al.* 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME. An assessment of the impacts of the 2013–2015 UME on common bottlenose dolphin stocks in the wNA is ongoing.

Table 3. Strandings of common bottlenose dolphins during 2011–2015 from North Carolina to New York that were ascribed to the Northern Migratory Coastal Stock, as well as number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Assignments to stock were based upon the understanding of the seasonal movements of this stock. However, in waters of North Carolina and Virginia there is likely overlap with other stocks during particular times of year. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016 (SER) and 6 June 2016 (NER)). Please note HI does not necessarily mean the interaction caused the animal's death.

State	2011			2012			2013			2014			2015		
Type	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD
North Carolina[a]	6[b]	16	22	11[c]	14	16	2[d]	21	36	3[e]	15	23	3[f]	10	23
Virginia[g]	7[h]	3	36	11[i]	7	37	11[j]	17	137	5[k]	5	44	9[l]	5	55
Maryland	2[m]	1	4	0	2	2	4[n]	19	43	0	1	6	0	2	8
Delaware	2[o]	2	6	1	3	11	3[p]	1	66	0	0	7	1	1	5
New Jersey	1[q]	14	4	2[r]	10	10	3[s]	58	92	0	2	16	0	0	24
New York	0	1	2	1[t]	2	4	0	12	26	1[u]	1	8	0	2	2
Annual Total	129			144			551			137			150		

[a] Strandings for North Carolina include data for November–April north of Cape Lookout when Northern Migratory Coastal animals may be in coastal waters. The stock identity of these strandings is highly uncertain and likely also includes animals from the NNCES Stock.
[b] Includes 5 FIs.
[c] Includes 10 FIs, 2 of which had markings indicative of interactions with gillnet gear (mortalities).
[d] Includes 2 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).
[e] Includes 3 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).
[f] Includes 2 FIs.
[g] Strandings from Virginia were ascribed to stock based upon both location and time of year. Some of the strandings ascribed to the Northern Migratory Coastal Stock could possibly be ascribed to the Southern Migratory Coastal Stock or NNCES Stock.
[h] Includes 6 FIs. One FI was an entanglement interaction in a Virginia pound net (mortality) and 2 FIs were mortalities with twisted twine markings indicative of interactions with Virginia pound net gear.
[i] Includes 1 mortality with evidence of a boat strike and 7 FIs. One FI was an entanglement interaction with hook and line gear (mortality) and 1 FI was a mortality with twisted twine markings indicative of interactions with Virginia pound net gear.
[j] Includes 8 FIs, 1 of which was a mortality with twisted twine markings indicative of interactions with Virginia pound net gear.
[k] Includes 3 FIs, 1 of which involved ingestion of hook and line gear (mortality). Another animal was released alive seriously injured following capture in a haul seine. Also includes 1 mortality with evidence of a boat strike.
[l] Includes 3 FIs. One FI was an entanglement interaction with commercial gillnet gear (mortality, Chesapeake Bay inshore gillnet fishery), and 2 FIs had markings indicative of interactions with gillnet gear (mortalities). Also includes 1 mortality with evidence of a boat strike.

[m] Includes 1 FI.
[n] Includes 3 FIs, 1 of which was an entanglement interaction with commercial gillnet gear (mortality, Chesapeake Bay inshore gillnet fishery).
[o] Includes 2 mortalities with evidence of a boat strike.
[p] Includes 1 FI.
[q] Includes 1 FI.
[r] Includes 1 mortality and 1 live animal with evidence of a boat strike.
[s] Includes 1 FI and 2 mortalities with evidence of a boat strike.
[t] Includes 1 FI.
[u] A mortality with evidence of a boat strike.

HABITAT ISSUES

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

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the Northern Migratory Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of common bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The Northern Migratory Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the Northern Migratory Coastal Stock is 48 and so the zero mortality rate goal, 10% of PBR, is 4.8. The documented mean annual human-caused mortality for this stock for 2011–2015 ranged between a minimum of 6.1 and a maximum of 13.2. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for the Northern Migratory Coastal Stock cannot be directly estimated because of the spatial overlap among the stocks of common bottlenose dolphins that occupy waters of North Carolina and Virginia resulting in uncertainty in the stock assignment of some takes, 2) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). The total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and therefore, cannot be considered to be insignificant and approaching a zero mortality and serious injury rate. The impacts of two large UMEs on the status of this stock are unknown. Analysis of trends in abundance suggests a probable decline in stock size between 2010–2011 and 2016, concurrent with a large UME in the area; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

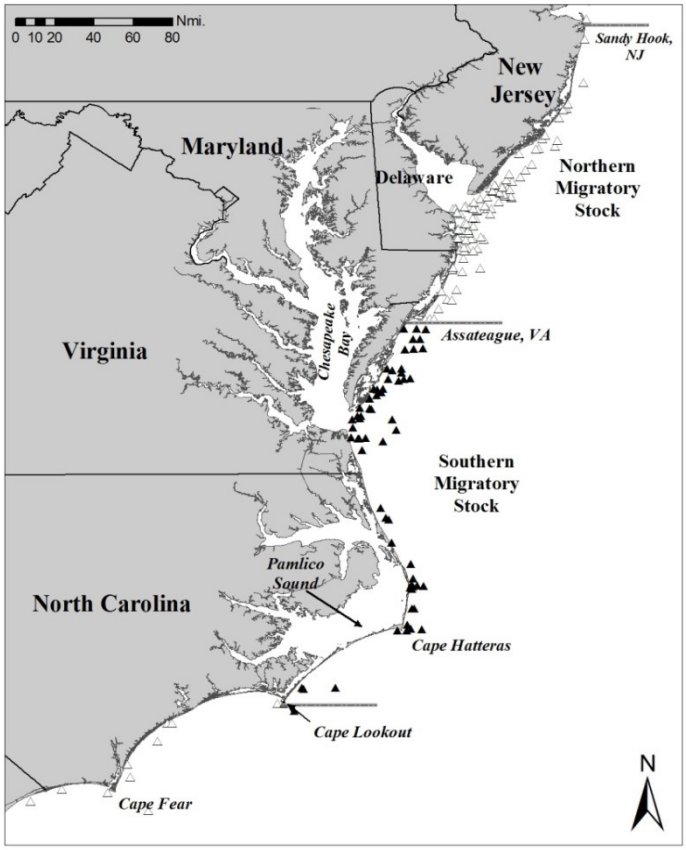

Figure 1. *The distribution of common bottlenose dolphin sightings in coastal waters from North Carolina to New Jersey during July-August 2002, 2004, 2010, 2011, and 2016. Sighting locations from aerial surveys are plotted as triangle symbols. Sightings ascribed to the Southern Migratory Coastal Stock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

The spatial distribution and migratory movements of the Southern Migratory Coastal Stock are poorly understood and have been defined based on movement data from satellite-tag telemetry and photo-ID studies, and stable isotope studies. The distribution of this stock is best described by satellite tag-telemetry data which provided evidence for a stock of dolphins migrating seasonally along the coast between North Carolina and northern Florida (Garrison *et al.* 2017b). Tag-telemetry data collected from two dolphins tagged in November 2004 just south of Cape Fear, North Carolina, suggested that, during October–

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

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

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

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

POPULATION SIZE

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

Background

Estimating the abundance of the Southern Migratory Coastal Stock is complicated by the spatiotemporal

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

Earlier abundance estimates (>8 years old)

Aerial surveys were conducted during the summers of 2002 and 2004. Survey tracklines for the 2002 and 2004 surveys were set perpendicular to the shoreline and effort was stratified into 0–20 m and 20–40 m strata with the majority of effort in the shallow depth stratum (Garrison *et al.* 2017a). The 2002 surveys employed two observer teams operating independently on the same aircraft to estimate the probability of detection on the trackline. This estimate was also applied to the 2004 survey to reduce bias in the resulting abundance estimate. The resulting abundance estimates from the 2002 and 2004 summer aerial surveys for the Southern Migratory Coastal Stock were 19,316 (CV=0.31) and 29,535 (CV=0.33), respectively (Garrison *et al.* 2017a). There were strong differences in spatial distribution between these two survey years, suggesting that the large difference in estimates was related to changes in distribution rather than population size of the stock (Garrison *et al.* 2017a). As recommended in the GAMMS II Workshop Report (Wade and Angliss 1997), these estimates are greater than eight years old and deemed unreliable and should not be used for PBR determinations. However, these estimates are included below in the assessment of trends for this stock.

Recent surveys and abundance estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters along the U.S. East Coast from southeastern Florida to Cape May, New Jersey, during the summers of 2010, 2011, and 2016 (Garrison *et al.* 2017a). The surveys were conducted along tracklines oriented perpendicular to the shoreline and spaced latitudinally at 20-km intervals, and covered waters from the shoreline to the continental shelf break (Garrison *et al.* 2017a).

The recent surveys were conducted using a two-team approach to develop estimates of detection probabilities using the independent observer approach with Distance analysis (Laake and Borchers 2004). The detection functions from each survey indicated a decreased probability of detection near the trackline. The sighting data were therefore "left-truncated" by analyzing only sightings occurring greater than 80 m from the trackline during the 2010 survey, 70 m during the 2011 survey, and 100 m from the trackline during the 2016 survey (see Buckland *et al.* 2001 for left-truncation methodology). The independent observer method assuming point independence was used to estimate detection probability on the trackline. This estimate accounts for the probability of detecting a marine mammal group conditional on it being available to both survey teams. Covariates that may influence detection probabilities (e.g., sea state, glare, cloud cover, visibility) were incorporated into both the mark-recapture and distance function components of the detection models (Laake and Borchers 2004; Garrison *et al.* 2017a). The resulting abundance estimates are negatively biased due to the effects of animals spending some time underwater where they are not available to the survey teams. However, due to the relatively short dive times of bottlenose dolphins (Klatsky *et al.* 2007) and the large group sizes, it is likely that this bias is small (Garrison *et al.* 2017a).

The abundance estimates for 2010, 2011, and 2016 summer aerial surveys were 9,217 (CV=0.51), 4,987 (CV=0.64), and 3,751 (CV=0.60), respectively (Garrison *et al.* 2017a). The 2016 estimate was used as the best estimate of the current population size for the stock due to possible effects from the 2013–2015 unusual mortality event. Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution. Another unquantified source of uncertainty in the abundance estimate is the potential overlap of this stock (during summer) with the NNCES Stock in near-shore ocean waters within 1 km from shore.

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

Month/Year	Area	N_{best}	CV
July–August 2002	Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)	19,316	0.31
July–August 2004	Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)	29,535	0.33
July–August 2010	Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)	9,217	0.51
July–August 2011	Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)	4,987	0.64
July–August 2016	Cape Lookout, North Carolina (34.6°N) to Assateague, Virginia (37.9°N)	3,751	0.60

Minimum Population Estimate

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

Current Population Trend

Available surveys allow a limited analysis of trend in population size for coastal stocks of common bottlenose dolphins. A standardized analytical approach accounting for variation in survey execution and environmental conditions was used to derive unbiased abundance estimates for each survey (Garrison *et al.* 2017a). A weighted generalized linear model was used to evaluate trends in population size by stock using abundance estimates from surveys conducted in the summers of 2002, 2004, 2010, 2011, and 2016. Abundance estimates were weighted by the inverse of their standard error, which reduces the influence of less certain estimates (Neter *et al.* 1983). Stock was treated as a fixed factor, and surveys were grouped into three periods to test for long term trends in population size: 2002–2004, 2010–2011, and 2016. Period was also included as a fixed factor in the model along with the interaction between stock and period. Contrasts were specified to test for differences in abundance between periods for each stock (Garrison *et al.* 2017a). For the Southern Migratory Coastal Stock, the resulting mean abundance estimate for 2002–2004 was 23,206 (CV=0.25), and that for 2010–2011 was 6,694 (CV=0.62). There was no significant difference between these estimates and the estimate of 3,751 (CV=0.60) for 2016. There is limited power to detect a significant change given the high CV of the estimates, interannual variability in spatial distribution and stock abundance between 2002 and 2004, and the availability of only one recent survey (Garrison *et al.* 2017a). However, see the Strandings section for a discussion of coast-wide trends in population size.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a "recovery" factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997; Wade 1998). The minimum population size of the Southern Migratory Coastal Stock of common bottlenose dolphins is 2,353. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.48 because the CV of the average mortality estimate is greater than 0.3 (Wade and Angliss 1997). PBR for this stock of common bottlenose dolphins is 23.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Southern Migratory Coastal Stock during 2011–2015 is unknown. The minimum mean annual fishery-related mortality and serious injury for observed fisheries and strandings identified as fishery-related ranged between 0 and 14.3 (CV=0.31). 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 2011–2015 ranged between 0 and 14.3 (Tables 2a, 2b and 2c). This range reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected and recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between this stock and other common bottlenose dolphin stocks throughout its range introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities and serious injuries were ascribed to a stock or stocks by comparing the season and geographic location of the take/stranding to the stock boundaries and geographic range delimited for each stock.

Fishery Information

There are 11 commercial fisheries that interact, or that potentially could interact, with this stock. These include the Category I mid-Atlantic gillnet fishery, nine Category II fisheries (Southeastern U.S. Atlantic shark gillnet, Southeast Atlantic gillnet, Chesapeake Bay inshore gillnet, Virginia pound net, Atlantic blue crab trap/pot, North Carolina roe mullet stop net, mid-Atlantic menhaden purse seine, mid-Atlantic haul/beach seine, and Southeastern U.S. Atlantic shrimp trawl fisheries), and the Category III Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery. Detailed fishery information is presented in Appendix III.

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the Southern Migratory Coastal Stock in the northern part of its range. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001; Lyssikatos and Garrison 2018). This fishery is observed by the Northeast and Southeast Fisheries Observer Programs. The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006, available at http://www.nmfs.noaa.gov/pr/pdfs/fr/fr71-24776.pdf). Mortality estimates for the period (2002–2006) immediately prior to implementation of the BDTRP and 2007–2011 are available in the 2015 stock assessment report for the Northern Migratory Coastal Stock (Waring *et al.* 2015). The current report covers the most recent available five-year estimate (NMFS 2016) for 2011–2015.

Mortality estimation for this stock is difficult because 1) observed takes are rare events, 2) the Southern Migratory Coastal, Northern Migratory Coastal, NNCES, and SNCES stocks of common bottlenose dolphin overlap in coastal waters of North Carolina and Virginia at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect bycatch events. To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2011–2015: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data and 2) a pooled CPUE approach (where all observer data from the most recent five years were combined into one sample to estimate CPUE; Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018).

During the most recent 5-year time period, 2011–2015, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 2.67% in state waters (0–3 miles from shore) and 5.36% in federal waters (3–200 miles from shore) (Lyssikatos and Garrison

2018). During these trips, observers documented one entangled dolphin (mortality) that may have been from the Southern Migratory Coastal Stock. The observed take (SEFOP) occurred off northern North Carolina in September 2014, and was ascribed to the NNCES and Southern Migratory Coastal stocks (Lyssikatos and Garrison 2018). The resultant 5-year mean minimum and maximum mortality estimates (2011–2015) for the Southern Migratory Coastal Stock were therefore 0 and 12.5 (CV=0.31) animals per year, respectively (Table 2a; Lyssikatos and Garrison 2018).

Historical and recent stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear attached (Byrd *et al.* 2014; Waring *et al.* 2015; Lyssikatos and Garrison 2018). In October 2011, the stranding network recovered a dead dolphin from a fisherman who had incidentally caught it in a small-mesh gillnet targeting spot in southern North Carolina during an unobserved trip. This animal was ascribed to the Southern Migratory Coastal and SNCES stocks. Because there is already an observer program-based bycatch estimate for the Southern Migratory Coastal Stock for the mid-Atlantic gillnet fishery, and the bycatch estimate was not zero, the additional recovered animal was not added to the bycatch estimate. However, the overall minimum annual mortality for this stock is likely not zero. During the current 5-year period there were also four common bottlenose dolphin strandings, all in North Carolina, with markings indicative of interaction with gillnet gear, but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. All four cases were ascribed to multiple stocks including the Southern Migratory Coastal Stock.

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

There have been no documented mortalities or serious injuries of common bottlenose dolphins associated with the Southeastern U.S. Atlantic Shark Gillnet or Southeast Atlantic Gillnet fisheries during 2011–2015 that could be ascribed to the Southern Migratory Coastal Stock (Gulak *et al.* 2012; Mathers *et al.* 2013; 2014; 2015; 2016). These fisheries target sharks and finfish in waters between North Carolina and southern Florida. The majority of fishing effort occurs in federal waters because Florida, Georgia, and South Carolina, with limited exception, prohibit the use of gillnets in state waters. The Southeast Gillnet Observer Program observes these fisheries year-round (e.g., Mathers *et al.* 2016).

Chesapeake Bay Inshore Gillnet

During 2011–2015, stranding data documented two interactions (mortalities) between common bottlenose dolphins and inshore gillnet gear in Chesapeake Bay. In 2013, in Maryland, a dead dolphin was recovered entangled in large-mesh gillnet gear. In 2015, in Virginia, another dead dolphin was recovered entangled in gillnet gear (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). Both of these animals were ascribed to the Northern and Southern Migratory Coastal stocks, and both are included in the annual human-caused mortality and serious injury total for this stock (Table 2b) as well as in the stranding database and stranding totals presented in Table 3 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). There is no observer coverage of this fishery within Maryland waters of Chesapeake Bay; however, within Virginia waters of Chesapeake Bay, there is a low level of observer coverage (<1%). No estimate of bycatch mortality is available for this fishery, and the documented interactions in this commercial gear represent a minimum known count of interactions in the last five years.

Virginia Pound Net

During 2011–2015, one stranded common bottlenose dolphin (mortality) was found entangled in pound net gear in Virginia. This animal was ascribed to the Northern and Southern Migratory Coastal stocks (in 2011; Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016), and is included in the annual human-caused mortality and serious injury total for this stock (Table 2b). During 2011–2015, an additional three dolphins stranded with twisted twine markings indicative of interactions with pound net gear, but it is unknown whether the interactions with the gear contributed to the death of these animals, and these cases are not included in the annual human-caused mortality and serious injury total for this stock. These three strandings were ascribed to the Southern and Northern Migratory Coastal stocks and the NNCES Stock. All of the strandings discussed here occurred inside estuarine waters near the mouth of the Chesapeake Bay in May, September, and October. The overall impact of the Virginia pound net fishery on the Southern Migratory Coastal Stock is unknown due to the limited information on the stock's movements. Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Atlantic Blue Crab Trap/Pot

During 2011–2015, stranding data documented one mortality and two serious injuries of common bottlenose dolphins from entanglements in trap/pot gear; all in Virginia. The mortality occurred during 2015 in commercial blue crab trap/pot gear. One serious injury occurred in 2014 in commercial blue crab trap/pot gear, and one occurred in 2015 in unidentified trap/pot gear. All three cases were ascribed to the Southern Migratory Coastal and NNCES stocks, and all are included in the annual human-caused mortality and serious injury total for this stock (Table 2b). These animals are also included in the stranding database and in the stranding totals presented in 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 18 May 2016 (SER) and 6 June 2016 (NER)). Because there is no observer program, it is not possible to estimate the total number of mortalities associated with crab traps/pots and these documented interactions in this commercial gear represent a minimum known count of interactions with this fishery. Stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

North Carolina Roe Mullet Stop Net

During 2011–2015, stranding data documented one mortality in which a common bottlenose dolphin was entangled in a stop net (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). This mortality, which occurred during November 2013, was ascribed to the NNCES and Southern Migratory Coastal stocks and is included in the annual human-caused mortality and serious injury total for this stock (Table 2b). In addition, a dead stranded dolphin with line markings indicative of interaction with stop net gear was recovered in October 2015 ~300 yards from a stop net, but it is unknown whether the interaction with gear contributed to the death of this animal, and this case is not included in the annual human-caused mortality and serious injury total for this stock. This animal was ascribed to multiple stocks: the Southern Migratory Costal, NNCES, and SNCES stocks. Both mortalities are included in the stranding database and in the stranding totals presented in Table 3 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). No estimate of bycatch mortality is available for the stop net fishery, and the confirmed interaction in this commercial gear represents a minimum known count of interactions with this fishery in the last five years. This fishery has not had regular, ongoing federal or state observer coverage. However, the NMFS Beaufort laboratory observed this fishery in 2001–2002 (Byrd and Hohn 2010), and Duke University observed the fishery in 2005–2006 (Thayer *et al.* 2007). Entangled dolphins were not documented during these formal observations, but historical takes of dolphins entangled in stop nets occurred in 1993 and 1999 (Byrd and Hohn 2010).

Mid-Atlantic Menhaden Purse Seine

During 2011–2015, there were no documented mortalities or serious injuries in mid-Atlantic menhaden purse seine gear of common bottlenose dolphins that could be ascribed to the Southern Migratory Coastal Stock. The mid-Atlantic menhaden purse seine fishery historically reported an annual incidental take of one to five common bottlenose dolphins (NMFS 1991, pp. 5–73). There has been very limited federal observer coverage since 2008. No observer coverage was allocated to this fishery during 2011 or during 2013–2015, and for 2012 only three trips were observed. Because there is no systematic observer program for this fishery, no estimate of bycatch mortality is available.

Mid-Atlantic Haul/Beach Seine

During 2011–2015, one serious injury of a common bottlenose dolphin occurred associated with the mid-Atlantic haul/beach seine fishery that could be ascribed to the Southern Migratory Coastal Stock. During 2014, a common bottlenose dolphin was found within a haul seine net in Virginia and released alive seriously injured (Maze-Foley and Garrison 2017). The animal was ascribed to the Northern and Southern Migratory Coastal stocks and NNCES Stock, and is included in the annual human-caused mortality and serious injury total for this stock (Table 2b) as well as in the stranding database and stranding totals presented in Table 3 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2012–2015. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Shrimp Trawl

During 2011–2015, there were no documented mortalities or serious injuries of common bottlenose dolphins associated with the shrimp trawl fishery that could be ascribed to the Southern Migratory Coastal Stock. There has been very little systematic observer coverage of this fishery in the Atlantic during the last decade.

Hook and Line (Rod and Reel)

During 2011–2015, stranding data documented four mortalities that could be ascribed to the Southern Migratory Coastal Stock for which hook and line gear entanglement or ingestion were recorded. For one mortality, available evidence suggested the hook and line gear interaction contributed to the cause of death (2012, Virginia). For one mortality, evidence suggested the hook and line gear interaction was not a contributing factor to cause of death (2012, North Carolina). For two mortalities, it could not be determined if the hook and line gear interaction contributed to cause of death (2011, Virginia; 2011, South Carolina). All four mortalities were ascribed to multiple stocks: the Northern Migratory Coastal, South Carolina/Georgia Coastal, and NNCES stocks. These mortalities were included in the stranding database and are included in the stranding totals presented in 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 18 May 2016 (SER) and 6 June 2016 (NER)). The 2012 mortality for which evidence suggested the gear contributed to the cause of death is included in the annual human-caused mortality and serious injury total for this stock (Table 2b).

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

Other Mortality

Historically, there have been occasional mortalities of common bottlenose dolphins during research activities (Waring *et al.* 2016); however, none were documented during 2011–2015 that could be ascribed to the Southern Migratory Coastal Stock. All mortalities and serious injuries from known human-caused sources for the Southern Migratory Coastal Stock are summarized in Tables 2a, 2b and 2c.

Table 2a. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock for commercial fisheries with ongoing, systematic, federal observer programs. Name of the fishery (Fishery), the years sampled (Years), the type of data used (Data Type), the annual percentage observer coverage (Observer Coverage), the observed serious injuries and mortalities recorded by on-board observers, and the mean annual estimate of mortality and serious injury (CV in parentheses) are provided. Minimum and maximum values are reported due to uncertainty in the assignment of mortalities to this particular stock because there is spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	Years	Data Type	Observer Coverage	Observed Serious Injury	Observed Mortality	Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data
Mid-Atlantic Gillnet	2011–2015	Obs. Data Logbook	2.0, 2.6, 3.1, 3.6, 5.6	0, 0, 0, 0, 0	0, 0, 0, 1, 0	Min=0 Max=12.5 (0.31)
Southeastern U.S. Atlantic Shark Gillnet	2011–2015	Obs. Data Logbook	NA due to uncertainty in reported effort	0,0,0,0,0	0,0,0,0,0	No estimate
Southeast Atlantic Gillnet	2011–2015	Obs. Data Logbook	NA due to uncertainty in reported effort	0,0,0,0,0	0,0,0,0,0	No estimate

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015)	**Min=0** **Max=12.5 (0.31)**

Table 2b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock during 2011–2015 for commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons. In addition, mortality due to research and other non-commercial fishery takes are included, as well as a total mean annual human caused mortality and serious injury summed from all sources.

Fishery	Years	Data Type	5-year Count Based on Stranding Data
Chesapeake Bay Inshore Gillnet	2011–2015	Limited Observer and Stranding Data	Min=0 Max=2
Virginia Pound Net[a]	2011–2015	Stranding Data	Min=0 Max=1
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	Min=0 Max=3
North Carolina Roe Mullet Stop Net[b]	2011–2015	Stranding Data	Min=0 Max=1
Mid-Atlantic Menhaden Purse Seine	2011–2015	Limited Observer and Stranding Data	0
Mid-Atlantic Haul/Beach Seine	2011–2015	Limited Observer and Stranding Data	Min=0 Max=1
Shrimp Trawl	2011–2015	Limited Observer and Stranding Data	0
Hook and Line[c]	2011–2015	Stranding Data	Min=0 Max=1
Mean Annual Mortality due to unobserved commercial fisheries (2011–2015)			**Min=0** **Max=1.8**

[a] Pound net interactions are included if the animal was found entangled in pound net gear. Strandings with twisted twine markings indicative of interactions with pound net gear are not included within the table. See "Virginia Pound Net" text for more details.
[b] Stop Net interactions are included if the animal was found entangled in stop net gear. Stranding with line markings indicative of interaction with stop net gear are not included within the table. See "North Carolina Roe Mullet Stop Net" text for more details.
[c] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.

Table 2c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern Migratory Coastal Stock during 2011–2015 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015) (Table 2a)	**Min=0** **Max=12.5 (0.31)**

Mean Annual Mortality due to unobserved commercial fisheries (2011–2015) (Table 2b)	**Min=0 Max=1.8**
Research Takes (5-year Min/Max Count)	0
Other takes (5-year Min/Max Count)	0
Mean Annual Mortality due to research and other takes (2011–2015)	**0**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)	**Min=0 Max=14.3**

Strandings

During 2011–2015, 965 common bottlenose dolphins stranded along the Atlantic coast between Florida and Virginia that could be ascribed to the Southern Migratory Coastal Stock (Table 3; Northeast Regional Marine Mammal Stranding Network; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016 (SER) and 6 June 2016 (NER)). It could not be determined if there was evidence of human interaction for 677 of these strandings, and for 203 it was determined there was no evidence of human interaction. The remaining 85 showed evidence of human interactions, of which 57 (67%) were fisheries interactions and four (5%) showed evidence of a boat strike (Table 3). It should be recognized that evidence of human interaction does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions due to spatial and temporal overlap among several common bottlenose dolphin stocks. Due to its migratory behavior, the Southern Migratory Coastal Stock can overlap with other common bottlenose dolphin stocks in every season. None of the 965 strandings ascribed to the Southern Migratory Coastal Stock were ascribed solely to this stock. Therefore, the counts in Table 3 likely include animals from other stocks and therefore overestimate the number of strandings attributable to the Southern Migratory Coastal Stock. Those strandings that could not be definitively ascribed to the Southern Migratory Coastal Stock alone are also included in the counts for these other stocks as appropriate. In addition, stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin, therefore it is possible that some of the reported strandings were of the offshore form though that number is likely to be low (Byrd *et al.* 2014).

This stock has also been impacted by three unusual mortality events (UMEs). Two events, one in 1987–1988 and one in 2013–2015, have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994; Morris *et al.* 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the Southern Migratory Coastal Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988), so impacts to the Southern Migratory Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988; Eguchi 2002). The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was 1,827 (http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 8 November 2016). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). The number of dolphins from the Southern Migratory Coastal Stock that died in this event is unknown. An analysis of trends in abundance for common bottlenose dolphins coast-wide (New Jersey to Florida) indicated a statistically significant decline in population size between 2011 and 2016 (Garrison *et al*. 2017a). A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model

included a linear term for survey year and an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant (p=0.0308) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse-variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison *et al.* 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME. An assessment of the impacts of the 2013–2015 UME on common bottlenose dolphin stocks in the wNA is ongoing. Finally, a UME was declared in South Carolina during February–May 2011. Six strandings assigned to the Southern Migratory Coastal Stock were considered to be part of the UME. The cause of this UME was undetermined.

Table 3. Strandings of common bottlenose dolphins during 2011–2015 from Maryland to Florida that were ascribed to the Southern Migratory Coastal Stock, as well as number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Assignments to stock were based upon the understanding of the seasonal movements of this stock; however, there is likely overlap with other stocks throughout the year. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016 (SER) and 6 June 2016 (GAR)). Please note HI does not necessarily mean the interaction caused the animal's death.

State	**2011**			**2012**			**2013**			**2014**			**2015**		
Type	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD
Maryland[a]	0	0	1	0	0	0	1[b]	1	8	0	0	6	0	0	2
Virginia[a]	7[c]	5	31	6[d]	5	32	15[e]	35	299	5[f]	5	49	5[g]	4	55
North Carolina[h]	10[i]	13	20	11[j]	10	12	10[k]	54	63	2[l]	25	24	7[m]	20	12
South Carolina[n] (Dec–Mar)	1[o]	2	5	0	5	2	0	5	12	0	6	7	0	1	2
Georgia[p] (Jan–Feb)	2	0	2	0	1	0	0	1	5	0	1	7	2[q]	2	4
Florida[p] (Jan–Feb)	0	1	4	0	0	2	1	0	3	0	1	6	0	0	2
Annual Total	104			86			513			144			118		

[a] Strandings from Virginia and Maryland were ascribed to stock based upon location and time of year with most occurring between May and September that could be ascribed to the Southern Migratory Coastal Stock. Some of these strandings could also be ascribed to the Northern Migratory Coastal Stock or NNCES Stock.
[b] Includes 1 fisheries interaction (FI) that was an entanglement interaction in commercial gillnet gear (mortality,

Chesapeake Bay inshore gillnet fishery).
[c] Includes 6 FIs. One FI was an entanglement interaction with hook and line gear (mortality), 1 was an entanglement interaction in a Virginia pound net (mortality), and 2 were mortalities with twisted twine markings indicative of interactions with Virginia pound net gear.
[d] Includes 3 FIs, 1 of which was an entanglement interaction with hook and line gear (mortality). Also includes 1 mortality with evidence of a boat strike.
[e] Includes 9 FIs, 1 of which was a mortality with twisted twine markings indicative of an interaction with Virginia pound net gear.
[f] Includes 3 FIs, 1 of which was an entanglement interaction with commercial trap/pot gear (released alive, seriously injured). Another animal was released alive seriously injured following capture in a haul seine. Also includes 1 mortality with evidence of a boat strike.
[g] Includes 3 FIs. One FI was an entanglement interaction with commercial blue crab trap/pot gear (mortality), 1 was an entanglement interaction with commercial gillnet gear (mortality, Chesapeake Bay inshore gillnet fishery), and 1 was an entanglement interaction with trap/pot gear (released alive, seriously injured).
[h] Strandings from North Carolina were ascribed based on location and time of year. During summer and fall, some of these strandings could also be ascribed to the NNCES or SNCES stocks.
[i] Includes 7 FIs, 1 of which was an entanglement interaction with commercial gillnet gear (mortality, mid-Atlantic gillnet fishery).
[j] Includes 8 FIs, 3 of which had markings indicative of interactions with gillnet gear (mortalities), and 1 in which an animal ingested hook and line gear (mortality).
[k] Includes 7 FIs, 1 of which was an entanglement in a stop net (mortality, North Carolina roe mullet stop net fishery). Also includes 2 mortalities with evidence of a boat strike.
[l] Includes 2 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).
[m] Includes 6 FIs. One FI had markings indicative of interactions with gillnet gear (mortality), and 1 had markings indicative of an entanglement in a stop net (mortality, North Carolina roe mullet stop net fishery).
[n] Strandings in coastal waters from South Carolina during December–March are potentially ascribed to the Southern Migratory Coastal Stock or the South Carolina/Georgia Coastal Stock.
[o] Includes 1 FI in which an animal ingested hook and line gear (mortality).
[p] Strandings in Georgia and northern Florida during January and February could be ascribed to the South Carolina/Georgia or the Northern Florida Coastal Stocks, respectively.
[q] Includes 1 FI which was an entanglement interaction with commercial blue crab trap/pot gear (released alive, seriously injured).

HABITAT ISSUES

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

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the Southern Migratory Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of common bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The Southern Migratory Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the Southern Migratory Coastal Stock is 23 and so the zero mortality rate goal, 10% of PBR, is 2.3. The documented mean annual human-caused mortality for this stock for 2011–2015 ranged between a minimum of 0 and a maximum of 14.3. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for the Southern

Migratory Coastal Stock cannot be directly estimated because of the spatial overlap of this stock with several other stocks of common bottlenose dolphins resulting in uncertainty in the stock assignment of takes, 2) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these biases and uncertainties, there is insufficient information to determine whether or not the total fishery-related mortality and serious injury is approaching a zero mortality and serious injury rate. The impacts of two large UMEs on the status of this stock are unknown. Although there was no statistically significant difference in abundance for this stock between the 2010–2011 and 2016 surveys, a statistically significant decline in population size of all common bottlenose dolphins in coastal waters from New Jersey to Florida between 2010–2011 and 2016 was detected (Garrison *et al*. 2017a), concurrent with a large UME in the area; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Western North Atlantic South Carolina/Georgia Coastal Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

Common bottlenose dolphins are found in coastal waters south of Cape Hatteras, North Carolina, year-round (Blaylock and Hoggard 1994; Garrison and Yeung 2001; Garrison *et al.* 2016; Caldwell 2016). Significant genetic differentiation was observed between animals sampled in coastal waters of Georgia and those sampled in coastal waters of Virginia north to New Jersey (Rosel *et al*. 2009) indicative of demographic independence between animals sampled in these two regions. Speakman *et al*. (2006), using

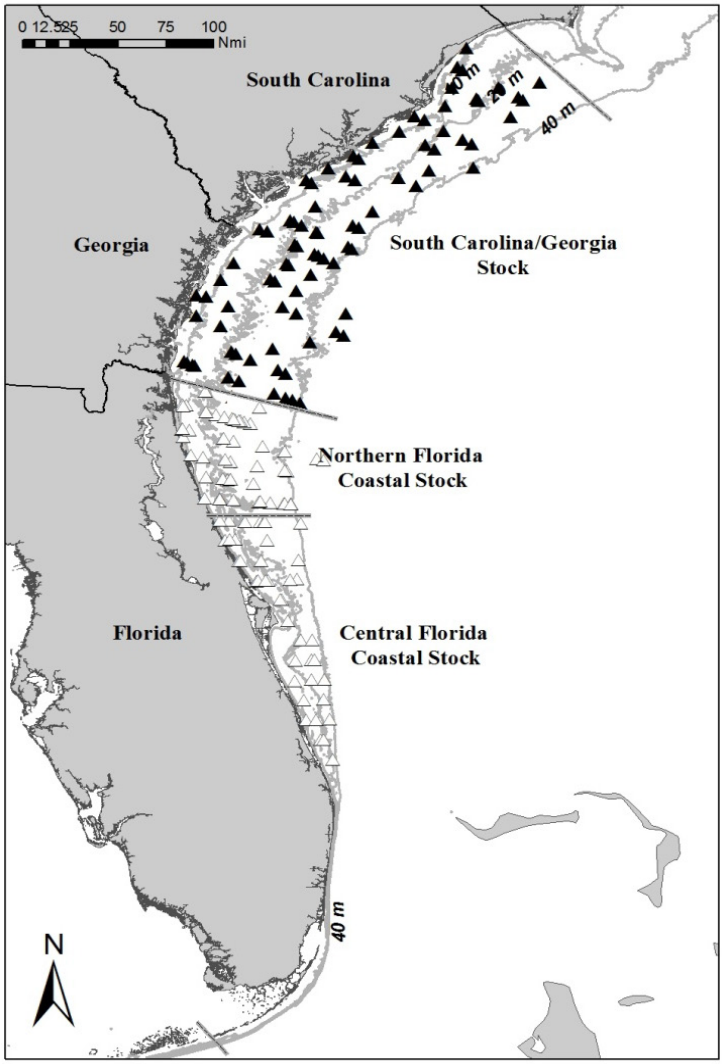

Figure 1. *The South Carolina/Georgia Coastal Stock of common bottlenose dolphins (Little River Inlet, South Carolina to the Georgia/Florida border). Symbols represent all sightings of bottlenose dolphin groups from NMFS 2010, 2011, and 2016 aerial surveys; dark symbols - groups within the boundaries of this stock. In waters >20 m, sightings may include the offshore morphotype of bottlenose dolphins. Horizontal gray lines intersecting the coast denote the stock boundaries.*

photo-ID studies, documented dolphins in coastal waters off Charleston, South Carolina, that are not known resident members of the local estuarine stock, revealing that the dolphins present in coastal waters are unlikely to be from the adjacent estuarine stocks. A coast-wide photo-ID catalog established across 15 sites from Cape May, New Jersey, to Cape Canaveral, Florida, found no matches between the northernmost and southernmost sites (Urian *et al.* 1999), and less movement was seen among southern sites than among northern sites (Urian *et al.* 1999), supporting the presence of stock structure in coastal waters of this region.

The South Carolina/Georgia Coastal Stock is best delimited as the dolphins of the coastal morphotype inhabiting coastal waters from the shoreline to approximately the 200-m isobath from the Little River Inlet, South Carolina (33.8°N), south to the Georgia/Florida border (30.7°N) (Figure 1). The northern and southern boundaries for this stock are provisional as the spatial extent of this stock is poorly understood. These boundaries were derived from the first delimitation of coastal stocks in 2002 (Waring *et al.* 2002) when the original single, coast-wide coastal stock suggested by Scott *et al.* (1988) was broken into seven management units (Waring *et al.* 2002). The offshore boundary was determined based on a combined genetic and logistic regression analysis that incorporated depth, latitude, and distance from shore to model the probability that a particular bottlenose dolphin group seen in coastal waters south of Cape Hatteras was of the coastal morphotype (Garrison *et al.* 2017a). Dolphins of the coastal morphotype were identified in waters out to 97 m depth. The logistic regression predicted that the majority of the coastal morphotype inhabits waters 0–20 m in depth and that the density of the coastal morphotype declines with increasing depth (Garrison *et al.* 2017a). South of Cape Hatteras in waters less than 20 m depth, 70% of the bottlenose dolphins were predicted to be of the coastal morphotype and fewer than 10% of the animals present beyond 35 m depth were predicted to be of the coastal morphotype (Garrison *et al.* 2017a).

During cold water months (best defined by January and February), this stock likely overlaps with the Southern Migratory Coastal Stock, which is thought to migrate south from waters of southern Virginia and north central North Carolina in the summer to waters south of Cape Fear, North Carolina, and as far south as coastal Florida during winter months (Garrison *et al*. 2017b). The spatiotemporal overlap of this stock with other stocks complicates the ability to definitively identify the offshore extension for the stock and the assignment of human-caused dolphin mortalities to stock at certain times of the year. While common bottlenose dolphins do exhibit significant site fidelity (e.g., Zolman 2002; Speakman *et al.* 2006), it is unlikely this stock contains multiple demographically independent populations as the area inhabited by the stock is relatively homogeneous, does not cross any known biogeographic breaks, nor cover multiple marine ecoregions (Spalding *et al.* 2007). However, to date, no analyses of stock structure within this stock have been performed.

It should be noted that dolphins in waters between 3 km from shore and the 200-m isobath from the Little River Inlet, South Carolina, to Cape Lookout, North Carolina, in summer are currently not contained with any delimited stock. These dolphins could be members of the South Carolina/Georgia Coastal Stock, but their affinities are currently unknown.

POPULATION SIZE

The best available abundance estimate for the South Carolina/Georgia Coastal Stock of common bottlenose dolphins in the western North Atlantic is 6,027 (CV=0.34; Table 1; Garrison *et al.* 2017a). This estimate is derived from aerial surveys conducted during the summer of 2016 covering coastal and shelf waters from Florida to New Jersey.

Background

Estimating the abundance of the South Carolina/Georgia Coastal Stock is complicated by the spatiotemporal overlap the stock has with the Southern Migratory Coastal Stock in cold water months and the potential for the presence, in nearshore coastal waters, of dolphins from estuarine common bottlenose dolphin stocks in South Carolina and Georgia. Summer surveys are best for estimating the abundance for this stock because it overlaps least with the Southern Migratory Coastal stock during warm water months. Using the logistic regression described above (Garrison *et al.* 2017a), abundance estimates for the South Carolina/Georgia Coastal Stock were made using sightings observed in the 0–200 m depth range during aerial summer surveys between the Little River Inlet, South Carolina (33.8°N), and the Georgia/Florida border (30.7°N). The regression model was used to estimate the probability that each sighting during the aerial survey is of the coastal (vs. offshore) morphotype. This probability and associated estimates of uncertainty were then incorporated into the abundance estimate for the coastal morphotype within the stock's range (Garrison *et al*. 2017a).

Earlier abundance estimates (>8 years old)

Aerial surveys were conducted during the summers of 2002 and 2004. Survey tracklines for the 2002 and 2004

surveys were set perpendicular to the shoreline and effort was stratified into 0–20 m and 20–40 m strata with the majority of effort in the shallow depth stratum (Garrison *et al.* 2017a). The 2002 surveys employed two observer teams operating independently on the same aircraft to estimate the probability of detection on the trackline. This estimate was also applied to the 2004 survey to reduce bias in the resulting abundance estimate. The resulting abundance estimates from the 2002 and 2004 summer aerial surveys for the South Carolina/Georgia Coastal Stock were 21,940 (CV=0.27) and 4,894 (CV=0.45), respectively (Garrison *et al.* 2017a). There were strong differences in spatial distribution between these two survey years, suggesting that the large difference in estimates was related to changes in distribution rather than population size of the stock (Garrison *et al.* 2017a). As recommended in the GAMMS II Workshop Report (Wade and Angliss 1997), these estimates are greater than eight years old and deemed unreliable and should not be used for PBR determinations. However, these estimates are included below in the assessment of trends for this stock.

Recent surveys and abundance estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters along the U.S. East Coast from southeastern Florida (26.9°N) to Sandy Hook, New Jersey (40.3°N), during the summers of 2010, 2011, and 2016 (see Garrison *et al.* 2017a for survey design). The surveys were conducted along tracklines spaced latitudinally at 20-km intervals and oriented perpendicular to the shoreline, and covered waters from the shoreline to the continental shelf break (Garrison *et al.* 2017a).

The recent surveys were conducted using a two-team approach to develop estimates of detection probabilities using the independent observer approach with Distance analysis (Laake and Borchers 2004). The detection functions from each survey indicated a decreased probability of detection near the trackline. The sighting data were therefore "left-truncated" by analyzing only sightings occurring greater than 80 m from the trackline during the 2010 survey, 70 m during the 2011 survey, and 100 m from the trackline during the 2016 survey (see Buckland *et al.* 2001 for left-truncation methodology). The independent observer method assuming point independence was used to estimate detection probability on the trackline. This estimate accounts for the probability of detecting a marine mammal group conditional on it being available to both survey teams. Covariates that may influence detection probabilities (e.g., sea state, glare, cloud cover, visibility) were incorporated into both the mark-recapture and distance function components of the detection models (Laake and Borchers 2004; Garrison *et al.* 2017a). In addition, the probability that an observed group was of the coastal morphotype was incorporated into the abundance estimates as noted above. The resulting abundance estimates are negatively biased due to the effects of animals spending some time underwater where they are not available to the survey teams. However, due to the relatively short dive times of bottlenose dolphins (Wells *et al.* 2013) and the large group sizes, it is likely that this bias is small (Garrison *et al.* 2017a).

The abundance estimates derived from the summer 2010, 2011, and 2016 surveys were 11,274 (CV=0.27), 7,406 (CV=0.53), and 6,027 (CV=0.34). The 2016 estimate was used as the best estimate of the current population size for the stock due to possible effects from the 2013–2015 unusual mortality event. Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution.

Table 1. Summary of abundance estimates for the western North Atlantic South Carolina/Georgia Coastal Stock of common bottlenose 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
July-August 2002	Little River Inlet, South Carolina (33.8°N) to the Georgia/Florida border (30.7°N)	21,940	0.27
July-August 2004	Little River Inlet, South Carolina (33.8°N) to the Georgia/Florida border (30.7°N)	4,894	0.45

July-August 2010	Little River Inlet, South Carolina (33.8°N) to the Georgia/Florida border (30.7°N)	11,274	0.27
July-August 2011	Little River Inlet, South Carolina (33.8°N) to the Georgia/Florida border (30.7°N)	7,406	0.53
July-August 2016	Little River Inlet, South Carolina (33.8°N) to the Georgia/Florida border (30.7°N)	6,027	0.34

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. The best estimate for the South Carolina/Georgia Coastal Stock is 6,027 (CV=0.34). The resulting minimum population estimate is 4,569.

Current Population Trend

Available surveys allow a limited analysis of trend in population size for coastal stocks of common bottlenose dolphins. A standardized analytical approach accounting for variation in survey execution and environmental conditions was used to derive unbiased abundance estimates for each survey (Garrison *et al.* 2017a). A weighted generalized linear model was used to evaluate trends in population size by stock using abundance estimates from surveys conducted in the summers of 2002, 2004, 2010, 2011, and 2016. Abundance estimates were weighted by the inverse of their standard error, which reduces the influence of less certain estimates (Neter *et al.* 1983). Stock was treated as a fixed factor, and surveys were grouped into three periods to test for long term trends in population size: 2002–2004, 2010–2011, and 2016. Period was also included as a fixed factor in the model along with the interaction between stock and period. Contrasts were specified to test for differences in abundance between periods for each stock (Garrison *et al.* 2017a). For the South Carolina/Georgia Coastal Stock, the resulting mean abundance estimate for 2002–2004 was 9,434 (CV=0.41), and that for 2010–2011 was 9,567 (CV=0.42). There was no significant difference between these estimates and the estimate of 6,027 (CV=0.34) for 2016. There is limited power to detect a significant change given the high CV of the estimates, interannual variability in spatial distribution and stock abundance between 2002 and 2004, and the availability of only one recent survey (Garrison *et al.* 2017a). However, see the Strandings section for a discussion of coast-wide trends in population size.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum productivity rate, and a "recovery" factor (MMPA Sec. 3. 16 U.S.C. 1362; Wade and Angliss 1997; Wade 1998). The minimum population size of the South Carolina/Georgia Coastal Stock of common bottlenose dolphins is 4,569. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because this stock is depleted. PBR for this stock of common bottlenose dolphins is 46.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Total mean annual human-caused mortality and serious injury for the South Carolina/Georgia Coastal Stock during 2011–2015 is unknown. The mean annual fishery-related mortality and serious injury for observed fisheries and strandings identified as fishery-related ranged between 1.0 and 1.2. Additional mean annual mortality and serious injury due to other human-caused sources (fishery research, recreational fishing gear) was 0.4. The minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2015 therefore ranged between 1.4 and 1.6 (Table 2). This range reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015),

3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between this stock and other common bottlenose dolphin stocks introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities were assigned to a stock or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock.

Fishery Information

There are six commercial fisheries that interact, or that potentially could interact, with this stock. These include four Category II fisheries (Southeastern U.S. Atlantic shark gillnet, Southeast Atlantic gillnet, Southeastern U.S. Atlantic shrimp trawl, and Atlantic blue crab trap/pot fisheries) and two Category III fisheries (Georgia cannonball jellyfish trawl, and Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fisheries). Limited or no observer data are available for these fisheries that may interact with this stock. Detailed fishery information is presented in Appendix III.

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

There have been no documented mortalities or serious injuries of common bottlenose dolphins associated with the Southeastern U.S. Atlantic Shark Gillnet or Southeast Atlantic Gillnet fisheries during 2011–2015 that could be ascribed to the South Carolina/Georgia Coastal Stock (Gulak *et al.* 2012; Mathers *et al.* 2013; 2014; 2015; 2016). These fisheries target sharks and finfish in waters between North Carolina and southern Florida. The majority of fishing effort occurs in federal waters because Florida, Georgia, and South Carolina, with limited exception, prohibit the use of gillnets in state waters. These fisheries use gillnets set in a sink (anchored), stab, set, strike, or drift fashion. The Southeast Gillnet Observer Program observes these fisheries year-round (e.g., Mathers *et al.* 2016).

Shrimp Trawl

During 2011–2015, one common bottlenose dolphin mortality occurred in association with the shrimp trawl fishery and was ascribed to the South Carolina/Georgia Coastal Stock. Through the Marine Mammal Authorization Program, a fisherman self-reported a mortality in 2013 off South Carolina in which a dolphin became entangled in his commercial shrimp trawl. This mortality was also included in the stranding database as well as in the annual human-caused mortality and serious injury total for this stock (Table 2). In addition, during 2015 off South Carolina, one common bottlenose dolphin was caught in the trawl net of a commercial shrimp trawl and released alive without serious injury (Maze-Foley and Garrison 2017). This animal was reported through the Marine Mammal Authorization Program. Prior to these interactions, the most recent documented interaction was another self-reported take in a commercial shrimp trawl in South Carolina during 2002. No other bottlenose dolphin mortality or serious injury has been reported to NMFS for the Atlantic shrimp trawl fishery. However, Greenman and McFee (2014) suggested shrimp trawl related mortality of bottlenose dolphins in South Carolina was underreported. There has been very little systematic observer coverage of this fishery during the last decade and the documented interactions in this commercial gear represents a minimum known count of interactions with this fishery.

Atlantic Blue Crab Trap/Pot

During 2011–2015, one mortality and one serious injury of common bottlenose dolphins were documented in association with commercial blue crab trap/pot gear. In 2011, one dolphin stranded dead with commercial blue crab pot trap/pot gear around its peduncle, and this animal was ascribed solely to the South Carolina/Georgia Coastal Stock. In addition, in 2015, a dolphin stranded alive entangled in commercial blue crab trap/pot gear. The animal was disentangled, released alive, and considered seriously injured (Maze-Foley and Garrison 2017). The dolphin was ascribed to the South Carolina/Georgia Coastal and Southern Migratory Coastal stocks. Both of these animals were included in the stranding database and in the stranding totals presented in Table 3 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016), and both were included in the annual human-caused mortality and serious injury total for this stock (Table 2). Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots and these documented interactions in this commercial gear represents a minimum known count of interactions with this fishery.

Georgia Cannonball Jellyfish Trawl

During 2011–2015, two common bottlenose dolphins ascribed to the South Carolina/Georgia Coastal Stock were observed incidentally killed by commercial fishing vessels trawling for cannonball jellyfish. These mortalities occurred in 2011 and 2012 off the Georgia coast and both were included in the annual human-caused mortality and

serious injury total for this stock (Table 2). Initially, this fishery was an experimental fishery but then became a bona fide commercial fishery in 2013 (Page 2015). The fishery operates from October–November to May–June, primarily in federal waters (3–200 nm) offshore of Georgia, with limited effort in Georgia's coastal state waters (Page 2015). Bycatch for this fishery was examined during December 2005–December 2012 by placing observers onboard commercial vessels operating in state and adjacent federal waters. The two bottlenose dolphin mortalities were observed during this study (Page 2015). The state of Georgia continued observer coverage until May 2014 and then ceased observing this fishery. There is no federal observer coverage of this fishery.

Hook and Line (Rod and Reel)

During 2011–2015, stranding data recorded one common bottlenose dolphin mortality for which hook and line gear ingestion was documented, but it could not be determined whether the interaction contributed to cause of death based on the available evidence so this mortality was not included in the annual human-caused mortality and serious injury total for this stock (Table 2). The dolphin was ascribed to the South Carolina/Georgia Coastal and Southern Migratory Coastal stocks. This mortality occurred in 2011 in South Carolina, and was included in the stranding database and in the stranding totals presented in Table 3 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016).

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

Other Mortality

During 2011–2015, there were four additional human-caused interactions that were all ascribed solely to the South Carolina/Georgia Coastal Stock: two mortalities and two animals released alive, one without serious injury and one for which it could not be determined if the animal was seriously injured. In 2012, one common bottlenose dolphin mortality occurred incidental to a research trawl off South Carolina. One mortality occurred during 2013 in a small-mesh recreational gillnet in South Carolina. Because this was a recreational, not commercial net, it does not fall under the Southeast Atlantic gillnet fishery. These two mortalities were included in the stranding database and in the annual human-caused mortality and serious injury total for this stock (Table 2). In addition, one bottlenose dolphin was released alive without serious injury following an entanglement in research bottom trawl gear during 2012. There was also an at-sea report in 2013 of an animal entangled in unidentified line, and it could not be determined if this animal was seriously injured. The live release and at-sea sighting are described by Maze-Foley and Garrison (2018). All mortalities and serious injuries from known sources for the South Carolina/Georgia Coastal Stock are summarized in Table 2.

Table 2. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the South Carolina/Georgia Coastal Stock. For fisheries with an ongoing, systematic, federal observer program, the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortality and serious injury recorded by on-board observers, and mean annual mortality and serious injury are provided. For fisheries that do not have an ongoing, systematic, federal observer program, counts of mortality and serious injury based on stranding data or fisherman self-reported takes via the Marine Mammal Authorization Program (MMAP) are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with the Southern Migratory Coastal Stock. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates. NA = not applicable.

Fishery	Years	Data Type	Observer Coverage	Observed Mortality and Serious Injury	Mean Annual Mortality and Serious Injury Based on Observer Data	5-year Min/Max Count Based on Stranding and/or MMAP Data
Southeastern U.S. Atlantic Shark Gillnet	2011–2015	Obs. Data Logbook	NA due to uncertainty in reported effort	0,0,0,0,0	Not estimated	NA

Southeast Atlantic Gillnet	2011–2015	Obs. Data Logbook	NA due to uncertainty in reported effort	0,0,0,0,0	Not estimated	NA
Shrimp Trawl	2011–2015	Limited Observer and MMAP fisherman self-reported takes	NA	NA	NA	1
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	NA	NA	NA	min=1 max=2
Georgia Cannonball Jellyfish Trawl[a]	2011–2012	Obs. Data Logbook	<5%	1,1,0,0, NA	0.6[a]	NA
Hook and Line[b]	2011–2015	Stranding Data	NA	NA	NA	0
Mean Annual Mortality due to commercial fisheries (2011–2015)					**0.6**	**min=0.4 max=0.6**
Research Takes (5-year Count)					1	
Other takes (recreational fishing gear; 5-year Count)					1	
Mean Annual Mortality due to research and other takes (2011–2015)					**0.4**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)					**min=1.4 max=1.6**	

[a] Observer coverage for this fishery occurred during 3.5 years only, and 2 mortalities were observed during this time. The 2 observed mortalities were divided by 3.5 years to calculate mean annual mortality.
[b] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.

Strandings

There were 273 stranded common bottlenose dolphins documented between 2011 and 2015 in the range of the South Carolina/Georgia Coastal Stock (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). It could not be determined whether there was evidence of human interaction for 173 of these strandings, and for 85 it was determined there was no evidence of human interaction. The remaining 15 showed evidence of human interaction, including eight fishery interactions (FIs). As mentioned above, two of the FIs were entanglements in commercial blue crab trap/pot gear, one consisted of ingested hook and line gear, and one was an entanglement in a commercial shrimp trawl. Two additional human interactions also mentioned above were an entanglement in recreational gillnet gear and an entanglement in a research trawl. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

It is worth noting that during winter months, the South Carolina/Georgia Coastal Stock overlaps with the Southern Migratory Coastal Stock and it is currently not possible to assign strandings to a single stock if they are recovered during the time of overlap. Therefore, the counts in Table 3 likely include some animals from the Southern Migratory Coastal Stock and overestimate the total number of strandings for the South Carolina/Georgia Coastal Stock. Of the 273 strandings ascribed to the South Carolina/Georgia Coastal Stock, 198 were ascribed solely to this stock.

This stock has been impacted by three unusual mortality events (UMEs). Two events, one in 1987–1988 and one in 2013–2015, have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994; Morris *et al.* 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the South Carolina/Georgia Coastal Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988), so impacts to the South Carolina/Georgia Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988; Eguchi 2002). The total number of stranded bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–

2015 UME was 1827, including 305 from South Carolina and Georgia (http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 8 November 2016). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). Fourteen strandings considered to be part of the UME were assigned to the South Carolina/Georgia Coastal Stock. An analysis of trends in abundance for common bottlenose dolphins coast-wide (New Jersey to Florida) indicated a statistically significant decline in population size between 2011 and 2016 (Garrison *et al*. 2017a). A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model included a linear term for survey year and an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant (p=0.0308) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse-variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison *et al.* 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME. An assessment of the impacts of the 2013–2015 UME on common bottlenose dolphin stocks in the wNA is ongoing. Finally, a UME was declared in South Carolina during February–May 2011. The cause of this UME was undetermined.

Table 3. Strandings of common bottlenose dolphins during 2011–2015 that were ascribed to the South Carolina/Georgia Coastal Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Assignments to stock were based upon the understanding of the seasonal movements of the coastal stocks. However, there is overlap between this stock and the Southern Migratory Coastal Stock during winter months, and it is currently not possible to distinguish between them. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016). Please note HI does not necessarily mean the interaction caused the animal's death.

State	**2011**			**2012**			**2013**			**2014**			**2015**		
Type	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD
South Carolina	3[a]	9	12	1[b]	16	8	2[c]	23	40	2[d]	14	20	0	6	8
Georgia	4[e]	2	14	0	3	8	1	8	37	0	2	15	2[f]	2	11
Annual Total	44			36			111			53			29		

[a] Includes 2 fishery interactions (FIs), 1 of which involved an animal that ingested hook and line gear (mortality).

[b] Includes 1 entanglement interaction in research trawl gear (mortality).

[c] Includes 2 FIs. One FI was an entanglement interaction with recreational gillnet gear (mortality), and 1 was an entanglement interaction with commercial shrimp trawl gear (mortality).

[d] Includes 2 FIs.

[e] Includes 1 FI, which was an entanglement interaction with commercial blue crab trap/pot gear (mortality).

[f] Includes 1 FI, which was an entanglement interaction with commercial blue crab trap/pot gear (released alive, seriously injured).

HABITAT ISSUES

The nearshore and estuarine habitats occupied by the coastal morphotype are adjacent to areas of high human population and some are highly industrialized. Studies have examined persistent organic pollutant concentrations in bottlenose dolphins inhabiting estuaries along the Atlantic coast and have found evidence of high blubber concentrations particularly near Brunswick, Georgia, Charleston, South Carolina, and Beaufort, North Carolina (Hansen *et al.* 2004; Balmer *et al.* 2011; Kucklick *et al.* 2011). The concentrations found in male dolphins from both of these sites exceeded toxic threshold values that may result in adverse effects on health or reproductive rates (Schwacke *et al.* 2002; Hansen *et al.* 2004; Balmer *et al.* 2011). Studies of contaminant concentrations relative to life history parameters showed higher levels of mortality in first-born offspring and higher contaminant concentrations in these calves and in primiparous females (Wells *et al.* 2005). Balmer *et al.* (2011) reported extremely high levels of total PCBs in dolphins sampled in estuarine waters near Brunswick, Georgia, where there is a point source Superfund site. Dolphins that have been sighted exclusively in estuarine waters 40 km north of Brunswick, near Sapelo Island, Georgia, also exhibit elevated PCB levels (Balmer *et al.* 2011). Movement of contaminated prey may be the source of exposure for the dolphins in Sapelo Island (Balmer *et al.* 2011), and, if those prey enter the coastal environment, the South Carolina/Georgia Coastal Stock could also be exposed to these unusually high contaminant levels. The exposure to environmental pollutants and subsequent effects on population health is an area of concern and active research.

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the South Carolina/Georgia Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The South Carolina/Georgia Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the South Carolina/Georgia Coastal Stock is 46 and so the zero mortality rate goal, 10% of PBR, is 4.6. The documented total mean annual human-caused mortality for this stock for 2011–2015 ranged between 1.4 and 1.6. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for the South Carolina/Georgia Coastal Stock cannot be directly estimated because of the spatial overlap among the stocks of bottlenose dolphins that occupy waters of South Carolina and Georgia resulting in uncertainty in the stock assignment of takes, 2) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these biases and uncertainties, there is insufficient information to determine whether or not the total fishery-related mortality and serious injury is approaching a zero mortality and serious injury rate. The stock may have been negatively impacted by the large 2013–2015 UME and the 2011 UME,

and some dolphins in the stock could contain high levels of PCBs from a Superfund site in Georgia. These factors could influence the recovery of this stock. Analysis of trends in abundance suggest a possible decline in stock abundance between 2002–2004 and 2016; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

Common bottlenose dolphins are found in coastal waters south of Cape Hatteras, North Carolina, to southern Florida year-round (Blaylock and Hoggard 1994; Garrison and Yeung 2001; Garrison *et al.* 2016; Caldwell 2016). Significant genetic differentiation was observed between animals sampled in coastal waters of Georgia and those sampled in the Jacksonville, Florida, area (Rosel *et al*. 2009), indicative of demographic independence between animals sampled in these two coastal regions. Genetic analyses of dolphins sampled in coastal and estuarine waters near both Jacksonville,

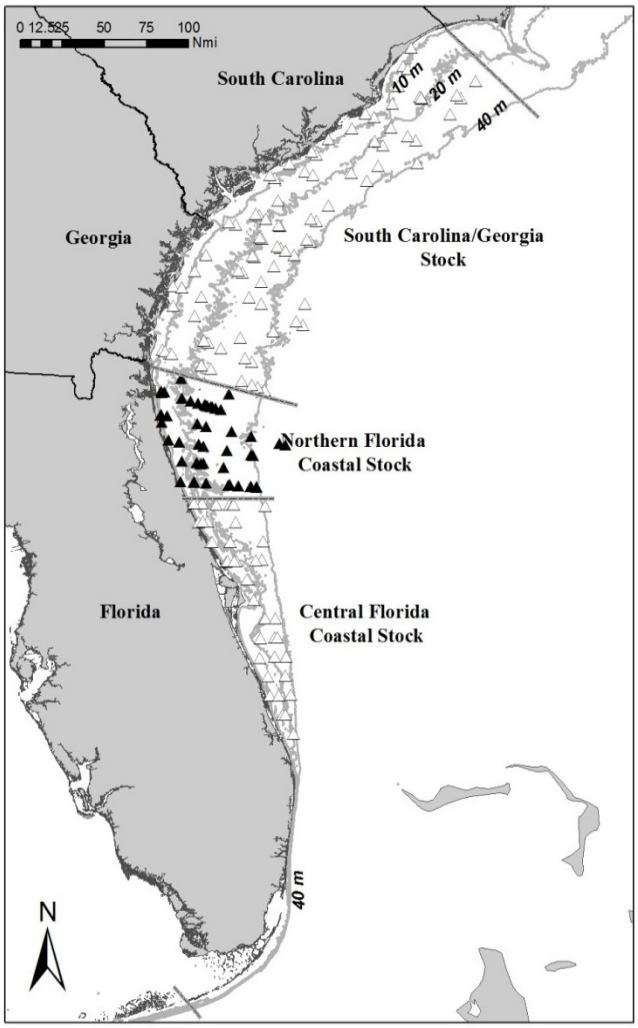

Figure 1. *The Northern Florida Coastal Stock of common bottlenose dolphins (Georgia/Florida border to 29.4°N). Symbols represent all sightings of bottlenose dolphin groups from NMFS 2010, 2011, and 2016 aerial surveys; dark symbols - groups within the boundaries of this stock. In waters > 20 m, sightings may include the offshore morphotype of bottlenose dolphins. Horizontal gray lines intersecting the coast denote the stock boundaries.*

Florida, and the Indian River Lagoon, Florida, revealed significant differentiation (Rosel *et al.* 2009; Richards *et al.* 2013), indicating the presence of demographically independent coastal and estuarine stocks along the Florida coast. Similarly, photo-ID studies in the same area distinguished between dolphins that used coastal waters and those using estuarine waters of the intracoastal waterway and St. John's River (Caldwell 2001; 2016).

The Northern Florida Coastal Stock is delimited as the dolphins of the coastal morphotype inhabiting coastal waters from the shoreline to approximately the 200-m isobath from the Georgia/Florida border (30.7°N) south to 29.4°N (Figure 1). The northern and southern boundaries for this stock are provisional as the spatial extent of this stock is poorly understood. These boundaries are derived from the first delimitation of coastal stocks in 2002 (Waring *et al.* 2002) when the original single, coast-wide coastal stock suggested by Scott *et al.* (1988) was broken into seven management units (Waring *et al.* 2002). The offshore boundary was determined based on a combined genetic and logistic regression analysis that incorporated depth, latitude, and distance from shore to model the probability that a particular bottlenose dolphin group seen in coastal waters south of Cape Hatteras was of the coastal morphotype (Garrison *et al.* 2017a). Dolphins of the coastal morphotype were identified in waters out to 97 m depth. The logistic regression predicted that the majority of the coastal morphotype inhabits waters 0–20 m in depth and that the density of the coastal morphotype declines with increasing depth (Garrison *et al.* 2017a). South of Cape Hatteras in waters less than 20 m depth, 70% of the bottlenose dolphins were predicted to be of the coastal morphotype and fewer than 10% of the animals present beyond 35 m depth were predicted to be of the coastal morphotype (Garrison *et al.* 2017a).

While common bottlenose dolphins do exhibit significant site fidelity, it is unlikely this stock contains multiple demographically independent populations because the habitat inhabited by the stock is relatively homogeneous, it does not cross any known biogeographic breaks nor cover multiple marine ecoregions (Spalding *et al.* 2007). Recently, Caldwell (2016) reported that three dolphins sighted behind trawling shrimp boats offshore of Jacksonville, Florida, were also photographed behind shrimp boats trawling off Hilton Head and/or Myrtle Beach, South Carolina. These sightings suggest the stock may cover a larger area. To date, no analyses of stock structure within this stock have been performed.

During cold water months, this stock likely overlaps with the Southern Migratory Coastal Stock, which is thought to migrate south from waters of southern Virginia and north central North Carolina in the summer to waters south of Cape Fear and as far south as coastal Florida during winter months (Garrison *et al*. 2017b). There is no firm boundary defining the offshore extent of the Northern Florida Coastal Stock and it overlaps to some degree with the Offshore Stock of common bottlenose dolphins (Garrison *et al.* 2017a). The spatiotemporal overlap of this stock with other stocks complicates the ability to definitively identify the offshore extension for the stock and the assignment of human-caused dolphin mortalities to stock at certain times of the year.

POPULATION SIZE

The best available abundance estimate for the Northern Florida Coastal Stock of common bottlenose dolphins in the western North Atlantic is 877 (CV=0.49; Table 1; Garrison *et al.* 2017a). This estimate is derived from aerial surveys conducted during the summer of 2016 covering coastal and shelf waters from Florida to New Jersey.

Background

Estimating the abundance of the Northern Florida Coastal Stock is complicated by the spatiotemporal overlap the stock has with the Southern Migratory Coastal Stock in cold water months, the potential for some overlap with the wNA Offshore Stock, and by the presence, in nearshore coastal waters, of dolphins from estuarine common bottlenose dolphin stocks in Florida (Mazzoil *et al.* 2011; Caldwell 2016). Summer surveys are best for estimating the abundance for this stock because it overlaps least with the Southern Migratory Coastal Stock during warm water months. Using the logistic regression described above (Garrison *et al.* 2017a), abundance estimates for the Northern Florida Coastal Stock were made using sightings observed in the 0–200 m depth stratum during summer aerial surveys between the Georgia/Florida border and 29.4°N. The regression model was used to estimate the probability that each sighting during the aerial survey is of the coastal (vs. offshore) morphotype. This probability and associated estimates of uncertainty were then incorporated into the abundance estimate for the coastal morphotype within the stock's range (Garrison *et al*. 2017a).

Earlier abundance estimates (>8 years old)

Aerial surveys were conducted during the summers of 2002 and 2004. Survey tracklines for the 2002 and 2004 surveys were set perpendicular to the shoreline and effort was stratified into 0–20 m and 20–40 m strata with the majority of effort in the shallow depth stratum (Garrison *et al.* 2017a). The 2002 surveys employed two observer teams operating independently on the same aircraft to estimate the probability of detection on the trackline. This

estimate was also applied to the 2004 survey to reduce bias in the resulting abundance estimate. The resulting abundance estimates from the 2002 and 2004 summer aerial surveys for the Northern Florida Coastal Stock were 299 (CV=0.84) and 2,130 (CV=0.41), respectively (Garrison *et al.* 2017a). There were strong differences in spatial distribution between these two survey years, suggesting that the large difference in estimates was related to changes in distribution rather than population size of the stock (Garrison *et al.* 2017a). As recommended in the GAMMS II Workshop Report (Wade and Angliss 1997), these estimates are >8 years old and deemed unreliable and should not be used for PBR determinations. However, these estimates are included below in the assessment of trends for this stock.

Recent surveys and abundance estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters along the U.S. East Coast from southeastern Florida (26.9°N) to Sandy Hook, New Jersey (40.3°N), during the summers of 2010, 2011, and 2016 (see Garrison *et al.* 2017a for survey design). The surveys were conducted along tracklines spaced latitudinally at 20-km intervals and oriented perpendicular to the shoreline, and covered waters from the shoreline to the continental shelf break (Garrison *et al.* 2017a).

The recent surveys were conducted using a two-team approach to develop estimates of detection probabilities using the independent observer approach with Distance analysis (Laake and Borchers 2004). The detection functions from each survey indicated a decreased probability of detection near the trackline. The sighting data were therefore "left-truncated" by analyzing only sightings occurring greater than 80 m from the trackline during the 2010 survey, 70 m during the 2011 survey, and 100 m from the trackline during the 2016 survey (see Buckland *et al.* 2001 for left-truncation methodology). The independent observer method assuming point independence was used to estimate detection probability on the trackline. This estimate accounts for the probability of detecting a marine mammal group conditional on it being available to both survey teams. Covariates that may influence detection probabilities (e.g., sea state, glare, cloud cover, visibility) were incorporated into both the mark-recapture and distance function components of the detection models (Laake and Borchers 2004; Garrison *et al.* 2017a). In addition, the probability that an observed group was of the coastal morphotype was incorporated into the abundance estimates as noted above. The resulting abundance estimates are negatively biased due to the effects of animals spending some time underwater where they are not available to the survey teams. However, due to the relatively short dive times of bottlenose dolphins (Wells *et al.* 2013) and the large group sizes, it is likely that this bias is small (Garrison *et al.* 2017a).

The abundance estimates derived from the summer 2010, 2011, and 2016 surveys were 4,355 (CV=0.45), 8,618 (CV=0.83), and 877 (CV=0.49). The 2016 estimate was used as the best estimate of the current population size for the stock due to possible effects from the 2013–2015 unusual mortality event (see Strandings section). Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution (See Current Population Trend section).

Table 1. Summary of abundance estimates for the western North Atlantic Northern Florida Coastal Stock of common bottlenose 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
July-August 2002	Georgia/Florida border (30.7°N) to 29.4°N latitude	299	0.84
July-August 2004	Georgia/Florida border (30.7°N to 29.4°N latitude	2,130	0.41
July-August 2010	Georgia/Florida border (30.7°N to 29.4°N latitude	4,355	0.45

July-August 2011	Georgia/Florida border (30.7°N to 29.4°N latitude	8,618	0.83
July-August 2016	Georgia/Florida border (30.7°N to 29.4°N latitude	877	0.49

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. The best estimate for the Northern Florida Coastal Stock is 877 (CV=0.49). The resulting minimum population estimate is 595.

Current Population Trend

Available surveys allow a limited analysis of trend in population size for coastal stocks of common bottlenose dolphins. A standardized analytical approach accounting for variation in survey execution and environmental conditions was used to derive unbiased abundance estimates for each survey (Garrison *et al.* 2017a). A weighted generalized linear model was used to evaluate trends in population size by stock using abundance estimates from surveys conducted in the summers of 2002, 2004, 2010, 2011, and 2016. Abundance estimates were weighted by the inverse of their standard error, which reduces the influence of less certain estimates (Neter *et al.* 1983). Stock was treated as a fixed factor, and surveys were grouped into three periods to test for long term trends in population size: 2002–2004, 2010–2011, and 2016. Period was also included as a fixed factor in the model along with the interaction between stock and period. Contrasts were specified to test for differences in abundance between periods for each stock (Garrison *et al.* 2017a). For the Northern Florida Coastal Stock, the resulting mean abundance estimate for 2002–2004 was 740 (CV=1.81), and that for 2010–2011 was 5,270 (CV=0.71). There was no significant difference between these estimates and the estimate of 877 (CV=0.49) for 2016. There is limited power to detect a significant change given the high CV of the estimates, interannual variability in spatial distribution and stock abundance between 2002 and 2004, and the availability of only one recent survey (Garrison *et al.* 2017a). However, see the Strandings section for a discussion of coast-wide trends in population size.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are not known for the wNA coastal morphotype. 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 Northern Florida Coastal Stock of common bottlenose dolphins is 595. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because this stock is depleted. PBR for this stock of common bottlenose dolphins is 6.0.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Northern Florida Coastal Stock during 2011–2015 is unknown. The mean annual fishery-related mortality and serious injury for strandings identified as fishery-related was 0. Mean annual mortality and serious injury due to other human-caused sources (fishery research and entanglements in unidentified gear and wire) was 0.6. Therefore, the minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2015 was also 0.6 (Table 2). This is a minimum estimate because 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected and recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, and 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). In the sections below, dolphin mortalities were assigned to a stock or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock.

Fishery Information

There are four commercial fisheries that interact, or that potentially could interact, with this stock. These include four Category II fisheries (Southeastern U.S. Atlantic shark gillnet, Southeast Atlantic gillnet, Southeastern U.S. Atlantic shrimp trawl, and Atlantic blue crab trap/pot fisheries) and one Category III fishery (Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery). Only limited observer data are available for these and other fisheries that may interact with this stock. Detailed fishery information is presented in Appendix III.

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

There have been no documented mortalities or serious injuries of common bottlenose dolphins associated with the Southeastern U.S. Atlantic Shark Gillnet or Southeast Atlantic Gillnet fisheries during 2011–2015 that could be ascribed to the Northern Florida Coastal Stock (Gulak *et al.* 2012; Mathers *et al.* 2013; 2014; 2015; 2016). These fisheries target sharks and finfish in waters between North Carolina and southern Florida. The majority of fishing effort occurs in federal waters because Florida, Georgia, and South Carolina, with limited exception, prohibit the use of gillnets in state waters. These fisheries use gillnets set in a sink (anchored), stab, set, strike, or drift fashion. The Southeast Gillnet Observer Program observes these fisheries year-round (e.g., Mathers *et al.* 2016).

Shrimp Trawl

During 2011–2015, there were no documented mortalities or serious injuries in association with the shrimp trawl fishery that could be ascribed to the Northern Florida Coastal Stock. While no interactions have been reported thus far, the potential for interactions exists in this area. Caldwell (2016) reported that bottlenose dolphins were regularly sighted behind trawling shrimp boats in coastal waters of northern Florida, and that the shrimp trawl fishing industry may influence movements of coastal dolphins. There has been very little systematic observer coverage of this fishery during the last decade, so it is not possible to estimate the total number of interactions or mortalities associated with the shrimp trawl fishery.

Atlantic Blue Crab Trap/Pot

During 2011–2015, there were no documented mortalities or serious injuries in crab trap/pot gear that could be ascribed to the Northern Florida Coastal Stock. The most recent documented interaction with this fishery was from 2009. Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots. Stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

Hook and Line (Rod and Reel)

During 2011–2015, there were no documented mortalities or serious injuries of common bottlenose dolphins involving hook and line gear entanglement or ingestion that could be ascribed to the Northern Florida Coastal Stock. The most recent documented interaction with hook and line gear was from 2010. It should be noted that, in general, it cannot be determined if 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.

Other Mortality

During 2011-2015, there was one common bottlenose dolphin mortality ascribed to the Northern Florida Coastal Stock that occurred incidental to fishery research. This mortality occurred in 2014 and resulted from an entanglement in the lazy line of a research trawl. Two additional mortalities occurred in 2013, likely as a result of entanglements. One entanglement involved green, braided rope of unknown origin that was bridled through the mouth of a dolphin, and the second involved conductor wire wrapped tightly around and cutting into the peduncle and flukes of a dolphin. These three mortalities were included in the stranding database and in the annual human-caused mortality and serious injury total for this stock (Table 2). All mortalities and serious injuries from known sources for the Northern Florida Coastal Stock are summarized in Table 2.

Table 2. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern Florida Coastal Stock. For fisheries with an ongoing, systematic, federal observer program, the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortality and serious injury recorded by on-board observers, and mean annual mortality and serious injury are provided. For fisheries that do not have an ongoing, systematic, federal observer program, minimum counts of mortality and serious injury based on stranding data or fisherman self-reported takes via the Marine Mammal Authorization Program (MMAP) are given. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates. NA = not applicable.

Fishery	Years	Data Type	Observer Coverage	Observed Mortality and Serious Injury	Mean Annual Mortality and Serious Injury Based on Observer Data	5-year Count Based on Stranding and/or MMAP Data
Southeastern U.S. Atlantic Shark Gillnet	2011–2015	Obs. Data Logbook	NA due to uncertainty in reported effort	0,0,0,0,0	Not estimated	NA
Southeast Atlantic Gillnet	2011–2015	Obs. Data Logbook	NA due to uncertainty in reported effort	0,0,0,0,0	Not estimated	NA
Shrimp Trawl	2011–2015	Limited Observer and MMAP fisherman self-reported takes	NA	NA	NA	0
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	NA	NA	NA	0
Hook and Line	2011–2015	Stranding Data	NA	NA	NA	0
Mean Annual Mortality due to commercial fisheries (2011–2015)					**0**	**0**
Research Takes (5-year Count)					1	
Other takes (entanglements in unidentified gear and conductor wire; 5-year Count)					2	
Mean Annual Mortality due to research and other takes (2011–2015)					**0.6**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)					**0.6**	

Strandings

During 2011–2015, 142 stranded common bottlenose dolphins were recovered within the range of the Northern Florida Coastal Stock (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). It could not be determined whether there was evidence of human

interaction for 117 of these strandings, and for 19 it was determined there was no evidence of human interaction. The remaining six showed evidence of human interactions, one of which was an entanglement in research trawling gear, as mentioned above. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

It is worth noting that during winter months, the Northern Florida Coastal Stock likely overlaps with the Southern Migratory Coastal Stock and it is currently not possible to distinguish between them. Therefore, the counts in Table 3 likely include some animals from the Southern Migratory Coastal Stock and overestimate the total number of strandings for the Northern Florida Coastal Stock. Of the 142 strandings ascribed to the Northern Florida Coastal Stock, 122 were ascribed solely to this stock.

This stock has also been impacted by two unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015. Both UMEs have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994; Morris *et al.* 2015) and both included deaths of dolphins in spatio-temporal locations that apply to the Northern Florida Coastal Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphins was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988), so impacts to the Northern Florida Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988; Eguchi 2002). The total number of stranded bottlenose dolphins from New York through central Florida (Brevard County) during the 2013–2015 UME was ~1827, including 319 from northern and central Florida (http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 8 November 2016). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). An analysis of trends in abundance for common bottlenose dolphins coast-wide (New Jersey to Florida) indicated a statistically significant decline in population size between 2011 and 2016 (Garrison *et al*. 2017a). A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model included a linear term for survey year and an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant (p=0.0308) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse-variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison *et al.* 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME. An assessment of the impacts of the 2013–2015 UME on common bottlenose dolphin stocks in the wNA is ongoing.

Table 3. Strandings of common bottlenose dolphins during 2011–2015 that were ascribed to the Northern Florida Coastal Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Assignments to stock were based upon the understanding of the seasonal movements of the coastal stocks. However, there is overlap between this stock and the Southern Migratory Coastal Stock during winter months, and it is currently not possible to distinguish between them. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016). Please note HI does not necessarily mean the interaction caused the animal's death.

State	**2011**			**2012**			**2013**			**2014**			**2015**		
Type	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD

Northern Florida Coastal Stock	2	4	13	0	1	15	3[a]	9	62	1[b]	4	23	0	1	4
Annual Total	19			16			74			28			5		

[a] Includes 2 HIs (mortalities) involving severe entanglements, one in unidentified gear and the other in conductor wire.

[b] This HI was an incidental take (mortality) in research trawl gear.

HABITAT ISSUES

The nearshore and estuarine habitats occupied by the coastal morphotype are adjacent to areas of high human population and some are highly industrialized. Studies that have examined persistent organic pollutant concentrations in bottlenose dolphins inhabiting estuaries along the Atlantic coast found evidence of high blubber concentrations, particularly near Brunswick, Georgia, Charleston, South Carolina, and Beaufort, North Carolina (Hansen *et al.* 2004; Balmer *et al.* 2011; Kucklick *et al.* 2011). The concentrations found in male dolphins from these sites exceeded toxic threshold values that may result in adverse effects on health or reproductive rates (Schwacke *et al.* 2002; Hansen *et al.* 2004; Balmer *et al.* 2011). Studies of contaminant concentrations relative to life history parameters showed higher levels of mortality in first-born offspring and higher contaminant concentrations in these calves and in primiparous females (Wells *et al.* 2005). Balmer *et al.* (2011) reported extremely high levels of total PCBs in dolphins sampled in estuarine waters near Brunswick, Georgia, where there is a point source Superfund site. Dolphins that were sighted exclusively in estuarine waters 40 km north of Brunswick, near Sapelo Island, Georgia, also exhibited elevated PCB levels (Balmer *et al.* 2011). Movement of contaminated prey may be the source of exposure for the dolphins in Sapelo Island (Balmer *et al.* 2011), and, if those prey enter the coastal environment, the Northern Florida Coastal Stock could also be exposed to these unusually high contaminant levels, as the northern boundary of the stock is approximately 50 km from Brunswick, Georgia. In fact, Watanabe *et al.* (2000) reported high levels of the PCB congeners associated with the signature PCB formulation from this Superfund site in one stranded dolphin from northern Florida. The exposure to environmental pollutants and subsequent effects on population health is an area of concern and active research.

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the Northern Florida Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. In addition, 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. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The Northern Florida Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the Northern Florida Coastal Stock is 6.0 and so the zero mortality rate goal, 10% of PBR, is 0.6. The documented total mean annual human-caused mortality for this stock for 2011–2015 was 0.6. However, this estimate is biased low for the following reasons: 1) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 2) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these biases and uncertainties, there is insufficient information to determine whether or not the total fishery-related mortality and serious injury is approaching a zero mortality and serious injury rate. The stock may have been negatively impacted by the large 2013–2015 UME and some dolphins in the stock could contain high levels of PCBs from a Superfund site in Georgia. Both of these factors could

influence the recovery of this stock. Analysis of trends in abundance suggest a probable decline in stock size between 2010–2011, and 2016; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Western North Atlantic Central Florida Coastal Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

Common bottlenose dolphins are found in coastal waters south of Cape Hatteras, North Carolina, to southern Florida year-round (Blaylock and Hoggard 1994; Garrison and Yeung 2001; Garrison *et al.* 2016; Mazzoil *et al.* 2016; Caldwell 2016). Significant genetic differentiation was observed between animals sampled in coastal waters of Georgia and those sampled in the Jacksonville, Florida, area (Rosel *et al*. 2009) indicative of demographic independence between animals sampled in these two coastal regions. Similarly, genetic analyses of dolphins sampled in

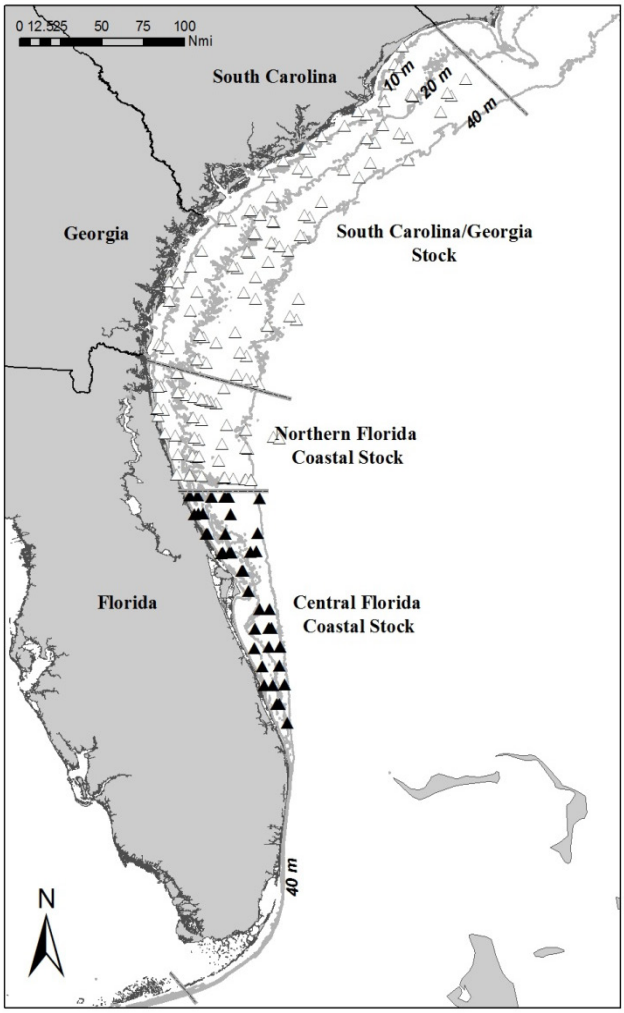

Figure 1. *The Central Florida Coastal Stock of common bottlenose dolphins (29.4°N to Vaca Key). Symbols represent all sightings of bottlenose dolphin groups from NMFS 2010, 2011, and 2016 aerial surveys; dark symbols - groups within the boundaries of this stock. In waters >20 m, sightings may include the offshore morphotype of bottlenose dolphins. Horizontal gray lines intersecting the coast denote the stock boundaries.*

coastal and estuarine waters near the Indian River Lagoon, Florida, revealed significant differentiation (Richards *et al.* 2013). Photo-ID studies in both Jacksonville and in central Florida near the Indian River Lagoon also distinguished between dolphins that used coastal waters and those using estuarine waters (Mazzoil *et al.* 2011; Caldwell 2016) indicating the presence of demographically independent coastal and estuarine stocks along the Florida coast.

The Central Florida Coastal Stock is delimited as the dolphins of the coastal morphotype inhabiting coastal waters from the shoreline to the 200-m isobath from 29.4°N south to the western end of Vaca Key (~24.7°N, 81.1°W) where the stock boundary for the Florida Keys Stock begins (Figure 1). There has been little study of bottlenose dolphin stock structure in coastal waters of central and southern Florida, and both the northern and southern boundaries for this stock are provisional as the spatial extent of this stock is poorly understood. The boundaries are derived from the first delimitation of coastal stocks in 2002 (Waring *et al.* 2002) when the original single, coast-wide coastal stock suggested by Scott *et al.* (1988) was broken into seven management units (Waring *et al.* 2002). The offshore boundary was determined based on a combined genetic and logistic regression analysis that incorporated depth, latitude, and distance from shore to model the probability that a particular bottlenose dolphin group seen in coastal waters south of Cape Hatteras was of the coastal morphotype (Garrison *et al.* 2017a). Dolphins of the coastal morphotype were identified in waters out to 97 m depth. The logistic regression predicted that the majority of the coastal morphotype inhabits waters 0–20 m in depth and that the density of the coastal morphotype declines with increasing depth (Garrison *et al.* 2017a). South of Cape Hatteras in waters less than 20 m depth, 70% of the bottlenose dolphins were predicted to be of the coastal morphotype and fewer than 10% of the animals present beyond 35 m depth were predicted to be of the coastal morphotype (Garrison *et al.* 2017a). These spatial patterns may not apply as well to the Central Florida Coastal Stock, however, as there is a significant change in the bathymetric slope and a close approach of the Gulf Stream to the shoreline south of Cape Canaveral.

It is plausible this stock contains multiple demographically independent populations because its range crosses a known biogeographic break at Cape Canaveral, Florida (Pelc *et al.* 2009), and appropriate coastal habitat is limited in southern Florida between West Palm Beach and Miami where the Gulf Stream comes close to shore. The lack of appropriate habitat in this region could serve as a barrier between members of this stock that inhabit coastal waters from Vaca Key eastward to approximately Miami, Florida, and those inhabiting coastal waters north of West Palm Beach where the shelf widens again.

There is no firm boundary defining the offshore extent of this stock and it overlaps to some degree with the Offshore Stock (Garrison *et al.* 2017a). This spatiotemporal overlap complicates the ability to definitively identify the offshore extent for the stock and the assignment of human-caused dolphin mortalities to stock at certain times of the year.

POPULATION SIZE

The best available abundance estimate for the Central Florida Coastal Stock of common bottlenose dolphins in the western North Atlantic is 1,218 (CV=0.35; Table 1; Garrison *et al.* 2017a). This estimate is derived from aerial surveys conducted during the summer of 2016 covering coastal and shelf waters from Florida to New Jersey.

Background

Estimating the abundance of the Central Florida Coastal Stock is complicated by the potential for the occasional presence, in nearshore coastal waters, of dolphins from estuarine common bottlenose dolphin stocks in Florida (Mazzoil *et al.* 2011; Caldwell 2016) as well as by possible spatiotemporal overlap with the wNA Offshore Stock of common bottlenose dolphins. Using the logistic regression described above (Garrison *et al.* 2017a), abundance estimates for the Central Florida Coastal Stock were made using sightings observed in the 0–200 m depth stratum during summer aerial surveys between 29.4°N and Ft. Pierce, Florida (26.9°N). The regression model was used to estimate the probability that each sighting during the aerial survey is of the coastal (vs. offshore) morphotype. This probability and associated estimates of uncertainty were then incorporated into the abundance estimate for the coastal morphotype within the stock range (Garrison *et al*. 2017a). The area of coastline between Fort Pierce, Florida, and Vaca Key, which lies within this stock's boundary, was not surveyed and so the resulting abundance estimates are negatively biased.

Earlier abundance estimates (>8 years old)

Aerial surveys were conducted during the summers of 2002 and 2004. Survey tracklines for the 2002 and 2004 surveys were set perpendicular to the shoreline and effort was stratified into 0–20 m and 20–40 m strata with the majority of effort in the shallow depth stratum (Garrison *et al.* 2017a). The 2002 surveys employed two observer teams operating independently on the same aircraft to estimate the probability of detection on the trackline. This

estimate was also applied to the 2004 survey to reduce bias in the resulting abundance estimate. The resulting abundance estimates from the 2002 and 2004 summer aerial surveys for the Central Florida Coastal Stock were 1,148 (CV=0.48) and 8,992 (CV=0.44), respectively (Garrison *et al.* 2017a). There were strong differences in spatial distribution between these two survey years, suggesting that the large difference in estimates was related to changes in distribution rather than population size of the stock (Garrison *et al.* 2017a). As recommended in the GAMMS II Workshop Report (Wade and Angliss 1997), these estimates are greater than eight years old and deemed unreliable and should not be used for PBR determinations. However, these estimates are included below in the assessment of trends for this stock.

Recent surveys and abundance estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters along the U.S. East Coast from southeastern Florida (26.9°N) to Sandy Hook, New Jersey (40.3°N), during the summers of 2010, 2011, and 2016 (see Garrison *et al.* (2017a) for survey design). The surveys were conducted along tracklines spaced latitudinally at 20-km intervals and oriented perpendicular to the shoreline, and covered waters from the shoreline to the continental shelf break (Garrison *et al.* 2017a).

The recent surveys were conducted using a two-team approach to develop estimates of detection probabilities using the independent observer approach with Distance analysis (Laake and Borchers 2004). The detection functions from each survey indicated a decreased probability of detection near the trackline. The sighting data were therefore "left-truncated" by analyzing only sightings occurring greater than 80 m from the trackline during the 2010 survey, 70 m during the 2011 survey, and 100 m from the trackline during the 2016 survey (see Buckland *et al.* 2001 for left-truncation methodology). The independent observer method assuming point independence was used to estimate detection probability on the trackline. This estimate accounts for the probability of detecting a marine mammal group conditional on it being available to both survey teams. Covariates that may influence detection probabilities (e.g., sea state, glare, cloud cover, visibility) were incorporated into both the mark-recapture and distance function components of the detection models (Laake and Borchers 2004; Garrison *et al.* 2017a). In addition, the probability that an observed group was of the coastal morphotype was incorporated into the abundance estimates as noted above. The resulting abundance estimates are negatively biased due to the effects of animals spending some time underwater where they are not available to the survey teams. However, due to the relatively short dive times of bottlenose dolphins (Wells *et al.* 2013) and the large group sizes, it is likely that this bias is small (Garrison *et al.* 2017a).

The abundance estimates derived from the summer 2010, 2011, and 2016 surveys were 18,221 (CV=0.74), 4,814 (CV=0.48), and 1,218 (CV=0.35). The 2016 estimate was used as the best estimate of the current population size for the stock due to possible effects from the 2013–2015 unusual mortality event. Uncertainties in the abundance estimate arise primarily from annual, and unquantified, variation in stock distribution.

Table 1. Summary of abundance estimates for the western North Atlantic Central Florida Coastal Stock of common bottlenose 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
July-August 2002	29.4°N Latitude to Ft. Pierce, Florida (26.9°N)	1,148	0.48
July-August 2004	29.4°N Latitude to Ft. Pierce, Florida (26.9°N)	8,992	0.44
July-August 2010	29.4°N Latitude to Ft. Pierce, Florida (26.9°N)	18,221	0.74
July-August 2011	29.4°N Latitude to Ft. Pierce, Florida (26.9°N)	4,814	0.48

July-August 2016	29.4°N Latitude to Ft. Pierce, Florida (26.9°N)	1,218	0.35

Minimum Population Estimate

The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. The best estimate for the Central Florida Coastal Stock is 1,218 (CV=0.35). The resulting minimum population estimate is 913.

Current Population Trend

Available surveys allow a limited analysis of trend in population size for coastal stocks of common bottlenose dolphins. A standardized analytical approach accounting for variation in survey execution and environmental conditions was used to derive unbiased abundance estimates for each survey (Garrison *et al.* 2017a). A weighted generalized linear model was used to evaluate trends in population size by stock using abundance estimates from surveys conducted in the summers of 2002, 2004, 2010, 2011, and 2016. Abundance estimates were weighted by the inverse of their standard error, which reduces the influence of less certain estimates (Neter *et al.* 1983). Stock was treated as a fixed factor, and surveys were grouped into three periods to test for long term trends in population size: 2002–2004, 2010–2011, and 2016. Period was also included as a fixed factor in the model along with the interaction between stock and period. Contrasts were specified to test for differences in abundance between periods for each stock (Garrison *et al.* 2017a). For the Central Florida Coastal Stock, the resulting mean abundance estimate for 2002–2004 was 2,108 (CV=0.99), and that for 2010–2011 was 6,777 (CV=0.63). There was no significant difference between these estimates and the estimate of 1,218 (CV=0.35) for 2016. There is limited power to detect a significant change given the high CV of the estimates, interannual variability in spatial distribution and stock abundance between 2002 and 2004, and the availability of only one recent survey (Garrison *et al.* 2017a). However, see the Strandings section for a discussion of coast-wide trends in population size.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are not known for the wNA coastal morphotype. 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 Central Florida Coastal Stock of common bottlenose dolphins is 913. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.5 because this stock is depleted. PBR for this stock of common bottlenose dolphins is 9.1.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Central Florida Coastal Stock during 2011–2015 is unknown. The mean annual fishery-related mortality and serious injury for strandings identified as fishery-related was 0.4. No additional mortality or serious injury was documented from other human-caused sources (e.g., fishery research) and therefore, the minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2015 was also 0.4 (Table 2). This is a minimum estimate because 1) not all fisheries that could interact with this stock are observed, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected and recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, and 4) the estimate includes a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). In the sections below, dolphin mortalities were assigned to a stock or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock.

Fishery Information

There are six commercial fisheries that interact, or that potentially could interact, with this stock. These include four Category II fisheries (Southeastern U.S. Atlantic shark gillnet, Southeast Atlantic gillnet, Atlantic blue crab trap/pot, and Southeastern U.S. Atlantic, Gulf of Mexico stone crab trap/pot fisheries) and two Category III fisheries

(Florida spiny lobster trap/pot, and the Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fisheries). Detailed fishery information is presented in Appendix III.

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

There have been no documented mortalities or serious injuries of common bottlenose dolphins associated with the Southeastern U.S. Atlantic Shark Gillnet or Southeast Atlantic Gillnet fisheries during 2011–2015 that could be ascribed to the Central Florida Coastal Stock (Gulak *et al.* 2012; Mathers *et al.* 2013; 2014; 2015; 2016). These fisheries target sharks and finfish in waters between North Carolina and southern Florida. The majority of fishing effort occurs in federal waters because Florida, Georgia, and South Carolina, with limited exception, prohibit the use of gillnets in state waters. These fisheries use gillnets set in a sink (anchored), stab, set, strike, or drift fashion. The Southeast Gillnet Observer Program observes these fisheries year-round (e.g., Mathers *et al.* 2016).

Trap/Pot

During 2011–2015, there were no documented mortalities or serious injuries in trap/pot gear that could be ascribed to the Central Florida Coastal Stock. The most recent documented interaction with trap/pot gear was from 2009. Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with trap/pot gear. Stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

Hook and Line (Rod and Reel)

During 2011–2015, stranding data documented three mortalities involving hook and line gear entanglement and/or ingestion that were ascribed to the Central Florida Coastal Stock. All three mortalities occurred in 2014. For two cases, available evidence from the stranding data suggested the hook and line gear contributed to the cause of death, and these two animals were included in the annual human-caused mortality and serious injury total for this stock (Table 2). For the remaining case, it could not be determined if the hook and line gear contributed to cause of death due to advanced decomposition. It should be noted that, in general, it cannot be determined if 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. The documented interactions in this gear represent a minimum known count of interactions with this fishery.

Other Mortality

Historically, there have been occasional mortalities of bottlenose dolphins during research activities (Waring *et al.* 2016); however, none were documented during 2011–2015 that could be ascribed to the Central Florida Coastal Stock. All mortalities and serious injuries from known sources for the Central Florida Coastal Stock are summarized in Table 2.

Table 2. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Central Florida Coastal Stock. For fisheries with an ongoing, systematic, federal observer program, the years sampled (Years), the type of data used (Data Type), the annual observer coverage (Observer Coverage), the observed mortality and serious injury recorded by on-board observers, and mean annual mortality and serious injury are provided. For fisheries that do not have an ongoing, systematic, federal observer program, minimum counts of mortality and serious injury based on stranding data are given. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates. NA = not applicable.

Fishery	Years	Data Type	Observer Coverage	Observed Mortality and Serious Injury	Mean Annual Mortality and Serious Injury Based on Observer Data	5-year Count Based on Stranding Data
Southeastern	2011–2015	Obs. Data	NA due to	0,0,0,0,0	Not estimated	NA

U.S. Atlantic Shark Gillnet		Logbook	uncertainty in reported effort			
Southeast Atlantic Gillnet	2011–2015	Obs. Data Logbook	NA due to uncertainty in reported effort	0,0,0,0,0	Not estimated	NA
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	NA	NA	NA	0
Hook and Line[a]	2011–2015	Stranding Data	NA	NA	NA	2
Mean Annual Mortality due to commercial fisheries (2011–2015)					**No estimate**	**0.4**
Research Takes (5-year Count)					0	
Other takes (5-year Count)					0	
Mean Annual Mortality due to research and other takes (2011–2015)					**0**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)					**0.4**	
[a] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.						

Strandings

During 2011–2015, 132 stranded common bottlenose dolphins were recovered within the range of the Central Florida Coastal Stock (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). It could not be determined if there was evidence of human interactions for 81 of these strandings, and for 48 it was determined there was no evidence of human interaction. The remaining three showed evidence of human interactions, all of which were fisheries interactions with hook and line gear. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

This stock has been impacted by three unusual mortality events (UME). Two events, one in 1987–1988 and one in 2013–2015, were attributed to morbillivirus epidemics (Lipscomb *et al.* 1994; Morris *et al.* 2015). When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988), so impacts to the Central Florida Coastal Stock alone are not known. However, it was estimated that between 10 and 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988; Eguchi 2002). The total number of stranded bottlenose dolphins from New York through central Florida (Brevard County) during the 2013–2015 UME was ~1827, including 319 from northern and central Florida (http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 8 November 2016). The southern end of Brevard County was delimited as the southernmost range of the UME, so approximately one-third of the Central Florida Coastal Stock range is found within this UME area. Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). An analysis of trends in abundance for common bottlenose dolphins coast-wide (New Jersey to Florida) indicated a statistically significant decline in population size between 2011 and 2016 (Garrison *et al*. 2017a). A weighted generalized linear model was used to evaluate trends in coast-wide population size based on aerial surveys conducted between 2002 and 2016 (see Population Size above for survey descriptions). The model included a linear term for survey year and

an interaction term to test for a difference in slope between 2002–2011 and 2011–2016. Estimates were weighted by the inverse of their standard error to reduce the influence of less certain estimates. There was no significant trend in population size between 2002 and 2011; however, there was a statistically significant (p=0.0308) change in slope between 2011 and 2016, indicating a decline in population size. The coast-wide inverse-variance weighted average estimate for coastal common bottlenose dolphins during 2011 was 41,456 (CV=0.30) while the estimate during 2016 was 19,470 (CV=0.23; Garrison *et al.* 2017a). It is possible that this apparent decline in common bottlenose dolphin abundance in coastal waters along the eastern seaboard is a result of the 2013–2015 UME. An assessment of the impacts of the 2013–2015 UME on common bottlenose dolphin stocks in the wNA is ongoing. Finally, a UME was also declared for the Indian River Lagoon, Florida, area from May to August 2008 and one dolphin from the Central Florida Coastal Stock was considered to be part of this UME (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 September 2012). The cause of this UME was undetermined.

Table 3. Strandings of common bottlenose dolphins during 2011–2015 that were ascribed to the Central Florida Coastal Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Assignments to stock were based upon the understanding of the seasonal movements of the coastal stocks. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016). Please note HI does not necessarily mean the interaction caused the animal's death.

State	**2011**			**2012**			**2013**			**2014**			**2015**		
Type	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD
Central Florida Coastal Stock	0	0	16	0	5	12	0	23	21	3[a]	15	26	0	5	6
Annual Total	16			17			44			44			11		

[a] Includes 3 fishery interactions, all of which involved ingestion of and/or entanglement in hook and line gear (mortalities).

HABITAT ISSUES

The nearshore and estuarine habitats occupied by the coastal morphotype are adjacent to areas of high human population and some are highly industrialized. Studies have examined persistent organic pollutant concentrations in bottlenose dolphins inhabiting estuaries along the Atlantic coast and have likewise found evidence of high blubber concentrations particularly near Brunswick, Georgia, Charleston, South Carolina, and Beaufort, North Carolina (Hansen *et al.* 2004; Balmer *et al.* 2011; Kucklick *et al.* 2011). Watanabe *et al.* (2000) also found high concentrations of PCBs and other organochlorine pesticides in livers of six dead, stranded dolphins collected along the Atlantic coast of Florida. The concentrations found in male dolphins from some sites exceeded toxic threshold values that may result in adverse effects on health or reproductive rates (Schwacke *et al.* 2002; Hansen *et al.* 2004; Balmer *et al.* 2011). Studies of contaminant concentrations relative to life history parameters showed higher levels of mortality in first-born offspring and higher contaminant concentrations in these calves and in primiparous females

(Wells *et al.* 2005). The exposure to environmental pollutants and subsequent effects on population health is an area of concern and active research.

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act, but the Central Florida Coastal Stock is a strategic stock due to its designation as depleted under the MMPA. From 1995 to 2001, NMFS recognized only the western North Atlantic Coastal Stock of bottlenose dolphins in the western North Atlantic, and this stock was listed as depleted as a result of a UME in 1988–1989 (64 FR 17789, April 6, 1993). The stock structure was revised in 2008, 2009, and 2010, to recognize resident estuarine stocks and migratory and resident coastal stocks. The Central Florida Coastal Stock retains the depleted designation as a result of its origin from the western North Atlantic Coastal Stock. This stock is presumed to be below OSP due to its designation as depleted. PBR for the Central Florida Coastal Stock is 9.1, so the zero mortality rate goal, 10% of PBR, is 0.9. The documented total mean annual human-caused mortality for this stock for 2011–2015 was 0.4. However, this estimate is biased low for the following reasons: 1) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 2) the estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these biases and uncertainties, there is insufficient information to determine whether or not the total fishery-related mortality and serious injury is approaching a zero mortality and serious injury rate. The impact to this stock of the 2013–2015 mid-Atlantic and the 2008 Indian River Lagoon UMEs is unknown. Analysis of trends in abundance suggests a possible decline in stock size between 2010–2011, and 2016; however, there is limited power to evaluate trends given uncertainty in stock distribution, lack of precision in abundance estimates, and a limited number of surveys.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

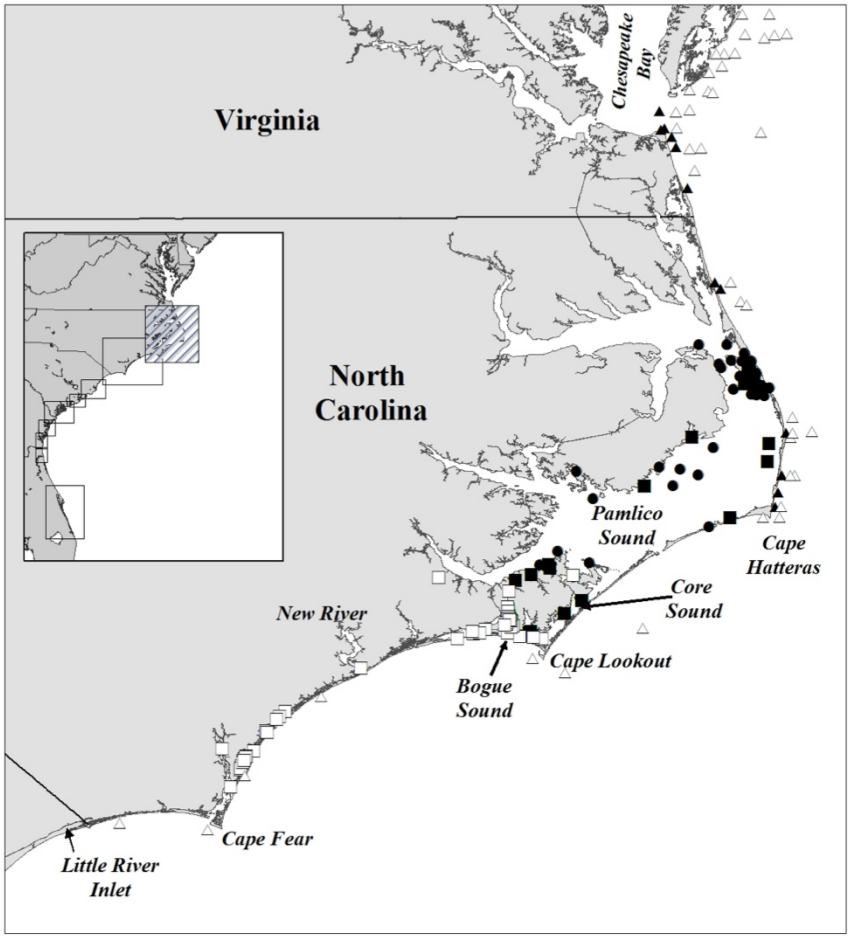

Figure 1. *The distribution of common bottlenose dolphins occupying coastal and estuarine waters in North Carolina and Virginia during July–August. Locations are shown from aerial surveys (triangles), satellite-linked telemetry (circles), and photo-identification studies (squares). Sightings assigned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

The Northern North Carolina Estuarine System (NNCES) Stock

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

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

POPULATION SIZE

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

Earlier abundance estimates (>8 years old)

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

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

Recent surveys and abundance estimates

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

Minimum Population Estimate

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the NNCES Stock during 2011–2015 is unknown. The mean annual fishery-related mortality and serious injury for observed fisheries, for strandings, and for at-sea observations identified as fishery-related ranged between 0.2 and 17.6. Additional mean annual mortality and serious injury due to other human-caused sources (fishery research, at-sea entanglements in unidentified gear) was 0.6. The minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2015 therefore ranged between 0.8 and 18.2 (Tables 1a, 1b and 1c). This range reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are detected or recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between the NNCES Stock and other common bottlenose dolphin stocks introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities were assigned to a stock

or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock.

Fishery Information

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

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the NNCES Stock. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001; Lyssikatos and Garrison 2018). This fishery is observed by the Northeast and Southeast Fisheries Observer Programs. The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006, available at http://www.nmfs.noaa.gov/pr/pdfs/fr/fr71-24776.pdf). Mortality estimates for the period (2002–2006) immediately prior to implementation of the BDTRP and 2007–2011 are available in the 2015 stock assessment report for the NNCES Stock (Waring *et al.* 2015). The current report covers the most recent available five-year estimate (NMFS 2016) for 2011–2015.

Mortality estimation for this stock is difficult because 1) observed takes are rare events, 2) the NNCES, Northern Migratory Coastal, Southern Migratory Coastal, and SNCES common bottlenose dolphin stocks overlap in coastal waters of North Carolina at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect rare bycatch events. To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2011–2015: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data; and 2) a pooled CPUE approach (where all observer data from the most recent 5 years were combined into one sample to estimate CPUE) (Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018). It should be noted that effort for internal North Carolina waters (i.e., Pamlico Sound Estuary) was not included in these analyses. Federal observer sampling rates in internal waters are low and insufficient to pool with bycatch rates coming from samples collected primarily in coastal/offshore waters. Internal waters are important habitat to the NNCES so this could lead to a downward bias in bycatch mortality estimates (see North Carolina Inshore Gillnet section below).

During the most recent five-year time period, 2011–2015, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 2.67% in state waters (0–3 miles from shore) and 5.36% in federal waters (3–200 miles from shore) (Lyssikatos and Garrison 2018). During this timeframe, two mortalities and one non-serious injury were observed. In January 2015, one mortality was observed by the NEFOP off Hatteras, North Carolina, entangled in a medium-mesh gillnet within 0.23 km of shore and was ascribed to the NNCES and Northern Migratory Coastal stocks (Lyssikatos and Garrison 2018) (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). The second mortality was observed by the SEFOP off the coast of northern North Carolina in September 2014, and this animal was ascribed to the NNCES and Southern Migratory Coastal stocks (Lyssikatos and Garrison 2018). The animal was observed entangled in a small-mesh gillnet. In February 2013, the NEFOP observed an animal entangled in a small-mesh gillnet off the coast of North Carolina that was released alive without serious injury, and, therefore, not included in the bycatch estimate (Wenzel *et al.* 2015). This animal was ascribed to the NNCES and Northern

Migratory Coastal stocks. The most recent five-year mean minimum and maximum mortality estimates (2011–2015) were 0 and 16.4 (CV=0.22) animals per year, respectively (Table 1a; Lyssikatos and Garrison 2018).

However, based on documented serious injury and mortality in this fishery from both federal observer coverage and other data sources, the mean annual minimum mortality is likely not zero. Historical stranding data have documented multiple cases of dead, stranded dolphins recovered with gillnet gear attached (Byrd *et al.* 2014; Waring *et al.* 2015). During 2011–2015, stranding data documented one mortality that was recovered in Roanoke Sound with medium-mesh gillnet gear entangled around its rostrum and flipper. The gear entangled around its flipper was attributed to the North Carolina inshore gillnet fishery and gear entangled around the rostrum was attributed to the mid-Atlantic gillnet fishery. This mortality is therefore reported in both fishery sections. The mortality is included within the annual human-caused mortality and serious injury total for the North Carolina inshore gillnet fishery (Table 1b). It may also be accounted for in the observer-based fishery bycatch estimate for the mid-Atlantic gillnet fishery. Four other dead, stranded common bottlenose dolphins were recovered with markings indicative of interaction with gillnet gear, but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. One of the four cases was ascribed to the NNCES Stock alone, one was ascribed to both the NNCES and Southern Migratory Coastal stocks, and two cases were ascribed to multiple stocks including the Northern and Southern Migratory Coastal stocks and NNCES Stock. Overall, the low level of observer coverage, rarity of observed takes, and the inability to definitively assign each observed take to stock are sources of uncertainty in the bycatch estimates for this fishery.

North Carolina Inshore Gillnet

During 2011–2015, one dead dolphin stranding, ascribed to the NNCES Stock, was recovered inshore with attached gillnet gear. This animal was recovered during 2011 in Roanoke Sound with two different types of medium-mesh gillnet gear entangled around its rostrum and flipper. The gear entangled around its flipper was attributed to the North Carolina inshore gillnet fishery, and gear entangled around the rostrum was attributed to the mid-Atlantic gillnet fishery. This mortality is therefore reported in both fishery sections. The mortality is included within the annual human-caused mortality and serious injury total for the North Carolina inshore gillnet fishery (Table 1b). It may also be accounted for in the observer-based fishery bycatch estimate for the mid-Atlantic gillnet fishery. This mortality was included in the stranding database and in the stranding totals presented in Table 2 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). The documented interaction in commercial gear represents a minimum known count of interactions with this fishery in the last five years.

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

North Carolina Long Haul Seine

There have been no documented interactions between common bottlenose dolphins of the SNCES Stock and the North Carolina long haul seine fishery during 2011–2015. The fishery includes fishing with long haul seine gear to target any species in waters off North Carolina, including estuarine waters in Pamlico and Core Sounds and their tributaries. There has not been federal observer coverage of this fishery.

Mid-Atlantic Haul/Beach Seine

During 2011–2015, stranding data documented one serious injury involving a common bottlenose dolphin and the mid-Atlantic haul/beach seine fishery in Virginia (Maze-Foley and Garrison 2017). The animal was ascribed to the Northern and Southern Migratory Coastal and NNCES stocks. The serious injury occurred during October 2014, and is included in the annual human-caused mortality and serious injury total for this stock (Table 1b) as well as in

the stranding database and in the stranding totals presented in Table 2 (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). The mid-Atlantic haul/beach seine fishery had limited observer coverage by the NEFOP in 2010–2011. No observer coverage was allocated to this fishery during 2012–2015. No estimate of bycatch mortality is available for this fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions in the last five years.

Virginia Pound Net

During 2011–2015, there were no documented mortalities or serious injuries in pound net gear of common bottlenose dolphins that could be ascribed to the NNCES Stock (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). However, during 2011–2015, two dolphin carcasses stranded with twisted twine markings indicative of interactions with Virginia pound net gear, but no gear was attached to the carcasses and it is unknown whether the interactions with the gear contributed to the death of these animals. These cases are not included in the annual human-caused mortality and serious injury total for this stock (Table 1b). Both strandings were ascribed to multiple stocks: the Northern and Southern Migratory Coastal and NNCES stocks. These two animals stranded inside estuarine waters near the mouth of the Chesapeake Bay in September and October of 2011. These mortalities were included in the stranding database and in the stranding totals presented in Table 2. Because there is no systematic observer program for the Virginia pound net fishery, no estimate of bycatch mortality is available. The overall impact of the Virginia Pound Net fishery on the NNCES Stock is unknown due to limited information on the extent to which the stock occurs within waters inside the mouth of the Chesapeake Bay.

North Carolina Roe Mullet Stop Net

During 2011–2015, stranding data documented one mortality in which a common bottlenose dolphin was found entangled and dead in a stop net (this animal was also self-reported by the fisherman per the Marine Mammal Authorization Program). This mortality occurred during November 2013, and the animal was ascribed to the NNCES and Southern Migratory Coastal stocks and is included in the annual human-caused mortality and serious injury total for this stock (Table 1b). In addition, in 2015 a dead dolphin with line markings indicative of interaction with stop net gear was recovered ~300 yards from a stop net, but it is unknown whether the interaction with gear contributed to the death of this animal, and this case is therefore not included in the annual human-caused mortality and serious injury total for this stock. This animal was ascribed to the NNCES, SNCES, and Southern Migratory Costal stocks. Both mortalities were included in the stranding database and in the stranding totals presented in Table 2 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). No estimate of bycatch mortality is available for the stop net fishery, and the documented interaction in this commercial gear represents a minimum known count of interactions with this fishery in the last five years. This fishery has not had regular, ongoing federal or state observer coverage. However, the NMFS Beaufort laboratory observed this fishery in 2001–2002 (Byrd and Hohn 2010), and Duke University observed the fishery in 2005–2006 (Thayer *et al.* 2007). Entangled dolphins were not documented during these formal observations, but two mortalities of dolphins due to entanglement in stop nets occurred in 1993 and 1999, and were documented by the stranding network in North Carolina (Byrd and Hohn 2010).

Atlantic Blue Crab Trap/Pot

During 2011–2015, stranding data documented one mortality and two serious injuries of common bottlenose dolphins entangled in trap/pot gear, all in Virginia. The mortality occurred during 2015 in commercial blue crab trap/pot gear. One serious injury occurred in 2014 in commercial blue crab trap/pot gear and one occurred in 2015 in unidentified trap/pot gear. All three cases were ascribed to the Southern Migratory Coastal and NNCES stocks, and all are included in the annual human-caused mortality and serious injury total for this stock (Table 1b). These animals were included in the stranding database and in the stranding totals presented in 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 18 May 2016 (SER) and 6 June 2016 (NER)). Because there is no observer program, it is not possible to estimate the total number of mortalities associated with crab traps/pots. However, stranding data indicate that interactions with trap/pot gear occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

North Carolina Pound Net

During 2011–2015, there were no documented mortalities or serious injuries in North Carolina pound net gear

of common bottlenose dolphins that could be ascribed to the NNCES Stock (Northeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 6 June 2016). The North Carolina pound net fishery is included within the Category III U.S. mid-Atlantic mixed species stop seine/weir/pound net fishery. The pound net is a common fishing gear used in portions of North Carolina's estuarine waters. However, the level of interaction with common bottlenose dolphins is unknown. Between 1997 and 2015, there has only been one documented mortality (2008) in North Carolina pound net gear, and this came from stranding data (Byrd *et al*. 2014). Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with this commercial gear.

Hook and Line (Rod and Reel)

During 2011–2015, stranding data included two mortalities ascribed to the NNCES Stock for which hook and line gear entanglement or ingestion were documented. For one of the mortalities, the stranding data suggested the hook and line gear interaction was not a contributing factor to cause of death (2012, North Carolina). For one mortality, it could not be determined whether the hook and line gear interaction contributed to cause of death (2011, Virginia). Neither of these mortalities is included in the annual human-caused mortality and serious injury total for this stock (Table 1b).

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

Other Mortality

There have been occasional incidental takes of common bottlenose dolphins during research activities. Two interactions with research gillnet gear were documented during 2011–2015 that were ascribed to the NNCES Stock: one mortality and one live release for which it could not be determined if the animal was seriously injured. The two animals were captured in 2012 in the same research sink gillnet targeting striped bass in North Carolina estuarine waters. Both research gillnet interactions were included in the stranding database and are included in Table 2. The mortality was included in the annual human-caused mortality and serious injury total for this stock (Table 1c).

In addition to animals included in the stranding database, during 2011–2015, there were at-sea observations in the NNCES Stock area of live common bottlenose dolphins entangled in unidentified line/fishing gear in two cases and one entangled in a sport toy flying ring (e.g., Aerobie or similar flying ring) in the third case. One observation occurred during 2011 and one during 2015, and both of these animals were considered seriously injured. Both were ascribed to the NNCES Stock alone and are included in the annual human-caused mortality and serious injury total for this stock (Table 1c). The remaining observation occurred during 2014 and it could not be determined if the animal was seriously injured (see Maze-Foley and Garrison 2017 for details on serious injury determinations). The 2014 observation was ascribed to the NNCES and SNCES stocks. All mortalities and serious injuries from known sources for the NNCES Stock are summarized in Tables 1a, 1b and 1c.

Table 1a. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern North Carolina Estuarine System Stock for the commercial mid-Atlantic gillnet fishery, which has an ongoing, systematic federal observer program. The years sampled (Years), the type of data used (Data Type), the annual percentage observer coverage (Observer Coverage), the observed serious injuries and mortalities recorded by on-board observers, and the mean annual estimate of mortality and serious injury (CV in parentheses) are provided. Minimum and maximum values are reported due to uncertainty in the assignment of mortalities to this particular stock because there is spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	Years	Data Type	Observer Coverage	Observed Serious Injury	Observed Mortality	Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data

Mid-Atlantic Gillnet	2011–2015	Obs. Data Logbook	2.0, 2.6, 3.1, 3.6, 5.6	0, 0, 0, 0, 0	0, 0, 0, 1, 1	Min=0 Max=16.4 (0.22)
Mean Annual Mortality due to the observed mid-Atlantic gillnet commercial fishery (2011–2015)						**Min=0** **Max=16.4 (0.22)**

Table 1b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern North Carolina Estuarine System Stock during 2011–2015 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury based on stranding data are given. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons. In addition, mortality due to research and other non-commercial fishery takes are included, as well as a total mean annual human caused mortality and serious injury summed from all sources.

Fishery	Years	Data Type	5-year Count Based on Stranding Data
North Carolina Inshore Gillnet	2011–2015	Limited Federal Observer and Stranding Data	1
North Carolina Long Haul Seine	2011–2015	Stranding Data	0
Mid-Atlantic Haul/Beach Seine	2011–2015	Limited Observer and Stranding Data	Min=0 Max=1
Virginia Pound Net[a]	2011–2015	Stranding Data	0
North Carolina Roe Mullet Stop Net[b]	2011–2015	Stranding Data	Min=0 Max=1
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	Min=0 Max=3
North Carolina Pound Net	2011–2015	Stranding Data	0
Hook and Line[c]	2011–2015	Stranding Data	0
Mean Annual Mortality due to unobserved commercial fisheries (2011–2015)			**Min=0.2** **Max=1.2**

[a] Pound net interactions are included if the animal was found entangled in pound net gear. Strandings with twisted twine markings indicative of interactions with pound net gear are not included within the table. See "Virginia Pound Net" text for more details.
[b] Stop Net interactions are included if the animal was found entangled in stop net gear. Stranding with line markings indicative of interaction with stop net gear are not included within the table. See "North Carolina Roe Mullet Stop Net" text for more details.
[c] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.

Table 1c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Northern North Carolina Estuarine System Stock during 2011–2015 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015) (Table 2a)	**Min=0** **Max=16.4 (0.22)**

Mean Annual Mortality due to unobserved commercial fisheries (2011–2015) (Table 2b)	**Min=0.2 Max=1.2**
Research Takes (5-year Min/Max Count)	1
Other takes (5-year Min/Max Count)	2
Mean Annual Mortality due to research and other takes (2011–2015)	**0.6**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)	**Min=0.8 Max=18.2**

Strandings

Between 2011 and 2015, 895 common bottlenose dolphins stranded along coastal and estuarine waters of North Carolina, Virginia, and Maryland that could be assigned to the NNCES Stock (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 18 May 2016 (SER) and 6 June 2016 (NER)). It could not be determined if there was evidence of human interaction (HI) for 635 of these strandings, and for 187 it was determined there was no evidence of human interaction. The remaining 73 showed evidence of human interactions (Table 2). Wells *et al.* (2015) estimated only one-third of common bottlenose dolphin carcasses in estuarine environments are recovered. In most cases, it was not possible to determine if an HI had occurred due to the decomposed state of the stranded animal. Of the 12 (of 156) estuarine strandings positive for HI, seven (58%) of them exhibited evidence of fisheries entanglement (e.g., entanglement lesions, attached gear), and two were incidental takes from research gillnet gear (described above). Evidence of human interaction does not indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

The assignment of animals to a single stock is impossible in some seasons and regions where stocks overlap, particularly in coastal waters of North Carolina and Virginia, and estuarine waters near Beaufort Inlet. Of the 895 strandings ascribed to the NNCES Stock, 156 were ascribed solely to this stock. It is likely, therefore, that the counts in Table 2 include some animals from the Southern Migratory Coastal, Northern Migratory Coastal, and SNCES stocks, and thereby overestimate the number of strandings for the NNCES Stock; those strandings that could not be definitively ascribed to the NNCES Stock were also included in the counts for these other stocks as appropriate. Stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin, therefore it is possible that some of the reported strandings were of the offshore form, though that number is likely to be low (Byrd *et al*. 2014).

This stock has also been impacted by two unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994; Morris *et al.* 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the NNCES Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988) and it was estimated that 10 to 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988; Eguchi 2002). Impacts to the NNCES Stock alone are not known. However, Scott *et al.* (1988) indicated that the observed mortalities from this event affected primarily coastal dolphins. The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was ~1827 (http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 8 November 2016). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, again suggesting that coastal stocks may have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). However, the habitat of the NNCES stock includes more nearshore coastal waters (in winter) than many estuarine stocks and so it may have been more heavily impacted by this UME than other estuarine stocks. An assessment of the impacts of the 2013–2015 UME to common bottlenose dolphin stocks in the wNA is

ongoing.

Table 2. Strandings of common bottlenose dolphins during 2011–2015 from North Carolina, Virginia, and Maryland that were ascribed to the Northern North Carolina Estuarine System (NNCES) Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Strandings observed in North Carolina are separated into those occurring within the Pamlico Sound estuarine system (Estuary) vs. coastal waters. Assignments to stock were based upon the understanding of the seasonal movements of this stock. However, particularly in coastal waters, there is likely overlap between the NNCES Stock and other common bottlenose dolphin stocks. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016 (SER) and 6 June 2016 (GAR)). Please note HI does not necessarily mean the interaction caused the animal's death.

State	**2011**			**2012**			**2013**			**2014**			**2015**		
Type	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD
North Carolina - Estuary	1[a]	3	8	3[b]	1	15	3[c]	13	45	2[d]	3	35	3[e]	3	18
North Carolina - Coastal	7[f]	20	25	12[g]	15	19	7[h]	46	69	2[i]	22	27	7[j]	15	25
Virginia[k]	5[l]	2	13	1[m]	2	12	12[n]	35	275	5[o]	3	16	3[p]	3	22
Maryland[k]	0	0	1	0	0	0	0	1	6	0	0	4	0	0	0
Annual Total	85			80			512			119			99		

[a] Includes 1 FI, an entanglement interaction with commercial gillnet gear (mortality, North Carolina inshore gillnet fishery).
[b] Includes 2 entanglement interactions in research sink gillnet gear (1 mortality; 1 released alive, could not be determined if seriously injured) and 1 FI.
[c] Includes 2 FIs.
[d] Includes 1 FI.
[e] Includes 2 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).
[f] Includes 4 FIs.
[g] Includes 9 FIs, 1 of which involved ingestion of hook and line gear (mortality), and 3 of which had markings indicative of interactions with gillnet gear (mortalities).
[h] Includes 4 FIs, 1 of which was an entanglement in a stop net (mortality, North Carolina roe mullet stop net fishery). Also includes 2 mortalities with evidence of a boat strike.
[i] Includes 2 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).
[j] Includes 6 FIs, 1 of which had markings indicative of an entanglement in a stop net (mortality, North Carolina roe mullet stop net fishery).

[k] Strandings from Virginia and Maryland include primarily waters inside Chesapeake Bay during late summer through fall. It is likely that the NNCES Stock overlaps with the Southern Migratory Coastal Stock in this area.
[l] Includes 5 FIs, 1 of which was an entanglement in hook and line gear (mortality). Two FIs were mortalities with twisted twine markings indicative of interaction with Virginia pound net gear.
[m] A mortality with evidence of a boat strike.
[n] Includes 7 FIs.
[o] Includes 3 FIs. One animal was released alive seriously injured following entanglement in commercial crab trap/pot gear. Another animal was released alive seriously injured following capture in a haul seine. Also includes 1 mortality with evidence of a boat strike.
[p] Includes 2 FIs. One FI was an entanglement in commercial blue crab trap/pot gear (mortality), and the other was an entanglement in unidentified trap/pot gear (released alive seriously injured).

HABITAT ISSUES

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

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act. However, this stock is considered strategic under the MMPA. PBR for the NNCES Stock is 7.8 and so the zero mortality rate goal, 10% of PBR, is 0.8. The documented mean annual human-caused mortality for this stock for 2011–2015 ranged between a minimum of 0.8 and a maximum of 18.2. However, these estimates are biased low for the following reasons: 1) the total U.S. human-caused mortality and serious injury for this stock cannot be directly estimated because of the spatial overlap of several stocks of common bottlenose dolphins in North Carolina and Virginia resulting in uncertainty in the stock assignment of some takes, 2) there are several commercial fisheries operating within this stock's boundaries that have little to no observer coverage, and 3) this mortality estimate incorporates a count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016). Given these uncertainties, and the fact that the maximum mean annual human-caused mortality and serious injury exceeds PBR, NMFS considers this stock strategic under the MMPA. The total fishery-related mortality and serious injury for this stock is not less than 10% of the calculated PBR and therefore, cannot be considered to be insignificant and approaching a zero mortality and serious injury rate. The impact of the 2013–2015 UME on the status of this stock is unknown. The status of this stock relative to OSP is unknown. There are insufficient data to determine the population trends for this stock.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

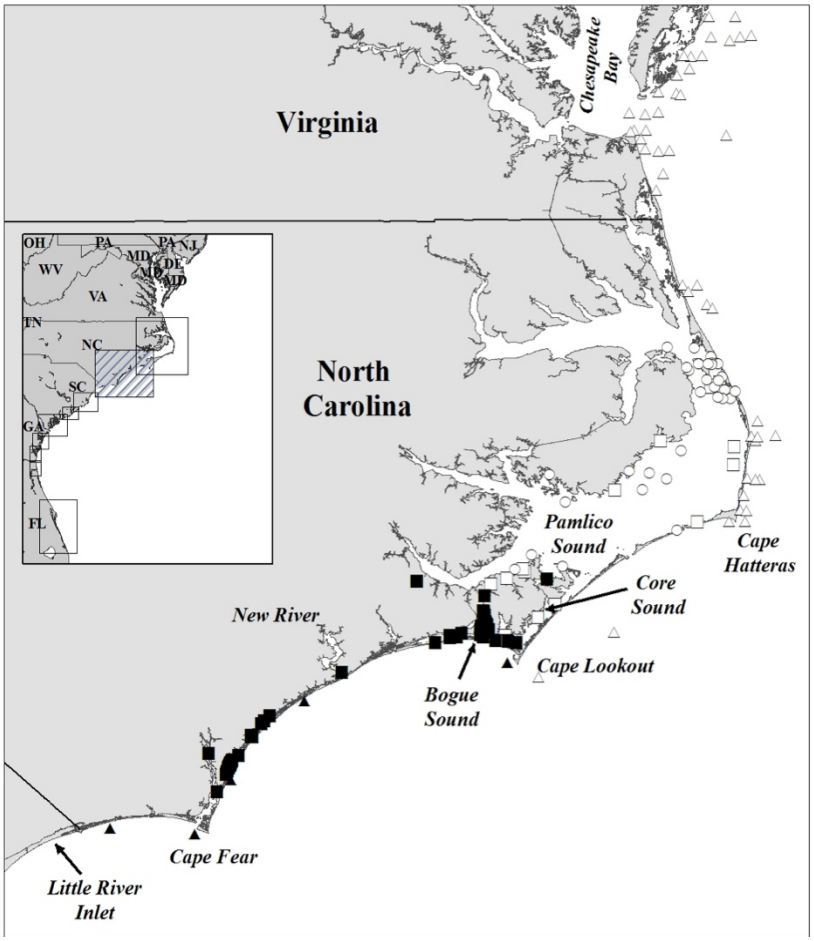

Figure 1. *The distribution of common bottlenose dolphins occupying coastal and estuarine waters in North Carolina and Virginia during the period July–September. Locations are shown from aerial surveys (triangles), satellite telemetry (circles) and photo-identification studies (squares). Sightings assigned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

The Southern North Carolina Estuarine System (SNCES) Stock is best defined as animals occupying estuarine and nearshore coastal waters (≤3 km from shore) between the Little River Inlet estuary (33.9°N), inclusive

of the estuary (near the North Carolina/South Carolina border), and the New River (34.5°N) during cold water months (best defined as January and February). Members of this stock do not undertake large-scale migratory movements. Instead, they expand their range only slightly northward during warmer months into estuarine waters and nearshore waters (≤3 km from shore) of southern North Carolina as far as central Core Sound and southern Pamlico Sound (Garrison *et al*. 2017b) (Figure 1). These movements and the range of this stock have been inferred from a combination of telemetry, photo-ID, and genetic data (Read *et al.* 2003; Rosel *et al.* 2009; Garrison *et al.* 2017b). Two animals tagged at Holden Beach, North Carolina, just south of Cape Fear during November 2004, remained within waters of southern and central North Carolina throughout the nine-month period their tags were operational (Garrison *et al.* 2017b). Eight animals tagged and/or freeze-branded near Beaufort, North Carolina, between 1995 and 2006 were documented, using long-term photo-ID studies, to have moved south and occupied estuarine and coastal waters near Cape Fear, south of the New River during cold water months (Garrison *et al.* 2017b). A photo-ID mark-recapture survey (Read *et al.* 2003) found little movement of marked animals between the northern portion of the survey area (northern Pamlico Sound, Roanoke Sound, Albemarle Sound, and Currituck Sound) and the southern portion (Southport, Cape Fear River, New River, and Bogue Sound). The authors suggested that movement patterns, differences in group sizes, and habitats are consistent with two stocks of animals occupying estuarine waters of North Carolina (Read *et al.* 2003). SNCES Stock animals have not been observed to move north of Cape Lookout in coastal waters nor into the main portion of Pamlico Sound during warm water months (Garrison *et al.* 2017b). Finally, genetic analysis of samples from animals in waters of southern North Carolina (including known SNCES animals based on live captures and strandings of unknown stock origin between Cape Lookout and the North Carolina/South Carolina border) demonstrated significant genetic differentiation from animals occupying waters from Virginia and further north and estuarine waters of South Carolina (Rosel *et al.* 2009).

The distribution of the SNCES Stock overlaps in certain seasons with several other common bottlenose dolphin stocks. During warm water months (best defined as July and August), this stock overlaps with the Northern North Carolina Estuarine System (NNCES) Stock in estuarine waters near Beaufort, North Carolina, and in southern Pamlico Sound (Garrison *et al.* 2017b). Because this stock also utilizes nearshore coastal waters along the coast of southern North Carolina, it also overlaps with the Southern Migratory Coastal Stock as this stock makes its seasonal migratory movements (Garrison *et al*. 2017b). The timing of the seasonal contraction (and expansion) of the range of the SNCES Stock, and therefore the degree of overlap with various stocks, likely occurs with some inter-annual variability related to seasonal changes in water temperatures and/or prey availability. Given the relatively small range of this stock and its seasonal movement, it is unlikely the stock contains multiple demographically independent populations; however, structure within this stock has not been investigated.

POPULATION SIZE

The current population size of the SNCES Stock is unknown because the survey data are more than eight years old (Wade and Angliss 1997).

Earlier abundance estimates (>8 years old)

Read *et al.* (2003) provided the first abundance estimate for common bottlenose dolphins occurring within the boundaries of the SNCES Stock. This estimate was based on a photo-ID mark-recapture survey of North Carolina waters inshore of the barrier islands, conducted during July 2000. Read *et al.* (2003) estimated the number of animals in the inshore waters of North Carolina occupied by the SNCES Stock at 141 (CV=0.15, 95% CI: 112–200). This estimate did not account for the portion of the stock that may have occurred in coastal waters. Summer aerial survey data from 2002 (Garrison *et al.* 2016) were therefore used to account for the portion of the stock in coastal waters. The abundance estimate for a 3-km strip from Cape Lookout to the North Carolina-South Carolina border was 2,454 (CV=0.53), yielding a total of 2,595 (CV=0.50). This estimate is likely positively biased as some animals in coastal waters may have belonged to a coastal stock.

A photo-ID mark-recapture study was conducted by Urian *et al.* (2013) in July 2006 using similar methods to those in Read *et al.* (2003) and included estuarine waters of North Carolina from, and including, the Little River Inlet estuary (near the North Carolina/South Carolina border) to, and including, Pamlico Sound. The 2006 survey also included coastal waters up to Cape Hatteras extending up to 1 km from shore. In order to estimate abundance for the SNCES Stock alone, only sightings south of 34°46’ N in central Core Sound were used. The resulting abundance estimate included a correction for the proportion of dolphins with non-distinct fins in the population. The abundance estimate for the SNCES Stock based upon photo-ID mark-recapture surveys in 2006 was 188 animals (CV=0.19, 95% CI: 118–257; Urian *et al*. 2013). This estimate is probably negatively biased as the survey covered waters only to 1 km from shore and did not include habitat in southern Pamlico Sound.

Minimum Population Estimate

The current minimum population estimate is unknown. The minimum population estimate is the lower limit of the two-tailed 60% confidence interval of the log-normally distributed best abundance estimate. This is equivalent to the 20th percentile of the log-normal distribution as specified by Wade and Angliss (1997).

Current Population Trend

A trend analysis has not been conducted for this stock. There are two abundance estimates from 2000/2002 and 2006. Methodological differences between the estimates need to be evaluated to quantify trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the SNCES Stock during 2011–2015 is unknown. The mean annual fishery-related mortality and serious injury estimated from observed fisheries and strandings identified as fishery-related ranged between 0.4 and 0.6. 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 2011–2015 ranged between 0.4 and 0.6 (Tables 1a, 1b and 1c). This range reflects several sources of uncertainty and is a minimum because 1) not all fisheries that could interact with this stock are observed and/or observer coverage is very low, 2) stranding data are used as an indicator of fishery-related interactions and not all dead animals are recovered by the stranding network (Peltier *et al*. 2012; Wells *et al*. 2015), 3) cause of death is not (or cannot be) routinely determined for stranded carcasses, 4) the estimate includes an actual count of verified human-caused deaths and serious injuries and should be considered a minimum (NMFS 2016), and 5) the spatiotemporal overlap between the SNCES Stock and other common bottlenose dolphin stocks introduces uncertainty in assignment of mortalities to stock. In the sections below, dolphin mortalities were assigned to a stock or stocks by comparing the time and geographic location of the mortality to the stock boundaries and geographic range delimited for each stock.

Fishery Information

There are six commercial fisheries that interact, or that potentially could interact, with this stock. These include the Category I mid-Atlantic gillnet fishery, four Category II fisheries (North Carolina inshore gillnet, Atlantic blue crab trap/pot, North Carolina long-haul seine, and North Carolina roe mullet stop net fisheries), and the Category III Atlantic Ocean, Gulf of Mexico, Caribbean commercial passenger fishing vessel (hook and line) fishery. Detailed fishery information is presented in Appendix III.

Mid-Atlantic Gillnet

The mid-Atlantic gillnet fishery operates along the coast from North Carolina through New York (2016 List of Fisheries) and overlaps with the SNCES Stock. North Carolina is the largest component of the mid-Atlantic gillnet fishery in terms of fishing effort and observed marine mammal takes (Palka and Rossman 2001; Lyssikatos and Garrison 2018). This fishery is observed by the Northeast and Southeast Fisheries Observer Programs. The Bottlenose Dolphin Take Reduction Team was convened in October 2001, in part, to reduce bycatch in gillnet gear. The Bottlenose Dolphin Take Reduction Plan (BDTRP) was implemented in May 2006 and resulted in changes to gillnet gear configurations and fishing practices (50 CFR 24776, April 26, 2006, available at http://www.nmfs.noaa.gov/pr/pdfs/fr/fr71-24776.pdf). Mortality estimates for the period (2002–2006) immediately prior to implementation of the BDTRP and 2007–2011 are available in the 2015 stock assessment report for the SNCES Stock (Waring *et al.* 2015). The current report covers the most recent available five-year estimate (NMFS 2016) for 2011–2015.

Mortality estimation for this stock is difficult because 1) observed takes are rare events, 2) the Northern Migratory, Southern Migratory, NNCES, and SNCES common bottlenose dolphin stocks overlap in coastal waters

off North Carolina and Virginia at different times of the year, and therefore it is not always possible to definitively assign every observed mortality, or extrapolated bycatch estimate, to a specific stock, and 3) the low levels of federal observer coverage in state waters are likely insufficient to consistently detect rare bycatch events. To help address the first problem, two different analytical approaches were used to estimate common bottlenose dolphin bycatch rates during the period 2011–2015: 1) a simple annual ratio estimator of catch per unit effort (CPUE = observed catch/observed effort) per year based directly upon the observed data; and 2) a pooled CPUE approach (where all observer data from the most recent five years were combined into one sample to estimate CPUE) (Lyssikatos and Garrison 2018). In each case, the annual reported fishery effort (defined as a fishing trip) was multiplied by the estimated bycatch rate to develop annual estimates of fishery-related mortality. Next, the two model estimates (and the associated uncertainty) were averaged, in order to account for the uncertainty in the two approaches, to produce an estimate of the mean mortality of common bottlenose dolphins for this fishery (Lyssikatos and Garrison 2018). To help address the second problem, minimum and maximum mortality estimates were calculated per stock to indicate the range of uncertainty in assigning observed takes to stock (Lyssikatos and Garrison 2018). Uncertainties and potential biases are described in Lyssikatos and Garrison (2018).

During the most recent five-year reporting period, 2011–2015, the combined average Northeast (NEFOP) and Southeast (SEFOP) Fisheries Observer Program observer coverage (measured in trips) for this fishery was 2.67% in state waters (0–3 miles from shore) and 5.36% in federal waters (3–200 miles from shore), respectively (Lyssikatos and Garrison 2018). This low level of observer coverage may result in small-sample bias in the bycatch estimate because the stock is small and PBR may be less than four (NMFS 2016). During this timeframe, no common bottlenose dolphin mortalities or serious injuries that could be attributed to the SNCES Stock were observed by the NEFOP or SEFOP. The most recent five-year mean minimum and maximum mortality estimates (2011–2015) were, therefore, both zero (Table 1a; Lyssikatos and Garrison 2018).

However, based on documented serious injury and mortality in this fishery during 2011–2015 from other data sources (see Table 1a), the mean annual minimum mortality is likely not zero. In October 2011, the stranding network recovered a dead dolphin from a fisherman who had incidentally caught it in a small-mesh gillnet in southern North Carolina during an unobserved trip targeting spot. This animal was ascribed to the Southern Migratory Coastal and SNCES stocks. In 2015, a stranded carcass was recovered with markings indicative of interaction with gillnet gear, but no gear was attached to the carcass and it is unknown whether the interaction with the gear contributed to the death of this animal. This case was ascribed to the SNCES and Southern Migratory Coastal stocks. Also in July 2015, through the Marine Mammal Authorization Program (MMAP), a fisherman self-reported an animal released alive following entanglement in his small-mesh gillnet in southern North Carolina. This animal was considered seriously injured (Maze-Foley and Garrison 2017) and was ascribed to the SNCES Stock. The 2011 stranding mortality and 2015 MMAP serious injury are included in the annual human-caused mortality and serious injury total for this stock since bycatch estimates for this stock based on observer program data were zero (Table 1a). Overall, the low level of observer coverage, rarity of observed takes, and the inability to definitively assign each observed take to stock are sources of uncertainty in the bycatch estimates for this fishery.

North Carolina Inshore Gillnet

During 2011–2015, there were no documented mortalities or serious injuries in inshore gillnet gear of common bottlenose dolphins that could be ascribed to the SNCES Stock (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). However, there were two cases documented in which a carcass stranded with markings indicative of interaction with gillnet gear but no gear was attached to the carcass and it is unknown whether the interactions with the gear contributed to the deaths of these animals. These cases occurred in 2012 and 2015, and both were ascribed to the SNCES and NNCES stocks. Neither of these mortalities are included in the annual human-caused mortality and serious injury total for this stock (Table 1b).

Current information about interactions between common bottlenose dolphins and the North Carolina inshore gillnet fishery is based solely on stranding data as no bycatch has been observed by state and federal observer programs. There was limited federal observer coverage (0.28%) of this fishery from May 2010 through March 2012, when the NMFS observed this fishery for the first time. No common bottlenose dolphin bycatch was recorded by federal observers. The low level of federal observer coverage in internal waters where the SNCES Stock resides is likely insufficient to detect bycatch events of common bottlenose dolphins if they were to occur in the inshore commercial gillnet fishery. The North Carolina Division of Marine Fisheries (NCDMF) has operated their own observer program since 2000 due to sea turtle bycatch in inshore gillnets. The NCDMF applied for and obtained an Incidental Take Permit (ITP) in September 2013 that covers gillnet fisheries in all internal state waters. This ITP

requires monitoring of gillnets statewide in internal waters with at least 7% observer coverage of large-mesh nets during spring, summer, and fall, and at least 1% observer coverage of small mesh nets during the same seasons (U.S. Dept. of Commerce 2013, Notice of permit issuance, Fed. Register 78: 57132–57133). No bycatch of common bottlenose dolphins has been recorded by state observers since they began monitoring in 2000.

Atlantic Blue Crab Trap/Pot

During 2011–2015, there were no documented mortalities or serious injuries in commercial blue crab trap/pot gear of common bottlenose dolphins that could be ascribed to the SNCES Stock (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). The most recent documented interaction was a 2009 mortality within the stranding database in which a common bottlenose dolphin was entangled in commercial blue crab trap/pot gear. The 2009 mortality was ascribed to the SNCES and Southern Migratory Coastal stocks. Because there is no systematic observer program, it is not possible to estimate the total number of interactions or mortalities associated with crab traps/pots. However, stranding data indicate that interactions occur at some unknown level in North Carolina (Byrd *et al*. 2014) and other regions of the southeast U.S. (Noke and Odell 2002; Burdett and McFee 2004).

North Carolina Long Haul Seine Fishery

There have been no documented interactions between common bottlenose dolphins of the SNCES Stock and the North Carolina long haul seine fishery during 2011–2015. The fishery includes fishing with long haul seine gear to target any species in waters off North Carolina, including estuarine waters in Pamlico and Core Sounds and their tributaries. There has not been federal observer coverage of this fishery.

North Carolina Roe Mullet Stop Net

During 2011–2015, stranding data documented no dolphins entangled in stop net gear that could be ascribed to the SNCES Stock. However, a dead stranded dolphin with line markings indicative of interaction with stop net gear was recovered in October 2015 ~300 yards from a stop net, but it is unknown whether the interaction with gear contributed to the death of this animal, and this case was not included in the annual human-caused mortality and serious injury total for this stock (Table 1b). This animal was ascribed to multiple stocks: the SNCES, NNCES, and Southern Migratory Costal stocks. This mortality is included in the stranding database and in the stranding totals presented in Table 2 (Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). No estimate of bycatch mortality is available for the stop net fishery. This fishery has not had regular, ongoing federal or state observer coverage. However, the NMFS Beaufort laboratory observed this fishery in 2001–2002 (Byrd and Hohn 2010), and Duke University observed the fishery in 2005–2006 (Thayer *et al.* 2007). Entangled dolphins were not documented during these formal observations, but two mortalities of dolphins due to entanglement in stop nets occurred in 1993 and 1999 and were documented by the stranding network in North Carolina (Byrd and Hohn 2010).

Hook and Line (Rod and Reel)

During 2011–2015, one live dolphin was observed at-sea (in 2013) entangled by a lure and monofilament line around its rostrum, deforming the rostrum and maxilla (Maze-Foley and Garrison 2017). This animal was ascribed to the SNCES stock alone and determined to have been seriously injured (Maze-Foley and Garrison 2017). This serious injury was included in the annual human-caused mortality and serious injury total for this stock (Table 1b).

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

Other Mortality

Historically, there have been occasional mortalities of common bottlenose dolphins during research activities (Waring *et al.* 2016); however, none were documented during 2011–2015 that were ascribed to the SNCES Stock.

In addition to animals included in the stranding database and the at-sea observation mentioned above (under Hook and Line), during 2011–2015, there was one at-sea observation of a live common bottlenose dolphin entangled in unidentified line (in 2014). It could not be determined if this animal was seriously injured or not (Maze-Foley and Garrison 2017). The animal was ascribed to the SNCES and NNCES stocks. All mortalities and serious injuries from known sources for the SNCES Stock are summarized in Tables 1a, 1b and 1c.

Table 1a. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern North Carolina Estuarine System Stock for the commercial mid-Atlantic gillnet fishery, which has an ongoing, systematic federal observer program. The years sampled (Years), the type of data used (Data Type), the annual percentage observer coverage (Observer Coverage), the observed serious injuries and mortalities recorded by on-board observers, and the mean annual estimate of mortality and serious injury (CV in parentheses) are provided. Counts of mortality and serious injury based on stranding data and fisherman self-reported takes via the Marine Mammal Authorization Program are also given for this fishery since bycatch estimates for this stock based on observer program data were zero. Minimum and maximum values are reported due to uncertainty in the assignment of mortalities to this particular stock because there is spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons.

Fishery	Years	Data Type	Observer Coverage	Observed Serious Injury	Observed Mortality	Mean Annual Estimated Mortality and Serious Injury (CV) Based on Observer Data
Mid-Atlantic Gillnet	2011–2015	Obs. Data Logbook	2.0, 2.6, 3.1, 3.6, 5.6	0, 0, 0, 0, 0	0, 0, 0, 0, 0	0
5-year Count Based on Stranding Data and Fisherman Self-Reported Takes via the Marine Mammal Authorization Program						Min=1 Max=2
Mean Annual Mortality due to the observed mid-Atlantic gillnet commercial fishery (2011–2015)						**Min=0.2** **Max=0.4**

Table 1b. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern North Carolina Estuarine System Stock during 2011–2015 from commercial fisheries that do not have ongoing, systematic federal observer programs. Counts of mortality and serious injury are based on stranding data. Minimum and maximum values are reported in individual cells when there is uncertainty in the assignment of mortalities to this particular stock due to spatial overlap with other common bottlenose dolphin stocks in certain areas and seasons. In addition, mortality due to research and other non-commercial fishery takes are included, as well as a total mean annual human caused mortality and serious injury summed from all sources.

Fishery	Years	Data Type	5-year Count Based on Stranding Data
North Carolina Inshore Gillnet[a]	2011–2015	Limited Federal Observer and Stranding Data	0
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	0

North Carolina Long Haul Seine	2011–2015	Stranding Data	0
North Carolina Roe Mullet Stop Net[b]	2011–2015	Stranding Data	0
Hook and Line[c]	2011–2015	Stranding Data and At-Sea Observation	1
Mean Annual Mortality due to unobserved commercial fisheries (2011–2015)			**0.2**

[a] North Carolina inshore gillnet interactions are included if the animal was found entangled in gillnet gear. Strandings with line markings indicative of interaction with gillnet gear are not included within the table. See "North Carolina Inshore Gillnet" text for more details.
[b] Stop net interactions are included if the animal was found entangled in stop net gear. Stranding with line markings indicative of interaction with stop net gear are not included within the table. See "North Carolina Roe Mullet Stop Net" text for more details.
[c] Hook and line interactions are counted here if the available evidence suggested the hook and line gear contributed to the cause of death. See "Hook and Line" text for more details.

Table 1c. Summary of the incidental mortality and serious injury of common bottlenose dolphins of the Southern North Carolina Estuarine System Stock during 2011–2015 from all sources, including observed commercial fisheries, unobserved commercial fisheries, and research and other takes. See the Annual Human-Caused Mortality and Serious Injury section for biases and limitations of mortality estimates.

Mean Annual Mortality due to the observed commercial mid-Atlantic gillnet fishery (2011–2015) (Table 2a)	**Min=0.2** **Max=0.4**
Mean Annual Mortality due to unobserved commercial fisheries (2011–2015) (Table 2b)	**0.2**
Research Takes (5-year Min/Max Count)	0
Other takes (5-year Min/Max Count)	0
Mean Annual Mortality due to research and other takes (2011–2015)	**0**
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)	**Min=0.4** **Max=0.6**

Strandings

Between 2011 and 2015, 80 common bottlenose dolphins stranded along coastal and estuarine waters of North Carolina that could be ascribed to the SNCES Stock (Table 2; Southeast Regional Marine Mammal Stranding Network; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). It could not be determined if there was evidence of human interaction for 26 of these strandings, and for 37 it was determined there was no evidence of human interaction. The remaining 17 showed evidence of human

interactions, including 16 fisheries interactions (FIs). One FI occurred in 2011 and involved a dolphin entangled in gillnet gear and reported to the stranding network, who recovered the carcass. The gillnet was targeting spot, and this take is included under the mid-Atlantic gillnet fishery (Table 1a). The remaining FIs could not be assigned to a specific fishery. It should be recognized that evidence of human interaction does not always indicate cause of death, but rather only that there was evidence of interaction with a fishery (e.g., line marks, net marks) or evidence of a boat strike, gunshot wound, mutilation, etc., at some point. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise to recognize signs of human interaction varies among stranding network personnel.

As described in the Stock Definition and Geographic Range section, there is spatiotemporal overlap between the SNCES Stock and the Southern Migratory Coastal Stock in coastal waters of southern North Carolina when the Southern Migratory Coastal Stock makes its seasonal migrations north and south. There is also overlap in waters from southern Pamlico Sound to Bogue Sound with the NNCES Stock during late summer and early fall. Therefore, assignment of animals to a single stock is impossible in some seasons and regions. Of the 80 strandings ascribed to the SNCES Stock, 12 were ascribed solely to this stock and two of those were identified as having evidence of fishery interaction. It is likely that the counts in Table 2 include some animals from the Southern Migratory Coastal and/or NNCES Stock and therefore overestimate the number of strandings for the SNCES Stock; those strandings that could not be solely ascribed to the SNCES Stock were also included in the counts for these other stocks as appropriate. In addition, stranded carcasses are not routinely identified to either the offshore or coastal morphotype of common bottlenose dolphin. Therefore, it is possible that some of the reported strandings recorded along the coast were of the offshore form, although that number is likely to be low (Byrd *et al*. 2014).

This stock has been impacted by two unusual mortality events (UMEs), one in 1987–1988 and one in 2013–2015, both of which have been attributed to morbillivirus epidemics (Lipscomb *et al.* 1994; Morris *et al.* 2015). Both UMEs included deaths of dolphins in spatiotemporal locations that apply to the SNCES Stock. When the impacts of the 1987–1988 UME were being assessed, only a single coastal stock of common bottlenose dolphin was thought to exist along the U.S. eastern seaboard from New York to Florida (Scott *et al.* 1988) and it was estimated that 10 to 50% of the coast-wide stock died as a result of this UME (Scott *et al.* 1988; Eguchi 2002). Impacts to the SNCES Stock alone are not known. However, Scott *et al.* (1988) indicated that the observed mortalities from this event affected primarily coastal rather than estuarine dolphins. The total number of stranded common bottlenose dolphins from New York through North Florida (Brevard County) during the 2013–2015 UME was ~1827 (http://www.nmfs.noaa.gov/pr/health/mmume/midatldolphins2013.html, accessed 8 November 2016). Most strandings and morbillivirus positive animals have been recovered from the ocean side beaches rather than from within the estuaries, suggesting that coastal stocks may have been more impacted by this UME than estuarine stocks (Morris *et al.* 2015). However, the habitat of the SNCES Stock includes more nearshore coastal waters than many estuarine stocks and so it may have been more heavily impacted by this UME than other estuarine stocks. An assessment of the impacts of the 2013–2015 UME to common bottlenose dolphin stocks in the wNA is ongoing.

Table 2. Strandings of common bottlenose dolphins during 2011–2015 from North Carolina that were ascribed to the Southern North Carolina Estuarine System (SNCES) Stock, including the number of strandings for which evidence of human interaction (HI) was detected and number of strandings for which it could not be determined (CBD) if there was evidence of HI. Strandings observed in North Carolina are separated into those occurring within estuaries vs. coastal waters. Assignments to stock were based upon the understanding of the seasonal movements of this stock. However, particularly in coastal waters, there is likely overlap between the SNCES Stock and other common bottlenose dolphin stocks. Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016). Please note HI does not necessarily mean the interaction caused the animal's death.

State	2011	2012	2013	2014	2015

Type	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD	HI Yes	HI No	CBD
North Carolina - Coastal	5[a]	4	3	3[b]	2	4	3[c]	15	9	0	9	1	3[d]	4	1
North Carolina - Estuary	0	1	1	0	0	1	2[e]	1	3	0	1	3	1[f]	0	0
Annual Total	14			10			33			14			9		

[a] Includes 4 FIs, 1 of which was an entanglement interaction with commercial gillnet gear (mortality, mid-Atlantic gillnet fishery).

[b] Includes 3 FIs, 1 of which had markings indicative of interactions with gillnet gear (mortality).

[c] Includes 3 FIs.

[d] Includes 3 FIs, 1 of which had markings indicative of an entanglement in a stop net (mortality, North Carolina roe mullet stop net fishery), and 1 of which had markings indicative of interactions with gillnet gear (mortality).

[e] Includes 2 FIs.

[f] Includes 1 FI, in which animal had markings indicative of interactions with gillnet gear (mortality).

HABITAT ISSUES

This stock inhabits areas with significant drainage from agricultural, industrial, and urban sources (Lindsey *et al.* 2014), and as such is exposed to contaminants in runoff from those sources. The blubber of 47 common bottlenose dolphins captured and released near Beaufort, North Carolina, contained levels of organochlorine contaminants, including DDT and PCBs, sufficiently high to warrant concern for the health of dolphins, and seven had unusually high levels of the pesticide methoxychlor (Hansen *et al.* 2004). Schwacke *et al.* (2002) found that the levels of polychlorinated biphenyls (PCBs) observed in female common bottlenose dolphins near Beaufort, North Carolina, would likely impair reproductive success, especially of primiparous females.

STATUS OF STOCK

Common bottlenose dolphins in the western North Atlantic are not listed as threatened or endangered under the Endangered Species Act. NMFS considers the SNCES Stock to be a strategic stock under the MMPA. An unbiased abundance estimate for this stock is unavailable, but the stock size is likely less than 200 given the restricted range of the stock and the best available abundance estimate (Urian *et al.* 2013). An annual average of 3.2 carcasses showing evidence of fishery interaction (primarily gillnet interactions, Table 2) are recovered within this stock's range. This high number is of concern, particularly in light of Wells *et al.* (2015) who estimated that only one-third of common bottlenose dolphin carcasses in estuarine environments are recovered. This suggests that annual human-caused mortality could approach 16 animals per year. While it is likely that not every dolphin with evidence of fishery interaction died as a result of that interaction, only five mortalities per year would place the stock at or above PBR if the minimum abundance (Nmin) is anything less than 500. Therefore, given the likely small stock size and the probable negative bias in the estimated total human-caused mortality, this stock is listed as strategic. The status of this stock relative to OSP is unknown. There is insufficient information available to determine whether the total fishery-related mortality and serious injury for this stock is insignificant and approaching a zero mortality and serious injury rate. The abundance of this stock is currently unknown and there are insufficient data to determine

population trends for this stock. The impact of the 2013–2015 UME to the status of this stock is unknown.

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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 and southern Bay of Fundy region, generally in waters less than 150 m deep (Gaskin 1977; Kraus *et al.* 1983; Palka 1995), with a few sightings 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. They are seen from the coastline to deep waters (>1800 m; Westgate *et al.* 1998), although the majority of the population is found over the continental shelf. 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. 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).

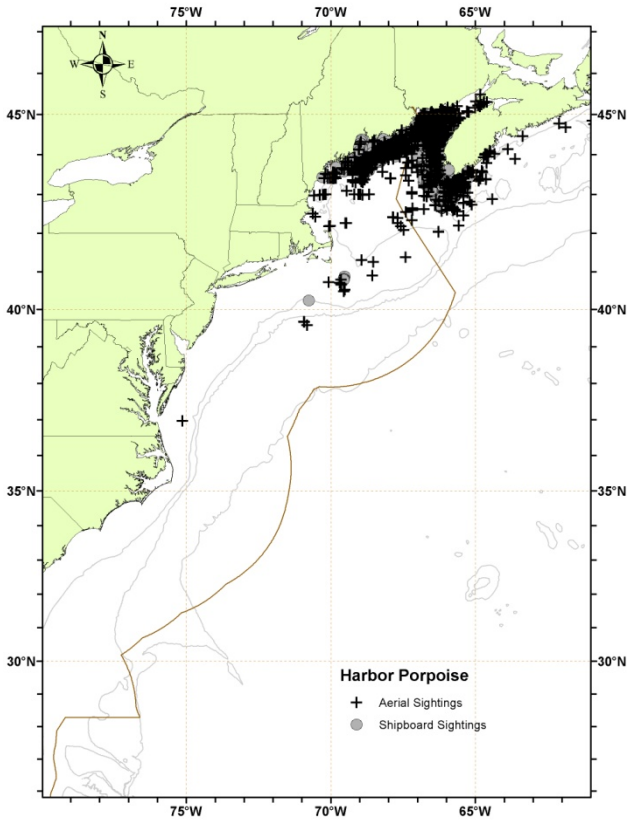

Figure 1. *Distribution of harbor porpoises from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, and 2011 and DFO's 2007 TNASS survey. Isobaths are the 100-m, 1000-m, and 4000-m depth contours.*

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.

POPULATION SIZE

The best current abundance estimate of the Gulf of Maine/Bay of Fundy harbor porpoise stock is from the 2011 survey: 79,883 (CV=0.32). Key uncertainties include: 1) the surveyed area may not have covered the entire area of the stock's habitat at the appropriate time of the year, and 2) the current abundance estimate did not account for availability bias due to submergence of animals. Without a correction for availability bias, the abundance estimate is expected to be biased low. Since the dive times of harbor porpoises are relatively short (~ 4 minutes), it is expected the bias is not large.

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.

Table 1. Summary of recent abundance estimates for the Gulf of Maine/Bay of Fundy harbor porpoise (*Phocoena phocoena phocoena*) by month, year, and area covered during each abundance survey and the resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV

Table 1. Summary of recent abundance estimates for the Gulf of Maine/Bay of Fundy harbor porpoise (*Phocoena phocoena phocoena*) by month, year, and area covered during each abundance survey and the resulting abundance estimate (N_{best}) and coefficient of variation (CV).

Month/Year	Area	N_{best}	CV
Jul-Aug 2011	Central Virginia to lower Bay of Fundy	79,883	0.32

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 79,883 (CV=0.32). The minimum population estimate for the Gulf of Maine/Bay of Fundy harbor porpoise is 61,415.

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).

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 61,415. 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 706.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual estimated average human-caused mortality and serious injury is 307 harbor porpoises per year (CV=0.16) from U.S. fisheries using observer data, of which 3 harbor porpoises were seriously injured. As the current abundance estimate is only for animals in U.S. waters, Canadian bycatch is not included in the human-caused mortality estimate.

A key uncertainty is the potential that the observer coverage in the Mid-Atlantic gillnet was not representative of the fishery during all times and places, since the observer coverage was relatively low, 0.02 – 0.06.

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 2014, 2015, 2016, Orphanides and Hatch 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.

Mid-Atlantic Gillnet

Annual bycatch is estimated using ratio estimator techniques (Hatch and Orphanides 2014, 2015, 2016; Orphanides and Hatch 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

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. Fishery-related bycatch rates were estimated using an annual stratified ratio-estimator (Chavez-Rosales 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.

CANADA

No current estimates exist, but harbor porpoise interactions have been documented in the Bay of Fundy sink gillnet fishery and in herring weirs. Trippel and Shepherd (2004) reported 52 observed harbor porpoise mortalities between the years 1998-2001 in the lower Bay of Fundy demersal gillnet fishery. Total estimated bycatch for that period was 171 animals. That fishery has declined since 2001 (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]	Observed Serious Injury [c]	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Combined Serious Injury	Estimated CVs	Mean Annual Combined Mortality
U.S.										
Northeast Sink Gillnet	11-15	Obs. Data, Weighout, Trip Logbook	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	66, 34, 20, 28, 23	0, 0, 0, 0, 0	273, 277, 399, 128, 177	273, 277, 399, 128, 177	.20, .59, .33, .27, .28	251 (0.18)
Mid-Atlantic Gillnet	11-15	Obs. Data Weighout	.02, .02, .03, .05, .06	0, 0, 0, 0, 0	11, 2, 1, 1, 2	0, 0, 0, 0, 0	123, 63, 19, 22, 33	123, 63, 19, 22, 33	.41, .83, 1.06, 1.03, 1.16	52 (0.34)
Northeast Bottom Trawl	11-15	Obs. Data	.26, .17, .15, .17,	1, 0, 0, 0, 0	1, 0, 1, 1, 4	3, 0, 0, 0, 0	2.9, 0, 7, 5.5, 3.71	5.9, 0, 7, 5.5, 3.71	.58, 0, .98, .86, .49	4.4 (0.42)

		Weighout	.19							
U.S. TOTAL	2011-2015									307 (0.16)

NA = Not available.

a. Observer data (Obs. Data) are used to measure bycatch rates; U.S. data are collected by the Northeast Fisheries Science Center (NEFSC) Sea Sampling Program and At-Sea Monitoring Program; Canadian data are collected by DFO. NEFSC collects Weighout (Weighout) landings data that are used as a measure of total effort for the U.S. gillnet fisheries. The Canadian DFO catch and effort statistical system collected the total number of trips fished by the Canadians (Can. Trips), which was the measure of total effort for the Canadian groundfish gillnet fishery. Mandatory vessel trip report (VTR) (Trip Logbook) data are used to determine the spatial distribution of fishing effort in the northeast sink gillnet fishery. Observed mortalities from herring weirs are collected by a cooperative program between fishermen and Canadian biologists (Coop. Data).

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 since 2011 using new guidelines and include both at-sea monitor and traditional observer data (Josephson *et al*. 2017)

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. In recent years 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.

During 2011, 164 harbor porpoises were reported stranded on Atlantic U.S. beaches. Of these, nine stranding mortalities were reported as having signs of human interaction, three of which were reported to be fishery interactions.

During 2012, 45 harbor porpoises were reported stranded on Atlantic U.S. beaches. Of these, 4 stranding mortalities were reported as having signs of human interaction, one of which was reported to be a fishery interaction.

During 2013, 102 harbor porpoises were reported stranded on Atlantic U.S. beaches. Of these, 9 stranding mortalities were reported as having signs of human interaction, three of which were reported to be fishery interactions.

During 2014, 39 harbor porpoises were reported stranded on Atlantic U.S. beaches. Of these, 5 stranding mortalities were reported as having signs of human interactions, one of which was reported to have been a fishery interaction.

During 2015, 44 harbor porpoises were reported stranded on Atlantic U.S. beaches. Of these, 2 stranding mortalities were reported as having signs of human interactions, neither of which were fishery interactions.

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 4. Harbor Porpoise (*Phocoena phocoena phocoena*) reported strandings along the U.S. and Canadian Atlantic coast, 2011-2015.

Area	Year					**Total**
	2011	2012	2013	2014	2015	
Maine[a,f]	15	7	7	5	2	**36**
New Hampshire	1	3	1	1	0	**6**
Massachusetts[a, d, e,f, g]	102	25	40	22	18	**207**
Rhode Island [b, d, h]	4	0	3	0	2	**9**
Connecticut [f]	0	0	1	0	0	**1**
New York[c,d, f]	11	3	15	1	3	**33**
New Jersey [f, g]	1	2	8	4	2	**17**
Delaware	0	0	2	0	0	**2**
Maryland	0	1	3	0	0	**4**
Virginia[d, g]	2	2	15	3	3	**25**
North Carolina[h]	28	2	7	11	14	**62**
TOTAL U.S.	**164**	**45**	**102**	**39**	**44**	**394**
Nova Scotia/Prince Edward Island[i]	13	6	21	9	13	**62**
Newfoundland and New Brunswick[i,j]	0	0	3	0	2	**5**
GRAND TOTAL	**177**	**51**	**126**	**48**	**59**	**461**

a. In Massachusetts in 2011, 5 animals were released alive and one taken to rehab. One Maine animal was taken to rehab in 2012. Three Massachusetts live strandings were taken to rehab in 2013 and 1 Maine animal was released alive.

b. In Rhode Island in 2011, one animal classified as human interaction due to fluke amputation.

c. One of the 2012 New York strandings classified as human interaction due to interaction with marine debris.

d. Nine total HI cases in 2011; 5 in Massachusetts, 1 in Rhode Island, 2 in New York and 1 in Virginia. Two of these Massachusetts animals and the Virginia animal were fishery interactions.

e. Four HI cases in 2012. One of these was a fishery interaction (Massachusetts).

f. 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.

g. 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.

h. Two HI cases in 2015: 1 in Rhode Island and 1 in North Carolina

i. Data supplied by Nova Scotia Marine Animal Response Society (pers. comm.). One of the 2012 animals trapped in mackerel net. Not included in count for 2014 are at least 8 animals released alive from weirs. One of the 2015 animals a suspected fishery interaction.

j. (Ledwell and Huntington 2011, 2012a, 2012b, 2013, 2014, 2015).

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, including 13 (4 released alive) in 2011, 6 in 2012, 21 in 2013, 9 in 2014 and 15 in 2015; Table 4).

Three dead stranded harbor porpoises were reported in 2013 by the Newfoundland and Labrador Whale Release and Strandings Program, and 0 in 2011, 2012, 2014 and 2015 (Ledwell and Huntington 2011, 2012a, 2012b, 2013; 2014; Table 3).

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.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The harbor seal (*Phoca 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 New Jersey from September through late May (Schneider and Payne 1983; Schroeder 2000). Scattered sightings and strandings have been recorded as far south as Florida (NOAA National Marine Mammal Health and Stranding Response Database, accessed 08 October 2015). A general southward movement from the Bay of Fundy to southern New England waters occurs in autumn and early winter (Rosenfeld *et al.* 1988; Whitman and Payne 1990; Jacobs and Terhune 2000). A northward movement from southern New England to Maine and eastern Canada occurs prior to the pupping season, which takes place from mid-May through June along the Maine coast (Richardson 1976; Wilson 1978; Whitman and Payne 1990; Waring *et al*. 2006). Earlier research identified no pupping areas in southern New England (Payne and Schneider 1984); however, more recent anecdotal reports suggest that some pupping is occurring at high-use haulout sites off Manomet, Massachusetts and the Isles of Shoals, Maine.

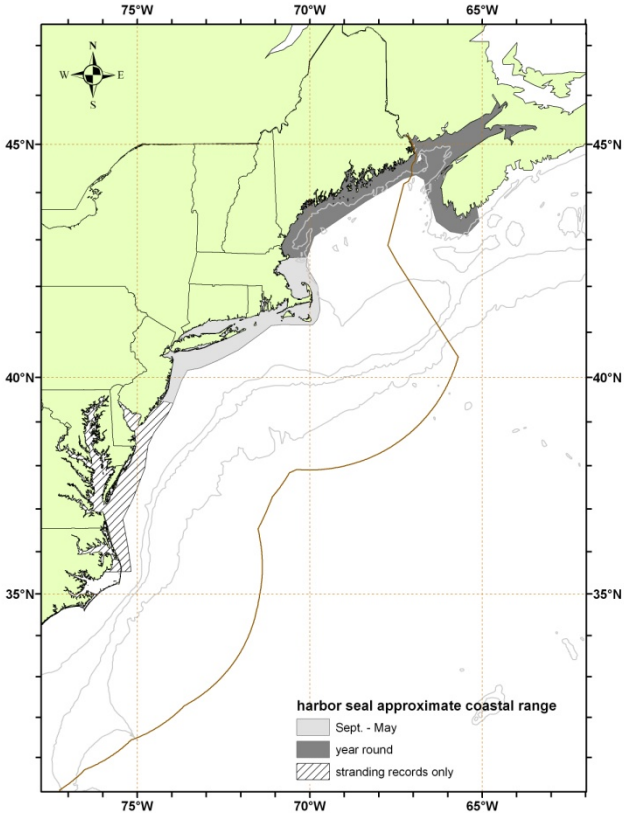

Figure 1. *Approximate coastal range of harbor seals, and distribution of harbor seal sightings at sea from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, and 2011. Isobaths are the 100-m, 1000-m, and 4000-m depth contours.*

Prior to the spring 2001 live-capture and radio-tagging of adult harbor seals (Waring *et al.* 2006), 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 2001 study established that adult animals also made this migration. Seventy-five percent (9/12) of the seals tagged in March in Chatham Harbor were detected at least once during the May/June 2001 abundance survey along the Maine coast (Gilbert *et al.* 2005; Waring *et al.* 2006). Similar findings were made in spring 2011 and 2012 (Waring *et al.* 2015).

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). 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.

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

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. The resultant estimate of harbor seal abundance was 5,834 (CV= 0.15; Waring *et al.* 2015).

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 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).

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 uncertainity 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. There has been very little systematic research conducted on fine-scale changes in habitat use, particularly in relation to the sympatric population of gray seals.

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 2011-2015 the total human caused mortality and serious injury to harbor seals is estimated to be 368 per year. The average was derived from two components: 1) 356 (CV=0.11; Table 2) from 2011–2015 observed fisheries; 2) 12 from 2011–2015 non-fishery-related, human interaction stranding and direct interaction mortalities (NOAA National Marine Mammal Health and Stranding Response Database, accessed 13 September 2016.

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. Williams (1999) aged 261 harbor seals caught in this fishery from 1991 to 1997, and 93% were juveniles (i.e., less than four years old). Revised serious injury guidelines were applied for this period (Josephson *et al.* 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. Analysis methodology and results can be found in Orphanides (2013), Hatch and Orphanides (2014, 2015, 2016), and Orphanides and Hatch (2017).

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 (2013), Hatch and Orphanides (2014, 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.

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 (Chavez-Rosales *et al*. 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 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 2011–2015 is calculated as 0.8 animal (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.

Mid-Atlantic Mid-water Trawl Fishery (Including Pair Trawl)

Two harbor seal mortalities were observed in this fishery in 2015. 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 2011–2015 is calculated as 0.4 animals (2 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 3 harbor seals were captured and released alive in 2011, 1 in 2012, 1 in 2013, and 0 in 2014 and 2015. In addition, 8 seals of unknown species were captured and released alive in 2011, 0 in 2012–2014, and 2 in 2015. One harbor seal and two unidentified seals were designated as serious injuries/mortalities in 2011, based on fisheries monitoring logs (Josephson *et al*. 2017). 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 2011–2015 is calculated as 0.2 animals (1 animal/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.

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[d]	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Annual Mortality
Northeast[c] Sink Gillnet	11–15	Obs. Data, Weighout, Logbooks	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	91, 37, 22, 59, 87	0, 0, 0, 0, 0	343, 252, 142, 390, 474	343, 252, 142, 390, 474	.19, .26, .31, .39, .17	321 (0.12)
Mid-Atlantic Gillnet	11–15	Obs. Data, Weighout	.02, .02, .03, .05, .06	0, 0, 0, 0, 0	2, 0, 0, 1, 5	0, 0, 0, 0, 0	21, 0, 0, 19, 48	21, 0, 0, 19, 48	.67, 0, 0, 1.06, .52	18 (0.40)
Northeast Bottom Trawl[d]	11–15	Obs. Data, Weighout	.26, .17, .15, .17, .19	0, 0, 0, 0, 0	3, 1, 1, 2, 0	0, 0, 0, 0, 0	9, 3, 4, 11, 0	9, 3, 4, 11, 0	.46, .81, .89, .63, 0	5.4 (.34)

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).

Mid-Atlantic Bottom Trawl	11–15	Obs. Data Dealer	.08, .05, .06, .08, 09	0, 0, 0, 0, 0	0, 3, 1, 2, 1	0, 0, 0, 0, 0	0, 23, 11, 10, 7	0, 23, 11, 10, 7	0, .96, .96, .95, 1	10 (.53)
Northeast Mid-water Trawl - Including Pair Trawl	11–15	Obs. Data Weighout Trip Logbook	.41, .45, .37, .42, 0.08	0, 0, 0, 0, 0	0, 1, 0, 1, 2	0, 0, 0, 0, 0	0, na, 0, na, na	0, na, 0, na, na	0, na, 0, na, na	0.8 (na)[c]
Mid-Atlantic Mid-water Trawl - Including Pair Trawl	11–15	Obs. Data Weighout Trip Logbook	.41, .21, .07, .05, .03	0, 0, 0, 0, 0	0, 0, 0, 0, 2	0, 0, 0, 0, 0	0, 0, 0, 0, na	0, 0, 0, 0, na	0, 0, 0, 0, na	0.4 (na)[c]
Herring Purse Seine	11–15	Obs. Data	.33, .17, .17, .08, .08	1, 0, 0, 0, 0	0, 0, 0, 0, 0	na, 0, 0, 0, 0	0, 0, 0, 0, 0	na, 0, 0, 0, 0	na, 0, 0, 0, 0	0.2 (na)[c]
TOTAL										356 (0.11)

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 2011-2015 includes samples collected from traditional fisheries observers in addition to fishery monitors through the Northeast Fisheries Observer Program (NEFOP).

c. Analyses of bycatch mortality attributed to the mid-water trawl fisheries and herring purse seine fishery for 2011–2015 have not been generated.

d. Serious injuries were evaluated for the 2011–2015 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2017)

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.

Other sources of harbor seal mortality include human interactions, storms, abandonment by the mother, disease (Anthony *et al.* 2012), and predation (Katona *et al.* 1993; NMFS unpublished data; Jacobs and Terhune 2000). Mortalities caused by human interactions include research mortalities, boat strikes, fishing gear interactions, oil spill/exposure, harassment, and shooting.

Harbor seals strand each year throughout their migratory range. Stranding data provide insight into some of

these sources of mortality. From 2011 to 2015, 1,404 harbor seal stranding mortalities were reported between Maine and Florida (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 September 2016). Seventy-three (5.2%) of the dead harbor seals stranded during this five-year period showed signs of human interaction (20 in 2011, 9 in 2012, 15 in 2013, 11 in 2014, and 18 in 2015), with 11 (0.8%) having some sign of fishery interaction (2 in 2011, 2 in 2012, 3 in 2013, 2 in 2014, and 2 in 2015). Five harbor seals during this period were reported as having been shot. 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.

An Unusual Mortality Event (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.

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).

Table 3. Harbor seal (*Phoca vitulina vitulina*) stranding mortalities along the U.S. Atlantic coast (2011-2015) with subtotals of animals recorded as pups in parentheses[a].

State	2011	2012	2013	2014	2015	Total
Maine[a]	147 (115)	131 (101)	99 (74)	127 (94)	73 (47)	577
New Hamphire[a]	77 (63)	24 (18)	16 (6)	38 (22)	56 (43)	211
Massachusetts[a]	133 (80)	54 (35)	95 (39)	58 (15)	81 (24)	421
Rhode Island	7 (0)	14 (0)	9 (3)	7 (1)	8 (0)	45
Connecticut	1 (1)	1 (1)	2 (1)	0	2 (1)	6
New York	17 (0)	14 (1)	11 (2)	13 (4)	21 (0)	76
New Jersey	10 (0)	7 (0)	4 (0)	2 (1)	9 (4)	32
Delaware	0	0	0	3 (0)	1 (0)	4
Maryland	1 (0)	0	1 (0)	2 (0)	0	4
Virginia	4 (0)	0	5 (0)	2 (0)	1 (0)	12
North Carolina	2 (0)	2 (0)	3 (0)	3 (1)	5 (2)	15
South Carolina	0	0	0	1 (0)	0	1

Total	**399**	**247**	**245**	**256**	**257**	**1404**
Unspecified seals (all states)	63	28	25	38	31	147
a. Unusual Mortality event (UME) declared for harbor seals in southern Maine to northern Massachusetts in 2011.						

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 2011–2015 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 western North Atlantic stock is equivalent to the Northwest Atlantic population, and ranges from New Jersey to Labrador (Davies 1957; Mansfield 1966; Katona *et al.* 1993; Lesage and Hammill 2001). This stock is separated by geography, differences in the breeding season, and mitochondrial and nuclear DNA variation from the northeastern Atlantic stocks (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). Outside the breeding period, there is overlap in the distribution of animals from the three colonies (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). 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 that recolonization by Canadian gray seals is the source of the U.S. population. 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) provide further evidence that there is movement of individuals between the U.S. and Canada. However, the percentage of time that individuals are resident in U.S. waters is unknown.

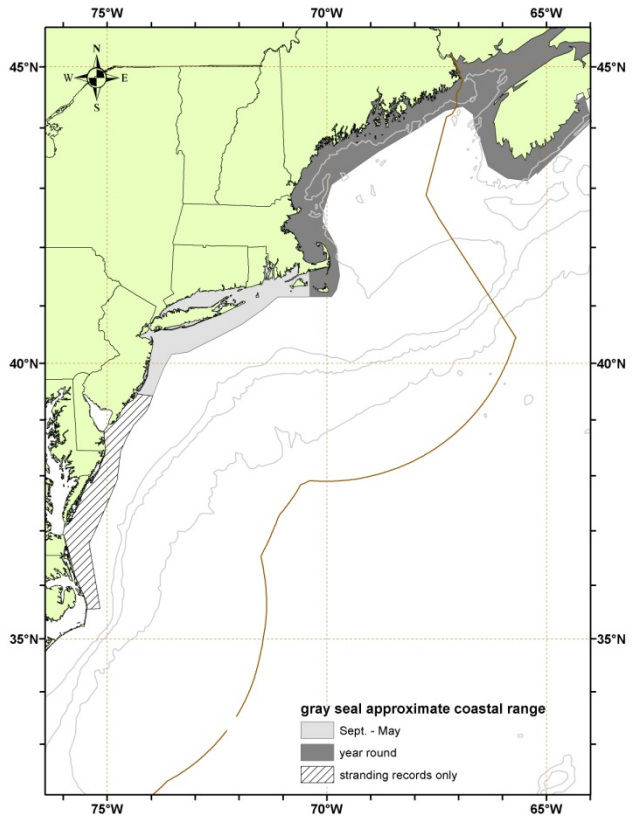

Figure 1. Approximate coastal range of gray seals. Isobaths are the 100-m, 1000-m, and 4000-m depth contours.

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.

POPULATION SIZE

Current estimates of the total western Atlantic gray seal population are not available; although estimates of portions of the stock are available for select time periods. Total pup production in 2016 at breeding colonies in Canada was 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 estimated Canadian gray seal population in 2016 was 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).

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. 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. 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. 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. Table 2 summarizes single-day pup counts from U.S. pupping colonies from 2001/2002 to 2015/2016 pupping periods. 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. 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.

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). 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 (95% CI: 22,162 – 33,215) animals. There is 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.

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	95% CI 263,000-458,000
2014[c]	Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island	505,000	95%CI=329,000-682,000
2016[d]	Gulf of St Lawrence + Nova Scotia Eastern Shore + Sable Island	424,300	95%CI=263,600-578,300
2016	U.S	27,131[e]	95% CI = 22,162 – 33,215

[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. 'CIP' = Counting in Progress. As single day pup counts, these counts do not represent the entire number of pups born in a pupping season. * indicates counts that need to be reviewed

	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	1704	9	Not surveyed	364	57	Not surveyed	Not surveyed
2007-08	2095	2	Not surveyed	466	59	Not surveyed	Not surveyed
2008-09	CIP	68	0	CIP	48	Not surveyed	Not surveyed
2009-10	1841	154	0	CIP	51	Not surveyed	Not surveyed
2010-11	3173	325	1	CIP	65	Not surveyed	112*
2011-12	2796*	80	8	CIP	41	2	57*

2012-13	2750	633	4	CIP	Not surveyed	Not surveyed	CIP
2013-14	3073	507	16	CIP	30	Not surveyed	201*
2014-15	1633	768	23	CIP	33	185	182*
2015-16	3787	935	32	1043	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 estimate 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

Gray seal abundance is likely increasing in the U.S. Atlantic Exclusive Economic Zone (EEZ), but the rate of increase is unknown.

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). 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. Pupping also occurs on Hay Island off Nova Scotia, in colonies off southwestern Nova Scotia, and in the Gulf of St. Lawrence. Pup production is 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); the realized growth rate for over two decades through 1997 was slightly higher.

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 2011–2015, the average annual estimated human caused mortality and serious injury to gray seals in the U.S. and Canada was 5,207 per year. The average was derived from six components: 1) 1,088 (CV=0.09) (Table 3) from the 2011–2015 U.S. observed fishery; 2) 7.8 from average 2011–2015 non-fishery related, human interaction stranding mortalities; 308 from the average 2011–2015 Canadian commercial harvest; 4) 132 from the average 2011–2015 DFO scientific collections; 5) 3,674 removals of nuisance animals in Canada (DFO 2017); and 6) 0.2 from U.S. research mortalities.

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. 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 2014, 2015, 2016, Orphanides and Hatch 2017). There were 9, 1, 8, 8, and 10 unidentified seals observed during 2011–2015, respectively. Since 1997 unidentified seals have not been prorated to a species. This is consistent with the treatment of other unidentified mammals that do not get prorated to a specific species. 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 2014, 2015, 2016; Orphanides and Hatch 2017). 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 2012 and one 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 2011–2015 is calculated as 0.4 animals (2 animals /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 34 gray seals were captured and released alive in 2011, 33 in 2012, 1 in 2013, and 2 in 2014, and 0 in 2015. In addition, during this time period 8 seals of unknown species were captured and released alive in 2011 and 2 in 2015 (Josephson *et al.* 2017).

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. Nineteen gray seal mortalities were observed in this fishery in 2011, 8 in 2012, 5 in 2013, 4 in 2014, and 2 in 2015.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

Three gray seal mortalities were observed in this fishery in 2011, 1 in 2012, 2 in 2013, 1 in 2014, and none in 2015. 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, the annual observer coverage, the serious injuries and mortalities recorded by on-board observers, the estimated annual 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]	Observed Serious Injury[c]	Observed Mortality	Estimated Serious Injury	Estimated Mortality	Estimated Combined Mortality	Estimated CVs	Mean Annual Combined Mortality
Northeast Sink Gillnet	11–15	Obs. Data,Weighout, Trip Logbook	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	222, 91, 69, 159, 131	0, 0, 0, 0, 0	1491, 542, 1127, 917, 1021	1491, 542, 1127, 917, 1021	.22, .19, .20, .14, .25	1020 (0.10)
Mid-Atlantic Gillnet	11–15	Obs. Data, Trip Logbook, Allocated Dealer Data	.02, .02, .03, .05, .06	0, 0, 0, 0, 0	2, 1, 0, 1, 1	0, 0, 0, 0, 0	19, 14, 0, 22, 15	19, 14, 0, 22, 15	.60, .98, 0, 1.09, 1.04	14 (0.48)
Northeast Bottom Trawl	11–15	Obs. Data,Trip Logbook	.26, 17, .15, .17, .19	0, 0, 0, 0, 0	19, 8, 5, 4, 2	0, 0, 0, 0, 0	58, 37, 20, 19, 23	58, 37, 20, 19, 23	.25, .49, .37, .45, .46	31 (0.16)
Mid-Atlantic Bottom Trawl	11–15	Obs. Data,Trip Logbook	.08, .05, .06, .08, .09	0, 0, 0, 0, 0	3, 1, 2, 1, 0	0, 0, 0, 0, 0	38, 42, 25, 7, 0	38, 42, 25, 7, 0	.54, .96, .67, .96, 0	22 (0.43)
Northeast Mid-water Trawl - Including Pair Trawl	11–15	Obs. Data, Trip Logbook	.41, .45, .37, .42, .08	0, 0, 0, 0, 0	0, 1, 1, 0, 0	0, 0, 0, 0, 0	0, na, na, 0, 0	0, na, na, 0, 0	0, na, na, 0, 0	0.4 (na) [d]
TOTAL										1,088 (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 2011–2015 period (Josephson *et al.* 2017)

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 have been reported at several major haul-out sites in the Gulf of Maine.

From 2011 to 2015, 570 gray seal stranding mortalities were recorded, extending from Maine to North Carolina (Table 4; NOAA National Marine Mammal Health and Stranding Response Database, accessed 16 September 2016). Most stranding mortalities were in Massachusetts, which is the center of gray seal abundance in U.S. waters. Sixty-nine (12%) of the total stranding mortalities showed signs of human interaction (20 in 2011, 4 in 2012, 17 in 2013, 8 in 2014, and 20 in 2015), 32 of which had some indication of fishery interaction (5 in 2011, 2 in 2012, 9 in 2013, 2 in 2014, and 14 in 2015). Eight gray seals are recorded in the stranding database during the 2011 to 2015 period as having been shot—7 in Massachusetts in 2011, none in 2012–2014, and 1 in Maine in 2015. 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.

CANADA

Between 2011 and 2015, the average annual human-caused mortality and serious injury to gray seals in Canadian waters from commercial harvest was 308 per year (http://www.dfo-mpo.gc.ca/fm-gp/seal-phoque/statistics-eng.htm accessed 1/4/2017), 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: 195 in 2011, 8 in 2012, 243 in 2013, 82 in 2014 and 1,145 in 2015. In addition, between 2011 and 2015, an average of 3,674 nuisance animals per year were killed. This included 1,722 in 2011, 5,428 in 2012, and 3,757 in 2013, 3,732 in 2014 and 3,732 in 2015 (DFO 2017). Lastly, DFO took 320 animals in 2011, 159 animals in 2012, 58 animals in 2013, 83 animals in 2014, and 42 animals in 2015 for scientific collections, for an annual average of 132 (DFO 2017).

Table 4. Gray seal (*Halichoerus grypus atlantica*) stranding mortalities along the U.S. Atlantic coast (2011-2015) with subtotals of animals recorded as pups in parentheses.

State	2011	2012	2013	2014	2015	Total
ME	4 (2)	10 (2)	9 (4)	3 (1)	5	31
NH	8 (1)	1 (1)	1 (0)	3 (2)	2	15
MA	89 (14)	38 (21)	82 (8)	62 (6)	77 (3)	348
RI	14 (2)	13 (5)	11 (2)	8 (1)	7 (1)	53
CT	2 (0)	0	0	0	0	2
NY	22 (6)	5 (3)	18 (5)	7 (4)	10	67
NJ	10 (0)	4 (0)	7 (2)	7 (6)	7 (6)	35

DE	0	0	0	3 (3)	3 (3)	6
MD	4 (2)	0	0	1 (0)	0	5
VA	1 (0)	0	0	0	3	4
NC	2 (2)	0	0	2 (2)	0	4
Total	156 (29)	71 (32)	128 (21)	96 (25)	114	570
Unspecified seals (all states)	63	28	25	38	31	176

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 2011–2015 average annual human-caused mortality and serious injury 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 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.

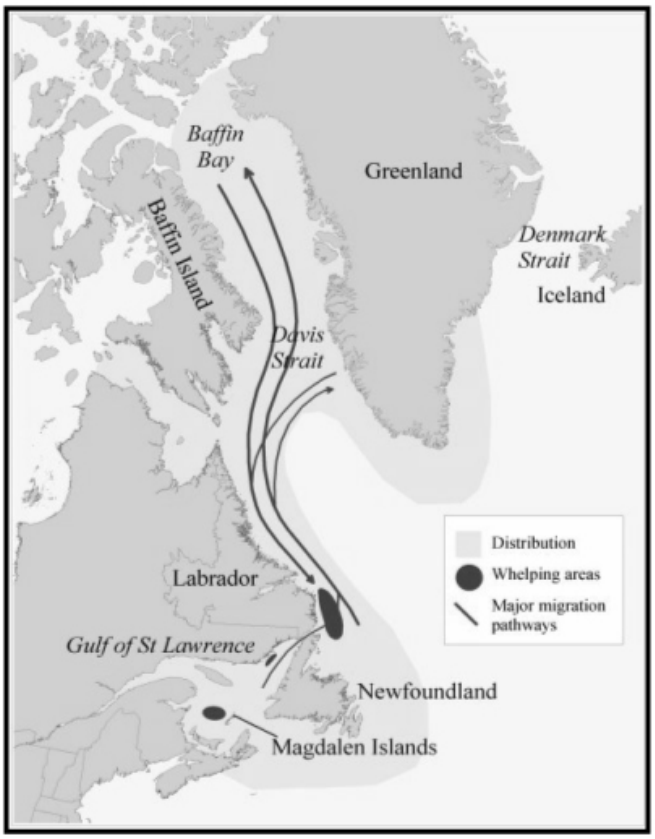

Figure 1: From: *Technical Briefing on the Harp Seal Hunt in Atlantic Canada*

http://www.dfo-mpo.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 extralimital 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).

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 (N_{best}) and confidence interval (CI)

Month/Year	Area	N_{best}	CI
2010	Front and Gulf	8.6-9.6 million	(95% CI 7.8-10.8 million)
2012	Front and Gulf	7.4 million	(95% CI 6.1-8.8 million)
2014	Front and Gulf	7.4 million	(95% CI 6.1 – 8.7 million)

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 is 7.4 million (95% CI 6.1-8.7 million; Hammill *et al.* 2014). 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 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. 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 2011–2015 the total estimated annual human caused mortality and serious injury to harp seals was 216,044. This is derived from three components: 1) an average catch of 215,998 seals from 2011-2015 by Canada and Greenland, including bycatch in the lumpfish fishery (Table 2a); 2) 43 harp seals (CV=0.24) from the observed U.S. fisheries (Table 2b); and 3) an average of 3 stranded seals from 2011-2015 that showed signs of non-fishing human interaction. 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 2011–2014, 27 mortalities were observed in the northeast sink gillnet fishery (Orphanides 2013, Hatch and Orphanides 2014; 2015; 2016, Orphanides and Hatch 2017). There were no observed injuries of harp seals in the Northeast region during 2011–2015 to assess using new serious injury criteria.

See Table 2b 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 harp seal mortality was observed in the Northeast bottom trawl fishery in 2011. Annual mortalities were estimated using annual stratified ratio-estimator methods (Chavez-Rosales *et al*. 2017). See Table 2b for bycatch estimates and observed mortality and serious injury for the current 5-year period, and Appendix V for historical bycatch information.

Table 2b. 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
Northeast Sink Gillnet[e]	11-15	Obs. Data, Trip Logbook, Allocated Dealer Data	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	4, 0, 2, 9, 12	0, 0, 0, 0, 0	14, 0, 22, 57, 119	14, 0, 22, 57, 119	.46, 0, .75, .42, .34	42 (0.24)

Northeast Bottom Trawl[d]	11-15	Obs. Data Weighout	.26, .17, .15, .17, .19	0, 0, 0, 0, 0	1, 0 , 0, 0, 0	0, 0, 0, 0, 0	0, 0, unk, 0, unk	.2.9, 0, 0, 0, 0	.81. 0, 0, 0, 0	0.6 (.81) [d]
TOTAL										43 (0.24)

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 2011–2015 period and include both at-sea monitor and traditional observer data (Josephson *et al.* 2017)

Other Mortality
U.S.

From 2011 to 2015, 279 harp seal stranding mortalities were reported (Table 3; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 13 September 2016). Sixteen (5.7%) of the mortalities during this five-year period showed signs of human interaction (7 in 2011, 1 in 2012, 2 in 2013, 4 in 2014 and 2 in 2015), 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 2a).

Table 2a. Summary of the Canadian directed catch and bycatch mortality of Northwest Atlantic harp seal (*Pagophilus groenlandicus*) by year

Fishery	**2011**	**2012**	**2013**	**2014**	**2015**	**Average**
Commercial catches[a]	40,389	71,460	90,703	54,830	35,304	**58,537**
Struck and lost[b]	77,156	64,664	86,970	66,946	81,609	**75,469**
Greenland subsistence catch[c]	73,277	59,124	80,102	62,147	68,662	**68,662**
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	**204,152**	**208,578**	**271,105**	**197,253**	**198,905**	**215,998**

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 are an average from 2011-2014
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)

Table 3. Harp seal (*Pagophilus groenlandicus*) stranding mortalities [a] along the U.S. Atlantic coast (2011–2015) with subtotals of animals recorded as pups in parentheses.

State	2011	2012	2013	2014	2015	Total
Maine	6	0	2	2 (1)	1	11
New Hampshire	0	0	1	0	0	1
Massachusetts	51 (1)	4	6 (1)	28	17	106
Rhode Island	7	0	1	9	4	21
Connecticut	4	0	0	0	0	4
New York	38 (1)	1	9	18	12	78
New Jersey	16	0	2	1	3	22
Delaware	2	0	1	0	0	3
Maryland	3	0	0	0	1	4
Virginia	5	0	1	9	4	21
North Carolina	3	0	2	1	2	8
Total	**135**	**5**	**23**	**68**	**44**	**279**
Unspecified seals (all states)	63	28	25	38	31	147

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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BRYDE'S WHALE (Balaenoptera edeni): Northern Gulf of MexicoStock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Bryde's whales are distributed worldwide in tropical and sub-tropical waters, but the taxonomy and number of species and/or subspecies of Bryde's whales in the world is currently a topic of debate (Kato and Perrin 2009; Rosel and Wilcox 2014). In the western Atlantic Ocean, Bryde's whales are reported from the Gulf of Mexico and the southern West Indies to Cabo Frio, Brazil (Leatherwood and Reeves 1983) but which subspecies the whales belong to in these different areas is unknown.. Sighting records and acoustic detections of Bryde's whales in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur almost exclusively in the northeastern Gulf in the De Soto canyon area, along the continental shelf break between 100 m and 400 m depth (Figure 1; Hansen *et al.* 1996; Mullin and Hoggard 2000; Mullin and Fulling 2004; Maze-Foley and Mullin 2006; Rice *et al.* 2014; Rosel and Wilcox 2014; Širović *et al.* 2014; Rosel *et al*. 2016; Soldevilla *et al.* 2017). Bryde's whales have been sighted in all seasons within the De Soto Canyon area (Mullin and Hoggard 2000; Maze-Foley and Mullin 2006; Mullin 2007; DWH MMIQT 2015). Genetic analysis suggests that Bryde's whales from the northern Gulf of Mexico represent a unique evolutionary lineage distinct from other recognized Bryde's whale subspecies, including those found in the southern Caribbean and southwestern Atlantic off Brazil (Rosel and Wilcox 2014). The geographic distribution of this Bryde's whale form has not yet been fully identified. Two strandings from the southeastern U.S. Atlantic coast share the same genetic characteristics with those from the northern Gulf of Mexico (Rosel and Wilcox 2014), but it is unclear whether these are extralimital strays (Mead 1977) or they indicate the population extends from the northeastern Gulf of Mexico to the Atlantic coast of the southern U.S. (Rosel and Wilcox 2014). There have been no confirmed sightings of Bryde's whales along the U.S. east coast during NMFS cetacean surveys (Rosel *et al*. 2016)

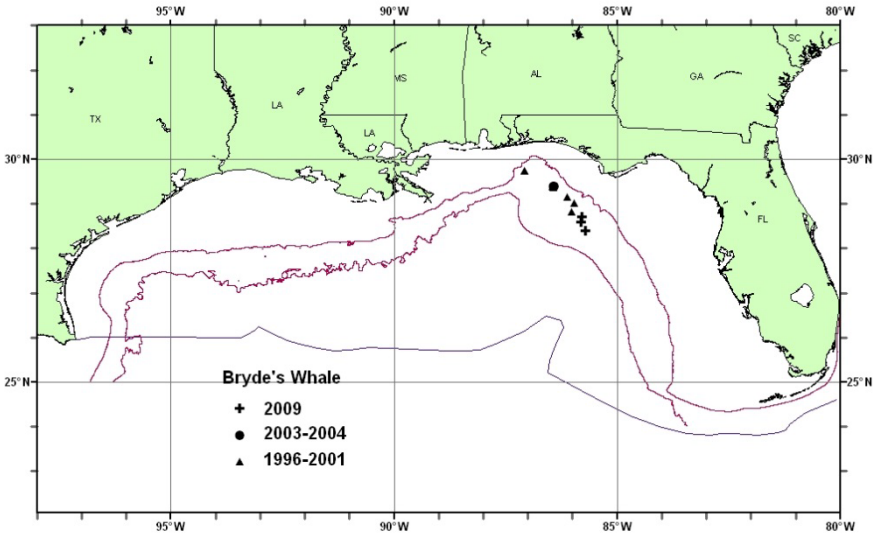

Figure 1. *Distribution of Bryde's whale sightings from SEFSC vessel surveys during spring 1996-2001, summer 2003 and spring 2004, and summer 2009. All the on-effort sightings are shown, though not all were used to estimate abundance. Solid lines indicate the 100m and 1,000m isobaths and the offshore extent of the U.S. EEZ.*

Historical whaling records from the 1800s suggest Bryde's whales may have been more common in the U.S. waters of the north central Gulf of Mexico and in the southern Gulf of Mexico in the Bay of Campeche (Reeves *et al.* 2011). At the time this report was written, there had yet to be a confirmed sightings in the north central or western Gulf despite NMFS survey effort in the area dating back to the early 1990s (e.g., Hansen *et al.* 1996; Mullin and Hoggard 2000; Mullin and Fulling 2004; Maze-Foley and Mullin 2006). A compilation of available records of cetacean sightings, strandings, and captures in Mexican waters of the southern Gulf of Mexico identified no Bryde's whales (Ortega-Ortiz 2002). There are insufficient data to determine whether it is plausible the stock contains multiple demographically independent populations that should be separate stocks.

POPULATION SIZE

The best abundance estimate available for northern Gulf of Mexico Bryde's whales is 33 (CV=1.07; Table 1). This estimate is from a summer 2009 oceanic survey covering waters from the 200-m isobath to the seaward extent of the U.S. EEZ.

Earlier abundance estimates

There are three previous estimates of abundance (Table 1). Please see Appendix IV for a summary of survey descriptions for earlier estimates.

Recent survey and abundance estimate

During summer 2009, a line-transect survey dedicated to estimating the abundance of oceanic cetaceans was conducted in the northern Gulf of Mexico (Garrison 2016). Survey lines were stratified in relation to depth and the location of the Loop Current. The abundance estimate for Bryde's whales in oceanic waters during 2009 was 33 (CV=1.07; Table 1).

Table 1. Summary of abundance estimates for northern Gulf of Mexico Bryde's 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
Apr-Jun 1991-1994	Oceanic waters	35	1.10
Apr-Jun 1996-2001 (excluding 1998)	Oceanic waters	40	0.61
Jun-Aug 2003, Apr-Jun 2004	Oceanic waters	15	1.98
Jun-Aug 2009	Oceanic waters	33	1.07

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 distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for Bryde's whales is 33 (CV=1.07). The minimum population estimate for the northern Gulf of Mexico is 16 Bryde's whales.

Current Population Trend

A trend analysis has not been conducted for this stock. Four point estimates of Bryde's whale abundance have been made based on data from line-transect surveys covering 1991–2009 (Table 1). The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. For example, the power to detect a precipitous decline in abundance (i.e., 50% decrease in 15 years) with estimates of low precision (e.g., CV>0.30) remains below 80% (alpha=0.30) unless surveys are conducted on an annual basis (Taylor *et al.* 2007). While not a trend analysis, it should be noted that research studies conducted under the Natural Resource Damage Assessment (NRDA) estimated there was up to a 22% decline in population size resulting from the *Deepwater Horizon* oil spill (see Habitat Issues section).

All verified Bryde's whale sightings have occurred in a very restricted area of the northeastern Gulf (Figure 1) during surveys that uniformly sampled the entire oceanic northern Gulf. Because the population size is small, in order to effectively monitor trends in Bryde's whale abundance in the future, other methods need to be used.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Current and maximum net productivity rates are unknown for this stock. For purposes of this assessment, the maximum net productivity rate was assumed to be 0.04. This value is based on theoretical modeling showing that cetacean populations likely do not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow *et al.* 1995). Between 1988 and 2015, there have been two documented strandings of calves (total length <700 cm) in the northern Gulf of Mexico (SEUS Historical Stranding Database unpublished data; NOAA

National Marine Mammal Health and Stranding Response Database unpublished data).

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of the minimum population size, one-half the maximum net productivity rate and a recovery factor (MMPA Sec. 3.16 U.S.C. 1362; Wade and Angliss 1997; Wade 1998). The minimum population size is 16. The maximum productivity rate is 0.04, the default value for cetaceans. The recovery factor is 0.1 because the stock is very small, exhibits very low genetic diversity, and appears to represent a unique and possibly endemic evolutionary lineage of Bryde's whale (Rosel and Wilcox 2014). PBR for the northern Gulf of Mexico Bryde's whale is 0.03, equivalent to one take every 33 years.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Gulf of Mexico Bryde's whale stock during 2011–2015 is unknown. There was no documented fishery-caused mortality or serious injury for this stock during 2011–2015. Mean annual mortality and serious injury during 2011–2015 due to other human-caused actions (the *Deepwater Horizon* oil spill) was 0.8. The minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2015 was, therefore, 0.8. This is considered a minimum mortality estimate as some fisheries with which the stock could interact have limited observer coverage. In addition, the likelihood is low that a whale killed at sea due to a fishery interaction or ship-strike will be recovered (Williams *et al.* 2011).

Fisheries Information

The commercial fisheries that potentially could interact with this stock in the Gulf of Mexico are the Category I Atlantic Ocean, Caribbean, Gulf of Mexico large pelagics longline fishery, and two Category III fisheries, the Southeastern U.S. Atlantic, Gulf of Mexico shark bottom longline/hook-and-line fishery and the Southeastern U.S. Atlantic, Gulf of Mexico, and Caribbean snapper-grouper and other reef fish bottom longline/hook-and-line fishery. See Appendix III for detailed fishery information. All three of these fisheries have observer programs, however observer coverage is limited for the two Category III fisheries. Pelagic swordfish, tunas, and billfish are the targets of the large pelagics longline fishery operating in the northern Gulf of Mexico. There has been no reported fishing-related mortality or serious injury of a Bryde's whale by this fishery during 1998–2015 (Yeung 1999; Yeung 2001; Garrison 2003; Garrison and Richards 2004; Garrison 2005; Fairfield Walsh and Garrison 2006; Fairfield-Walsh and Garrison 2007; Fairfield and Garrison 2008; Garrison *et al.* 2009; Garrison and Stokes 2010; 2012a, b; 2013; 2014; 2016; 2017). For the two category III bottom longline/hook-and-line fisheries, the target species are large and small coastal sharks and reef fishes such as snapper, grouper, and tilefish. There has been no reported fishery-related mortality or serious injury of a Bryde's whale by either of these fisheries (e.g., Scott-Denton *et al.* 2011; Hale *et al.* 2012; Gulak *et al.* 2013; 2014; Enzenauer *et al.* 2015; 2016).

Two other commercial fisheries that overlap to a small degree with the primary Bryde's whale habitat in the northeastern Gulf of Mexico are the Category III Gulf of Mexico butterfish trawl fishery and Category II Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl fishery (Rosel *et al.* 2016). No interactions have been documented for either of these fisheries.

Other Mortality

There were two reported strandings of Bryde's whales in the Gulf of Mexico during 2011–2015 (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). During 2012, two Bryde's whale strandings occurred in Louisiana. It could not be determined if there was evidence of human interaction for these strandings. Both whales were in a state of advanced decomposition when observed. Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury, particularly for offshore species such as Bryde's 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.

An Unusual Mortality Event (UME) was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). It includes cetaceans that stranded prior to the *Deepwater Horizon* (DWH) oil spill (see "Habitat Issues" below), during the spill, and after. The two Bryde's whale strandings in 2012 are considered to be part of this UME.

A population model was developed to estimate the injury and time to recovery for stocks affected by the DWH oil spill, taking into account long-term effects resulting from mortality, reproductive failure, reduced survival rates,

and the proportion of the stock exposed to DWH oil (DWH MMIQT 2015). Based on the population model, it was projected that 3.8 Bryde's whales died during 2011–2015 due to elevated mortality associated with oil exposure and that the stock experienced a 22% maximum reduction in population size due to the oil spill (DWH MMIQT 2015). The DWH Marine Mammal Injury Quantification Team cautioned that the capability of Bryde's whales to recover from the DWH oil spill is unknown because the population models do not account for stochastic processes and genetic effects (DWH MMIQT 2015), to which small populations are highly susceptible (Shaffer 1981; Rosel and Reeves 2000). The population model used to predict Bryde's whale mortality due to the DWH event has a number of sources of uncertainty. Model parameters (e.g., survival rates, reproductive rates, and life-history parameters) were derived from literature sources for Bryde's whales occupying waters outside of the Gulf of Mexico. In addition, proxy values for the effects of DWH oil exposure on both survival rates and reproductive success were applied based upon estimated values for common bottlenose dolphins in Barataria Bay. Finally, there was no estimation of uncertainty in model parameters or outputs.

Ship strikes may pose a threat to this stock. In 2009, a Bryde's whale was found floating in the Port of Tampa, Tampa Bay, Florida. The whale had evidence of pre-mortem and post-mortem blunt trauma, and was determined to have been struck by a ship, draped across the bow, and carried into port.

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). Shortly after the oil spill, the NRDA process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals. These studies estimated that 48% of the Bryde's whale stock was exposed to oil, that 22% (95% CI: 10–31) of females suffered from reproductive failure, and 18% (95% CI: 7–28) of the population suffered adverse health effects (DWH MMIQT 2015). A population model estimated the stock experienced a maximum 22% reduction in population size (see Other Mortality section above).

STATUS OF STOCK

The Bryde's whale is not listed as threatened or endangered under the Endangered Species Act, but the northern Gulf of Mexico stock is considered strategic under the MMPA because minimum total mean annual human-caused mortality and serious injury exceeds PBR. In addition, the stock is very small and exhibits very low genetic diversity, which place the stock at great risk of demographic stochasticity. The restricted range places it at risk of environmental stochasticity. The DWH oil spill is estimated to have resulted in a 22% maximum decline in population size for this stock. In April 2015, NMFS made a positive 90-day finding on a petition to list the Gulf of Mexico Bryde's whale as endangered under the ESA (NMFS 2015) and a proposed rule to list was published in December 2016 (NMFS 2016). The status of this stock relative to OSP is unknown. There are insufficient data to determine population trends for this stock.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*)
Barataria Bay Estuarine System 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. Until this effort is completed and 31 individual reports are available, some of the basic information presented in this report will also be included in the report: "Northern Gulf of Mexico Bay, Sound and Estuary Stocks".

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 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). In many cases, residents occur predominantly within estuarine waters, with limited movements through passes to the Gulf of Mexico (Shane 1977; Shane 1990; 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 presences of relatively 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. The Sellas *et al.* (2005) findings support the identification of BSE populations distinct from those occurring in adjacent Gulf coastal waters. 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

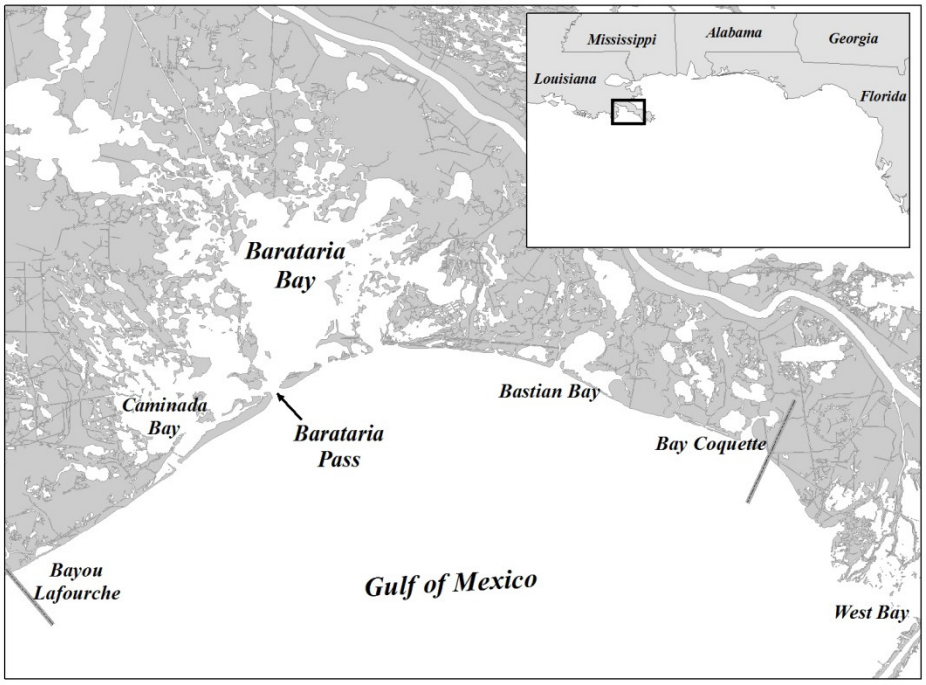

Figure 1. *Geographic extent of the Barataria Bay Estuarine System Stock, located on the coast of Louisiana. The borders are denoted by solid lines.*

2007; Rosel *et al*. 2009).

Barataria Bay is a shallow (mean depth = 2 m) estuarine system located in central Louisiana. It is bounded in the west by Bayou Lafourche, in the east by the Mississippi River delta and in the south by the Grand Terre barrier islands. Barataria Bay is approximately 110 km in length and 50 km in width at its widest point where it opens into the Gulf of Mexico (Conner and Day 1987). This estuarine system is connected to the Gulf of Mexico by a series of passes: Caminada Pass, Barataria Pass, Pass Abel, and Quatre Bayou Pass. The margins of Barataria Bay include marshes, canals, small embayments, and channels. Bay waters are turbid, and salinity varies widely from south to north with the more saline, tidally influenced portions in the south and freshwater lakes in the north (U.S. EPA 1999; Moretzsohn *et al*. 2010). Barataria Bay, together with the Timbalier-Terrebonne Bay system (referred to as the Barataria-Terrebonne National Estuary Program), has been selected as an estuary of national significance by the Environmental Protection Agency National Estuary Program (see http://www.btnep.org/BTNEP/home.aspx). The marshes and swamp forests which characterize Barataria Bay supply breeding and nursery grounds for an assortment of commercial and recreational species of consequence, such as finfish, shellfish, alligators, songbirds, geese, and ducks (U.S. EPA 1999; Moretzsohn *et al*. 2010).

The Barataria Bay Estuarine System (BBES) Stock was delimited in the first stock assessment reports published in 1995 (Blaylock et al. 1995). The stock area includes Caminada Bay, Barataria Bay east to Bastian Bay, Bay Coquette, and Gulf coastal waters extending 1 km from the shoreline (Figure 1). During June 1999–May 2002, Miller (2003) conducted 44 boat-based, photo-ID surveys in lower Barataria and Caminada Bays. Dolphins were present year-round, and 133 individual dolphins were identified. One individual was sighted six times, 42% were sighted two to six times, and 58% were sighted only once. More recently, Wells *et al.* (2017) deployed satellite-linked transmitters on 44 bottlenose dolphins captured within Barataria Bay during capture-release health assessments in August 2011, June 2013, and June 2014. It should be noted that the majority of tags were placed on animals captured in western Barataria Bay (see Wells *et al.* 2017 for tag deployment locations). Dolphins are known to inhabit eastern Barataria Bay (e.g., see Figure 1 in Rosel *et al.* 2017), but were not captured for tagging in far eastern waters due to logistical reasons. The tracking data found that the tagged dolphins remained within Barataria Bay, with a few animals occasionally entering coastal waters but venturing, on average, only out to approximately 1.7 km from shore (Wells *et al.* 2017). Telemetry data revealed three distinct ranging patterns for dolphins within the Bay, referred to as Island, West, and East. Island dolphins typically ranged near the western barrier islands of Grand Terre and Grande Isle and the nearby passes and Gulf waters within a few kilometers from the shoreline. West dolphins typically ranged in estuarine waters in the western portion of the Bay, such as Caminada Bay, West Champagne Bay, and Bassa Bassa Bay, as well as estuarine waters near Grand Isle and nearby Gulf waters within a few kilometers from the shoreline. East dolphins typically ranged in estuarine waters near the eastern barrier islands of East Grand Terre and Grand Pierre and in coastal marshes in eastern Barataria Bay. Tagged dolphins had relatively small home ranges (mean <70 km^2, Wells *et al.* 2017) within the BBES Stock area and displayed year-round, multi-year site fidelity to these home ranges, providing strong evidence of a year-round resident population in Barataria Bay. Molecular genetic analysis of population structure supported the telemetry data. Significant genetic differentiation was found at nuclear microsatellite DNA markers between dolphins sampled in Barataria Bay and those representing the Western Coastal Stock of common bottlenose dolphins that were sampled in coastal waters >2.5 km from shore outside of Barataria Bay (Rosel *et al.* 2017). In addition, the genetic analysis also suggested that there may be further partitioning within Barataria Bay (Rosel *et al.* 2017) similar to what was described from the telemetry data of Wells *et al*. (2017). Together the movement and genetic data provide strong evidence that the dolphins within Barataria Bay represent a demographically independent population separate from the dolphins inhabiting coastal waters. Both datasets also suggest it is plausible the BBES Stock contains multiple demographically independent populations, but further work is needed to better understand how the habitat is partitioned within the bay.

Dolphins residing in the estuaries southeast of this stock between BBES and the Mississippi River mouth (West Bay) are not currently covered in any stock assessment report. There are insufficient data to determine whether animals in this region exhibit affiliation to the BBES Stock or should be delineated as their own stock. Further research is needed to establish affinities of dolphins in this region and could result in revision to the eastern and/or western BBES Stock boundary. During 2011–2015, no bottlenose dolphins were reported stranded in this region.

POPULATION SIZE

The best available abundance estimate for the BBES Stock of common bottlenose dolphins is 2,306 (95% CI: 2,014–2,603; CV=0.09), which is the average of 10 vessel-based capture-recapture photo-ID surveys conducted from June 2010 to May 2014 (McDonald *et al.* 2017).

Earlier abundance estimates (>8 years old)

Miller (2003) conducted boat-based, photo-ID surveys in lower Barataria and Caminada Bays from June 1999 to May 2002. Miller (2003) identified 133 individual dolphins, and using closed-population unequal catchability models in the program CAPTURE, produced an abundance estimate of 138–238 (95% CI: 128–297) for the study area. Miller's (2003) estimate covered only a portion of the area of the BBES Stock and did not include a correction for the unmarked portion of the population. Therefore, the estimate is considered negatively biased.

Recent surveys and abundance estimates

McDonald *et al.* (2017) conducted vessel-based capture-recapture (C-R) photo-ID surveys from June 2010 to May 2014 to estimate density and abundance of common bottlenose dolphins within Barataria Bay during and after the *Deepwater Horizon* (DWH) oil spill. The study area included ~27% of the stock's area including the estuarine waters from the barrier islands of Grand Isle and Grande Terre, Louisiana, north and west into the main waters of Barataria Bay (McDonald *et al.* 2017). A spatially-explicit robust-design C-R model was used to estimate survival and density for each of 10 primary survey periods, and density and abundance estimates were adjusted for the proportion of the population that had non-distinctive fins. Suitable common bottlenose dolphin habitat (defined as average salinity >7.89 ppt) within the stock area was defined based upon a combined analysis of tag telemetry data (Wells *et al.* 2017) and average salinity maps (Hornsby *et al.* 2017). Common bottlenose dolphin density differed significantly among habitats near barrier islands, the eastern portion of the bay, and the western portion of the bay during the C-R study. Therefore, three habitat-specific densities from the surveyed area were estimated and these were then each appropriately expanded to the entire available suitable dolphin habitat in Barataria Bay (McDonald *et al.* 2017). Extrapolation of density estimates was therefore informed by habitat preferences of dolphins within Barataria Bay and did not include areas dominated by fresh water or shallow, marsh habitats that are not suitable dolphin habitats. Primary period abundances ranged from 1,303 dolphins (95% CI: 1,164–1,424) in June 2010 to 3,150 dolphins (95% CI: 2,759–3,559) in April 2014. The mean abundance for the BBES Stock estimated across the 10 C-R surveys was 2,306 dolphins (95% CI: 2,014–2,603; CV=0.09; McDonald *et al.* 2017). There were no clear seasonal or interannual temporal patterns in abundance. Key uncertainties in this abundance estimate include use of extrapolation from the surveyed area to a total stock abundance based on a preferred habitat model (McDonald *et al.* 2017; Hornsby *et al.* 2017). Also, the surveys for this abundance estimate were conducted during the DWH oil spill event and therefore may not accurately represent the post oil-spill abundance as it does not account for mortality that occurred after 2014 due to the spill.

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 2,306 (CV=0.09). The minimum population estimate for the BBES Stock is 2,138 bottlenose dolphins.

Current Population Trend

There are insufficient data to assess population trends for this stock. While not a trend analysis, it should be noted it was projected that there was up to a 51% decline in population size resulting from the DWH oil spill (see Habitat Issues section).

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). The current productivity rate may be compromised by the DWH oil spill as Lane *et al.* (2015) and Kellar *et al.* (2017) reported negative reproductive impacts (see Habitat Issues section).

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 BBES Stock of common bottlenose dolphins is 2,138. 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 Louisiana BSE stocks is greater than 0.8 (Wade and Angliss 1997). PBR for this stock of common bottlenose dolphins is 17.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the BBES Stock of common bottlenose dolphins during 2011–2015 is unknown. The mean annual fishery-related mortality and serious injury during 2011–2015 for strandings and at-sea observations identified as fishery-related was 0.8 (see Shrimp Trawl section for additional fishery-related mortality). Additional mean annual mortality and serious injury during 2011–2015 due to other human-caused sources (fishery research, at-sea entanglements, gunshot wounds, and DWH oil spill) was 159. The minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2015 was therefore 160 (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, 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), 5) the estimate does not include shrimp trawl bycatch (see Shrimp Trawl section), and 6) various assumptions were made in the population model used to estimate population decline for the northern Gulf of Mexico BSE stocks impacted by the DWH oil spill.

Fishery Information

There are four commercial fisheries that interact, or that potentially could interact, with this stock. These include two Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl and Gulf of Mexico menhaden purse seine fisheries) 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 does 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 Louisiana from Sabine Lake east to Barataria Bay, not just the BBES Stock. The mortality estimate for Louisiana BSE stocks for the years 2010–2014 was 61 (CV=1.4; Soldevilla *et al.* 2016). If all of the mortality occurred in Barataria Bay, the mortality estimate would exceed PBR for this stock; however, because bycatch for the BBES Stock alone cannot be quantified at this time, the mortality estimate is not included in the annual human-caused mortality and serious injury total for this stock. It should also be noted that this mortality estimate does not include skimmer trawl effort, which accounts for >48% of shrimp fishery effort in Louisiana, Alabama, and Mississippi inshore waters, because Observer Program coverage of skimmer trawls is limited. Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015; 2016).

Menhaden Purse Seine

During 2011–2015 there were no documented interactions between the menhaden purse seine fishery and the BBES Stock. The menhaden purse seine fishery operates in Gulf of Mexico coastal waters just outside the barrier islands of Barataria Bay (Smith *et al.* 2002). It has the potential to interact with dolphins of this stock that use nearshore coastal waters. Interactions have been reported for nearby coastal and estuarine stocks (Waring *et al.* 2015). Without a systematic observer program, it is not possible to obtain statistically reliable information for this fishery on the number of sets annually, the incidental take and mortality rates, and the stocks from which bottlenose dolphins are being taken.

Blue Crab Trap/Pot

During 2011–2015 there were two documented interactions in commercial blue crab trap/pot gear. There was one documented mortality (in 2011) of a common bottlenose dolphin in trap/pot gear, and one live animal (in 2012) was disentangled from trap/pot gear and released alive without serious injury following mitigation efforts (Maze-Foley and Garrison 2017). Both the mortality and live release were included in the stranding database (NOAA

National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016) and in the totals presented in Table 2. The mortality was included in the annual human-caused mortality and serious injury total for this stock (Table 1). There is no systematic observer coverage of crab trap/pot fisheries, so it is not possible to quantify total mortality. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Hook and Line (Rod and Reel)

During 2011–2015, four interactions with hook and line gear were documented within the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016; Table 2). In 2011, hook and line gear entanglement or ingestion were documented for one mortality and one animal released alive without serious injury following mitigation efforts (Maze-Foley and Garrison 2017). For the 2011 mortality, available evidence from the stranding data suggested the hook and line gear interaction 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). In 2013, two animals were released alive without serious injury following mitigation efforts after being found with hook and line gear on them during a live-capture-release health assessment (Maze-Foley and Garrison 2017).

In addition to animals included in the stranding database, during 2011–2015, there were two at-sea observations in Barataria Bay of dolphins entangled in fishing gear (monofilament line). The observations occurred during 2011 and 2012, and both dolphins were considered seriously injured (Maze-Foley and Garrison 2017), and both were 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 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. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Other Mortality

A population model was developed to estimate long-term injury to stocks affected by the DWH oil spill (see Habitat Issues section), taking into account long-term effects resulting from mortality, reproductive failure, and reduced survival rates (DWH MMIQT 2015; Schwacke *et al.* 2017). For the BBES Stock, the model predicted the stock experienced a 51% (95% CI: 32–72) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; DWH NRDAT 2016; Schwacke *et al.* 2017), and for the years 2011–2015, the model projected 789 mortalities (Table 1). This population model has a number of sources of uncertainty. The baseline population size was estimated from studies initiated after initial exposure to DWH oil occurred. Therefore, it is possible that the pre-spill population size was larger than this baseline level and some mortality occurring early in the event was not quantified. The duration of elevated mortality and reduced reproductive success after exposure is unknown, and expert opinion was used to predict the rate at which these parameters would return to baseline levels. Where possible, uncertainty in model parameters was included in the estimates of excess mortality by re-sampling from statistical distributions of the parameters (DWH MMIQT 2015; DWH NRDAT 2016; Schwacke *et al.* 2017).

During 2011–2015, one mortality was documented in Barataria Bay (in 2015) as a result of entanglement in a fishery research gillnet, and this animal was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016) and in the totals presented in Table 2, as well as in the annual human-caused mortality and serious injury total for this stock (Table 1).

During 2011–2015, there were two at-sea observations, one during 2013 and one during 2015, in Barataria Bay, of dolphins entangled around the head by constricting straps, potentially a plastic packing strap in one case. Both of these animals were considered seriously injured (Maze-Foley and Garrison 2017), and both were included in the annual human-caused mortality and serious injury total for this stock (Table 1).

NOAA's Office of Law Enforcement has been investigating increased reports from along the northern Gulf of Mexico coast of violence against bottlenose dolphins, including shootings via guns and bows and arrows, throwing pipe bombs and cherry bombs, and stabbings (Vail 2016). During 2011–2015, one mortality was attributed to a shooting. A small caliber bullet was found lodged in the lung of a carcass recovered during 2012, and the gunshot wound was determined to be the cause of death. This animal was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016) and in the totals presented in Table 2, as well as in the annual human-caused mortality and serious injury total for this stock (Table 1). From recent cases that have been prosecuted, it has been shown that fishermen became frustrated and retaliated against dolphins for removing bait or catch, or depredating, their fishing gear. It is unknown whether

the 2012 shooting involved depredation.

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 to the dolphin's activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning, or the illegal feeding, of wild 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 BBES Stock are summarized in Table 1.

Table 1. Summary of the incidental mortality and serious injury of common bottlenose dolphins (*Tursiops truncatus*) of the Barataria Bay Estuarine System (BBES) Stock. For the shrimp trawl fishery, the bycatch mortality for the BBES Stock alone cannot be quantified at this time and the state-wide mortality estimate for Louisiana 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	2011–2015	Pilot Observer Program (2011); MMAP fisherman self-reported takes	NA	0
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	NA	1
Hook and Line	2011–2015	Stranding Data and At-Sea Observations	NA	3
Mean Annual Mortality due to commercial fisheries (2011–2015)			**0.8**	
Research Takes (5-year Count)			1	
Other Takes (at-sea entanglements, gunshot wound; 5-year Count)			3	
Mortality due to DWH (5-year Projection)			789	
Mean Annual Mortality due to research takes, other takes, and DWH (2011–2015)			**159**	

Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)	160

Strandings

During 2011–2015, 98 common bottlenose dolphins were reported stranded within the BBES area (Table 2; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). It could not be determined if there was evidence of human interaction for 72 of these strandings. For eight dolphins, no evidence of human interaction was detected. Evidence of human interactions was detected for 18 stranded dolphins, seven of which stranded visibly oiled. In addition, there were two entanglements with commercial blue crab trap/pot gear (described above), four entanglements with hook and line gear (described above), one incidental take in a fishery research gillnet (described above), three animals with evidence of gunshot wounds, and one animal with evidence of a boat strike (Table 2). Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement, or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

The BBES Stock has been affected by two bottlenose dolphin die-offs or Unusual Mortality Events (UME). A UME occurred from January through May 1990, included 344 bottlenose dolphin strandings in the northern Gulf of Mexico (Litz *et al.* 2014), and may have affected the BBES Stock because strandings were reported in the Barataria Bay area during the time of the event. However, there is no information available on the impact of the event on the BBES Stock. 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). 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; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). This UME included cetaceans that stranded prior to the *Deepwater Horizon* 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 (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015a; Colegrove *et al.* 2016; DWH NRDAT 2016; see "Habitat Issues" below). During 2011–2014, nearly all stranded dolphins from this stock were considered to be part of the UME (see Table 2).

Table 2. Common bottlenose dolphin strandings occurring in the Barataria Bay Estuarine System Stock area from 2011 to 2015, 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 18 May 2016). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2011	2012	2013	2014	2015	Total
Barataria Bay Estuarine System Stock	Total Stranded	36[a]	18[a]	18[a]	15[a]	11	98
	Human Interaction						
	---Yes	11[b]	2[c]	3[d]	1[e]	1[f]	18
	---No	6	2	0	0	0	8
	---CBD	19	14	15	14	10	72

[a] All strandings were part of the ongoing UME event in the northern Gulf of Mexico.
[b] Includes 6 animals stranded visibly oiled (mortalities), 1entanglement interaction in commercial blue crab pot gear (mortality), and 2 entanglement interactions with hook and line gear (1 mortality and 1 animal released alive without serious injury [the live animal also had evidence of a boat strike]), and 1 animal with a gunshot wound (mortality).
[c] Includes 1 animal that stranded visibly oiled and with evidence of gunshot wounds (mortality), and 1 entanglement interaction with commercial blue crab trap/pot gear (released alive without serious injury).
[d] Includes 2 entanglement interactions with hook and line gear (both were released alive without serious injury).
[e] One animal with evidence of gunshot wounds (mortality)
[f] One entanglement interaction in research gillnet gear (mortality).

HABITAT ISSUES

Issues Related to the DWH Oil Spill

The DWH 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). A substantial number of beaches and wetlands along the Louisiana coast experienced heavy or moderate oiling (OSAT-2 2011; Michel *et al.* 2013). The heaviest oiling in Louisiana occurred on the tip of the Mississippi Delta, west of the Mississippi River in Barataria, Terrebonne and Timbalier Bays, and to the east of the river on the Chandeleur Islands (Michel *et al.* 2013).

A suite of research efforts indicate the DWH oil spill negatively affected the BBES Stock of common bottlenose dolphins. Capture-release health assessments and analysis of stranded dolphins during the oil spill both found evidence of moderate to severe lung disease and compromised adrenal function (Schwacke *et al.* 2014; Venn-Watson *et al.* 2015a). Based on data collected during a health assessment in Barataria Bay in 2011, 48% of the dolphins sampled were given a guarded or worse health prognosis, and 17% were given a poor prognosis, indicating that they would likely not survive (Schwacke *et al.* 2014). Subsequent health assessments in 2013 and 2014 revealed that the percentage of the population with a guarded or worse health prognosis decreased from levels measured in 2011 but still remained elevated when compared to the Sarasota Bay, Florida, reference site (DWH NRDAT 2016; Smith *et al.* 2017). Pulmonary abnormalities and impaired stress response were still detected four years after the DWH oil spill (Smith *et al.* 2017). De Guise *et al.* (2017) suggested immune systems were weakened due to the DWH oil exposure, most noticeably in 2011 compared to subsequent years. Stranding rates in the northern Gulf of Mexico were also higher in the years following the oil spill than previously recorded (Litz *et al.* 2014; Venn-Watson *et al.* 2015b) and 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; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). Investigations to date have determined that the DWH oil spill is the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015a; Colegrove *et al.* 2016). During 2011–2014, 87 stranded dolphins from this stock were considered to be part of the UME. Rosel *et al.* (2017) used genetic assignment tests to estimate stock of origin for stranded dolphins recovered between 2010 and 2013 in the estuary and along the coast of Barataria Bay and found that 83–84% of the stranded dolphins sampled originated from the BBES Stock, while the rest were assigned to the adjacent Western Coastal Stock. Balmer *et al.* (2015) suggested it is unlikely that persistent organic pollutants (POP) significantly contributed to the unusually high stranding rates following the DWH oil spill because POP concentrations from six northern Gulf sites were comparable to or lower than those previously measured by Kucklick *et al.* (2011) from southeastern U.S. sites; however, the authors cautioned that potential synergistic effects of oil exposure and POPs should be considered as the extra stress from oil exposure added to the background POP levels could have intensified toxicological effects. Morbillivirus infection, brucellosis, and biotoxins were also ruled out as a primary cause of the UME (Venn-Watson *et al.* 2015a).

Reproductive success also was compromised after the oil spill. Kellar *et al.* (2017) reported a reproductive success rate for Barataria Bay of 0.185, meaning that less than one in five detected pregnancies resulted in a viable calf. This rate was much lower than the expected rate, 0.647, based on previous work in non-oiled reference areas (Kellar *et al.* 2017). In addition, Lane *et al.* (2015) monitored 10 pregnant dolphins in Barataria Bay and determined that only 20% (95% CI: 2.50–55.6%) produced viable calves, as compared with a reported pregnancy success rate of 83% in a reference population. The reproductive failure rates are also consistent with findings of Colegrove *et al.*

(2016) who examined perinate strandings in Louisiana, Mississippi, and Alabama during 2010–2013 and found that common bottlenose dolphins were prone to late-term failed pregnancies and occurrence of *in utero* infections, including pneumonia and brucellosis.

Congruent with evidence for compromised health and poor reproductive success in Barataria Bay dolphins, McDonald *et al.* (2017) reported low survival rate estimates for these dolphins. Estimated survival rates in the first three years following the DWH oil spill using data from C-R photo-ID surveys ranged from 0.80 to 0.85 (McDonald *et al.* 2017), and are lower than those reported previously for other southeastern U.S. estuarine areas, such as Charleston, South Carolina (0.95; Speakman *et al.* 2010), or Sarasota Bay, Florida (0.96; Wells and Scott 1990).

Other Habitat Issues

Like much of coastal southeastern Louisiana, the Barataria Bay Basin has experienced significant wetland loss resulting in more open water and less marsh habitat (CPRA 2017). Subsidence, sea-level rise, storms, winds and tides, and human activities including levee construction and loss of sediment input, and channelization (navigational channels and oil and gas canals), all play a role in the habitat degradation (CPRA 2017). The impact to bottlenose dolphins from these changes to the habitat are unknown, although the marshes do serve as important nursery areas for many fish and invertebrates that may be prey species (CPRA 2017). The State of Louisiana has a wetland restoration master plan for the area to build and maintain land (CPRA 2017), which could result in additional changes to the Barataria Bay habitat, including significant and prolonged reductions in salinity levels. Bottlenose dolphins are typically found in salinities ranging from 20–35 ppt and can experience significant health impacts and/or death due to low salinity exposure (e.g., Andersen 1973; Holyoake *et al.* 2010).

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act. Because the estimate of human-caused mortality and serious injury exceeds PBR, NMFS considers this a strategic stock under the MMPA. The documented mean annual human-caused mortality for this stock for 2011–2015 was 160. However, it is likely the estimate of annual fishery-caused mortality and serious injury is biased low as indicated above (see Annual Human-Caused Mortality and Serious Injury), and there are uncertainties in the population model used to estimate population decline due to the DWH oil spill, also indicated above (see Habitat Issues). Because a UME of unprecedented size and duration (March 2010–July 2014) has impacted the northern Gulf of Mexico, including Barataria Bay, and because the health assessment findings of Schwacke *et al.* (2014) and others indicate compromised health and reproductive success of dolphins sampled within Barataria Bay as a result of the DWH oil spill, NMFS finds cause for concern about this stock. The DWH damage assessment estimated that the stock experienced a 51% (95% CI: 32–72) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; Schwacke *et al.* 2017). It is therefore likely that this stock is below its optimum sustainable population (NMFS 2016). In the absence of any additional non-natural mortality or restoration efforts, the DWH damage assessment estimated this stock will take 39 years to recover to pre-spill population size (DWH MMIQT 2015). The total human-caused mortality and serious injury for this stock is unknown but at a minimum is greater than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. There are insufficient data to determine population trends for this stock.

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COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*)
Mississippi Sound, Lake Borgne, Bay Boudreau 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. Until this effort is completed and 31 individual reports are available, some of the basic information presented in this report will also be included in the report: "Northern Gulf of Mexico Bay, Sound and Estuary Stocks".

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are distributed throughout the bays, sounds, and estuaries of the northern 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). In many cases, residents occur predominantly within estuarine waters, with limited movements through passes to the Gulf of Mexico (Shane 1977; Shane 1990; 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 concept of relatively discrete, demographically independent bay, sound and estuary (BSE) stocks (Duffield and Wells 2002; Sellas *et al.* 2005). Sellas *et al.* (2005) examined population subdivision among 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 structure among all areas on the basis of both mitochondrial DNA control region sequence data and nine nuclear microsatellite loci. The Sellas *et al.* (2005) findings support the identification of BSE populations distinct from those occurring in adjacent Gulf coastal waters. 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).

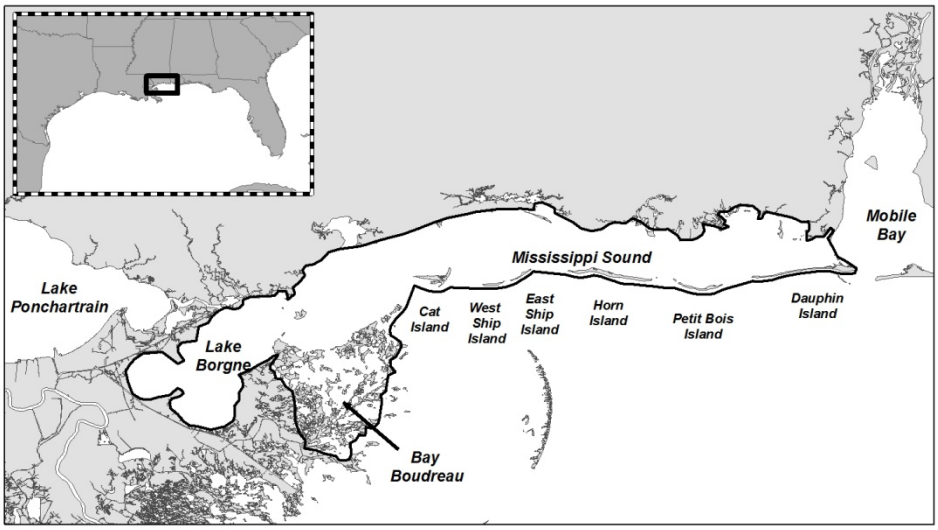

Figure 1. *Geographic extent of the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, located on the coasts of Alabama, Mississippi and Louisiana.*

The Mississippi Sound, Lake Borgne, Bay Boudreau Stock was delimited in the first stock assessment reports published in 1995 (Blaylock *et al.* 1995). The stock area (Figure 1) is complex with an estimated surface area of

3,711 km^2 (Scott *et al*. 1989), including adjacent Gulf coastal waters extending 1 km from Mississippi Sound barrier islands and passes. Mississippi Sound itself has a surface area of about 2,100 km^2 (Eleuterius 1978a,b) and is bounded by Mobile Bay in the east, Lake Borgne in the west, and the opening to Bay Boudreau in the southwest. It is bordered to the north by the mainlands of Louisiana, Mississippi and Alabama and to the south by six barrier islands: Cat, West Ship, East Ship, Horn, Petit Bois and Dauphin islands (Eleuterius 1978b), and in the extreme west, by Louisiana marshes. Mississippi Sound is an open embayment with large passes between the barrier islands allowing broad access to the Gulf of Mexico, including two dredged shipping channels. Average depth at mean low water is 2.98 m, and tides are diurnal with an average range of 0.57 m (Eleuterius 1978b). Sea surface temperature ranges seasonally from 9˚C to 32˚C (Christmas 1973). Salinity patterns are complex, varying seasonally with managed outputs from the Mississippi River, and there are multiple sharp salinity fronts; however, measurements of 20–35 ppt are typical (Kjerfve 1986). The bottom type is soft substrate consisting of mud and/or sand (Moncreiff 2007). Lake Borgne and Bay Boudreau are part of the Pontchartrain Basin and are remnants of the Saint Bernard lobe of the Mississippi River Delta that existed until about 2000 years ago when the Mississippi River changed course (Roberts 1997; Penland *et al.* 2013). Lake Borgne has an average depth of 3 m and an average salinity of 7 ppt (USEPA 1999). Bay Boudreau is a large shallow complex in the Saint Bernard marshes and consists of marshes, bayou, shallow bays, and points (Penland *et al*. 2013).

The Mississippi Sound, Lake Borgne, Bay Boudreau Stock area ("MS Sound Region") configuration is, in part, a result of the management of the live-capture fishery for bottlenose dolphins (Scott 1990). Mississippi Sound was once the site of the largest live-capture fishery of bottlenose dolphins in North America (Reeves and Leatherwood 1984). Between 1973 and 1988, of the 533 bottlenose dolphins removed from southeastern U.S. waters, 202 were removed from Mississippi Sound and adjacent waters (Scott 1990). In 1989, the Alliance of Marine Mammal Parks and Aquariums declared a self-imposed moratorium on the capture of bottlenose dolphins in the Gulf of Mexico (Corkeron 2009). Passage of the Marine Mammal Protection Act in 1972 and the concomitant need to manage the live-capture fishery for bottlenose dolphins was the impetus for much of the earliest bottlenose dolphin research in the MS Sound Region. This work focused on estimating the abundance of bottlenose dolphins (see below) and, to a lesser extent, on stock structure research primarily to provide live-capture quota recommendations (Scott 1990). To gather baseline biological data and study dolphin ranging patterns, 57 bottlenose dolphins were captured from Mississippi Sound, freeze-branded and released during 1982–1983 (Solangi and Dukes 1983; Lohoefener *et al*. 1990*a*). Re-sighting efforts for these dolphins conducted from 1982–1985 by Lohoefener *et al.* (1990a) suggested at least some individual dolphins exhibited fidelity for specific areas within Mississippi Sound.

The first dedicated photo-ID effort in the area undertaken by Hubard *et al.* (2004) during 1995–1996 suggested that some individual dolphins, seen multiple times, displayed spatial and temporal patterns of site fidelity, and some dolphins showed preferences to different habitats, particularly barrier islands, channels, or mainland coasts. Some individuals were seen in the same seasons both years, while others were seen in multiple seasons with a gap during winter months (Hubard *et al.* 2004). Also, two dolphins freeze branded during the live capture performed by Solangi and Dukes (1983) were re-sighted by Hubard *et al.* (2004).

During 2004–2007, Mackey (2010) followed dolphins in a portion of Mississippi Sound near and on both the Gulf and sound sides of the barrier islands and along the Gulfport Shipping Channel, and identified three different residency patterns. Of the 687 dolphins identified in those surveys, 71 (10%) were classified as year-round residents, 109 (16%) as seasonal residents, and 498 (73.5%) as transients. These patterns may not be representative of the entire MS Sound Region. Dolphins sighted near the barrier islands adjacent to or within the range of the Northern Coastal Stock of bottlenose dolphins may have a higher probability of being transient. Outside of the ship channel, a small proportion of the dolphins sighted by Mackey (2010) were from the interior two-thirds of Mississippi Sound (adjacent to the mainland) where dolphins may have quite different residency patterns. Mackey (2010) also identified two animals that were freeze-branded during the live captures 20 years earlier (Solangi and Dukes 1983).

Sinclair (2016) conducted photo-ID surveys in four zones within central Mississippi Sound during 2002–2005 to examine group sizes and movement patterns. The zones included one inner-sound zone near the mainland coast, two outer-sound zones near two barrier islands, and one coastal Gulf zone adjacent to the barrier island. Mean group sizes were significantly larger in summer, in outer-sound zones, and when a calf was present within the group. Limited movements were detected between the inner sound and other zones; however, movements between the outer sound and coastal waters were common.

Sinclair (2016), Mackey (2010), and Hubard *et al.* (2004) all noted low re-sighting rates of dolphins with a high percentage of dolphins seen only on one occasion. Both Mackey (2010) and Hubard *et al.* (2004) suggested dolphins move out of the Sound into deeper Gulf of Mexico waters during winter months, whereas Sinclair (2016) suggested that as dolphins are present year-round, it is the reverse and dolphins are moving into the sound in warm months,

coinciding with the active seasons of the menhaden and shrimp fisheries.

Establishing residency patterns in the MS Sound Region using photo-ID studies that cover large study areas will be difficult because of the large number of dolphins that inhabit the area and its open geography. Nevertheless, studies to date indicate that, similar to other Gulf of Mexico BSE areas, some individuals are long-term inhabitants of the MS Sound Region. In addition, photo-ID data indicate distinct ranging and habitat usage patterns, suggesting that the stock may contain multiple demographically independent populations. The stock boundaries are subject to change upon further study of dolphin residency patterns in estuarine waters of Alabama, Mississippi, and Louisiana.

POPULATION SIZE

The best available abundance estimate for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of common bottlenose dolphins is 3,046 (95% CI: 2,702–3,293; CV=0.06), based on a January 2012 vessel-based capture-recapture photo-ID survey (Mullin *et al.* 2017).

Earlier abundance estimates (>8 years old)

Aerial and small boat surveys conducted in the MS Sound Region covered different portions of the region and yielded a wide range of abundance estimates for bottlenose dolphins. Because of the differences in techniques and areas surveyed, it is very difficult to compare results. Early aerial surveys were conducted from single-engine high-winged aircraft that were not designed to estimate abundance for the Mississippi Sound or the MS Sound Region (Leatherwood *et al.* 1978; Thompson 1982). The studies also produced negatively biased results because the strip of transect directly under the aircraft was not observed. Scott *et al.* (1989) attempted to correct this bias by utilizing an aircraft with a glass bubble nose and placing an observer in it to observe the trackline at all times. Their estimates for the MS Sound Region ranged from 205 in winter to 858 in summer. (Abundances for Mississippi Sound only ranged from 136 dolphins in winter to 719 dolphins in summer). Boat-based mark-recapture surveys using dolphins freeze-branded during a previous live-capture study were performed by Lohoefener *et al.* (1990a) to assess the impacts of removing 30 dolphins from the population for captivity. The pre-removal estimate was 2,392 dolphins, and the post-removal estimate was 7,052 dolphins (Lohoefener *et al.* 1990a), but these were probably not accurate estimates, as too many assumptions of mark-recapture analysis were likely violated in this study (Lohoefener *et al.* 1990a). Boat-based line-transect abundance surveys of Mississippi Sound (about 55% of the MS Sound Region) were carried out by Lohoefener *et al*. (1990b) in 1984 and 1985, yielding much higher abundance estimates than aerial strip- or line-transect surveys and suggesting a seasonal shift in bottlenose dolphin abundance. For the entire Sound, abundance estimates were 2,400 and 500 dolphins for summer and winter, respectively. Another series of line-transect aerial surveys were performed in fall of 1992 by Blaylock and Hoggard (1994), where the abundance was reported as 1,401 for the MS Sound Region. Two additional abundance estimates from Mississippi Sound were boat-based line-transect surveys and only covered a portion of Mississippi Sound. Hubard *et al.* (2004) surveyed an area that was roughly one-quarter the size of the entire Sound. Again, abundances were found to fluctuate seasonally with higher abundances observed in summer months in 1995 (584 dolphins) and 1996 (555 dolphins) versus winter 1995–1996 months (268 dolphins). Miller *et al.* (2013) reported abundance estimates for a study area in eastern Mississippi Sound roughly 2,104 km^2 in size that included areas up to 15 km south of the barrier islands. Abundance estimates were 2,255 dolphins in summer 2007 and 1,413 dolphins in winter 2007–2008 (Miller *et al.* 2013).

Recent surveys and abundance estimates

The Southeast Fisheries Science Center conducted aerial surveys of continental shelf waters (shoreline to 200 m depth) along the U.S. Gulf of Mexico coast from the Florida Keys to the Texas/Mexico border during spring (March–April) 2011, summer (July–August) 2011, fall (October–November) 2011, and winter (January–February) 2012 (see Garrison 2017 for survey design and abundance estimation method). Each of these surveys was conducted using a two-team approach to develop estimates of visibility bias using the independent observer approach with Distance analysis (Laake and Borchers 2004). A model for the probability of detection on the trackline as a function of sighting conditions (sea state, glare, water color, etc.) was developed using data across all four surveys (Garrison 2017). The abundance estimates for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of bottlenose dolphins were based upon trackline and sightings in waters along the Alabama, Mississippi, and Louisiana coasts inside of the barrier islands. The surveys did not include trackline in Lake Borgne, but the estimated density was extrapolated to include the entire stock area. The seasonal abundance estimates for this stock were: spring – 2,395 (CV=0.42), summer – 1,709 (CV=0.59), fall – 1,140 (CV=0.41), and winter – 900 (CV=0.63). As with other BSE stocks, it is possible that there is movement of animals from coastal waters into the MS Sound Region on a seasonal basis. In order to assure that the abundance estimate for the stock reflects primarily resident animals, the lowest seasonal estimate (winter) was used to determine N_{best} for this stock. The resulting best estimate of abundance for

the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of common bottlenose dolphins from these aerial surveys was 900 (CV=0.63).

Pitchford *et al.* (2016) conducted vessel-based line-transect surveys from December 2011 to November 2013 in Lake Borgne and Mississippi Sound, excluding the far eastern waters of Mississippi Sound within Alabama. Density and population size was estimated for each season (winter, December–February; spring, March–May; summer, June–August; and fall, September–November) across the two years. Density estimates varied by stratum and season from 0.27 dolphins/km^2 (CV%=31.3) in spring 2013 to 1.12 dolphins/km^2 (CV%=21.6) in spring 2012 (Pitchford *et al.* 2016). The population estimates ranged from 738 (95% CI: 397–1369) in spring 2013 to 3,236 (95% CI: 1927–4627) in spring 2012 (Pitchford *et al.* 2016). According to Pitchford *et al.* (2016) differences in density estimates among central and eastern Mississippi Sound strata compared to the westernmost Mississippi Sound stratum and Lake Borgne stratum suggested animals use the westernmost portions of the study area during the warmer seasons of summer and fall, and also suggested the Mississippi Sound region is dynamic with respect to environmental variables that affect dolphin distribution and occurrence. The population size estimates of Pitchford *et al.* (2016) were negatively biased for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock because estimates did not include the easternmost waters of Mississippi Sound nor the waters of Bay Boudreau.

Vessel-based capture-recapture (C-R) photo-ID surveys were conducted from June 2010 to May 2012 to estimate density and abundance of bottlenose dolphins within Mississippi Sound during and after the *Deepwater Horizon* (DWH) oil spill (Mullin *et al.* 2017). The study area included waters of Mississippi Sound between the mainland and Horn and Petit Bois islands, as well as nearshore coastal waters south of Horn and Petit Bois islands (see Mullin *et al.* 2017). Density estimates and abundance were adjusted for the proportion of the population that had distinctive fins. Similar to McDonald *et al.* (2017), a spatial covariate, consisting of Island and Inshore strata, was used to estimate separate densities for these two strata. Abundance estimates were extrapolated by multiplying the total area of each stratum by the stratum density. Only a small part of Lake Borgne west of Half Moon Island was included in the extrapolation area as the remainder of Lake Borgne was presumed to be poor dolphin habitat (based on average low salinity). None of Bay Boudreau was included in the extrapolation. Resulting abundance estimates ranged from 4610 (95% CI: 4,271−4,865) during July 2011 to 3046 (95% CI: 2,702–3,293) during January 2012, for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock. While abundances for the stock were generally similar across the primary periods, previous studies consistently showed cold and warm seasons with low and high abundances, respectively. In order to assure that the abundance estimate for the stock reflects primarily resident animals, the lowest seasonal estimate (January 2012), 3,046 (CV=0.06), was used to determine N_{best} for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of common bottlenose dolphins. This estimate is negatively biased because it does not include an estimate for most of Lake Borgne nor any of Bay Boudreau. Key uncertainties in this abundance estimate include use of extrapolation from the surveyed area to a total stock abundance based on a preferred habitat model (McDonald *et al.* 2017; Hornsby *et al.* 2017). Also, this abundance estimate was made during the DWH oil spill event and may not accurately represent the post oil-spill abundance as it does not account for mortality due to the spill that occurred after 2012.

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 distributed abundance estimate as specified by Wade and Angliss (1997). The best estimate of abundance for this stock of common bottlenose dolphins is 3,046 (CV=0.06). The minimum population estimate for the stock is 2,896 common bottlenose dolphins.

Current Population Trend

There are insufficient data to determine the population trends for this stock. While not a trend analysis, it should be noted it was projected that there was up to a 62% decline in population size resulting from the DWH oil spill (see Habitat Issues section).

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). The current productivity rate may be compromised by the DWH oil spill as Kellar *et al.* (2017) reported negative reproductive impacts from the spill (see Habitat Issues section).

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 common bottlenose dolphins in the MS Sound Region is 2,896. 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 Mississippi and Alabama BSE stocks is greater than 0.8 (Wade and Angliss 1997). PBR for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of bottlenose dolphins is 23.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

The total annual human-caused mortality and serious injury for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock during 2011–2015 is unknown. The mean annual fishery-related mortality and serious injury during 2011–2015 for strandings and at-sea observations identified as fishery-related was 1.0 (see Shrimp Trawl section for additional fishery-related mortality). Additional mean annual mortality and serious injury during 2011–2015 due to other human-caused sources (fishery research, sea turtle relocation trawling, gunshot wounds, and DWH oil spill) was 309. The minimum total mean annual human-caused mortality and serious injury for this stock during 2011–2015 was therefore 310 (Table 1). This is likely 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, 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), 5) the estimate does not include shrimp trawl bycatch (see Shrimp Trawl section), and 6) various assumptions were made in the population model used to estimate population decline for the northern Gulf of Mexico BSE stocks impacted by the DWH oil spill.

Fishery Information

There are four commercial fisheries that interact, or that potentially could interact, with this stock. These include two Category II fisheries (Southeastern U.S. Atlantic, Gulf of Mexico shrimp trawl, and Gulf of Mexico menhaden purse seine fisheries) 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 does not extend into BSE waters, therefore time-area stratified bycatch rates were extrapolated into inshore waters to estimate the most recent five-year unweighted mean mortality estimate for 2010–2014 based on inshore fishing effort (Soldevila *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 all inshore waters of both Mississippi and Alabama and thus all their respective BSE stocks, not just the Mississippi Sound, Lake Borgne, Bay Boudreau Stock. The mortality estimate for Mississippi/Alabama BSE stocks (from Mississippi River Delta east to Mobile Bay, Bonsecour Bay) was 27 (CV=1.1) dolphins per year. If all of the mortality occurred in the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, the mortality estimate would nearly equal PBR for this stock; however, because bycatch for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock alone cannot be quantified at this time, the mortality estimate is not included in the annual human-caused mortality and serious injury total for this stock. It should also be noted that this mortality estimate does not include skimmer trawl effort, which accounts for >48% of shrimp fishery effort in Louisiana, Alabama, and Mississippi inshore waters, because Observer Program coverage of skimmer trawls is limited. Limitations and biases of annual bycatch mortality estimates are described in detail in Soldevilla *et al.* (2015; 2016).

Menhaden Purse Seine

During 2011–2015, there were two mortalities (2012) and one animal released alive without serious injury (2011) documented within waters of the MS Sound Region that involved the menhaden purse seine fishery. The two mortalities were included in the annual human-caused mortality and serious injury total for this stock (Table 1).

There is currently no observer program for the Gulf of Mexico menhaden purse seine fishery; however, incidental takes during 2011–2015 have been reported via two sources. First, during 2011, a pilot observer program operated from May through September and observers documented three dolphins trapped within purse seine nets. All three were released alive without serious injury (Maze-Foley and Garrison 2017). Two of the three dolphins were trapped within a single purse seine within waters of the Western Coastal Stock, and the third animal was trapped in waters of the MS Sound Region. Second, through the Marine Mammal Authorization Program (MMAP), there have been 13 self-reported incidental takes (all mortalities) of common bottlenose dolphins in northern Gulf of Mexico coastal and estuarine waters by the menhaden purse seine fishery during 2000–2015. These takes likely affected the following stocks: Western Coastal Stock; Northern Coastal Stock; Mississippi Sound, Lake Borgne, Bay Boudreau Stock; and Mississippi River Delta Stock. Specific self-reported takes under the MMAP likely involving the MS Sound Region are as follows: two dolphins were reported taken in a single purse seine during 2012 in waters of Mississippi Sound; one take of a single unidentified dolphin was reported during 2002 in waters of Mississippi Sound; and during 2000, three bottlenose dolphins were reported taken in a single purse seine in waters of Mississippi Sound.

Without an ongoing observer program, it is not possible to obtain statistically reliable information for this fishery on the number of sets annually, the incidental take and mortality rates, and the stocks from which bottlenose dolphins are being taken. The documented interactions in this commercial gear represent a minimum known count of interactions in the last five years.

Blue Crab Trap/Pot

During 2011–2015, there were two mortalities of common bottlenose dolphins for which commercial blue crab trap/pot gear entanglement were documented within the stranding data. One mortality occurred during 2011 and another in 2014. Both mortalities were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016) and in the totals presented in Table 2, as well as in the annual human-caused mortality and serious injury total for this stock (Table 1). There is no systematic observer coverage of crab trap/pot fisheries, so it is not possible to quantify total mortality. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Hook and Line (Rod and Reel)

During 2011–2015, there were six mortalities of common bottlenose dolphins for which hook and line gear entanglement or ingestion were documented within the stranding data. Three mortalities occurred in 2011, one in 2012, one in 2013 (in Lake Pontchartrain), and one in 2015. For one of these mortalities (2015), available evidence from the stranding record suggested the hook and line gear interaction contributed to the cause of death. For two mortalities (both 2011), available evidence suggested the hook and line gear interaction was not a contributing factor to cause of death. For three mortalities (2011, 2012, 2013), based on available evidence, it could not be determined if the hook and line gear interaction contributed to cause of death. These mortalities were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016) and in the totals presented in Table 2. The 2015 mortality for which evidence suggested the gear contributed to the cause of death 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 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. The documented interactions in this gear represent a minimum known count of interactions in the last five years.

Other Mortality

A population model was developed to estimate the injury in lost cetacean years and time to recovery for stocks affected by the DWH oil spill (see Habitat Issues section), taking into account long-term effects resulting from mortality, reproductive failure, and reduced survival rates (DWH MMIQT 2015; Schwacke *et al.* 2017). For the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, this model predicted the stock will have experienced a 62% (95% CI: 43–83) maximum reduction in population size (DWH MMIQT 2015; Schwacke *et al.* 2017), and for the years 2011–2015, the model projected 1539 mortalities (Table 1). This population model has a number of sources of uncertainty. The baseline population size was estimated from studies initiated after initial exposure to DWH oil occurred. Therefore, it is possible that the pre-spill population size was larger than this baseline level and some mortality occurring early in the event was not quantified. The duration of elevated mortality and reduced

reproductive success after exposure is unknown, and expert opinion was used to predict the rate at which these parameters would return to baseline levels. Where possible, uncertainty in model parameters was included in the estimates of excess mortality by re-sampling from statistical distributions of the parameters (DWH MMIQT 2015; DWH NRDAT 2016; Schwacke *et al.* 2017).

In both 2013 and 2014, single dolphins were entrapped during research skimmer trawl operations within the MS Sound Region. The 2014 interaction resulted in a mortality, and for the 2013 interaction, the animal was released alive, and it could not be determined if the animal was seriously injured (Maze-Foley and Garrison 2017). An additional mortality was documented in 2011 in the MS Sound Region as a result of an entanglement in a fishery research gillnet. All of these interactions were included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016) and in the totals presented in Table 2. The two mortalities were included in the annual human-caused mortality and serious injury total for this stock (Table 1).

As part of its annual coastal dredging program, the Army Corps of Engineers conducts sea turtle relocation trawling during hopper dredging as a protective measure for marine turtles. During 2011–2015, one bottlenose dolphin mortality was documented during 2011 in the MS Sound Region incidental to relocation trawling activities. This mortality was included in the stranding database (NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016) and in the totals presented in Table 2, as well as in the annual human-caused mortality and serious injury total for this stock (Table 1).

NOAA's Office of Law Enforcement has been investigating increasing numbers of reports from the northern Gulf of Mexico coast of violence against bottlenose dolphins, including shootings using guns and bows and arrows, throwing pipe bombs and cherry bombs, and stabbings (Vail 2016). During 2011–2015, three mortalities were attributed to shootings. In the summer of 2012, a shrimp fisherman knowingly shot a dolphin with a shotgun while shrimping and was subsequently convicted under the MMPA. In addition, two carcasses recovered in 2012 by the stranding network had gunshot wounds that caused the deaths of these animals (included in Table 2; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, accessed 18 May 2016). All three gunshot mortalities were included in the annual human-caused mortality and serious injury total for this stock (Table 1). From recent cases that have been prosecuted, it has been shown that fishermen became frustrated and retaliated against dolphins for removing bait or catch from (depredating) their fishing gear (Vail 2016).

Depredation is a growing problem in the Gulf of Mexico 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 to the dolphin's activity patterns, such as decreases in natural foraging (Powell and Wells 2011). It has been suggested that provisioning, or the illegal feeding, of wild 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 Florida and Texas (Bryant 1994; Samuels and Bejder 2004; Cunningham-Smith *et al.* 2006; Powell and Wells 2011). To date there are no reports within the literature of provisioning in the Mississippi Sound region.

All mortalities and serious injuries from known sources for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock are summarized in Table 1.

Table 1. Summary of the incidental mortality and serious injury of common bottlenose dolphins (*Tursiops truncatus*) of the Mississippi Sound, Lake Borgne, Bay Boudreau Stock. For the shrimp trawl fishery, the bycatch mortality for the Mississippi Sound, Lake Borgne, Bay Boudreau Stock alone cannot be quantified at this time and the mortality estimate for Mississippi and Alabama 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	2011–2015	Pilot Observer Program (2011); MMAP fisherman self-reported takes	NA	2
Atlantic Blue Crab Trap/Pot	2011–2015	Stranding Data	NA	2
Hook and Line	2011–2015	Stranding Data and At-Sea Observations	NA	1
Mean Annual Mortality due to commercial fisheries (2011–2015)			**1.0**	
Research Takes (5-year Count)			2	
Other Takes (sea turtle relocation trawling, gunshot wounds; 5-year Count)			4	
Mortality due to DWH (5-year Projection)			1539	
Mean Annual Mortality due to research takes, other takes, and DWH (2011–2015)			**309**	
Minimum Total Mean Annual Human-Caused Mortality and Serious Injury (2011–2015)			**310**	

Strandings

During 2011–2015, 306 common bottlenose dolphins were reported stranded within the Mississippi Sound, Lake Borgne, Bay Boudreau Stock area (Table 2; NOAA National Marine Mammal Health and Stranding Response Database unpublished data, 18 May 2016). Of those 306, 21 dolphins stranded within Lake Pontchartrain, which is connected to Lake Borgne. It is likely the stranded animals in Lake Pontchartrain were members of this stock. It could not be determined if there was evidence of human interaction for 281 of these strandings. For eight dolphins, no evidence of human interaction was detected. Evidence of human interactions was detected for 17 stranded dolphins. Human interactions were from numerous sources, including six entanglements with hook and line gear (described above), two entanglements with commercial blue crab trap/pot gear (described above), one incidental take in a research gillnet, one incidental take during turtle relocation trawling, and two incidental takes during research skimmer trawling, three mortalities with evidence of gunshot wounds, and one animal with evidence of a boat strike (Table 2). Stranding data probably underestimate the extent of human and fishery-related mortality and serious injury because not all of the dolphins that die or are seriously injured in human interactions wash ashore, or, if they do, they are not all recovered (Peltier *et al*. 2012; Wells *et al*. 2015). Additionally, not all carcasses will show evidence of human interaction, entanglement or other fishery-related interaction due to decomposition, scavenger damage, etc. (Byrd *et al*. 2014). Finally, the level of technical expertise among stranding network personnel varies widely as does the ability to recognize signs of human interaction.

The Mississippi Sound, Lake Borgne, Bay Boudreau Stock has been affected by three bottlenose dolphin die-offs or Unusual Mortality Events (UMEs). From January through May 1990, a total of 344 bottlenose dolphins stranded in the northern Gulf of Mexico including Mississippi. Overall this represented a two-fold increase in the prior maximum recorded number of strandings for the same period, but in some locations (i.e., Alabama) strandings were 10 times the average number. 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). In 1996 a UME was declared for

bottlenose dolphins in Mississippi when 31 bottlenose dolphins stranded during November and December. The cause was not determined, but a *Karenia brevis* (red tide) bloom was suspected to be responsible (Litz *et al.* 2014). A UME was declared for cetaceans in the northern Gulf of Mexico beginning 1 March 2010 and ending 31 July 2014 (Litz *et al.* 2014; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). It includes cetaceans that stranded prior to the DWH oil spill (see "Habitat Issues" below), during the spill, and after. Exposure to the DWH oil spill was determined to be the primary underlying cause of the elevated stranding numbers in the northern Gulf of Mexico after the spill (e.g., Schwacke *et al.* 2014; Venn-Watson *et al.* 2015a; Colegrove *et al.* 2016; DWH NRDAT 2016). During 2011–2014, nearly all stranded dolphins from this stock were considered to be part of the UME (see Table 2).

Table 2. Common bottlenose dolphin strandings occurring in the Mississippi Sound, Lake Borgne, Bay Boudreau Stock area from 2011 to 2015, including the number of strandings for which evidence of human interaction was detected and number of strandings for which it could not be determined (CBD) if there was evidence of human interaction (HI). Data are from the NOAA National Marine Mammal Health and Stranding Response Database (unpublished data, accessed 18 May 2016). Please note HI does not necessarily mean the interaction caused the animal's death.

Stock	Category	2011	2012	2013	2014	2015	Total
Mississippi Sound, Lake Borgne, Bay Boudreau Stock	Total Stranded	114[a]	47[a]	62[a]	53[b]	30	306
	Human Interaction						
	---Yes	8[c]	3[d]	3[e]	2[f]	1[g]	17
	---No	6	2	0	0	0	8
	---CBD	100	42	59	51	29	281

[a] All strandings were part of the UME event in the northern Gulf of Mexico.
[b] 46 strandings were part of the UME event in the northern Gulf of Mexico.
[c] Includes 3 entanglement interactions (mortalities) with hook and line fishing gear, 1 entanglement interaction (mortality) with commercial blue crab trap/pot gear, 1 mortality incidental to sea turtle relocation trawling, 1 entanglement interaction (mortality) with a research gillnet, and 1 mortality with evidence of a boat strike.
[d] Includes 1entanglement interaction (mortality) with hook and line fishing gear and 2 mortalities with evidence of gunshot wounds.
[e] Includes 1 entanglement interaction (mortality, Lake Pontchartrain) with hook and line fishing gear, 1 interaction with a research skimmer trawl (CBD if seriously injured), and 1 mortality with evidence of a gunshot wound.
[f] Includes 1 entanglement interaction with commercial blue crab trap/pot gear (mortality) and 1 entanglement interaction with a research skimmer trawl (mortality).
[g] An entanglement interaction with hook and line fishing gear (mortality).

HABITAT ISSUES

Issues Related to the DWH Oil Spill

The DWH 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). Within the region occupied by the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of common bottlenose dolphins, light to trace oil was reported along the majority of Mississippi's mainland coast, and heavy to light oiling occurred on Mississippi's barrier islands (Michel *et al.* 2013). Shortly after the oil spill, the Natural Resource Damage Assessment (NRDA) process was initiated under the Oil Pollution Act of 1990. A variety of NRDA research studies were conducted to determine potential impacts of the spill on marine mammals.

Stranding rates in the northern Gulf of Mexico rose significantly in the years of and following the DWH oil spill to levels higher than previously recorded (Litz *et al.* 2014; Venn-Watson *et al.* 2015b) and 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; http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm, accessed 1 June 2016). The primary cause for the UME was attributed to exposure to the DWH oil spill (Venn-Watson *et al.* 2015a; Colegrove *et al.*

2016; DWH NRDAT 2016) as other possible causes (e.g., morbillivirus infection, brucellosis, and biotoxins) were ruled out (Venn-Watson *et al.* 2015a).

A suite of research efforts indicated the DWH oil spill negatively affected the Mississippi Sound, Lake Borgne, Bay Boudreau Stock of common bottlenose dolphins (Schwacke *et al.* 2014; Venn-Watson *et al.* 2015a; Colegrove *et al.* 2016). Capture-release health assessments and analysis of stranded dolphins during the oil spill both found evidence of moderate to severe lung disease and compromised adrenal function (Schwacke *et al.* 2014; Venn-Watson *et al.* 2015a). In addition, low serum cortisol levels were found in Mississippi Sound dolphins (Smith *et al.* 2017). Compared to animals from Sarasota Bay, Florida, the percentage of the population with a guarded or worse health prognosis was 24% higher in Mississippi Sound (DWH MMIQT 2015; Smith *et al.* 2017). In addition, De Guise *et al.* (2017) suggested immune systems were weakened due to the DWH oil exposure.

Reproductive success also was compromised after the oil spill. Kellar *et al.* (2017) estimated the reproductive success rate of bottlenose dolphins in Mississippi Sound during and following the DWH oil spill at 0.222, meaning only about one in five detected pregnancies resulted in a viable calf. This rate was much lower than the expected rate, 0.647, based on previous work in non-oiled reference areas (Kellar *et al.* 2017). The elevated reproductive failure rate determined for Mississippi Sound following the DWH spill is consistent with previous research on mammals demonstrating a connection between petroleum exposure and reproductive impairments, and was not thought to be caused by other possible agents, namely persistent organic pollutants, *Brucella* spp., or biotoxins (Kellar *et al.* 2017). The reproductive failure rates are also consistent with findings of Colegrove *et al.* (2016) who examined perinate strandings in Louisiana, Mississippi, and Alabama during 2010–2013 and found that bottlenose dolphins were prone to late-term failed pregnancies and *in utero* infections, including pneumonia and brucellosis.

Congruent with evidence for compromised health and poor reproductive success, low survival rates were reported for bottlenose dolphins in Mississippi Sound following the DWH oil spill based on C-R photo-ID surveys (DWH MMIQT 2015; DWH NRDAT 2016). The estimated survival rate in the first year after the spill (July 2010–July 2011) was 0.73 and the rate for the second period (July 2011–January 2012) was 0.78. These survival rates are much lower than those reported previously for other southeastern U.S. estuarine areas, such as Charleston, South Carolina (0.95; Speakman *et al.* 2010), or Sarasota Bay, Florida (0.96; Wells and Scott 1990).

Finally, Balmer *et al.* (2015) indicated it is unlikely that persistent organic pollutants (POPs) significantly contributed to the unusually high stranding rates following the DWH oil spill. POP concentrations in dolphins sampled between 2010 and 2012 at six northern Gulf sites that experienced DWH oiling were comparable to or lower than those previously measured by Kucklick *et al.* (2011) from southeastern U.S. sites; however, the authors cautioned that potential synergistic effects of oil exposure and POPs should be considered as the extra stress from oil exposure added to the background POP levels could have intensified toxicological effects.

Other Habitat Issues

Environmental contaminants have been an issue of concern for bottlenose dolphins throughout the southeastern U.S., including Mississippi Sound, prior to the DWH oil spill. As mentioned above, Kucklick *et al.*(2011) examined POPs (PCBs, chlordanes, mirex, DDTs, HCB and dieldrin) and polybrominated diphenyl ether (PBDE) concentrations from bottlenose dolphin blubber samples collected during 2000–2007 from 14 locations, including Mississippi Sound, 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 Mississippi Sound had relatively high concentrations of some pollutants, like PBDEs, HCB, mirex and DDTs, and more intermediate concentrations of dieldrin, PCBs and chlordanes, when compared to dolphins sampled from the other 13 locations (Kucklick *et al.* 2011). The more recent work of Balmer *et al.* (2015) (discussed above) found lower levels of POPs in Mississippi Sound when compared to the results of Kucklick *et al.* (2011); however, future research is necessary to identify which stock(s) were sampled in this region to fully assess changes in POPs over time.

The presence of vessels may impact bottlenose dolphin behavior in bays, sounds, and estuaries. Miller *et al.* (2008) investigated the immediate responses of bottlenose dolphins to "high-speed personal watercraft" (i.e., boats) in Mississippi Sound. They found an immediate impact on dolphin behavior demonstrated by an increase in traveling behavior and dive duration, and a decrease in feeding behavior for non-traveling groups. The findings suggested dolphins attempted to avoid high-speed personal watercraft. It is unclear whether repeated short-term effects will result in long-term consequences like reduced health and viability of dolphins. Further studies are needed to determine the impacts throughout the Gulf of Mexico.

STATUS OF STOCK

Common bottlenose dolphins are not listed as threatened or endangered under the Endangered Species Act.

Because the estimate of human-caused mortality and serious injury exceeds PBR, this is a strategic stock under the MMPA. The documented mean annual human-caused mortality for this stock for 2011–2015 was 310. However, it is likely the estimate of annual fishery-caused mortality and serious injury is biased low as indicated above (see Annual Human-Caused Mortality and Serious Injury), and there are uncertainties in the population model used to estimate population decline due to the DWH oil spill, also indicated above (see Habitat Issues). Because a UME of unprecedented size and duration (1 March 2010 through 31 July 2014) has impacted the northern Gulf of Mexico, including the Mississippi Sound, Lake Borgne, Bay Boudreau Stock, NMFS finds cause for concern about this stock. The DWH damage assessment projected that the stock will have experienced a 62% (95% CI: 43–83) maximum reduction in population size due to the oil spill (DWH MMIQT 2015; Schwacke *et al.* 2017). It is therefore likely that this stock is below its optimum sustainable population (NMFS 2016). In the absence of any additional non-natural mortality or restoration efforts, the DWH damage assessment estimated this stock will take 46 years to recover to pre-spill population size (DWH MMIQT 2015). It is plausible that this stock contains multiple demographically independent populations. Total fishery-related mortality and serious injury for this stock is not known, but at a minimum is greater than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. There are insufficient data to determine population trends for this stock.

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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	2011-2015	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	273(.20), 277(.59), 399(.33), 128(.27), 177(.28)	251(.18)	706
Atlantic white sided dolphin	2011-2015	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	18(.43), 9(.92), 4(1.03), 10(.66), 0	8.2(.34)	304
Common dolphin	2011-2015	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	49(.71), 95(.40), 104(.46), 111(.47), 55(.54)	86(.23)	557
Long-finned pilot whale	2010-2014	.17, .19, .15, .11, .18		3(.82), 0, 0, 0, 0	0.6(.82)	35
Risso's dolphin	2011-2015	.19, .15, .11, .18, .14		0, 6(.87), 23(1.0), 0, 0	5.8 (.79)	126
Bottlenose dolphin (offshore)	2010-2014	.17, .19, .15, .11, .18		0, 0, 0, 26(.95), 0	5.2(.95)	561
Harbor seal	2011-2015	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	343(.19), 252(.26), 142(.31), 390(.39), 474(.17)	321 (.12)	2,006
Gray seal	2011-2015	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	1550(.22), 542(.19), 1127(.20), 917(.14), 1021(.25)	1020(.10)	1,389
Harp seal	2011-2015	.19, .15, .11, .18, .14	0, 0, 0, 0, 0	14(.46), 0, 22(.75), 57(.42), 119(.34)	42(.24)	unk
Gillnet Fisheries:US Mid-Atlantic gillnet						
Harbor porpoise	2011-2015	.02, .02, .03, .05, .06		123(.41), 63(.83), 19(1.06), 22(1.03), 33(1.16)	52(.34)	706
Common dolphin	2011-2015	.02, .02, .03, .05 .06	0, 0, 0, 0, 0	29(.53), 15(.93), 62(.67), 17(.86), 30(.86)	31(.34)	557
Harbor seal	2011-2015	.02, .02, .03, .05 .06	0, 0, 0, 0, 0	21(.67), 0, 0, 19(1.06), 48(.52)	18(.40)	2,006

Category, Fishery, Species	Yrs. observed	observer coverage	Est. SI by Year (CV)	Est. Mortality by Year (CV)	Mean Annual Mortality	PBR
Gray Seal	2011-2015	.02, .02, .03, .05 .06	0, 0, 0, 0, 0	19(.60), 14(98), 0, 22(1.09), 15(1.04)	14(.48)	1,389
Longline Fisheries: Pelagic longline (excluding NED-E)						
Risso's dolphin	2011-2015	.09, .07, .09, .10, .12	12(.63), 15 (1.0), 1.9(1.0), 7.7(1.0), 8.4 (.71)	0, 0, 0, 0, 0	8.9(.44)	126
Short-finned pilot whale	2010-2014	.08, .09, .07, .09, .10	127(.78), 286 (.29), 170(.33), 124(.32), 233(.24)	0, 19, 0, 0, 0	192 (.17)	159
Long-finned pilot whale	2010-2014	08, .09, .07, .09, .10	0, 0, 0, 0, 9.6	0, 0, 0, 0, 0	1.9(.43)	35
Bottlenose dolphin (offshore)	2010-2014	.08, .09, .07, .09, .10	0,0, 61.8(.68), 0,0	0, 0, 0, 0, 0	12.4(.68)	561
Kogia spp.	2010-2014	.08, .09, .07, .09, .10	0, 17, 0, 0, 0	0, 0, 0, 0, 0	3.5(1.0)	21
Common dolphin	2011-2015	.09, .07, .09, .10, .12	0, 0, 0, 0, 9.05	0, 0, 0, 0, 0	1.8(1.0)	557
CATEGORY II						
Mid-Atlantic Mid-Water Trawl – Including Pair Trawl						
Harbor Seal	2011-2015	.41, .21, .07, .05, .03	0, 0, 0, 0, 0	0, 0, 0, 0, na	0.4(na)	2,006
Trawl Fisheries:Northeast bottom trawl						
Harp seal	2011-2015	.26, .17, .15, .17 .19	0, 0, 0, 0, 0	2.9(.81), 0, 0, 0, 0	0.6(.81)	unk
Harbor seal	2011-2015	.26, .17, .15, .17 .19	0, 0, 0, 0, 0	9(.58), 3(1), 4(.96), 11(.63), 0	5.3(.34)	2,006
Gray seal	2011-2015	.26, .17, .15, .17 .19	0, 0, 0, 0, 0	58(.25), 37(.49), 30(.37), 19(.45), 23(.46)	31(.16)	1,389
Risso's dolphin	2011-2015	.26, .17, .15, .17 .19	0, 0, 0, 0, 0	0, 0, 0, 4.2(.91), 0	0.8 (.91)	126
Bottlenose dolphin (offshore)	2010-2014	.16, .26, .17, .15, .17	0, 0, 0, 0, 0	4(.53),10(.84), 0, 0, 0	2.8(.62)	561

Category, Fishery, Species	Yrs. observed	observer coverage	Est. SI by Year (CV)	Est. Mortality by Year (CV)	Mean Annual Mortality	PBR
Long-finned pilot whale	2010-2014	.16, .26, .17, .15, .17	6, 12, 10, 0, 6	30 (43), 55(.18), 33(.32), 16(.42), 25(.44)	33(.15)	35
Common dolphin	2011-2015	.26, .17, .15, .17 .19	0, 0, 0, 0, 0	73(.32), 42(.47), 17(.54), 17(.53), 22(.45)	34 (.22)	557
Atlantic white-sided dolphin	2011-2015	.26, .17, .15, .17 .19	3, 0, 0, 0, 0	140(.24), 27(.47), 33(.31), 16(.5), 15(.52)	46(.17)	304
Harbor porpoise	2011-2015	.26, .17, .15, .17, .19	3, 0, 0, 0, 0	0, 2.9(.58), 0, 7(.98), 5.5(.86), 3.7(.49)	4.4(.49)	706
Mid-Atlantic Bottom Trawl						
Common dolphin	2011-2015	.08, .05, .06, .08 .09	8, 7, 0, 24, 0	263(.25), 316(.26), 269(.29), 305(.29), 259(.32)	285 (.12)	57
Atlantic white-sided dolphin	2011-2015	.08, .05, .06, .08, .09	0, 0, 0, 0, 0	0, 0, 0, 9.7(.94), 0	1.9(.94)	304
Risso's dolphin	2011-2015	.08, .05, .06, .08 .09	0, 0, 0, 0, 27	27(.68), 7.6(1.0), 42(.71), 21(.93), 13(.63)	28 (.35)	126
Bottlenose dolphin (offshore)	2010-2014	.06, .08, .05, .06, .08	0, 0, 0, 0, 0	20(.34),34(.31), 16(1.0), 0, 25(.66)	19(.28)	561
Harbor seal	2011-2015	.08, .05, .06, .08 .09	0, 0, 0, 0, 0	0, 23(1), 11(.96), 10(.95), 7	10(0.53)	2,006
Gray seal	2011-2015	.08, .05, .06, .08 .09	0, 0, 0, 0, 0	38(.54) , 42(.96), 25(.67), 7(.96), 0	22(.43)	1,389
Northeast Mid-Water Trawl Including Pair Trawl						
Long -finned pilot whale	2010-2014	.41, .17, .45, .37, .42	0, 0, 0, 0, 0	0, 1, 1, 3, 4	1.8(na)	35
Common dolphin	2011-2015	.41, .45, .37, .42, .08	0, 0, 0, 0, 0	0, 1, 0, 0, 0	0.2(na)	557
Harbor seal	2011-2015	.41, .45, .37, .42, .08	0, 0, 0, 0, 0	0, na, 0, na, na	0.8(na)	2,006
Gray seal	2011-2015	.41, .45, .37, .42, .08	0, 0, 0, 0, 0	0, na, na, 0, 0	0.4(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, US East Coast, and adjacent Canadian Maritimes, 2011–2015, with number of events attributed to entanglements or vessel collisions by year.

Stock	Mean annual mortality and SI rate (PBR[1] for reference)	Entanglements			Vessel Collisions		
		Annual rate (US waters / Canadian waters/unknown first sighted in US/unknown first sighted in Canada)	Confirmed mortalities (2011, 2012, 2013, 2014, 2015)	Confirmed SIs (2011, 2012, 2013, 2014, 2015)	Annual rate (US waters / Canadian waters/unknown first sighted in US/unknown first sighted in Canada)	Confirmed mortalities (2011, 2012, 2013, 2014, 2015)	Confirmed SIs (2011, 2012, 2013, 2014, 2015)
Western North Atlantic right whale *(Eubalaena glacialis)*	5.36 (0.9)	4.55 (0.40/ 0.00/ 2.45/ 1.70)	(1, 2, 0, 2, 0)	(4.5, 2, .75, 7, 3.5)	.81 (0.81/ 0.00/ 0.00/ 0.00)	(1, 0, 0, 0, 0)	(2, 1, 0, 0, 0)
Gulf of Maine humpback whale *(Megaptera novaeangliae)*	8.25 (3.7)	6.45 (1.65/ 0.3/ 4.15/ 0.35)	(0, 0, 2, 2, 1)	(7.75, 4.75, 1.75, 5.5,7.5)	1.8 (1.80/ 0.00/ 0.00/ 0.00)	(3, 0, 2, 0, 4)	0
Western North Atlantic fin whale *(Balaenoptera physalus)*	2.65 (2.5)	1.05 (0.20/ 0.8[2]/ 0.85/ 0)	(3, 0, 0, 1, 0)	(1, 0.75, 1, 1.5, 1)	1.6 (1.6/ 0.00/ 0.00/ 0.00)	(1, 4, 1, 2, 0)	0
Nova Scotian sei whale *(B. borealis)*	0.8 (0.5)	0	0	0	0.8 (0.80/ 0.00/ 0.00/ 0.00)	(1, 0, 0, 3, 0)	0
Canadian East Coast minke whale *(B. acutorostrata)*	9.15 (14)	7.75 (1.9/ 3.25/ 2.6/ 0.00)	(4, 6, 1, 4, 7)	(2, 5, 7, 3, 1)	1.4 (1.2/ 0.2/ 0.00/ 0.00)	(3, 1, 0, 2, 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 2003).

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-GARFO 2018). 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 forms are available online.

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.sefsc.noaa.gov/fisheries/logbook.htm.

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-GARFO 2018). 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 forms are available online.

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. 2011 Northeast sink gillnet observed hauls (A) and incidental takes (B).

Figure 2. 2012 Northeast sink gillnet observed hauls (A) and incidental takes (B).

Figure 3. 2013 Northeast sink gillnet observed hauls (A) and incidental takes (B).

Figure 4. 2014 Northeast sink gillnet observed hauls (A) and incidental takes (B).

Figure 5. 2015 Northeast sink gillnet observed hauls (A) and incidental takes (B).

Figure 6. 2011 mid-Atlantic coastal gillnet observed hauls (A) and incidental takes (B).

Figure 7. 2012 mid-Atlantic coastal gillnet observed hauls (A) and incidental takes (B).

Figure 8. 2013 mid-Atlantic coastal gillnet observed hauls (A) and incidental takes (B).

Figure 9. 2014 mid-Atlantic coastal gillnet observed hauls (A) and incidental takes (B).

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Figure 11. 2011 mid-Atlantic bottom trawl observed tows (A) and incidental takes (B).

Figure 12. 2012 mid-Atlantic bottom trawl observed tows (A) and incidental takes (B).

Figure 13. 2013 mid-Atlantic bottom trawl observed tows (A) and incidental takes (B).

Figure 14. 2014 mid-Atlantic bottom trawl observed tows (A) and incidental takes (B).

Figure 15. 2015 mid-Atlantic bottom trawl observed tows (A) and incidental takes (B).

Figure 16. 2011 Northeast bottom trawl observed tows (A) and incidental takes (B).

Figure 17. 2012 Northeast bottom trawl observed tows (A) and incidental takes (B).

Figure 18. 2013 Northeast bottom trawl observed tows (A) and incidental takes (B).

Figure 19. 2014 Northeast bottom trawl observed tows (A) and incidental takes (B).

Figure 20. 2015 Northeast bottom trawl observed tows (A) and incidental takes (B).

Figure 21. 2011 Northeast mid-water trawl observed tows (A) and incidental takes (B).

Figure 22. 2012 Northeast mid-water trawl observed tows (A) and incidental takes (B).

Figure 23. 2013 Northeast mid-water trawl observed tows (A) and incidental takes (B).

Figure 24. 2014 Northeast mid-water trawl observed tows (A) and incidental takes (B).

Figure 25. 2015 Northeast mid-water trawl observed tows (A) and incidental takes (B).

Figure 26. 2011 mid-Atl. mid-water trawl observed tows (A) and incidental takes (B).

Figure 27. 2012 mid-Atl. mid-water trawl observed tows (A) and incidental takes (B).

Figure 28. 2013 mid-Atl. mid-water trawl observed tows (A) and incidental takes (B).

Figure 29. 2014 mid-Atl. mid-water trawl observed tows (A) and incidental takes (B).

Figure 30. 2015 mid-Atl. mid-water trawl observed tows (A) and incidental takes (B).

Figure 31. 2011 Atlantic herring purse seine observed hauls (A) and incidental takes (B).

Figure 32. 2012 Atlantic herring purse seine observed hauls (A) and incidental takes (B).

Figure 33. 2013 Atlantic herring purse seine observed hauls (A) and incidental takes (B).

Figure 34. 2014 Atlantic herring purse seine observed hauls (A) and incidental takes (B).

Figure 35. 2015 Atlantic herring purse seine observed hauls (A) and incidental takes (B).

Figure 36. 2011Observed sets and marine mammal interactions in the pelagic longline fishery - U.S. Atlantic coast.

Figure 37. 2012 Observed sets and marine mammal interactions in the pelagic longline fishery - U.S. Atlantic coast.

Figure 38. 2013 Observed sets and marine mammal interactions in the pelagic longline fishery - U.S. Atlantic coast.

Figure 39. 2014 Observed sets and marine mammal interactions in the pelagic longline fishery - U.S. Atlantic coast.

Figure 40. 2015 Observed sets and marine mammal interactions in the pelagic longline fishery - U.S. Atlantic coast.

Figure 41. 2011 Observed sets and marine mammal interactions in the pelagic longline fishery - Gulf of Mexico.

Figure 42. 2012 Observed sets and marine mammal interactions in the pelagic longline fishery - Gulf of Mexico.

Figure 43. 2013 Observed sets and marine mammal interactions in the pelagic longline fishery - Gulf of Mexico.

Figure 44. 2014 Observed sets and marine mammal interactions in the pelagic longline fishery - Gulf of Mexico.

Figure 45. 2015 Observed sets and marine mammal interactions in the pelagic longline fishery - Gulf of Mexico.

Figure 1. 2011 Northeast sink gillnet observed hauls (A) and observed takes (B).

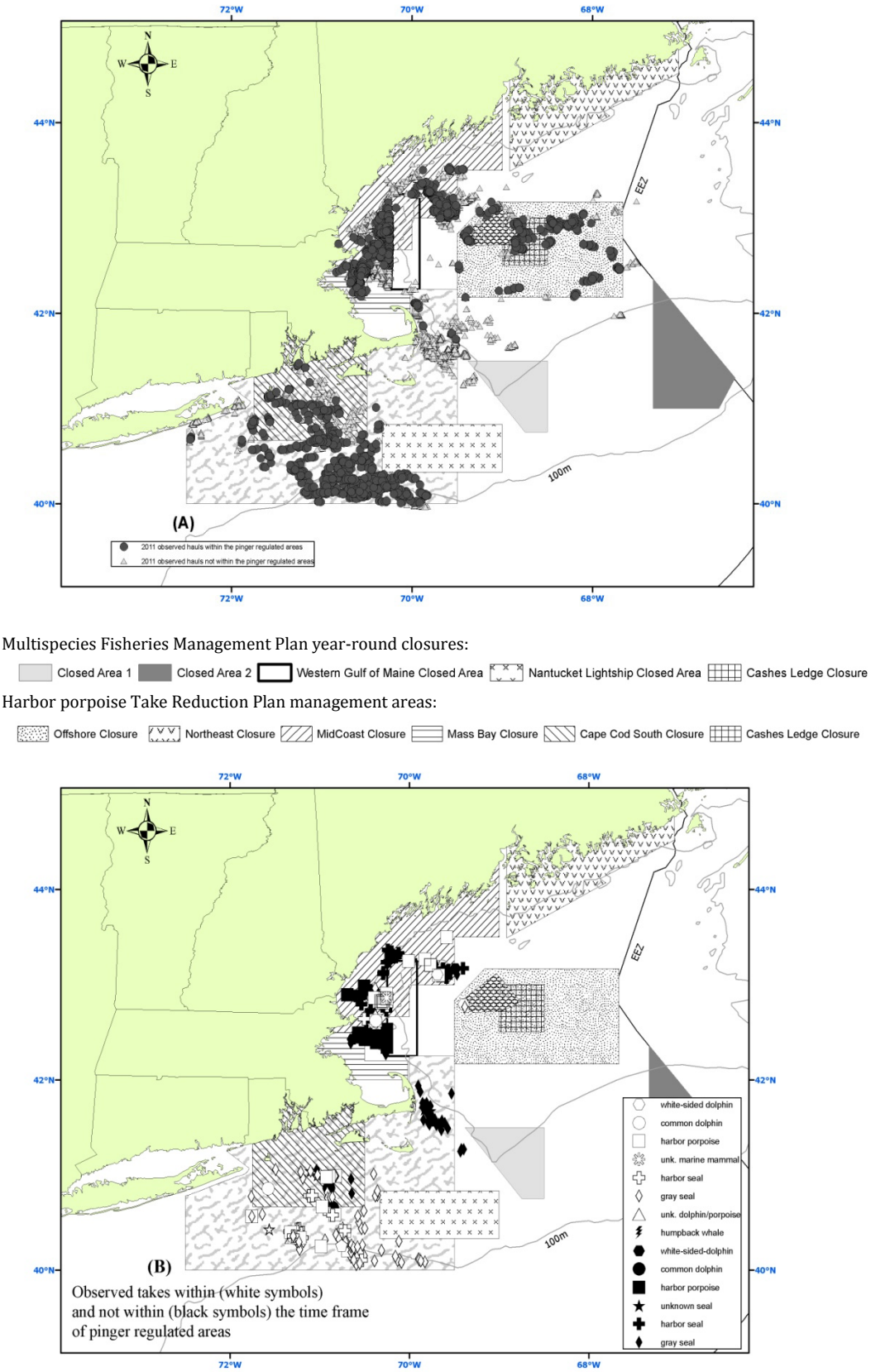
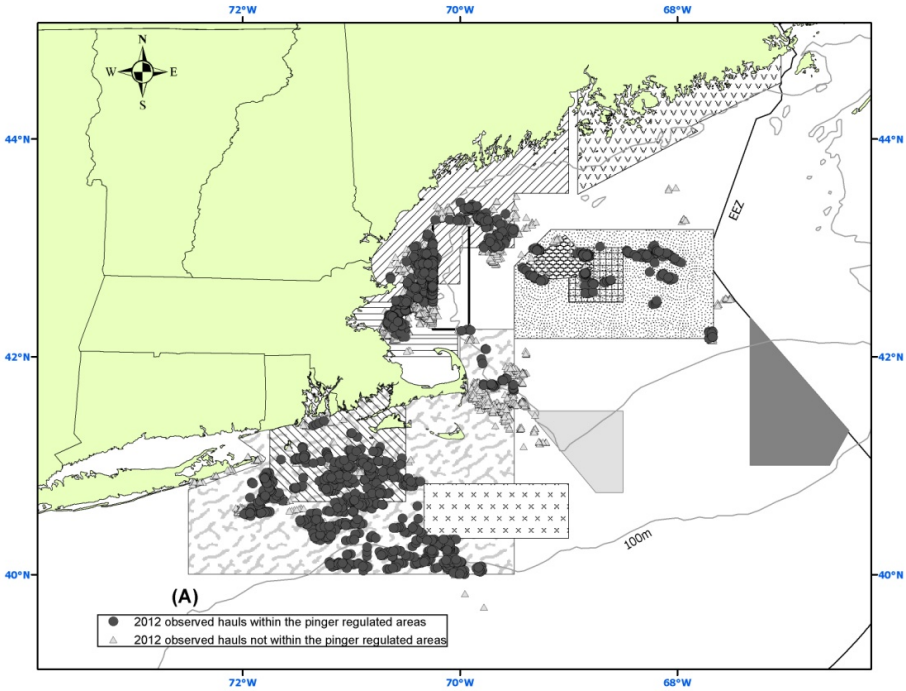


Figure 2. 2012 Northeast sink gillnet observed hauls (A) and observed takes (B).



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

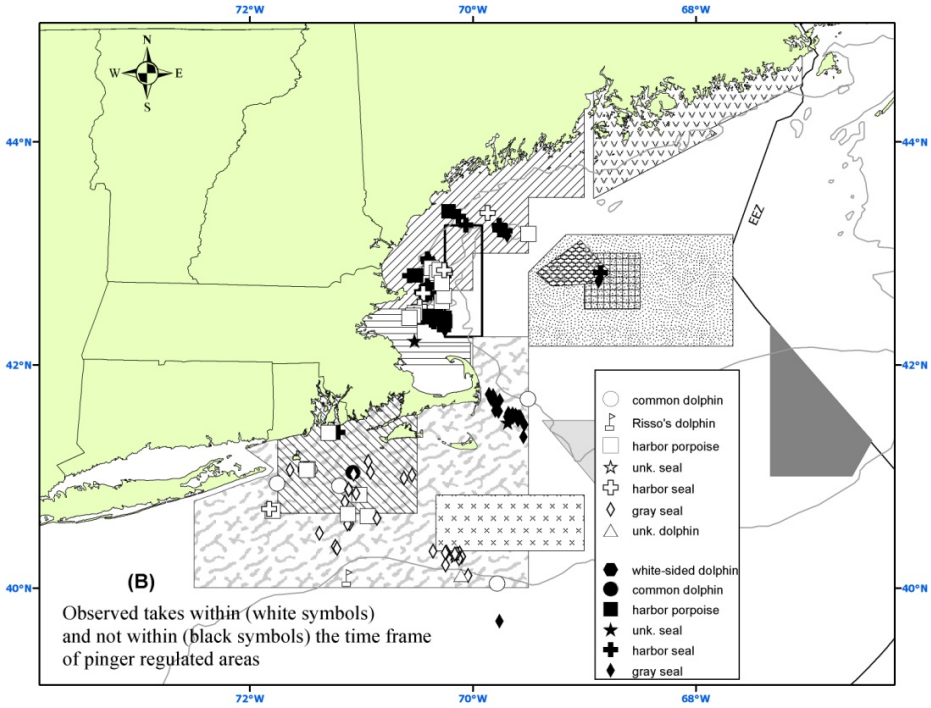
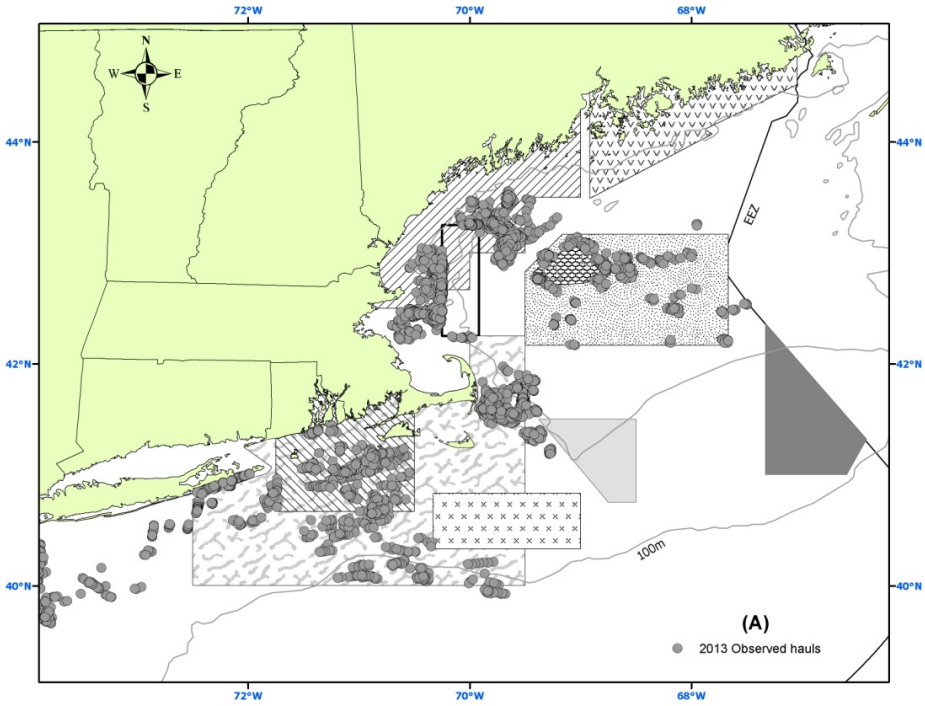


Figure 3. 2013 Northeast sink gillnet observed hauls (A) and observed takes (B).



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

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Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cape Cod South Closure | Cashes Ledge Closure

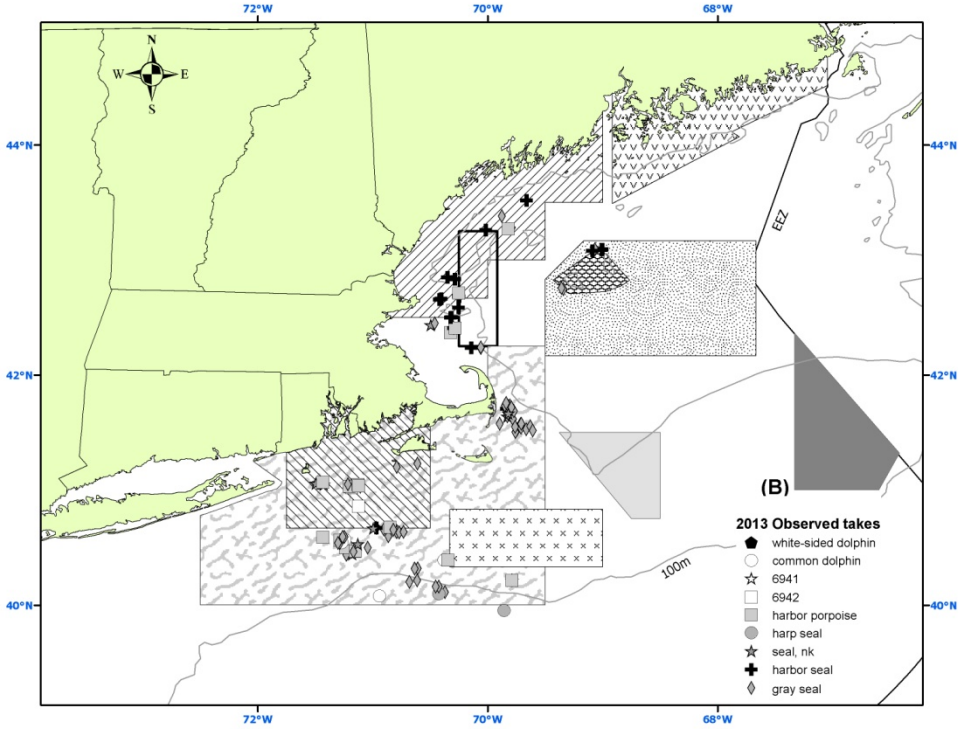
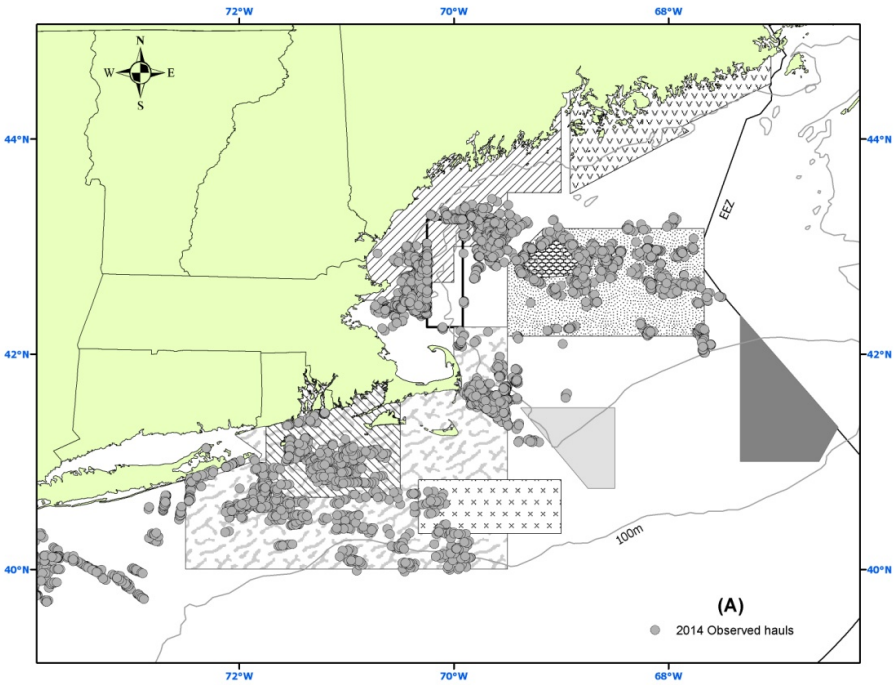


Figure 4. 2014 Northeast sink gillnet observed hauls (A) and observed takes (B).



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Harbor porpoise Take Reduction Plan management areas:

Offshore Closure | Northeast Closure | MidCoast Closure | Mass Bay Closure | Cape Cod South Closure | Cashes Ledge Closure

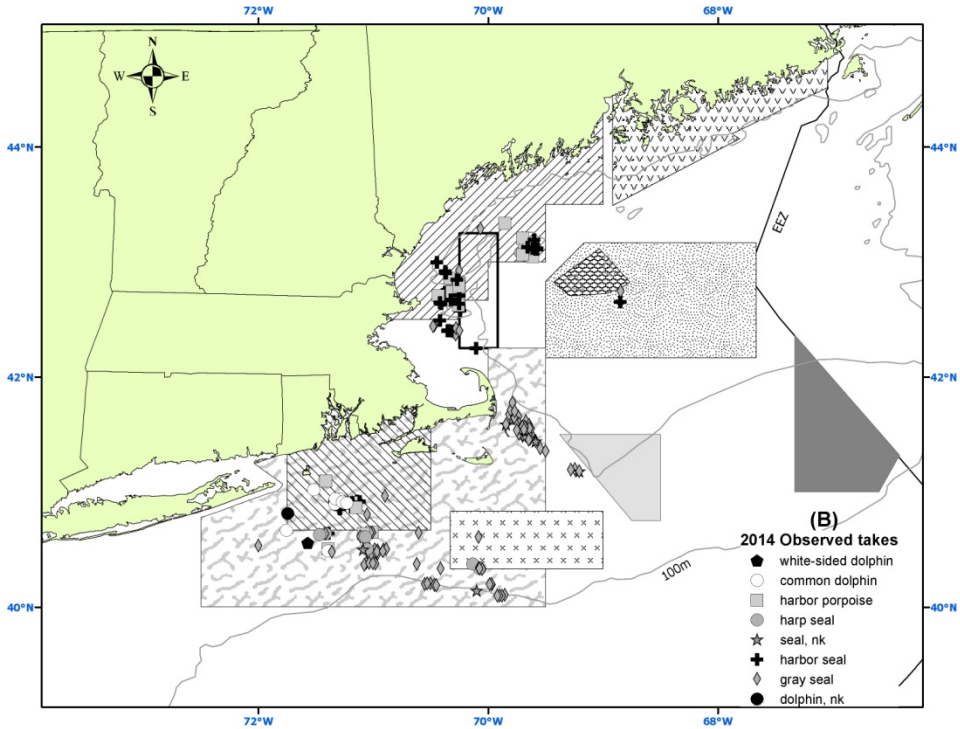
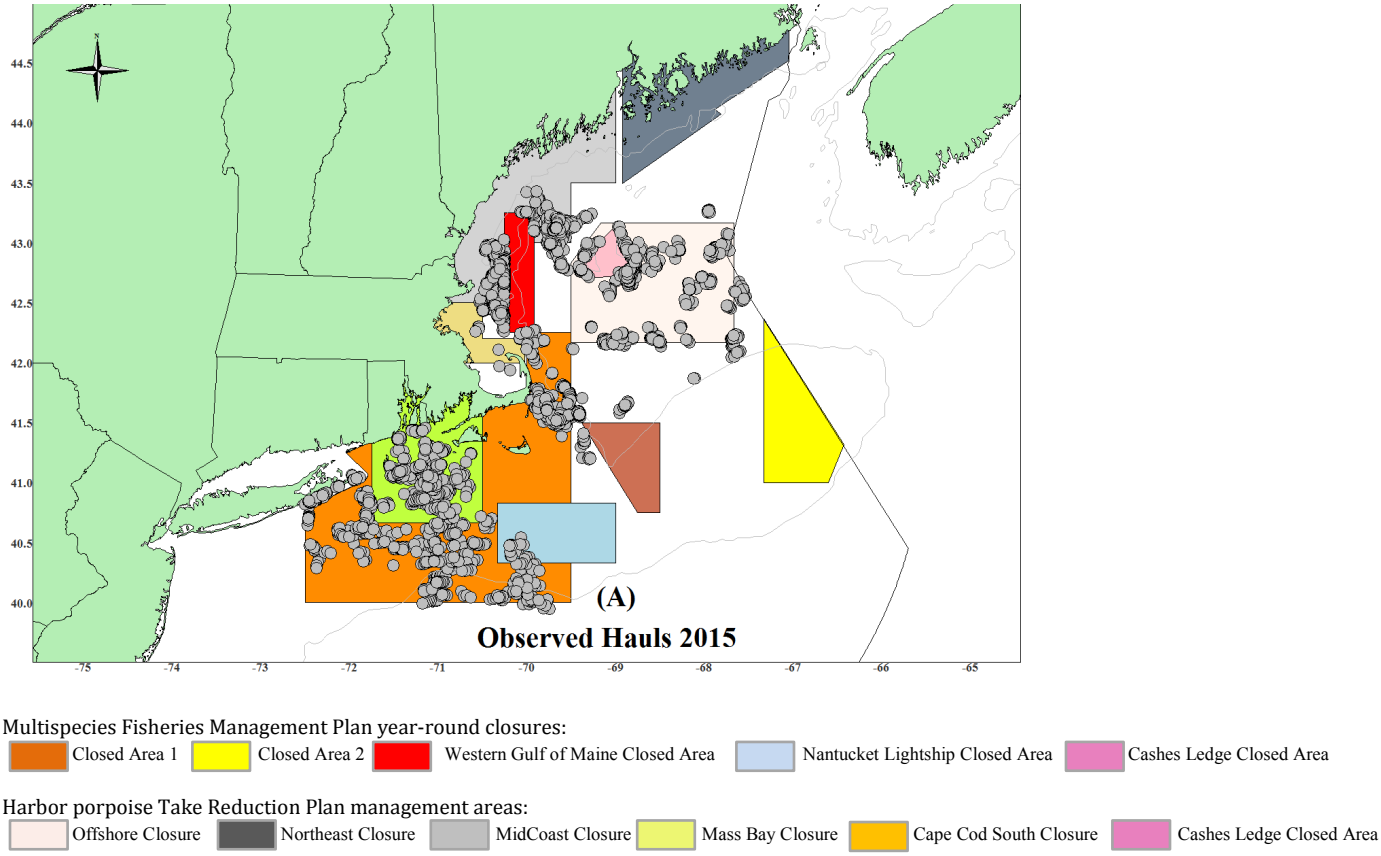


Figure 5. 2015 Northeast sink gillnet observed hauls (A) and observed takes (B).



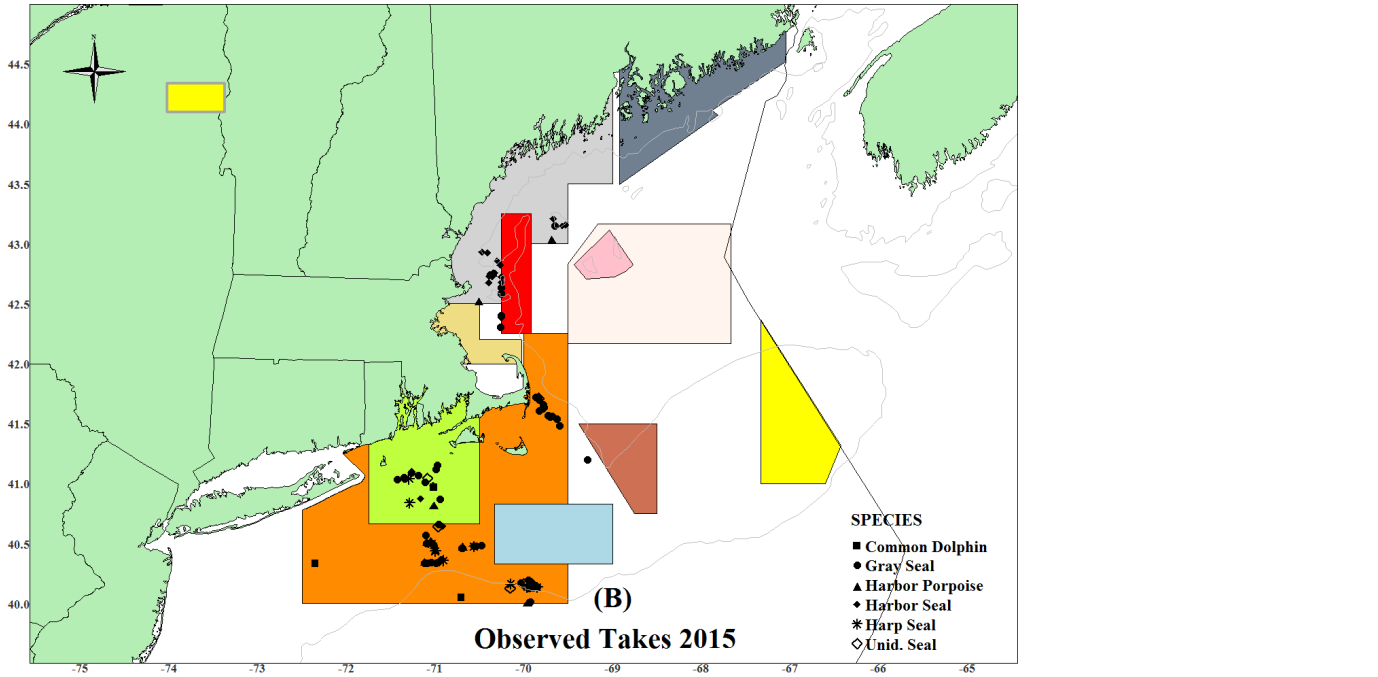
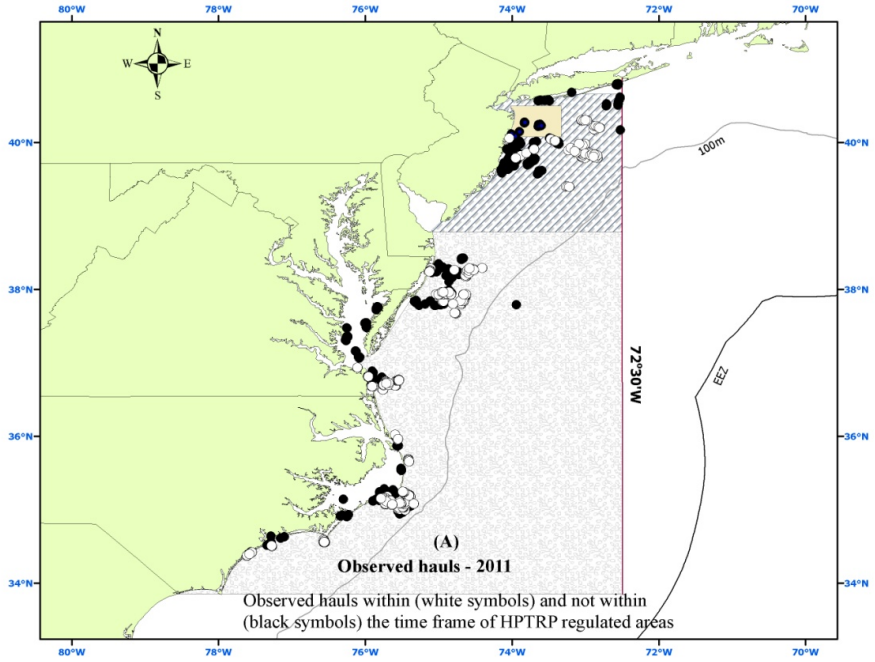


Figure 6. 2011 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

Southern mid-Atlantic waters | New Jersey Mudhole | waters off New Jersey

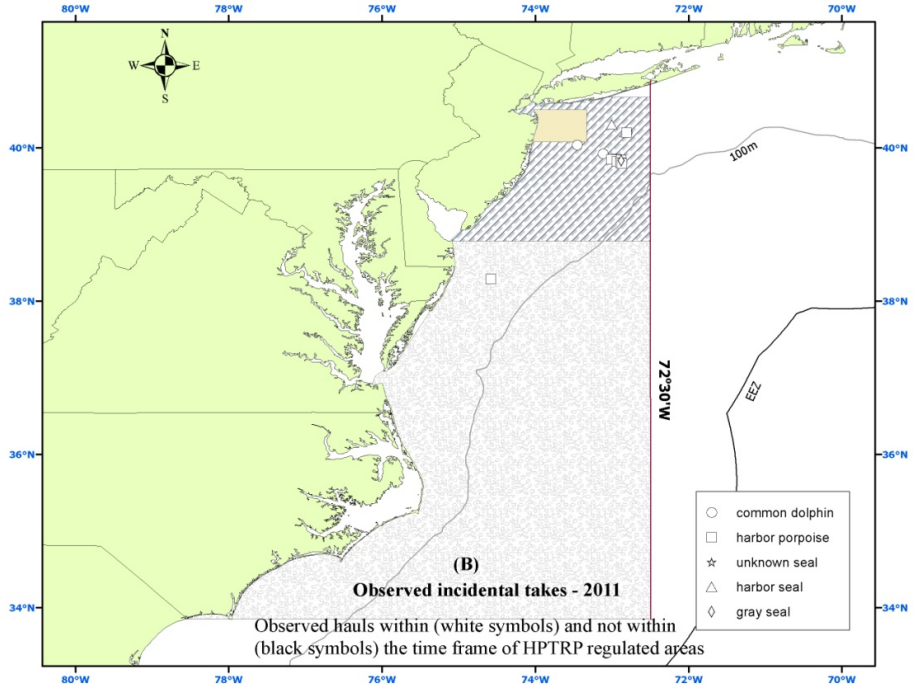
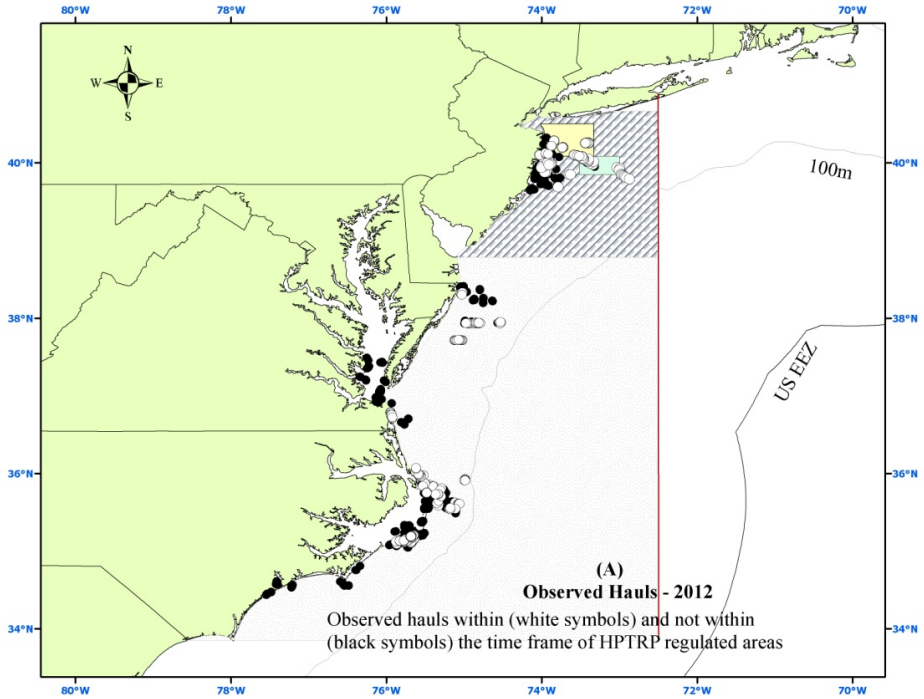


Figure 7. 2012 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

Southern mid-Atlantic waters | New Jersey Mudhole | waters off New Jersey

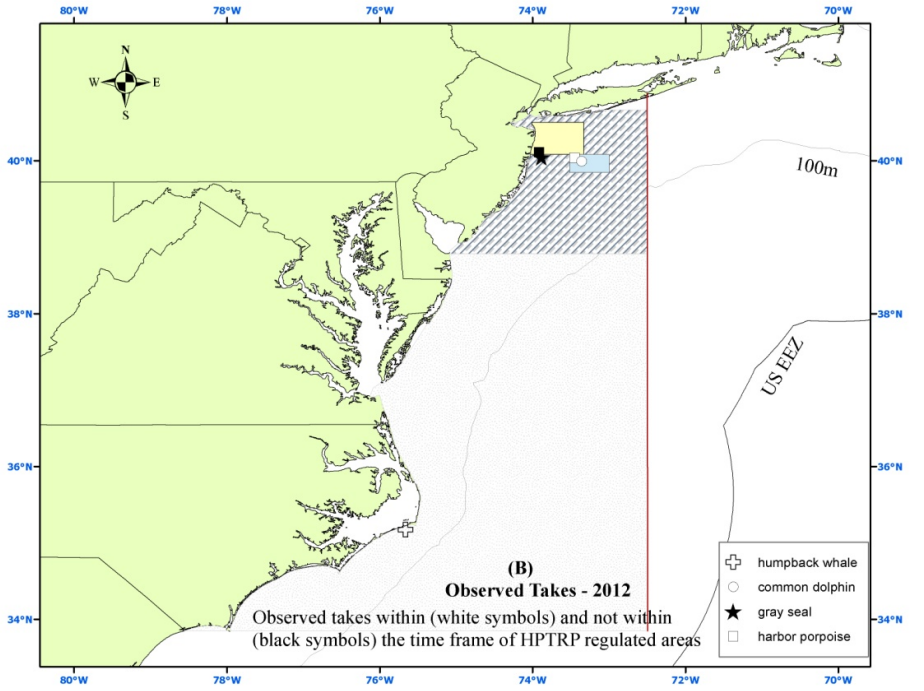


Figure 8. 2013 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).

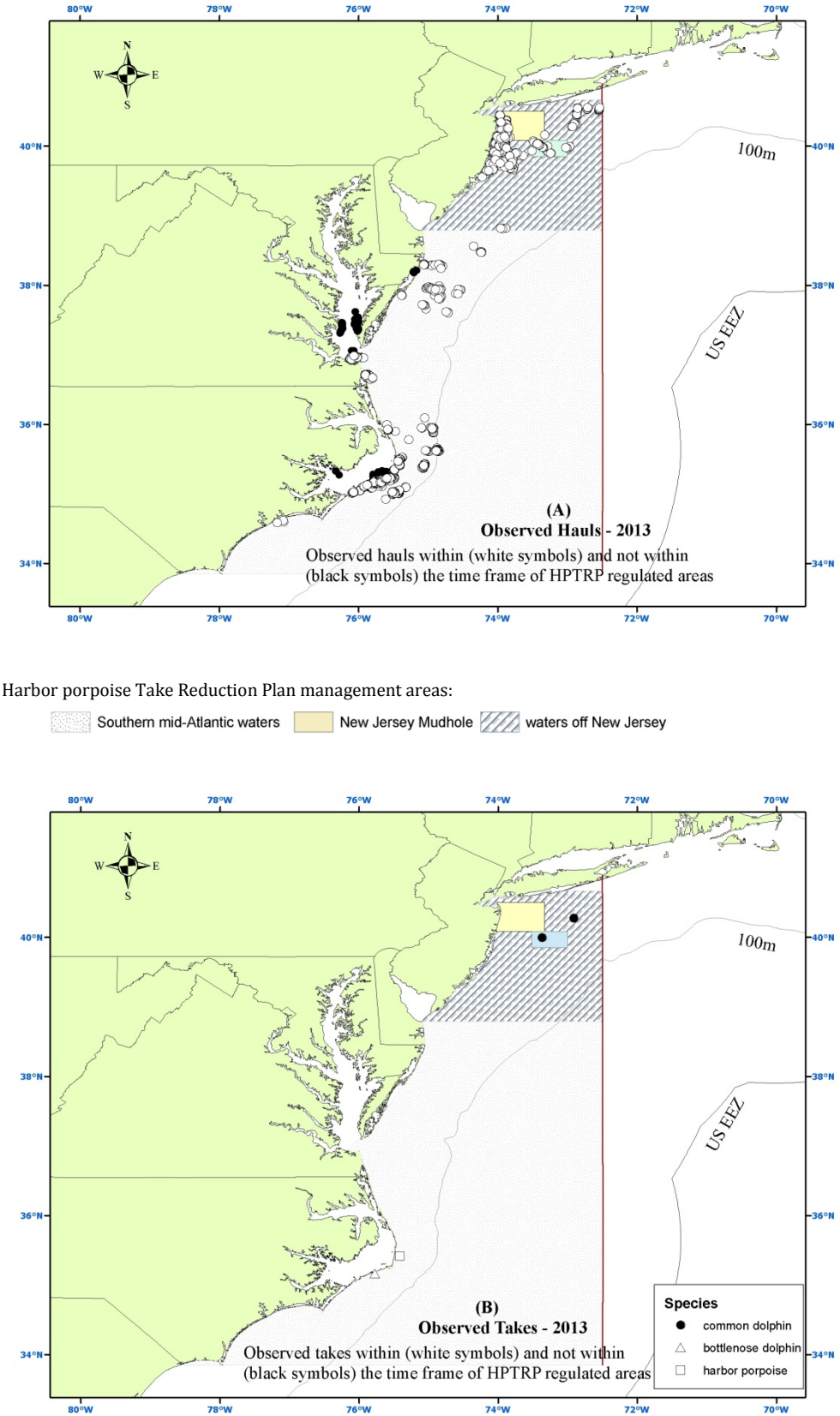
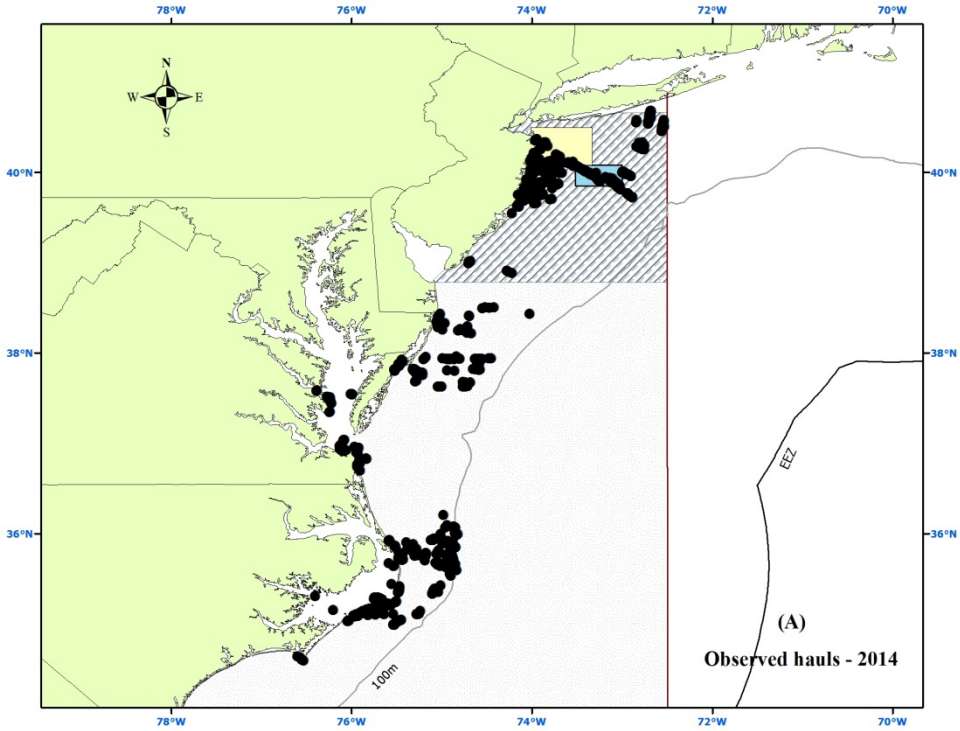


Figure 9. 2014 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

Southern mid-Atlantic waters | New Jersey Mudhole | waters off New Jersey

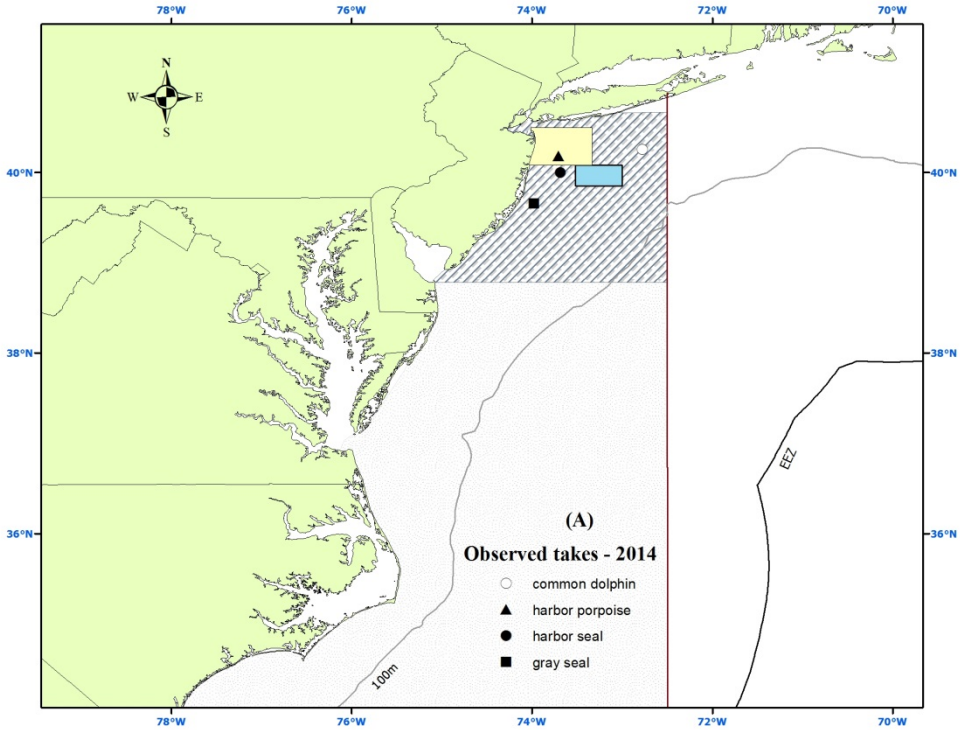
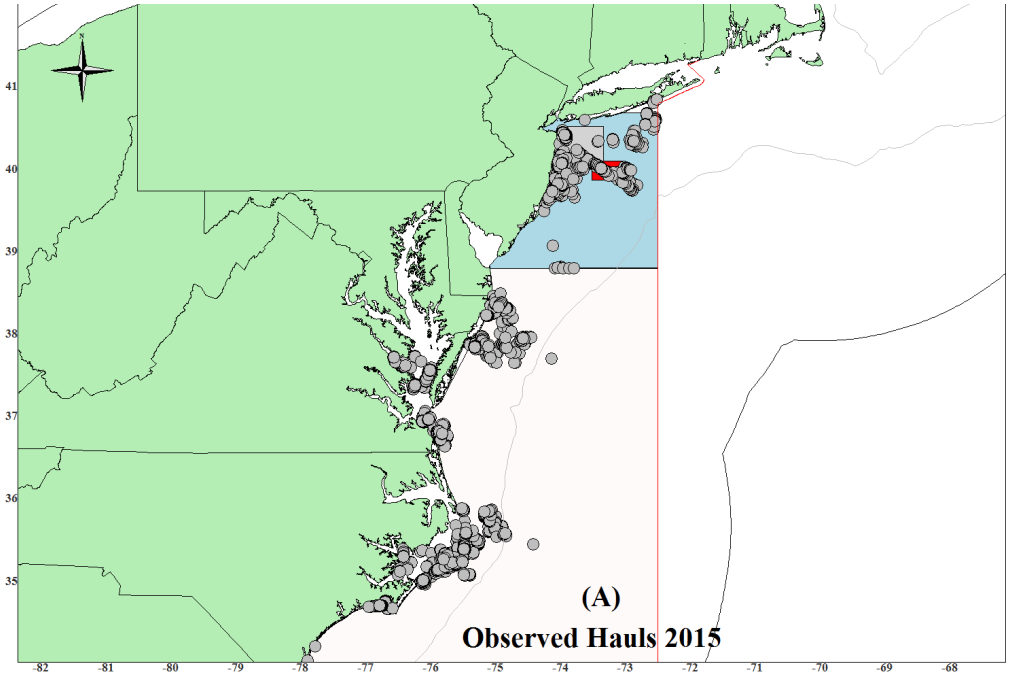


Figure 10. 2015 Mid-Atlantic gillnet observed hauls (A) and observed takes (B).



Harbor porpoise Take Reduction Plan management areas:

Southern mid-Atlantic waters | New Jersey Mudhole | Mudhole South | waters off New Jersey

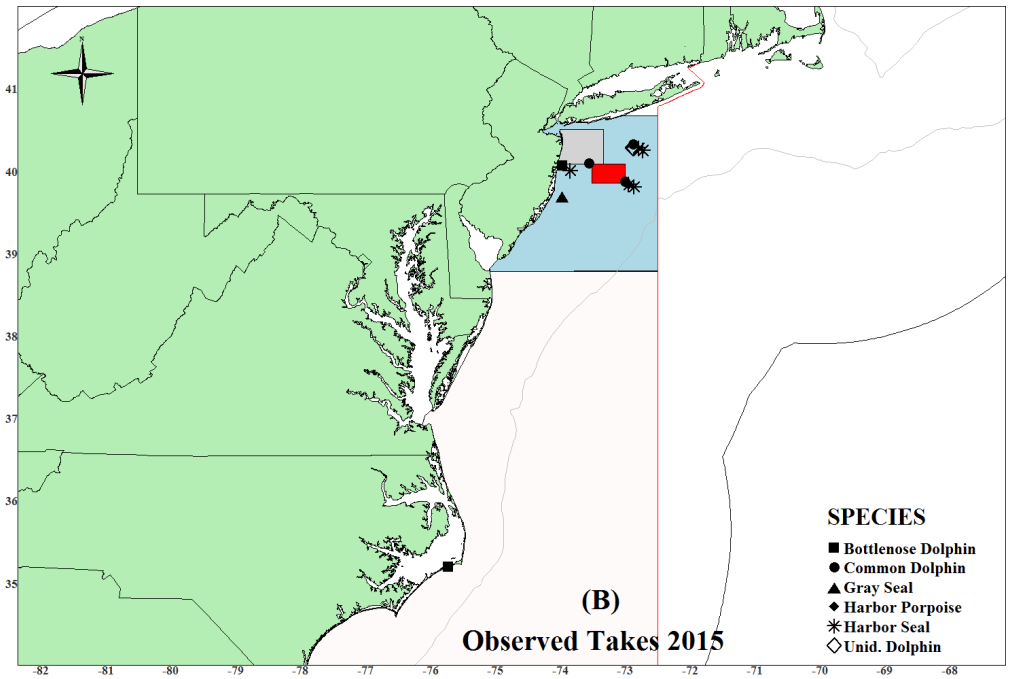


Figure 11. 2011 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

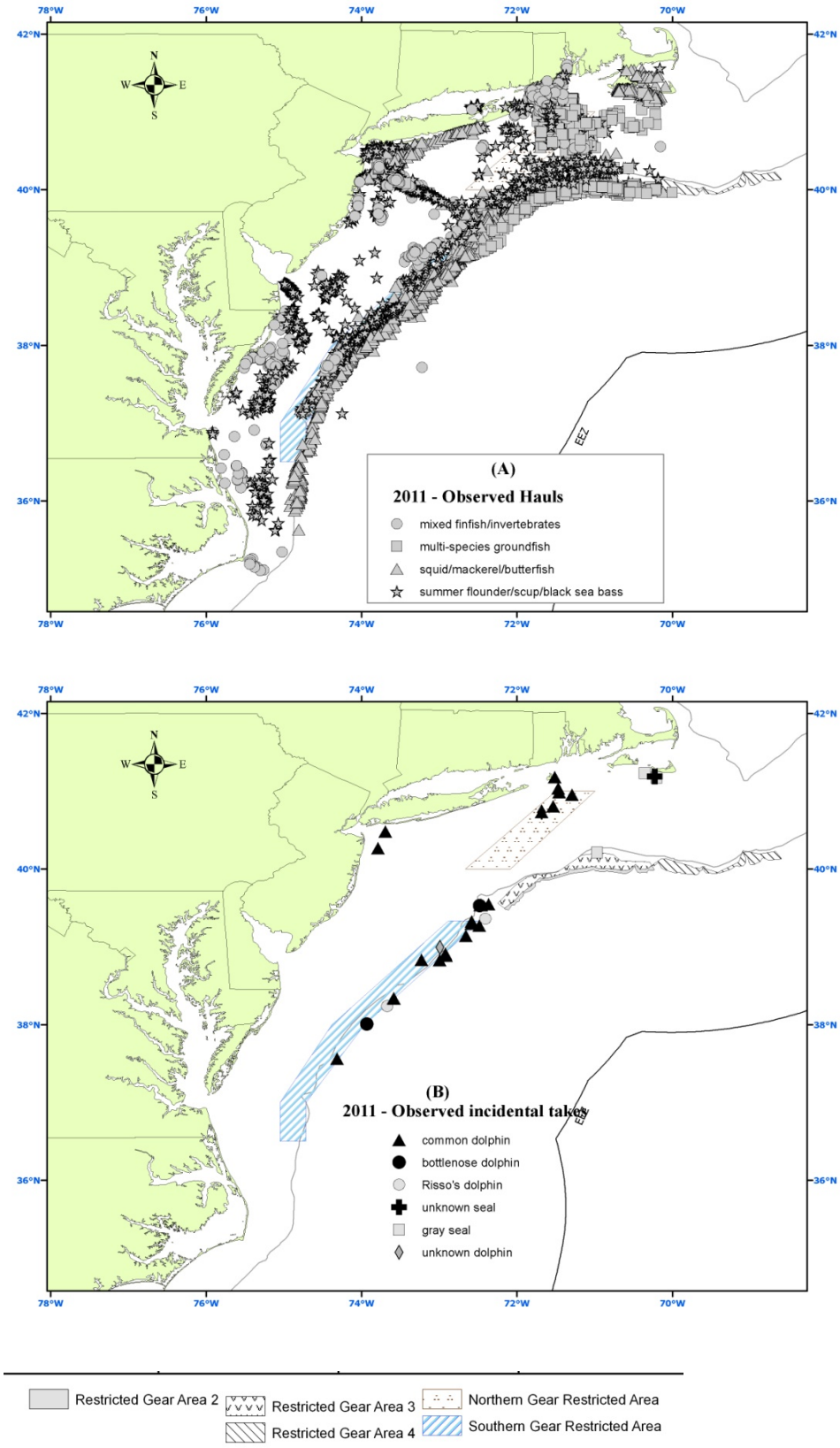


Figure 12. 2012 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

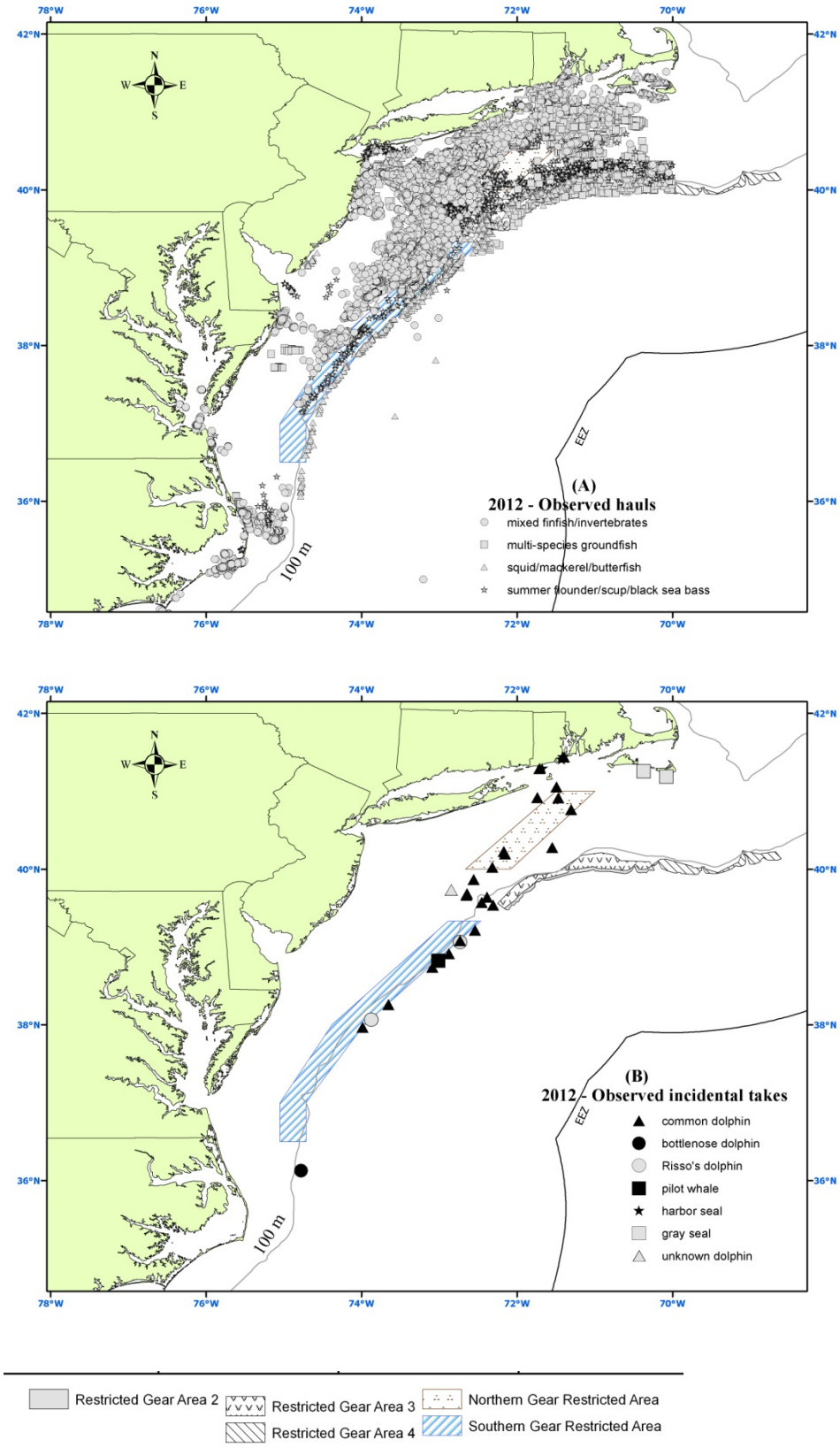


Figure 13. 2013 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

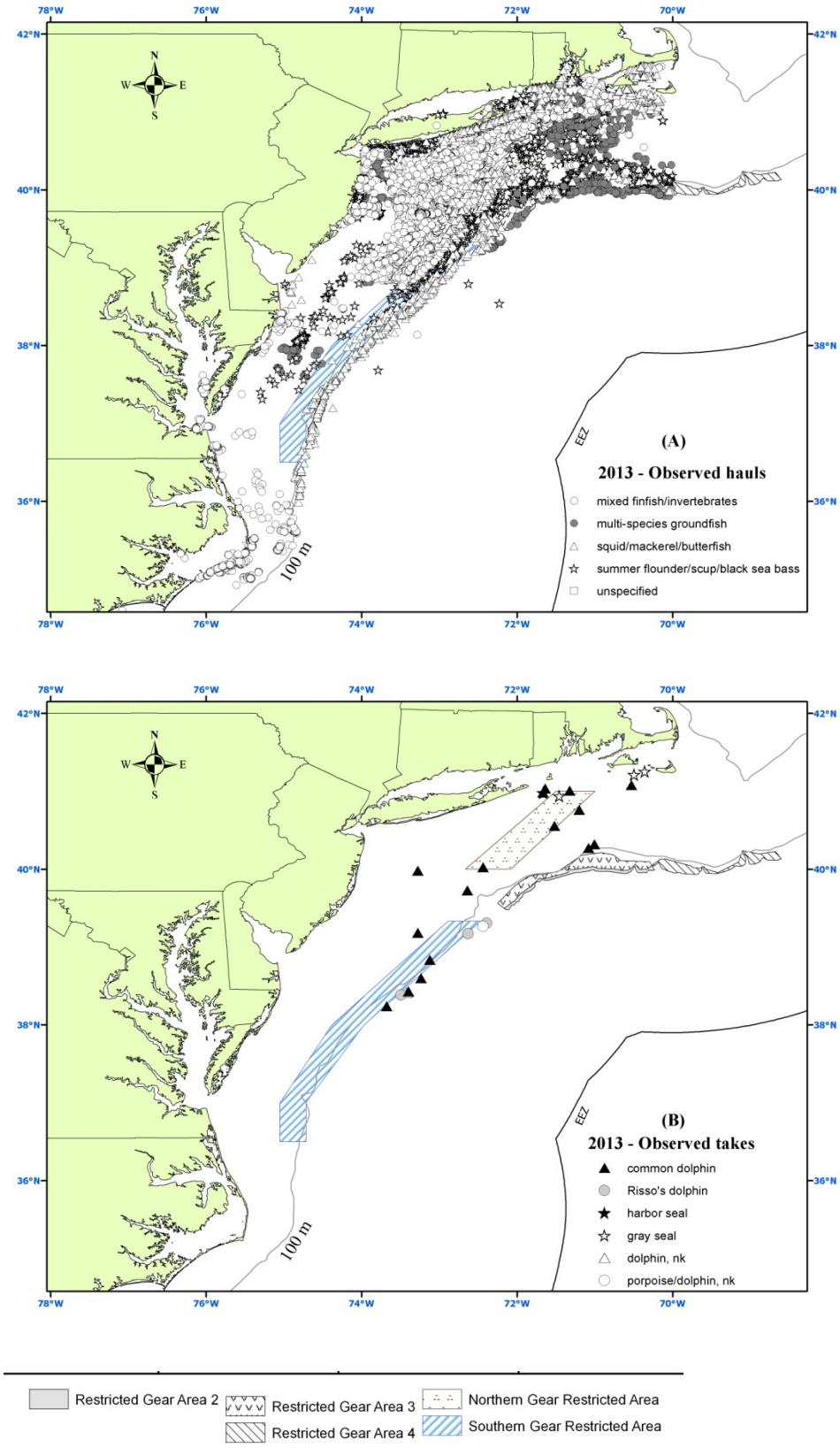


Figure 14. 2014 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

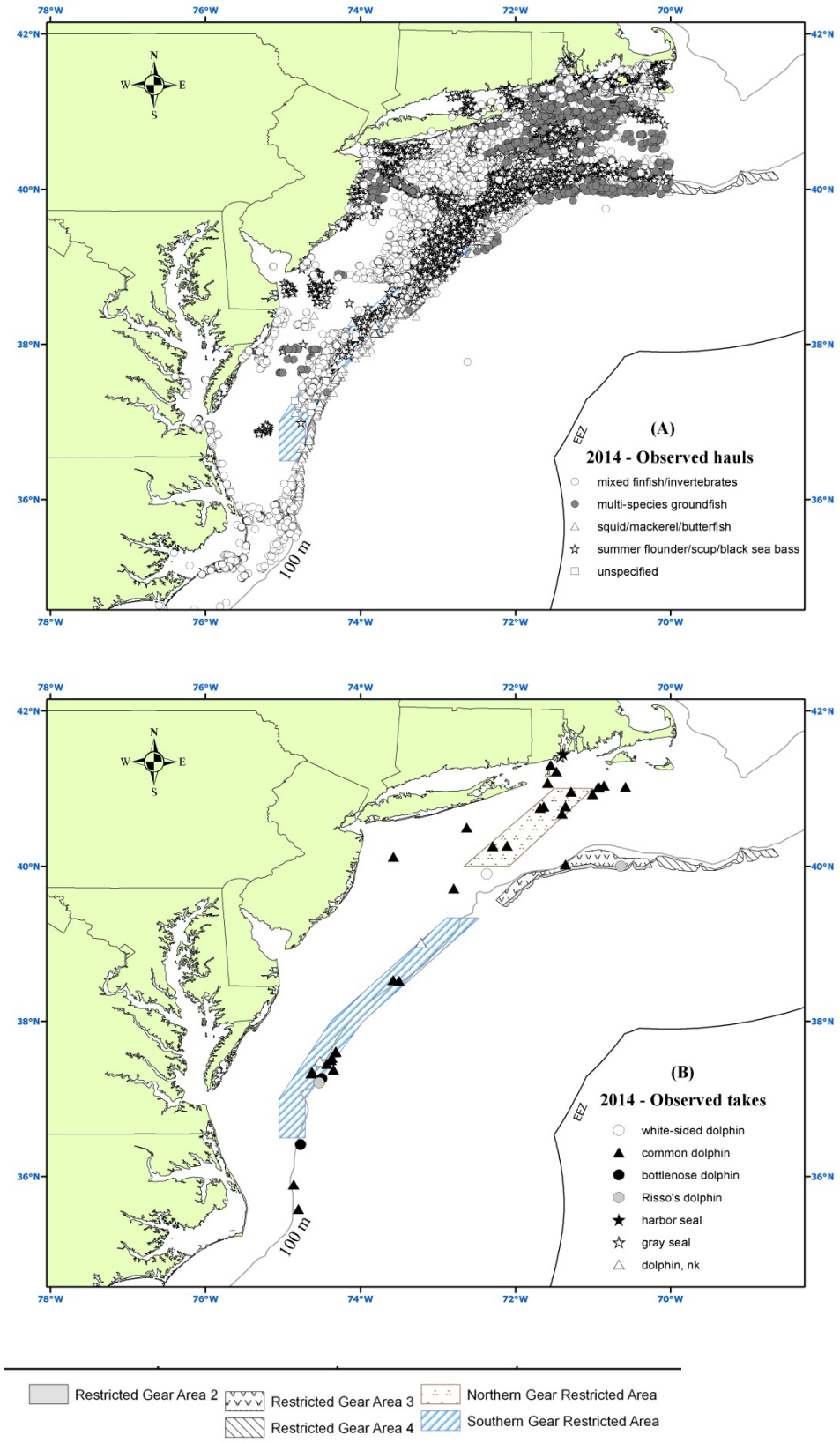


Figure 15. 2015 Mid-Atlantic bottom trawl observed tows (A) and observed takes (B).

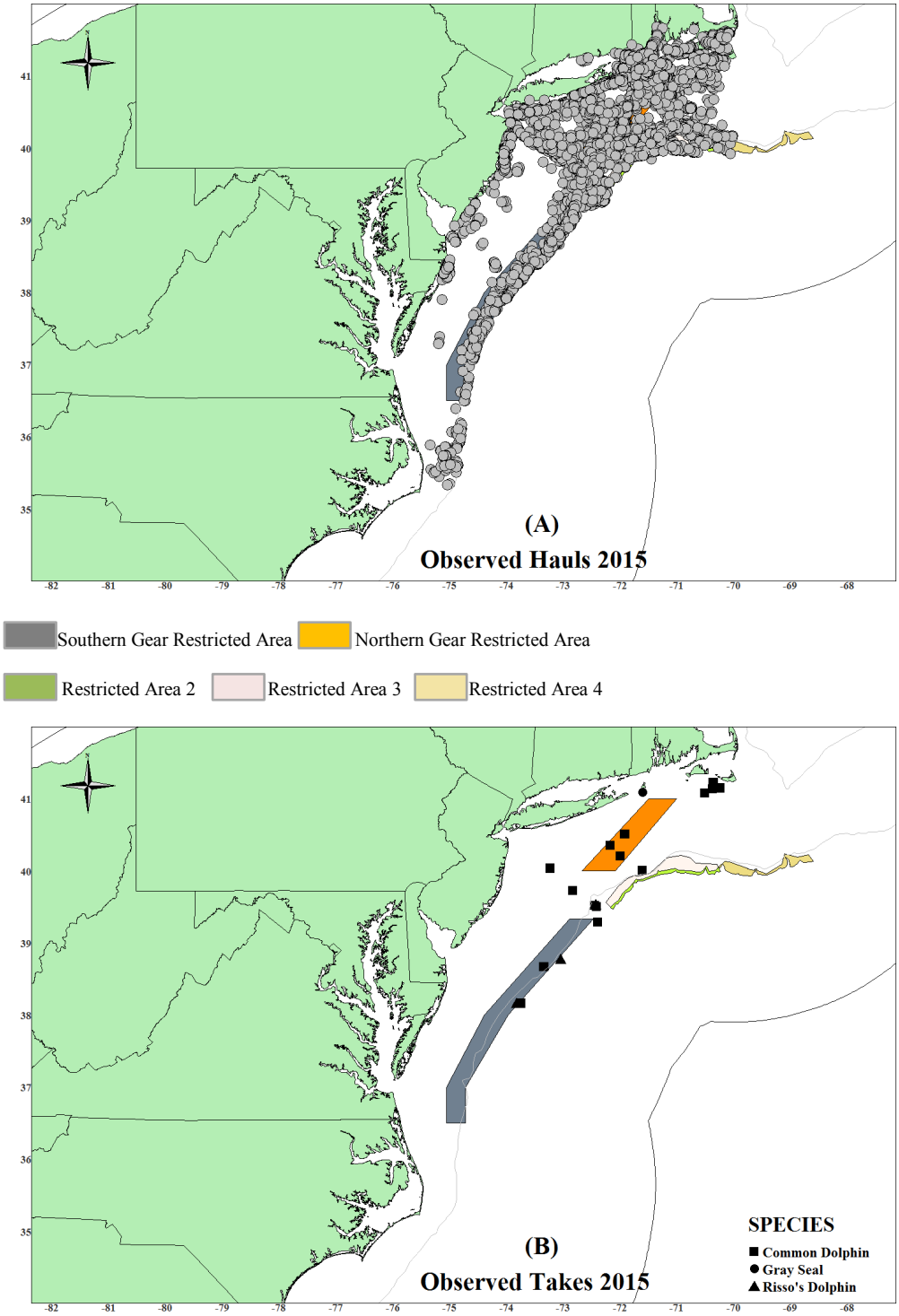
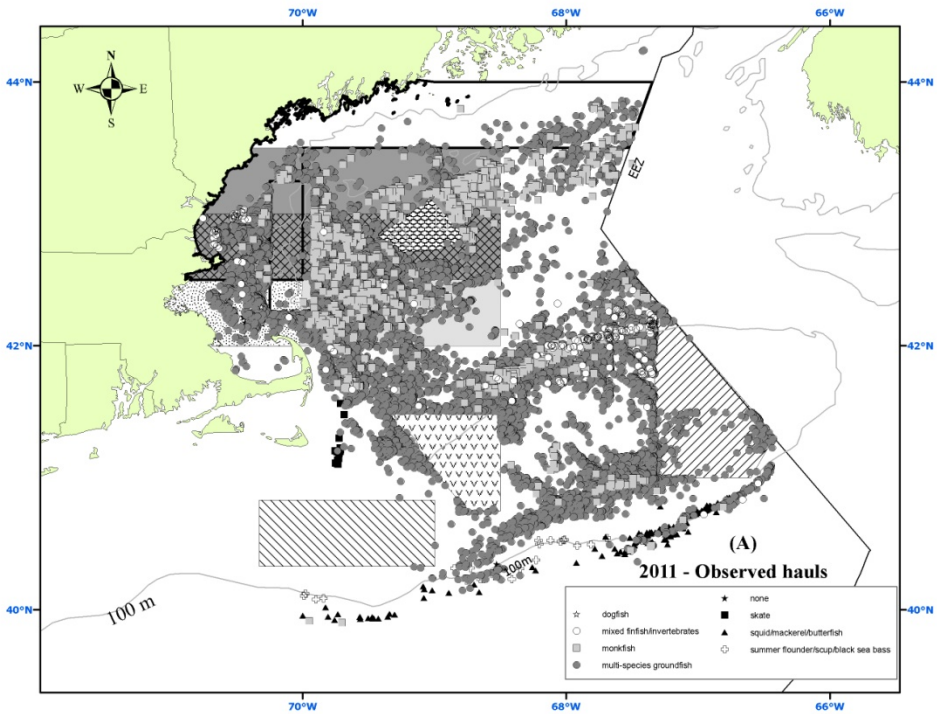
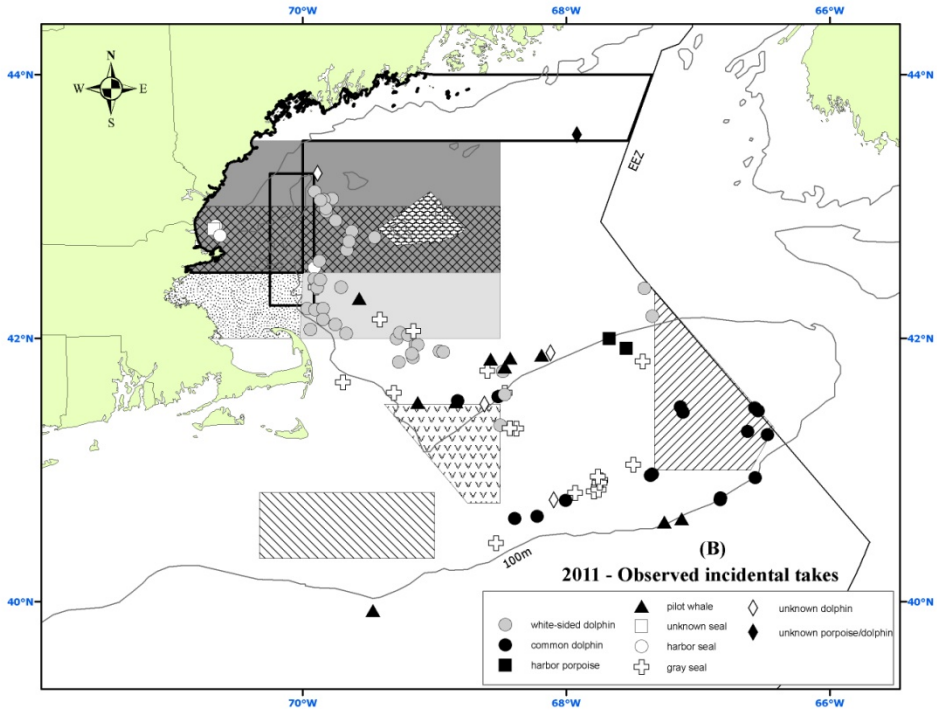


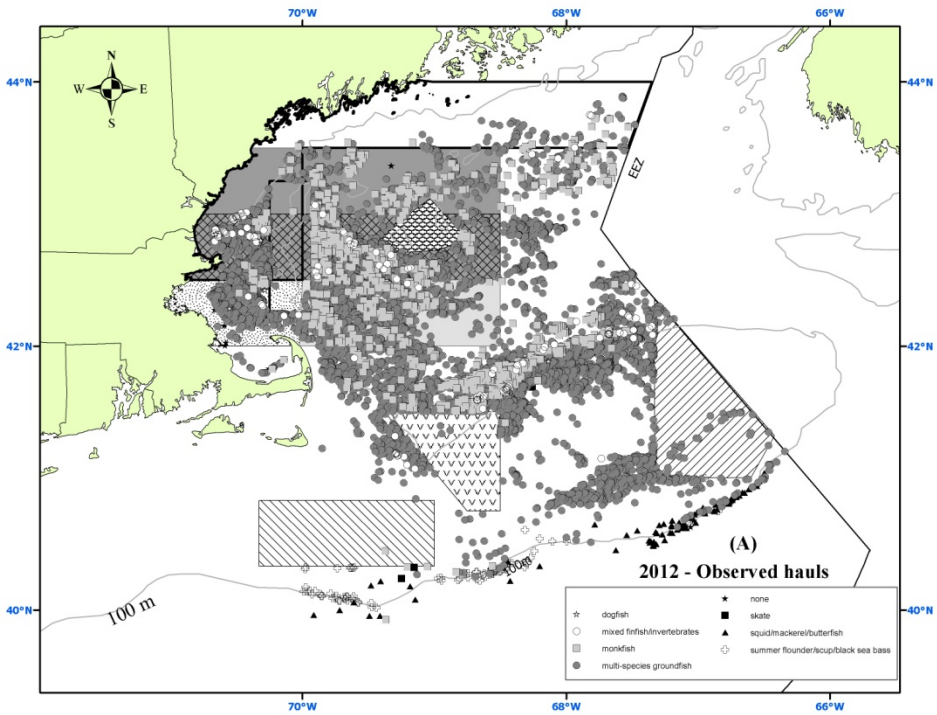
Figure 16. 2011 Northeast bottom trawl observed tows (A) and observed takes (B).

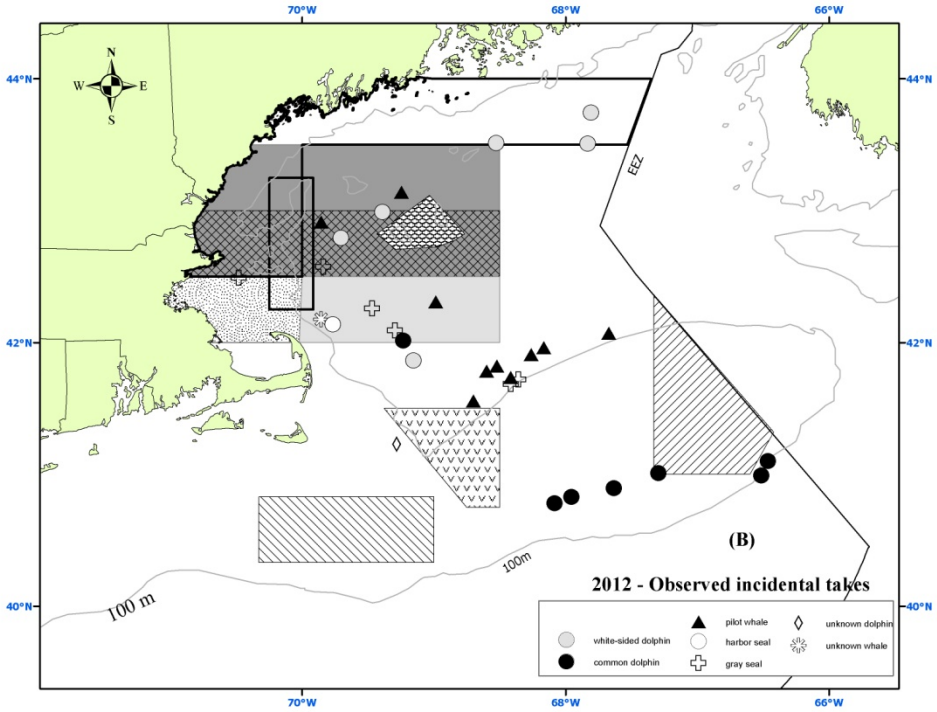




Closed Area 1 | Rolling Closure Area 1 | Rolling Closure Area 3 | Rolling Closure Area 5
Closed Area 2 | Rolling Closure Area 2 | Rolling Closure Area 4 | Western Gulf of Maine Closed Area
Cashes Ledge | Nantucket Lightship Closed Area

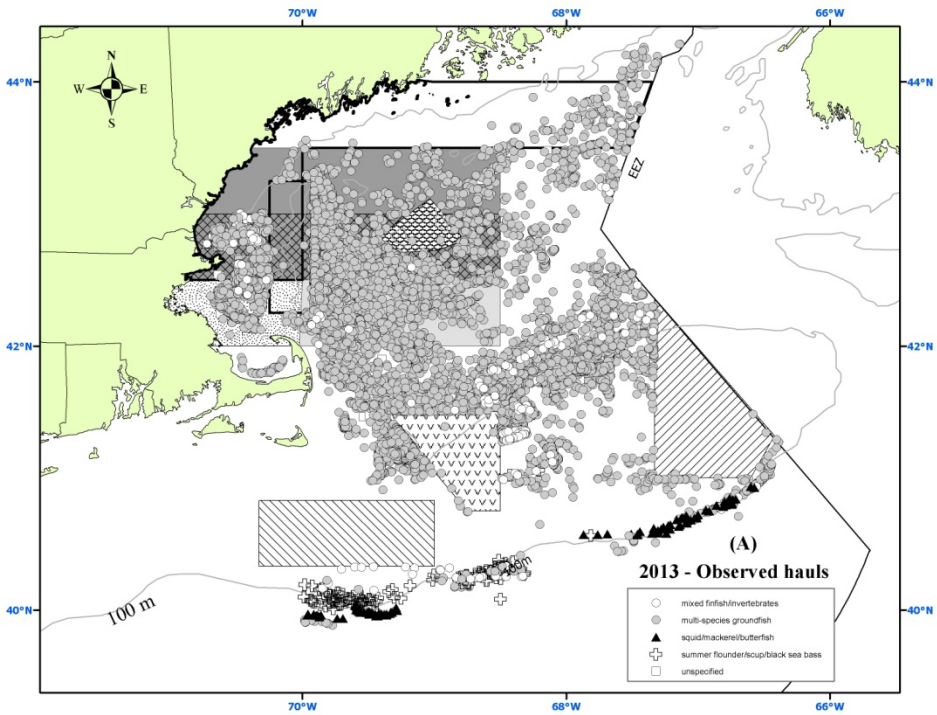
Figure 17. 2012 Northeast bottom trawl observed tows (A) and observed takes (B).

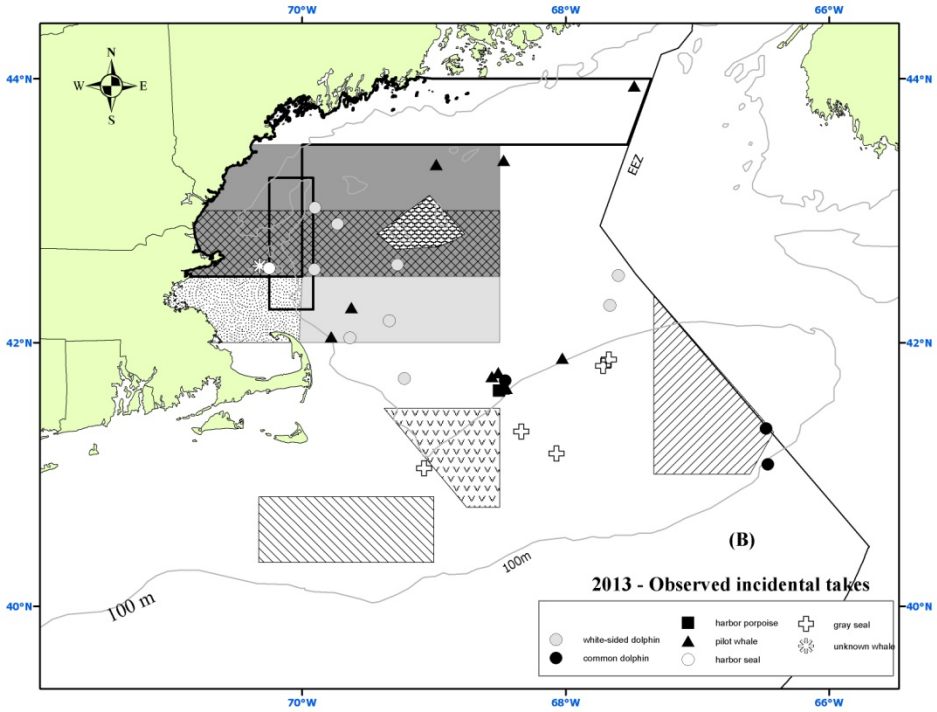




Closed Area 1
Rolling Closure Area 1
Rolling Closure Area 3
Rolling Closure Area 5
Closed Area 2
Rolling Closure Area 2
Rolling Closure Area 4
Western Gulf of Maine Closed Area
Cashes Ledge
Nantucket Lightship Closed Area

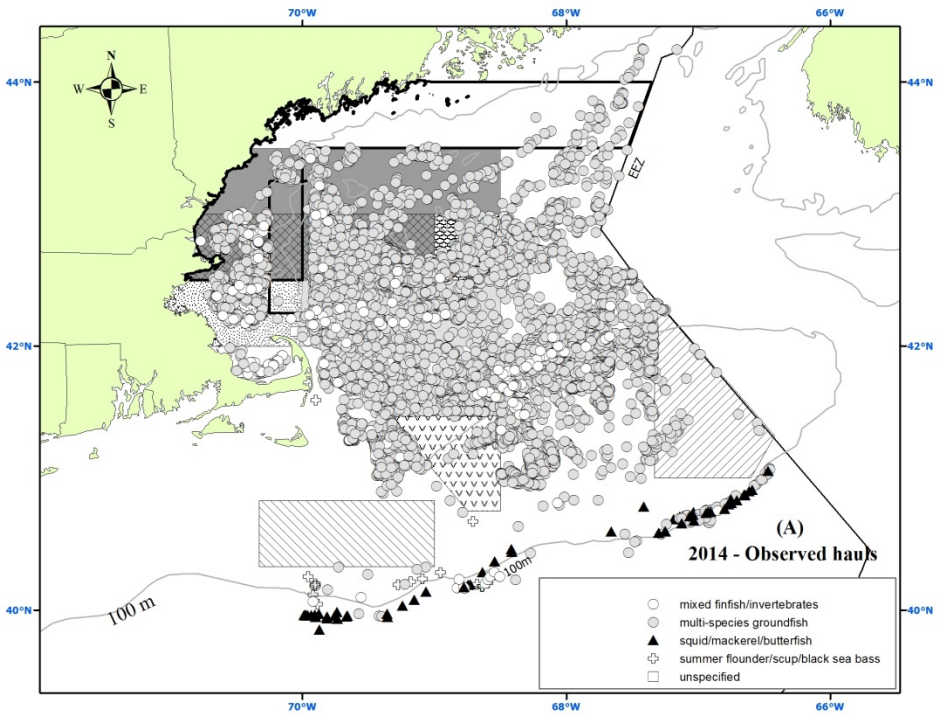
Figure 18. 2013 Northeast bottom trawl observed tows (A) and observed takes (B).

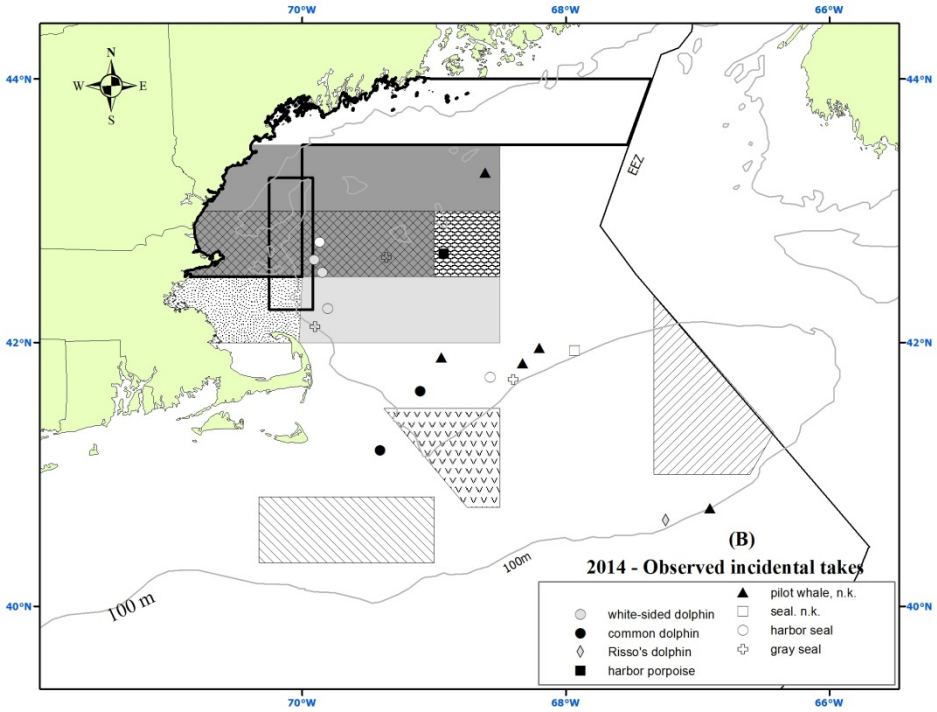




Closed Area 1 | Rolling Closure Area 1 | Rolling Closure Area 3 | Rolling Closure Area 5
Closed Area 2 | Rolling Closure Area 2 | Rolling Closure Area 4 | Western Gulf of Maine Closed Area
Cashes Ledge | Nantucket Lightship Closed Area

Figure 19. 2014 Northeast bottom trawl observed tows (A) and observed takes (B).





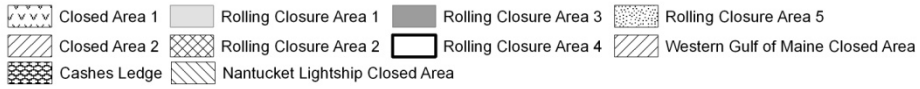


Figure 20. 2015 Northeast bottom trawl observed tows (A) and observed takes (B).

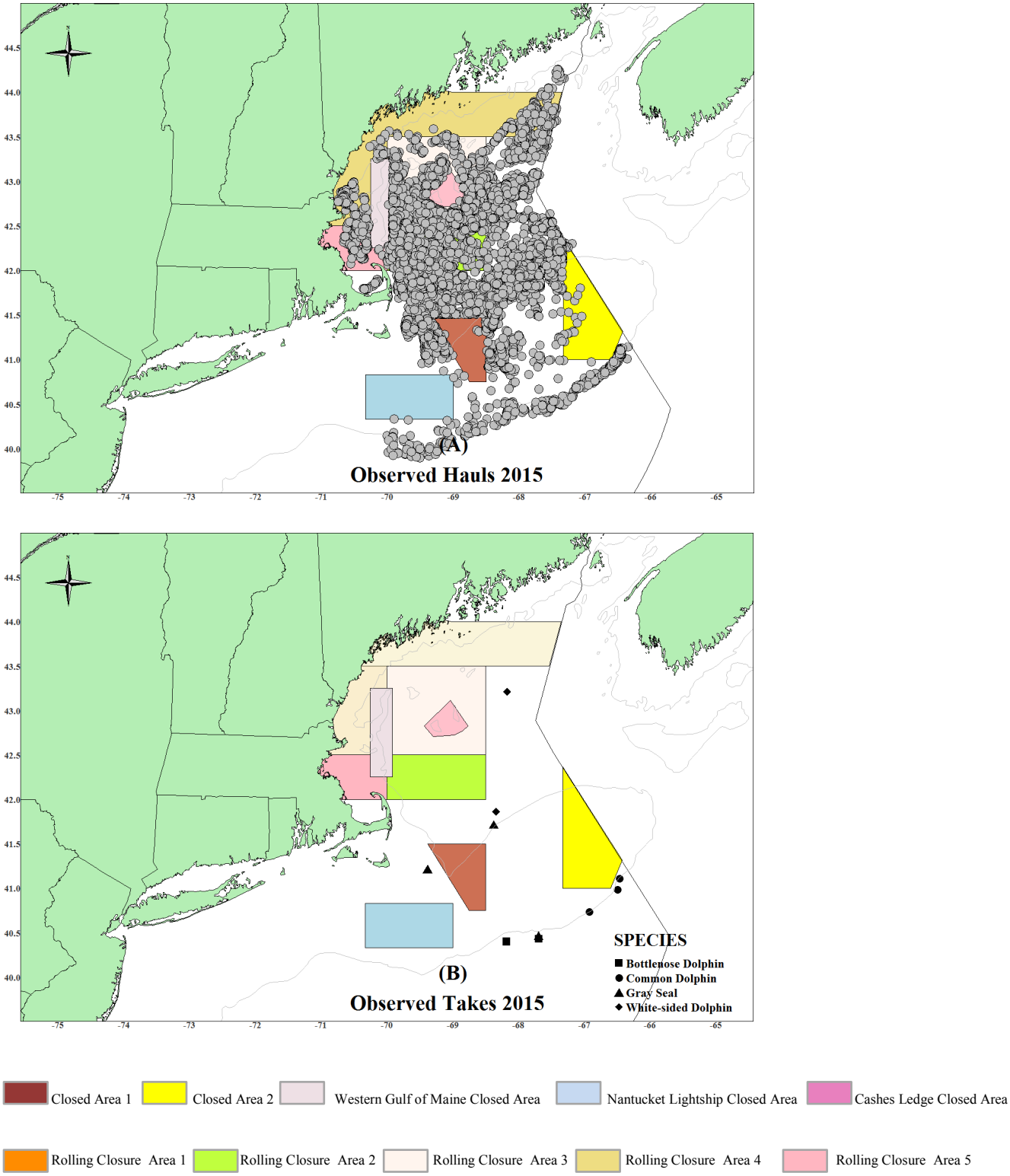


Figure 21. 2011 Northeast mid-water trawl observed tows (A) and observed takes (B).

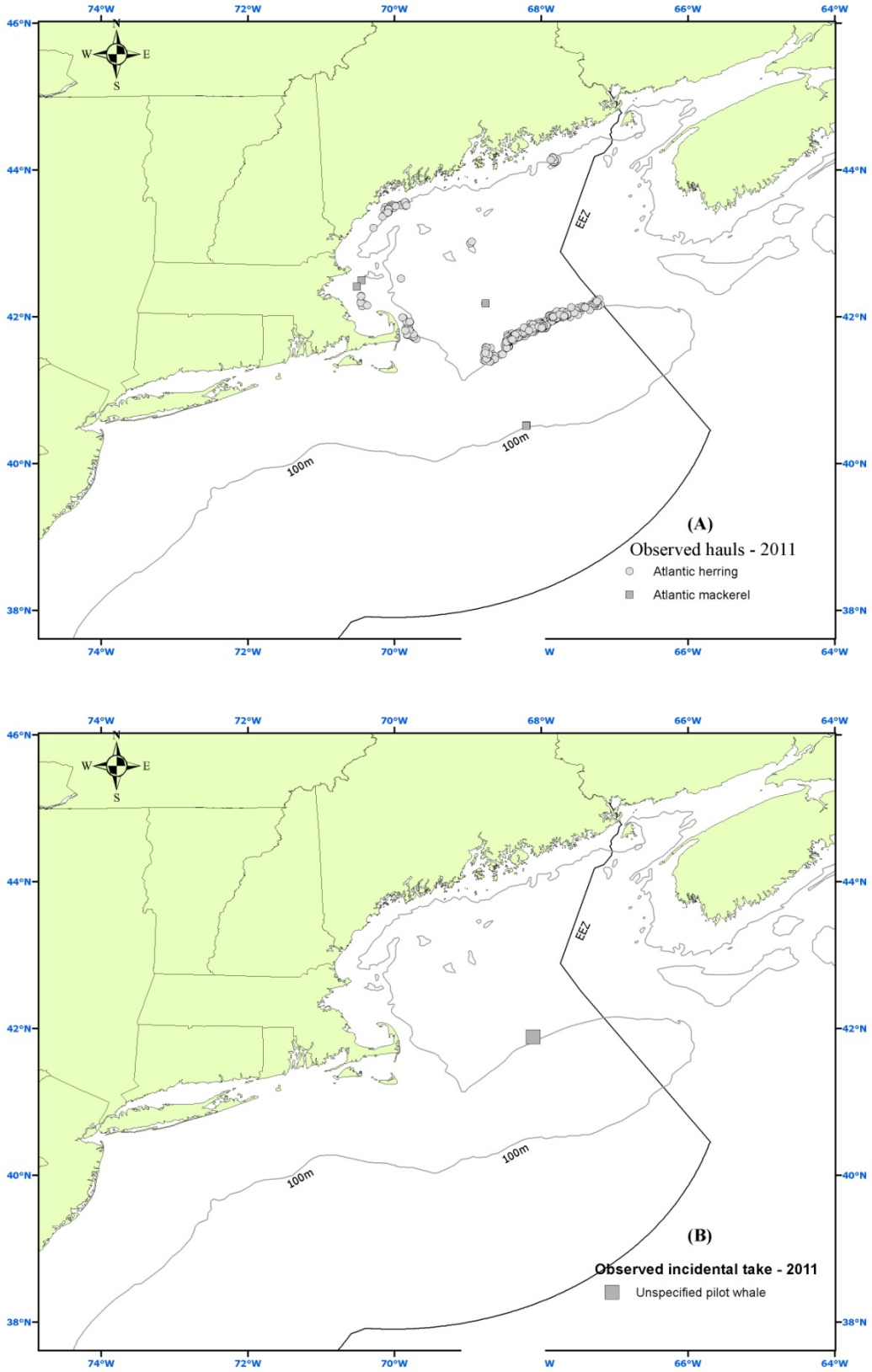


Figure 22. 2012 Northeast mid-water trawl observed tows (A) and observed takes (B).

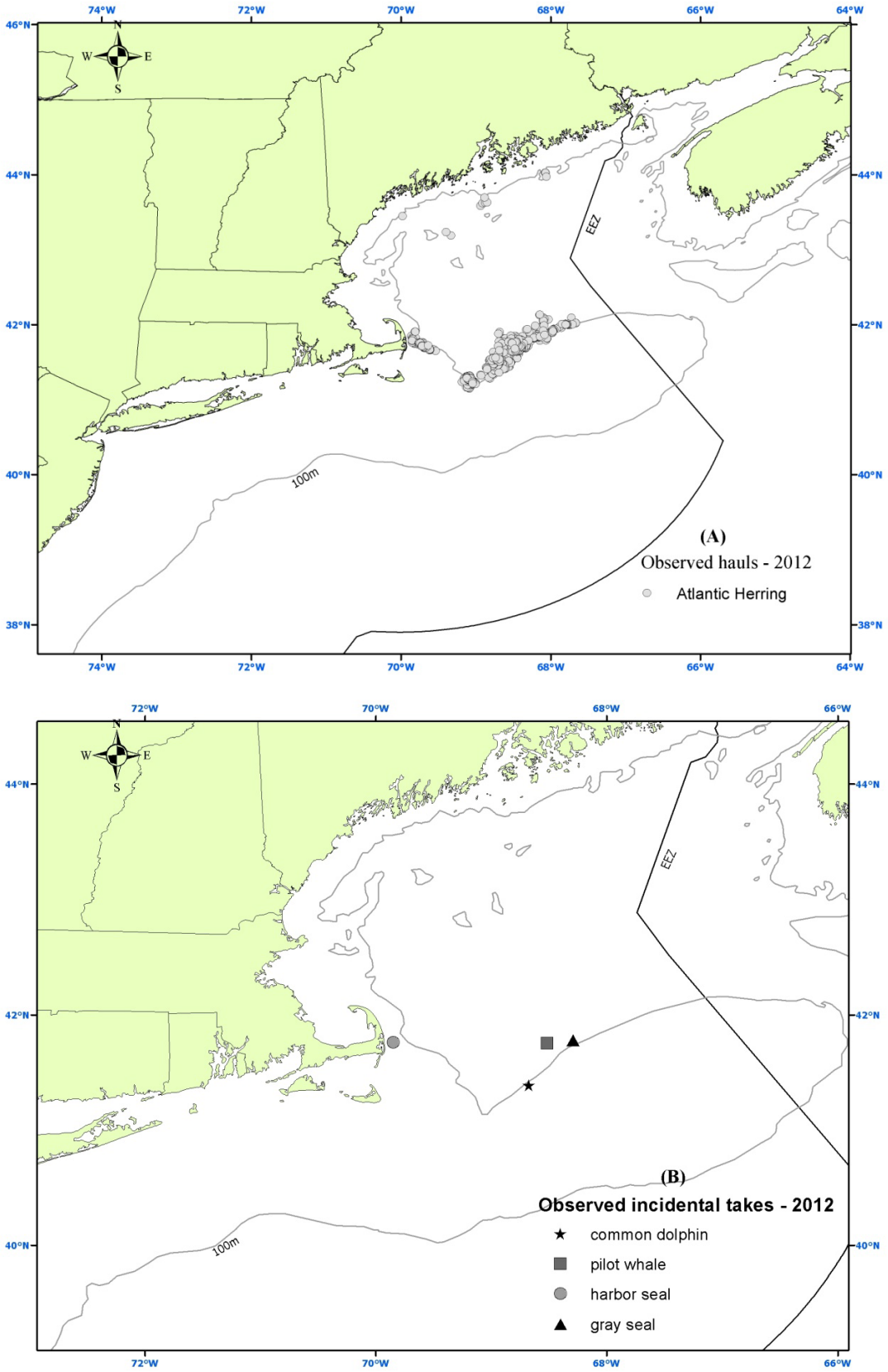


Figure 23. 2013 Northeast mid-water trawl observed tows (A) and observed takes (B).

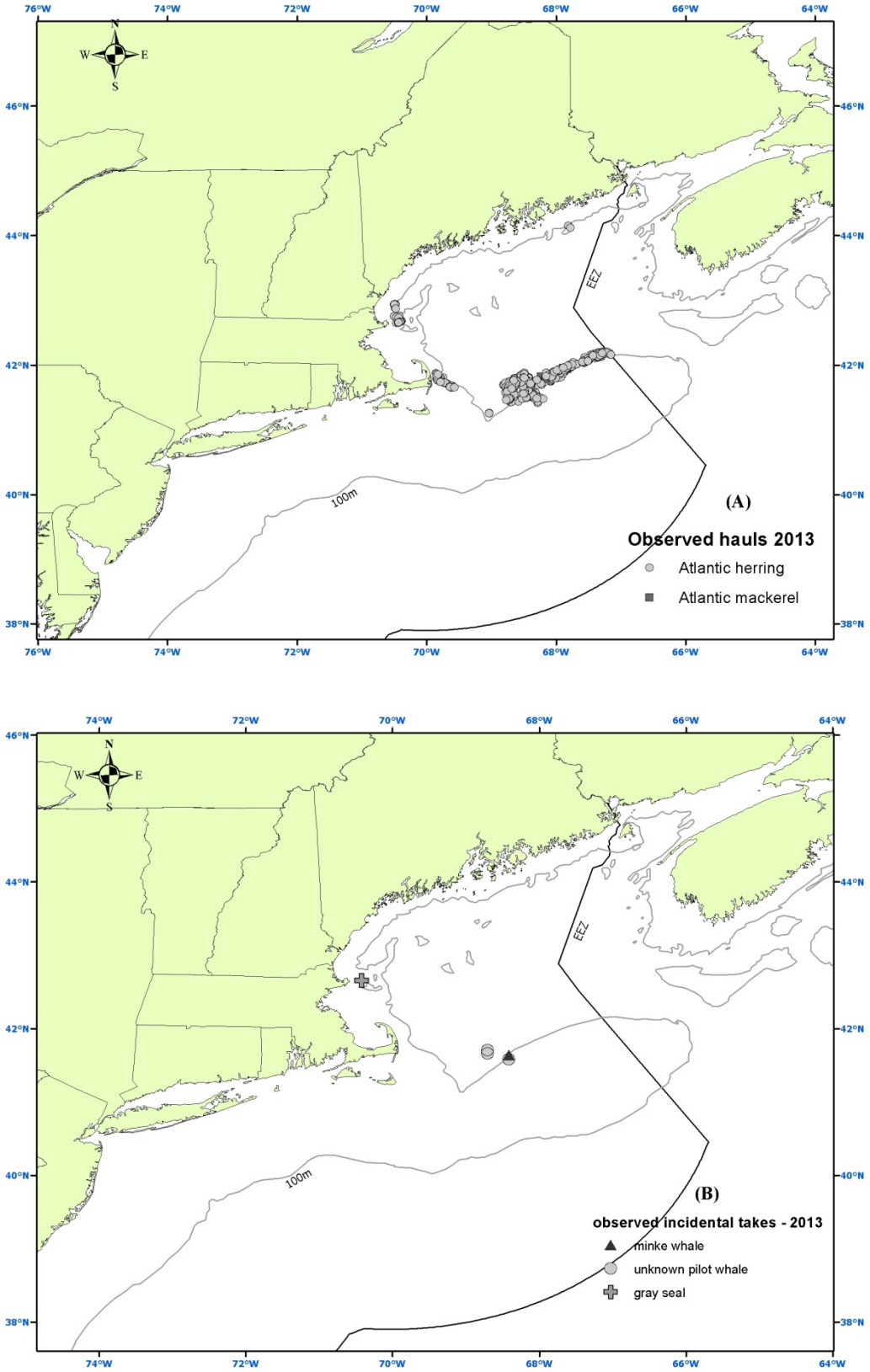


Figure 24. 2014 Northeast mid-water trawl observed tows (A) and observed takes (B).

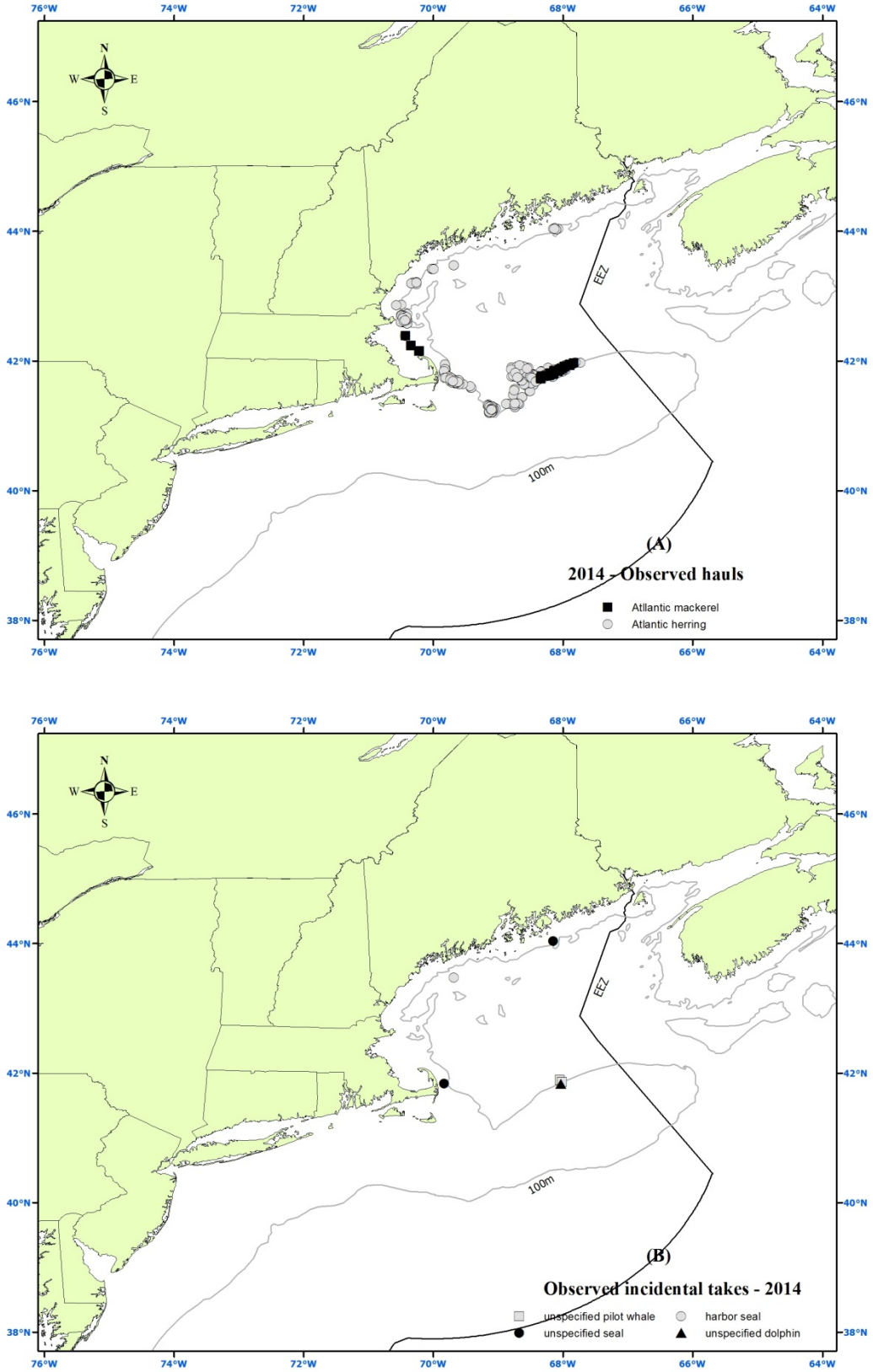


Figure 25. 2015 Northeast mid-water trawl observed tows (A) and observed takes (B).

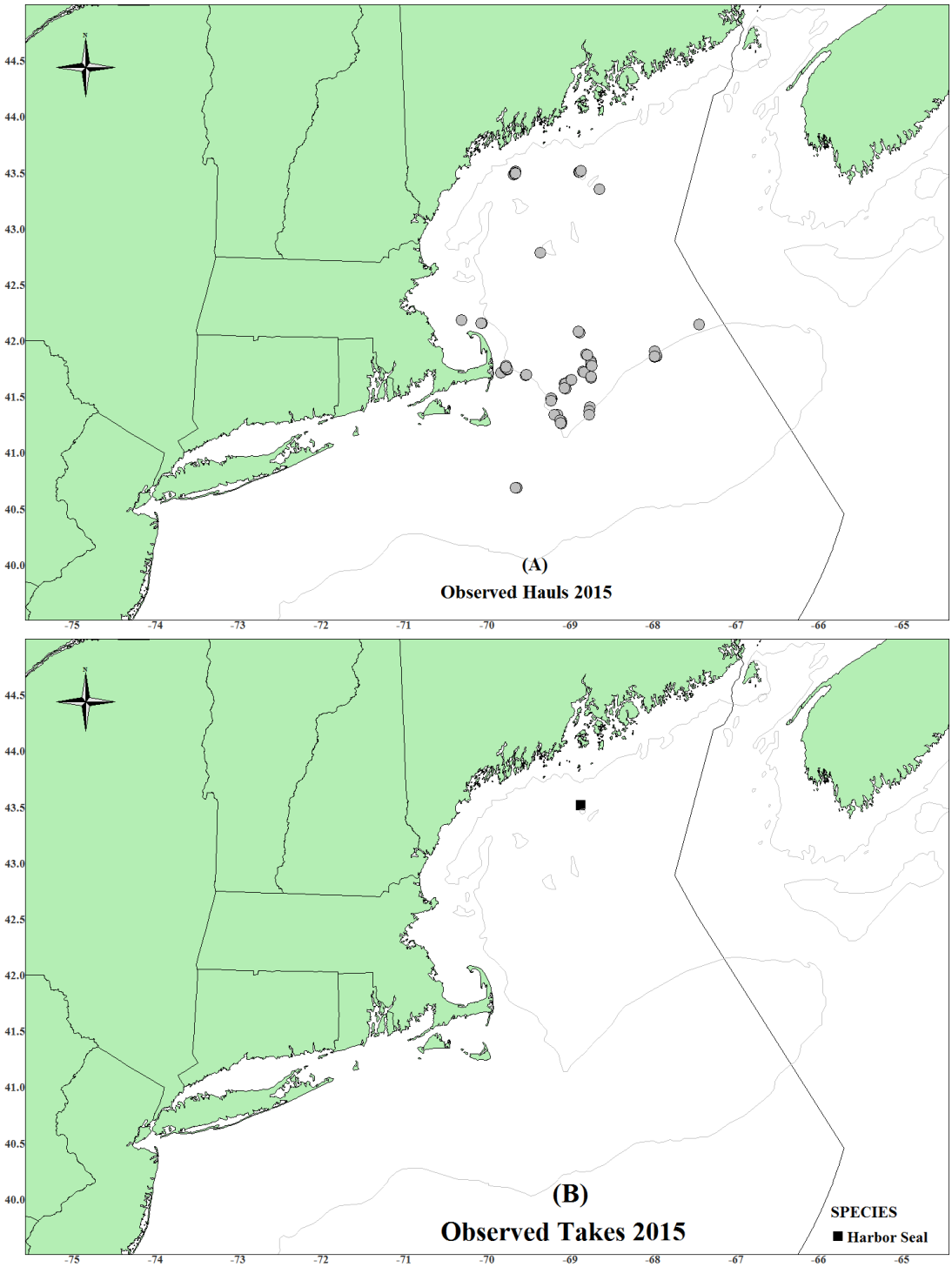


Figure 26. 2011 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).

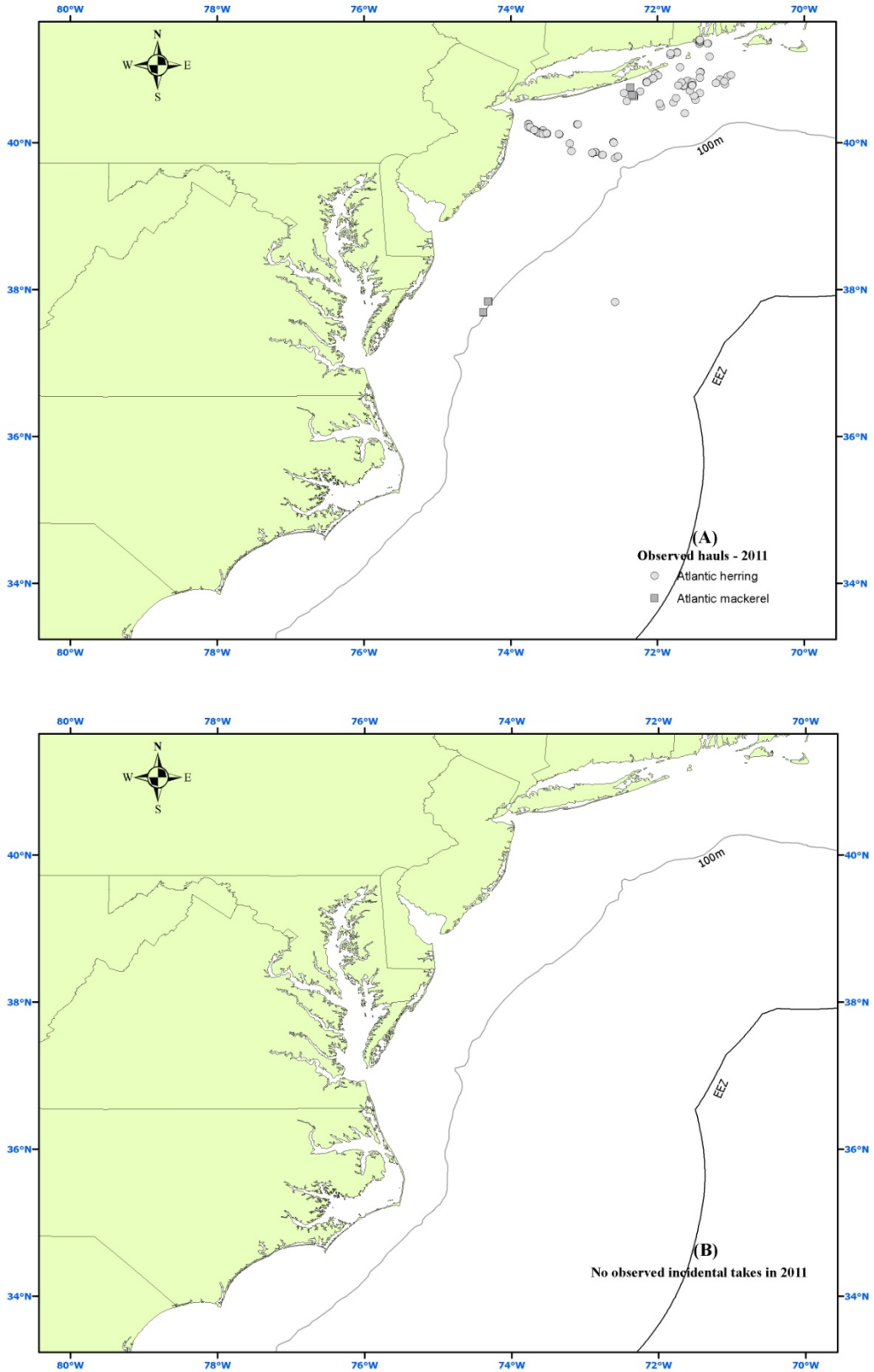


Figure 27. 2012 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).

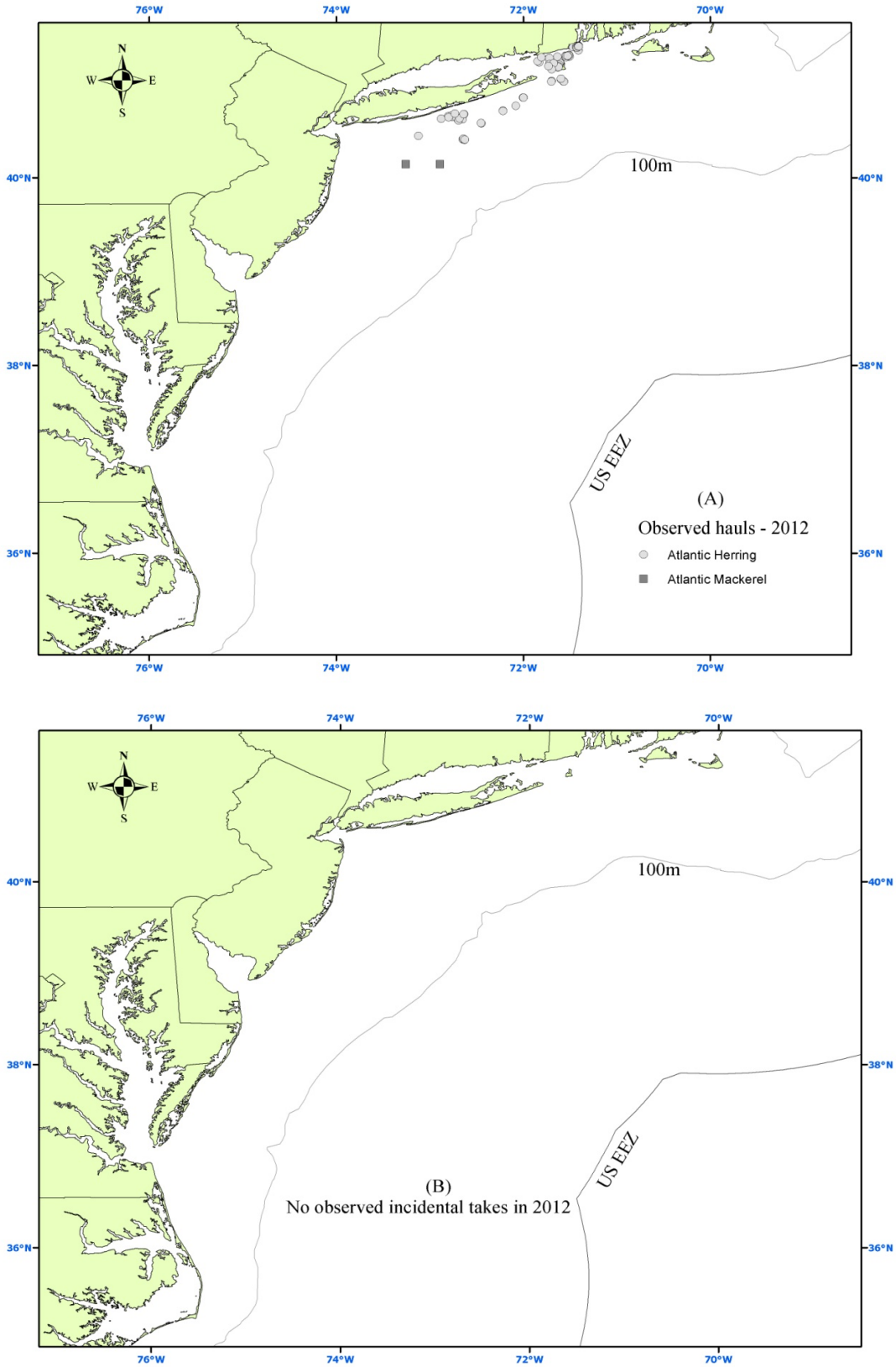


Figure 28. 2013 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).

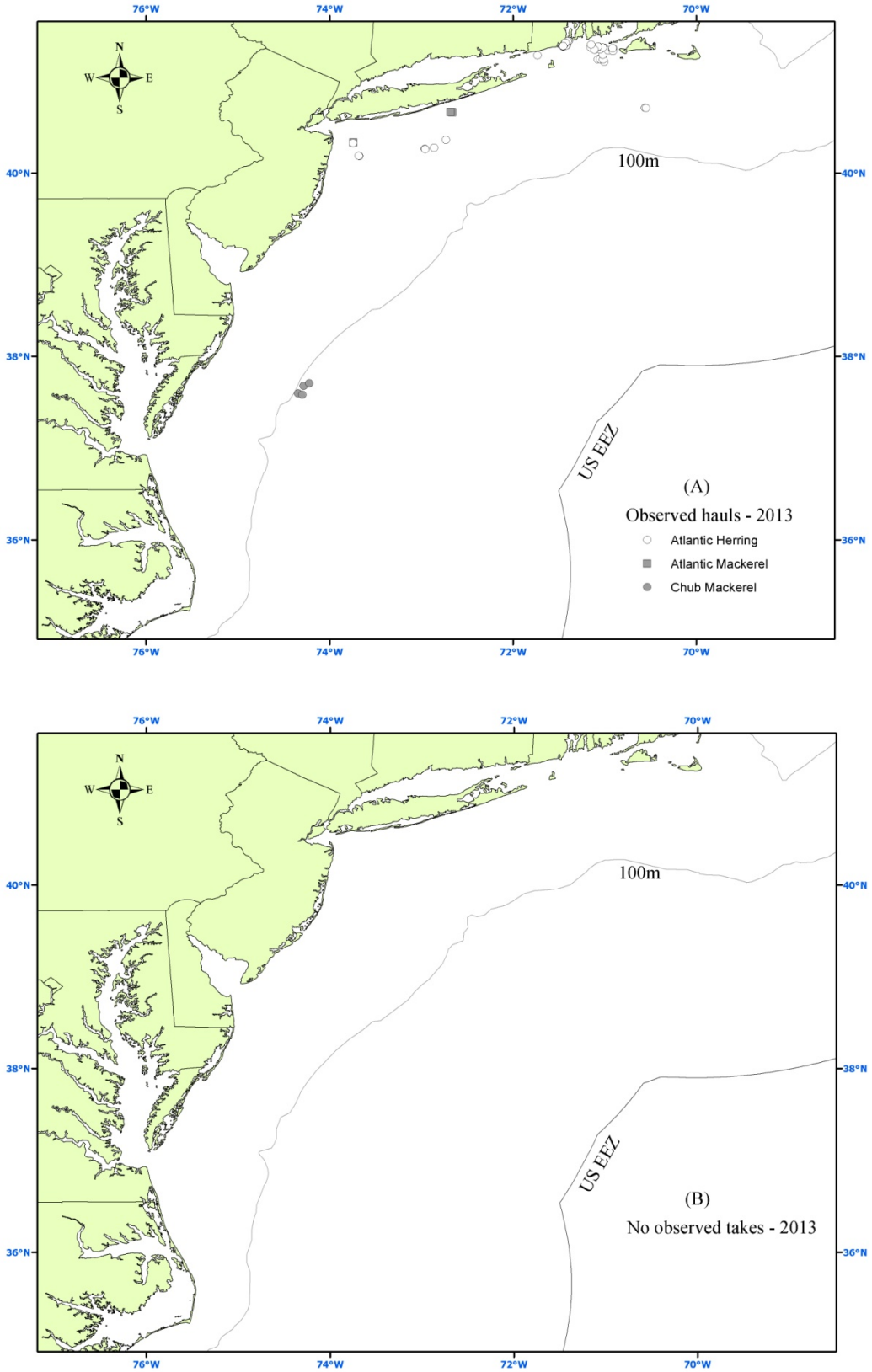
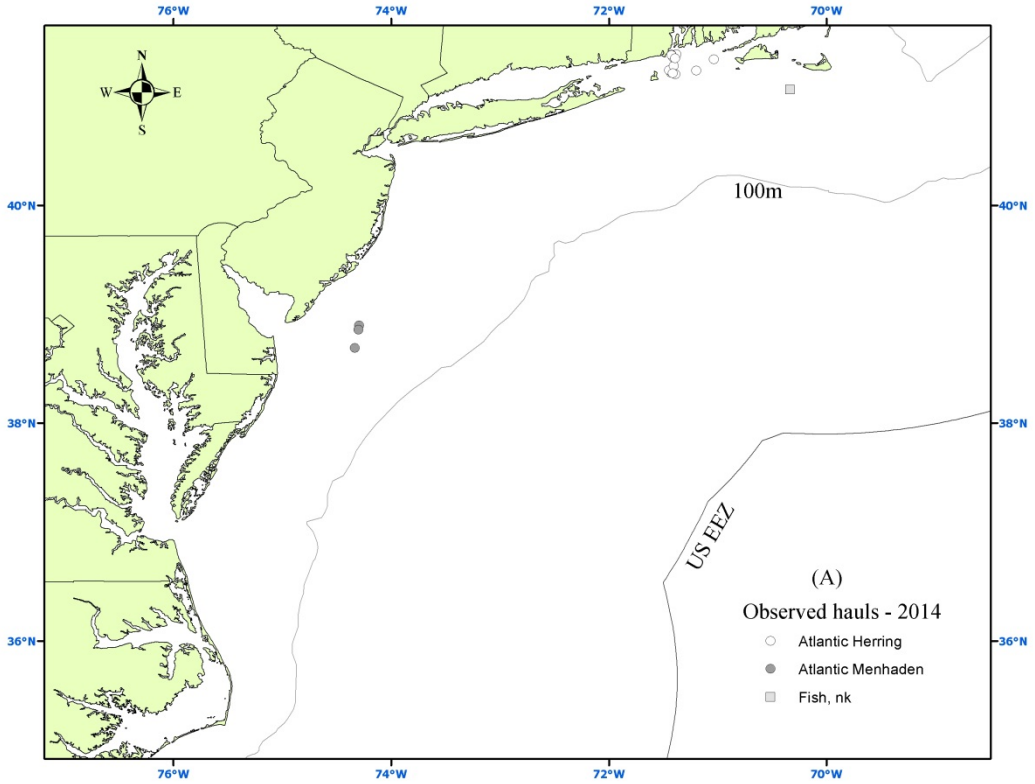


Figure 29. 2014 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).



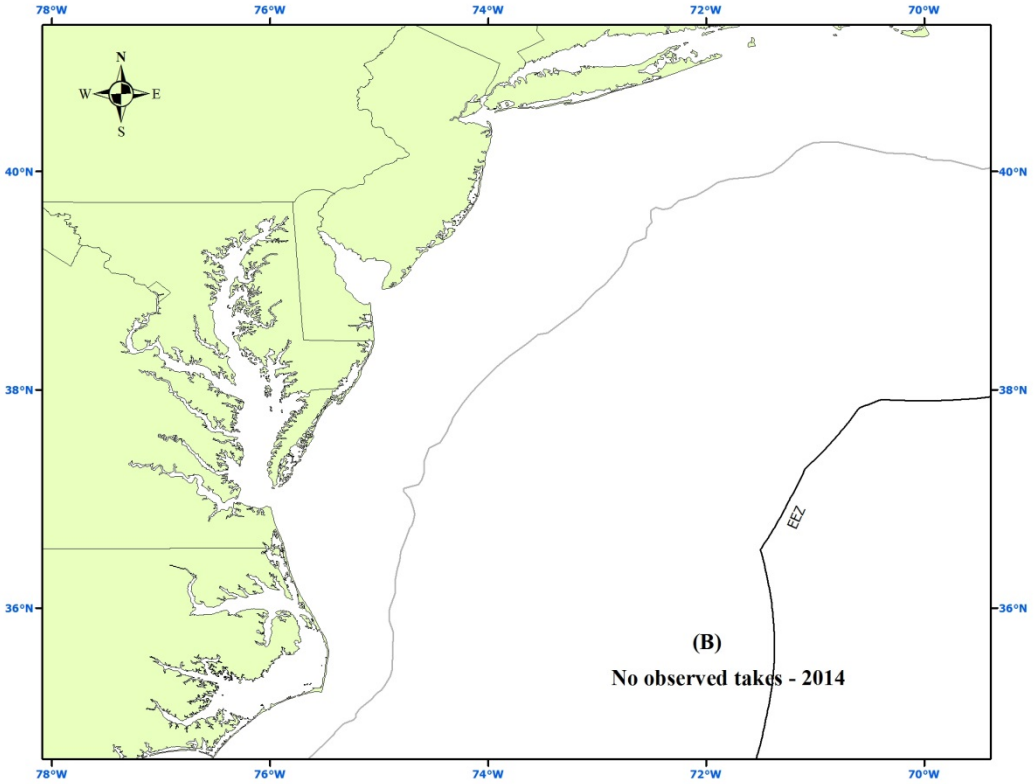


Figure 30. 2015 Mid-Atlantic mid-water trawl observed tows (A) and observed takes (B).

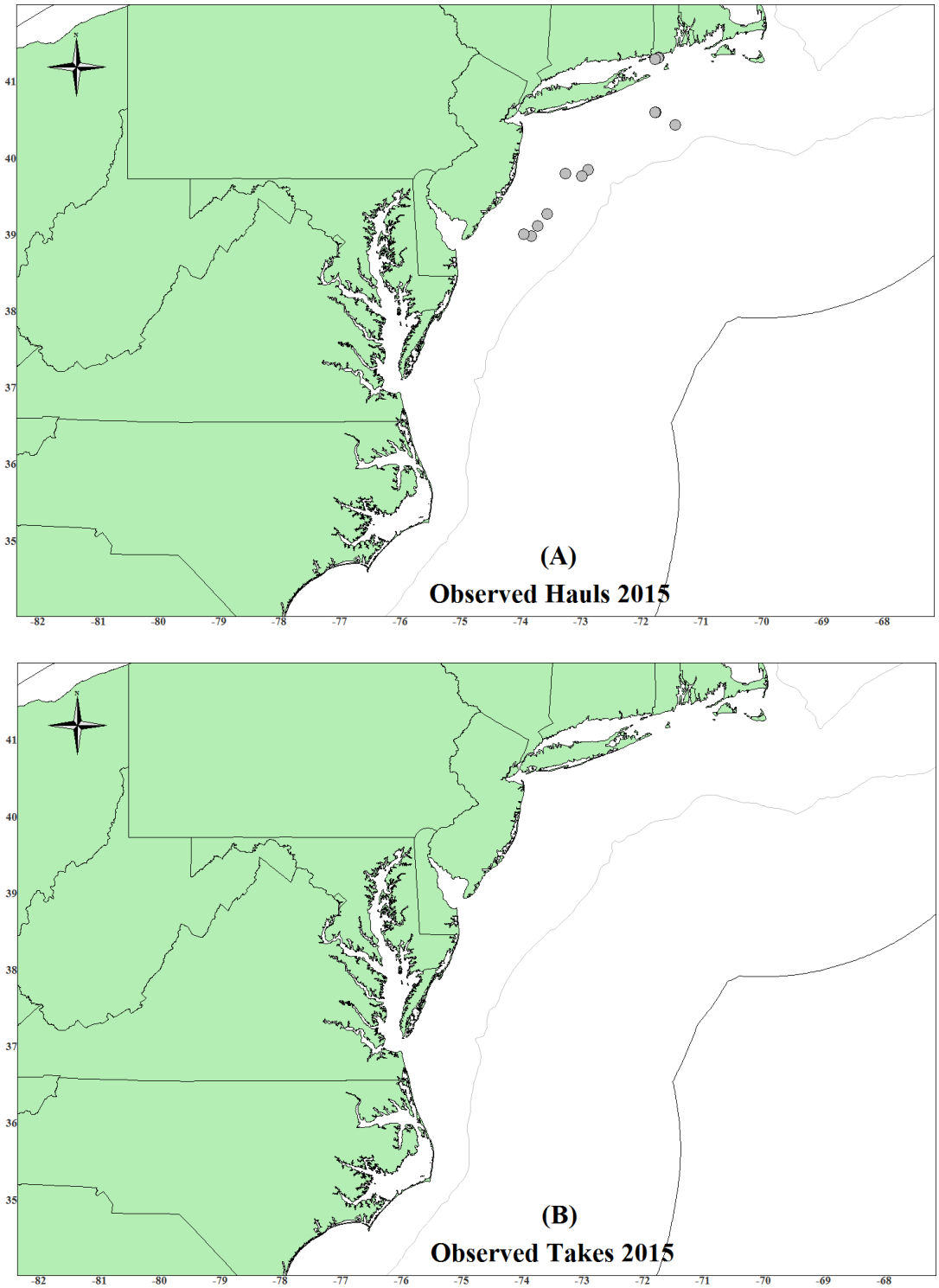


Figure 31. 2011 Herring Purse Seine observed hauls (A) and observed takes (B).

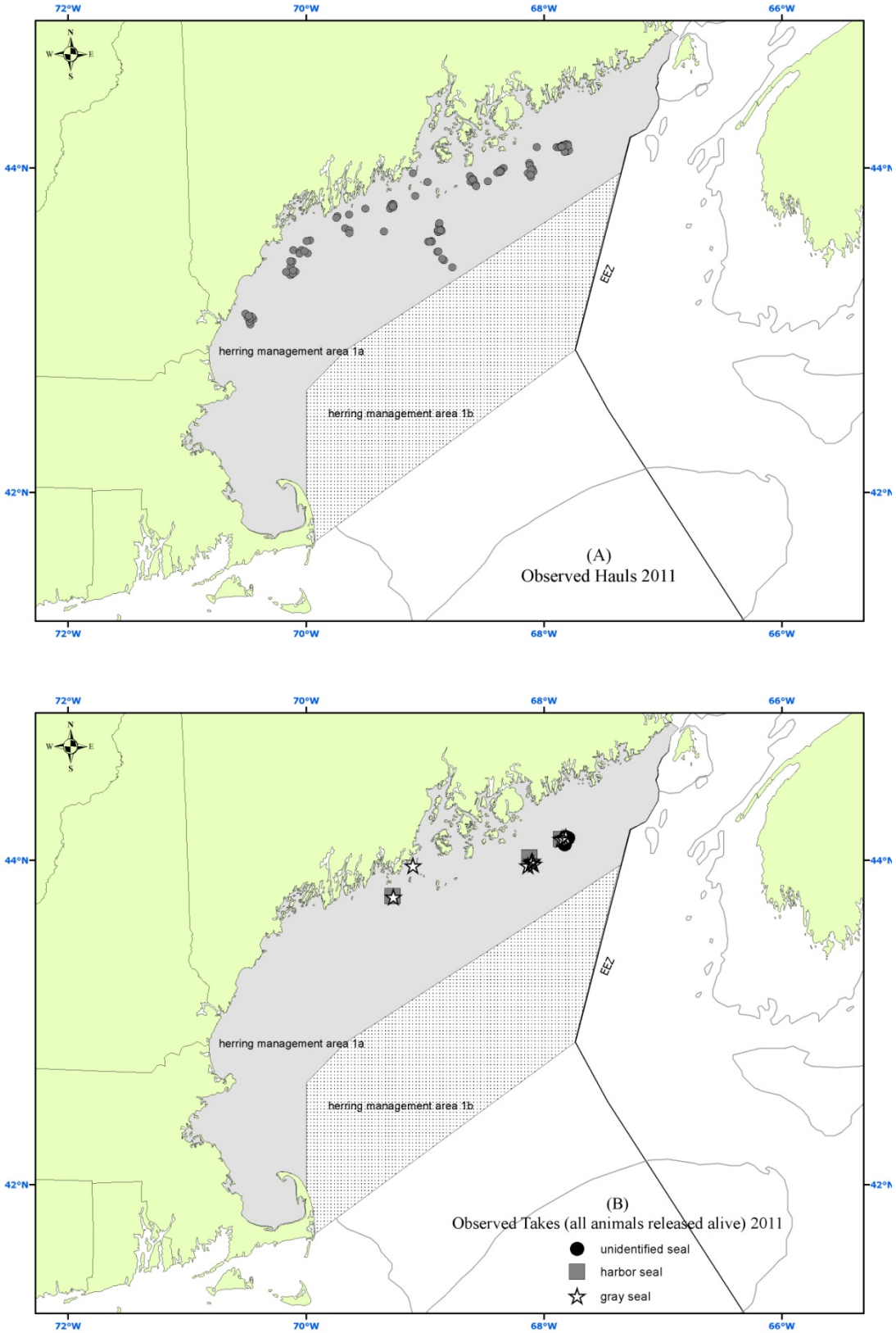


Figure 32. 2012 Herring Purse Seine observed hauls (A) and observed takes (B).

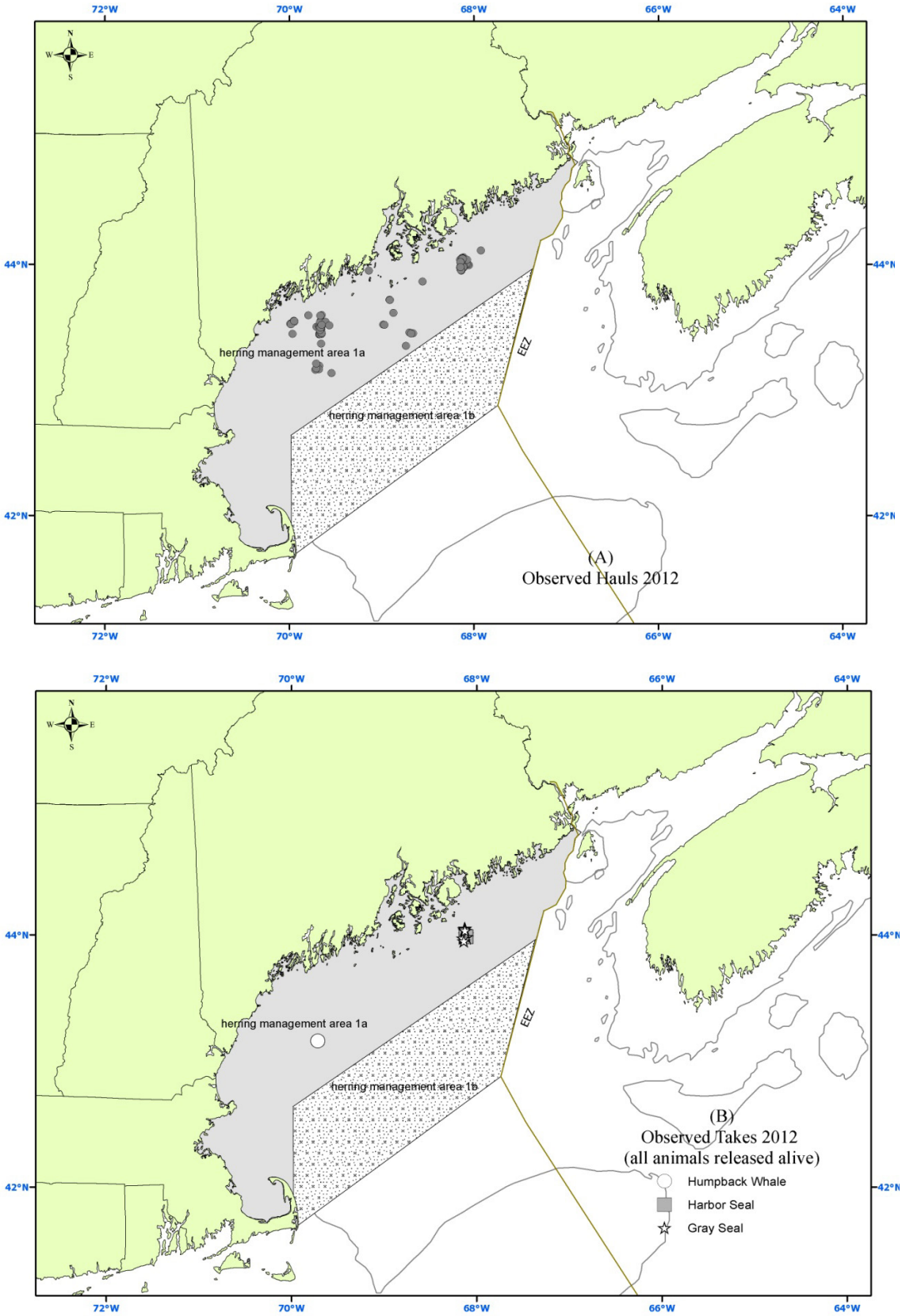


Figure 33. 2013 Herring Purse Seine observed hauls (A) and observed takes (B).

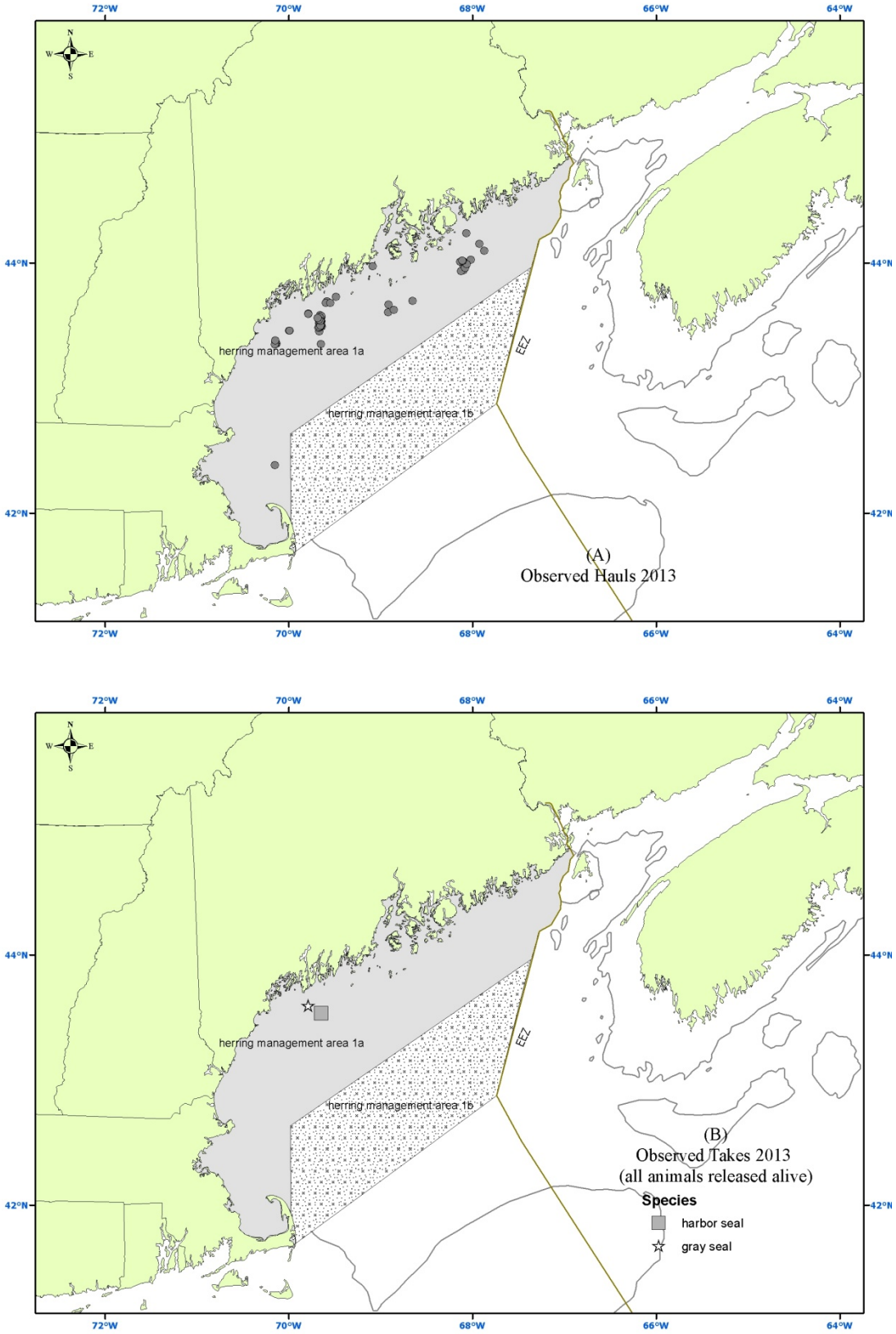


Figure 34. 2014 Herring Purse Seine observed hauls (A) and observed takes (B).

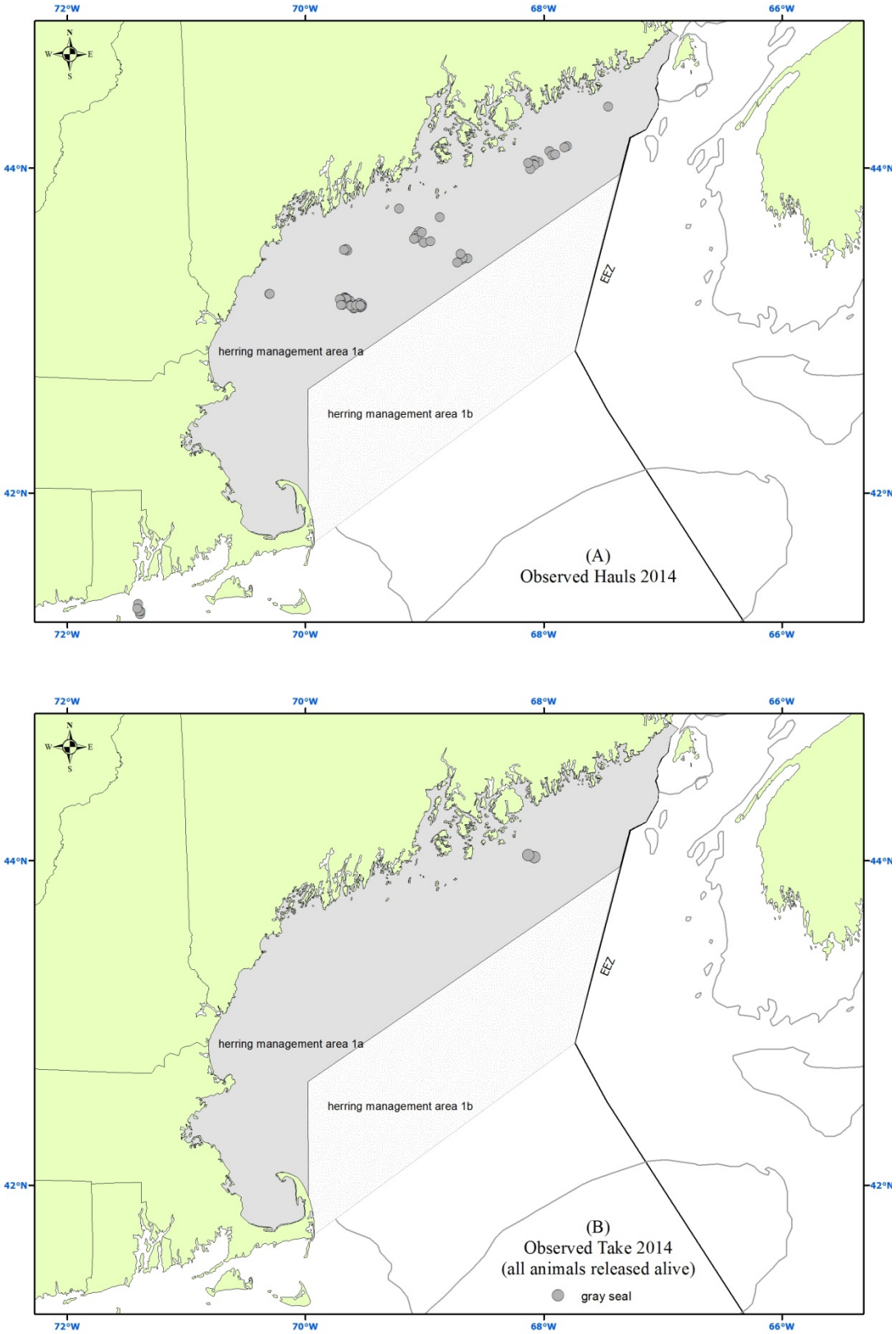


Figure 35. 2015 Herring Purse Seine observed hauls (A) and observed takes (B).

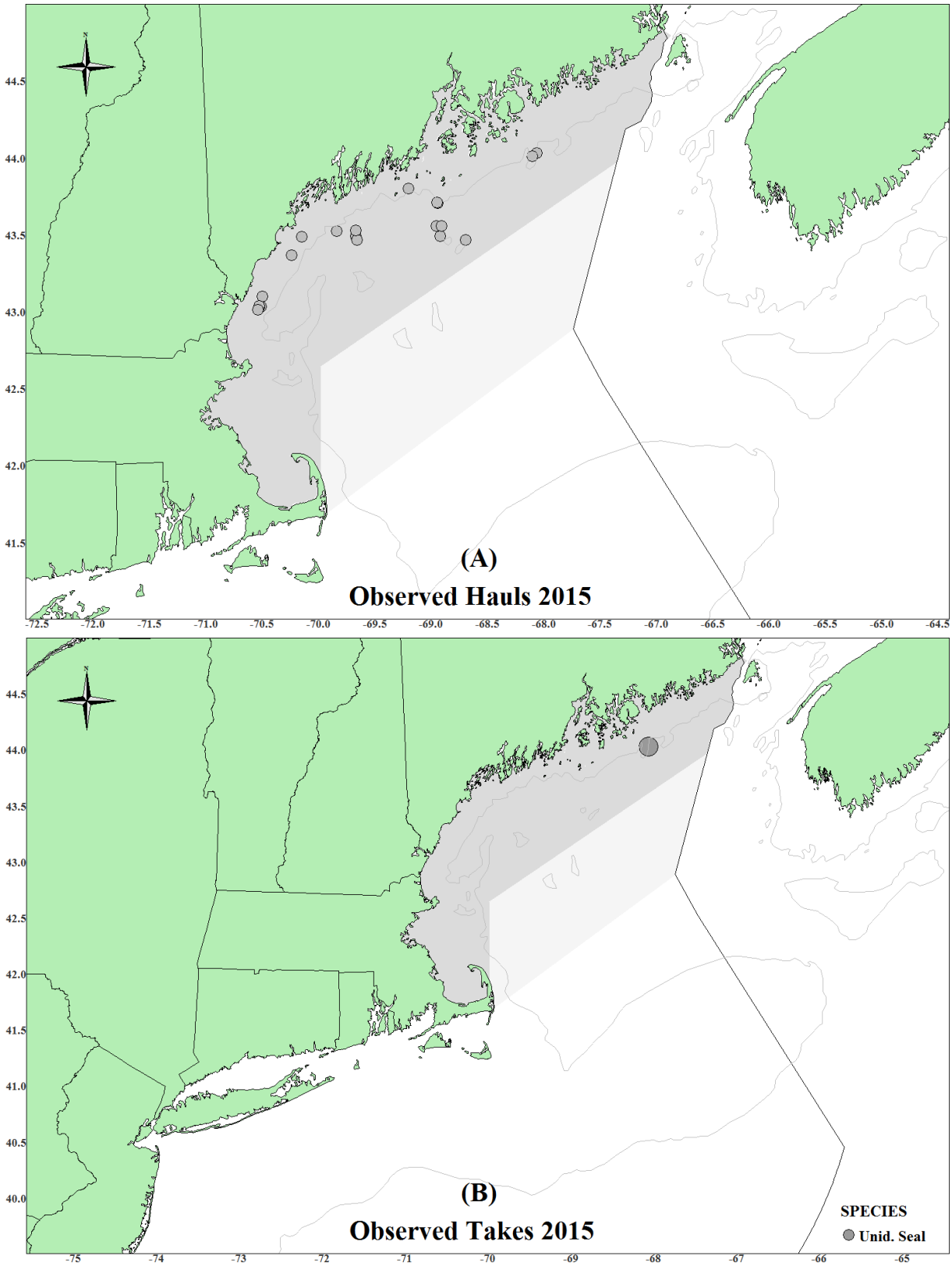


Figure 36. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2011. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

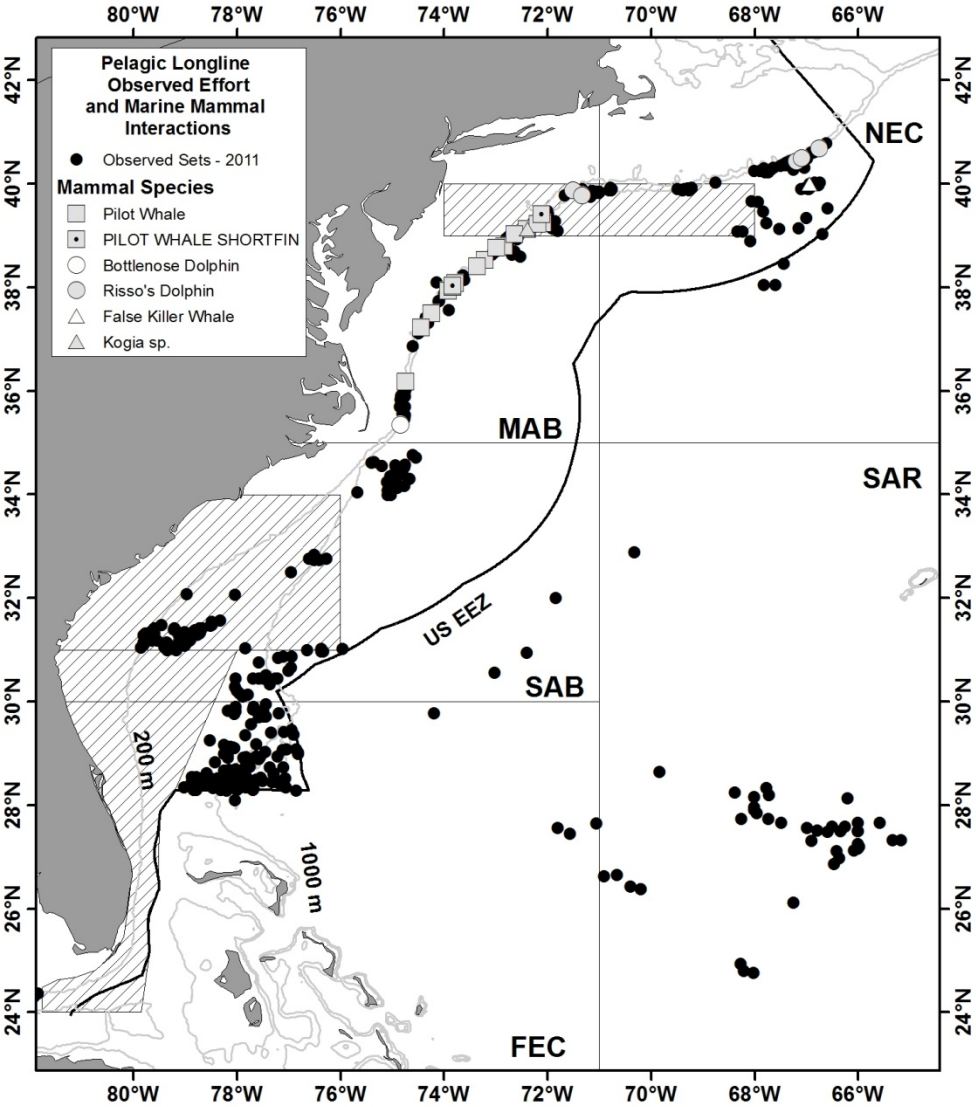


Figure 37. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2012. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

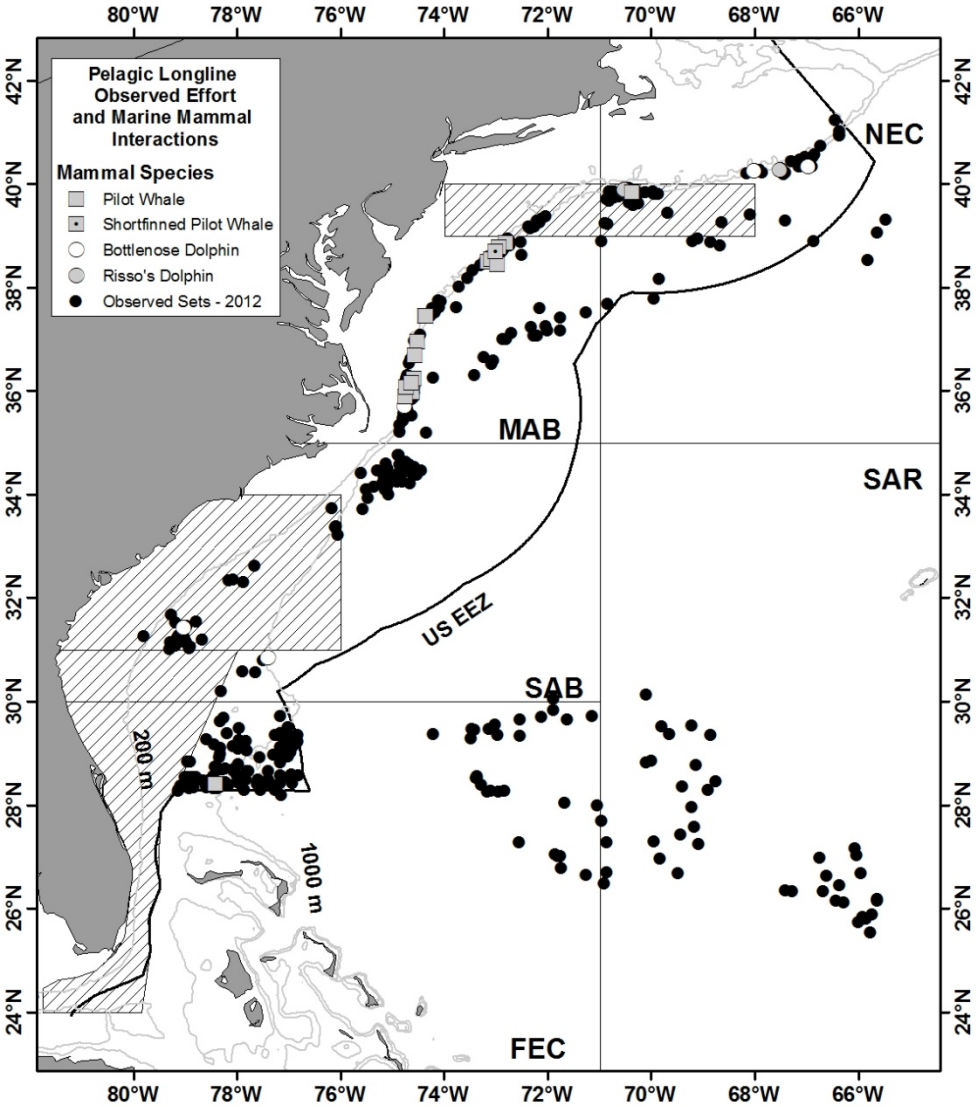


Figure 38. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2013. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

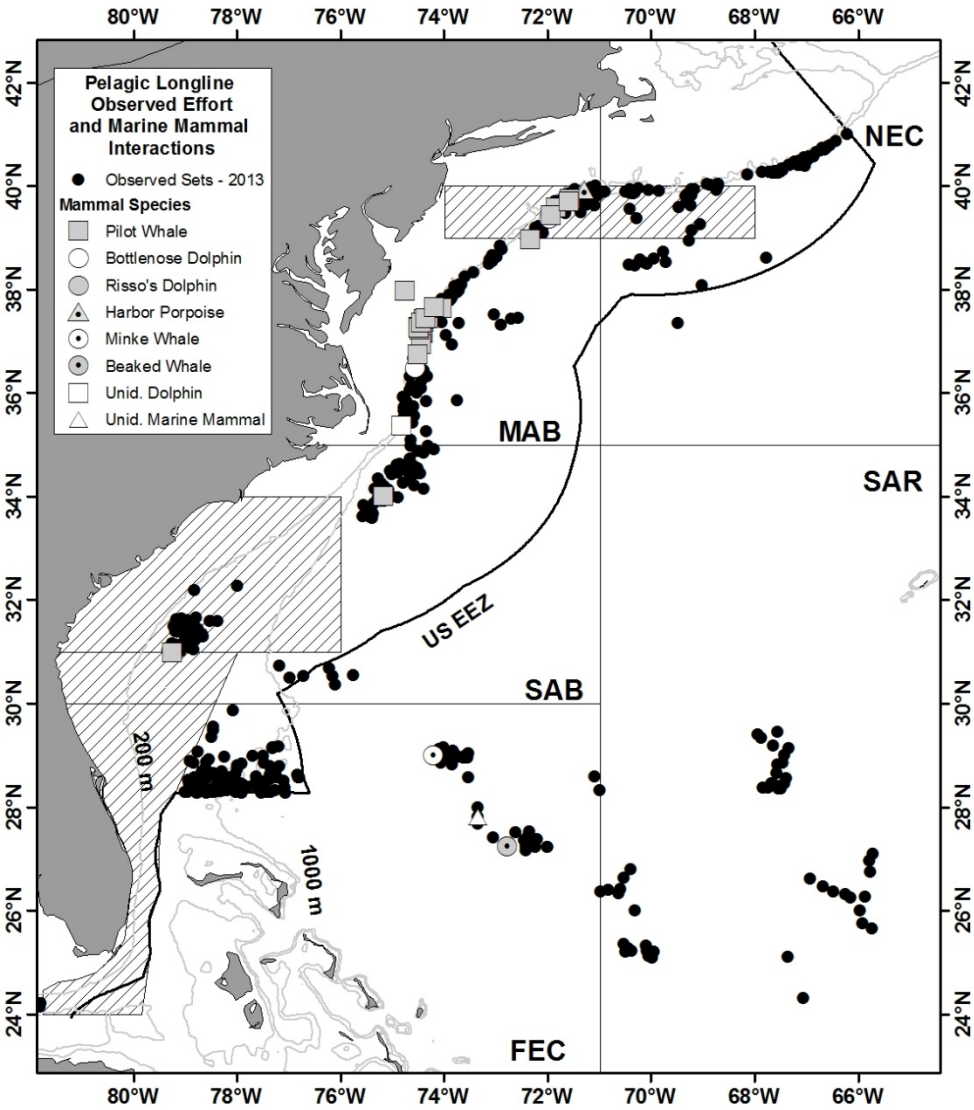


Figure 39. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2014. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

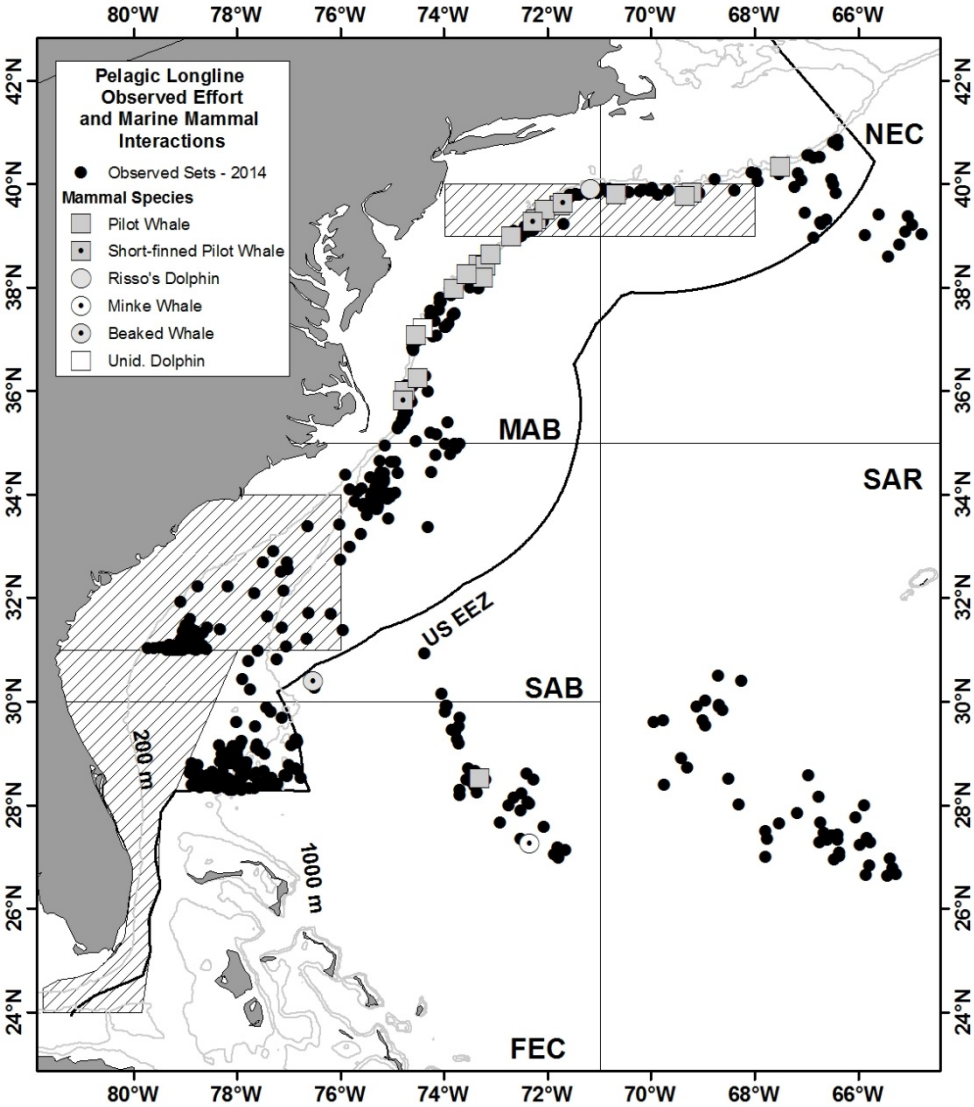


Figure 40. Observed sets and marine mammal interactions in the Pelagic longline fishery along the U.S. Atlantic coast during 2015. The boundaries of the Florida East Coast (FEC), South Atlantic Bight (SAB), Mid-Atlantic Bight (MAB), Northeast Coastal (NEC), and Sargasso Sea (SAR) fishing areas are shown. Seasonal closed areas instituted in 2001 under the HMS FMP are shown as hatched areas.

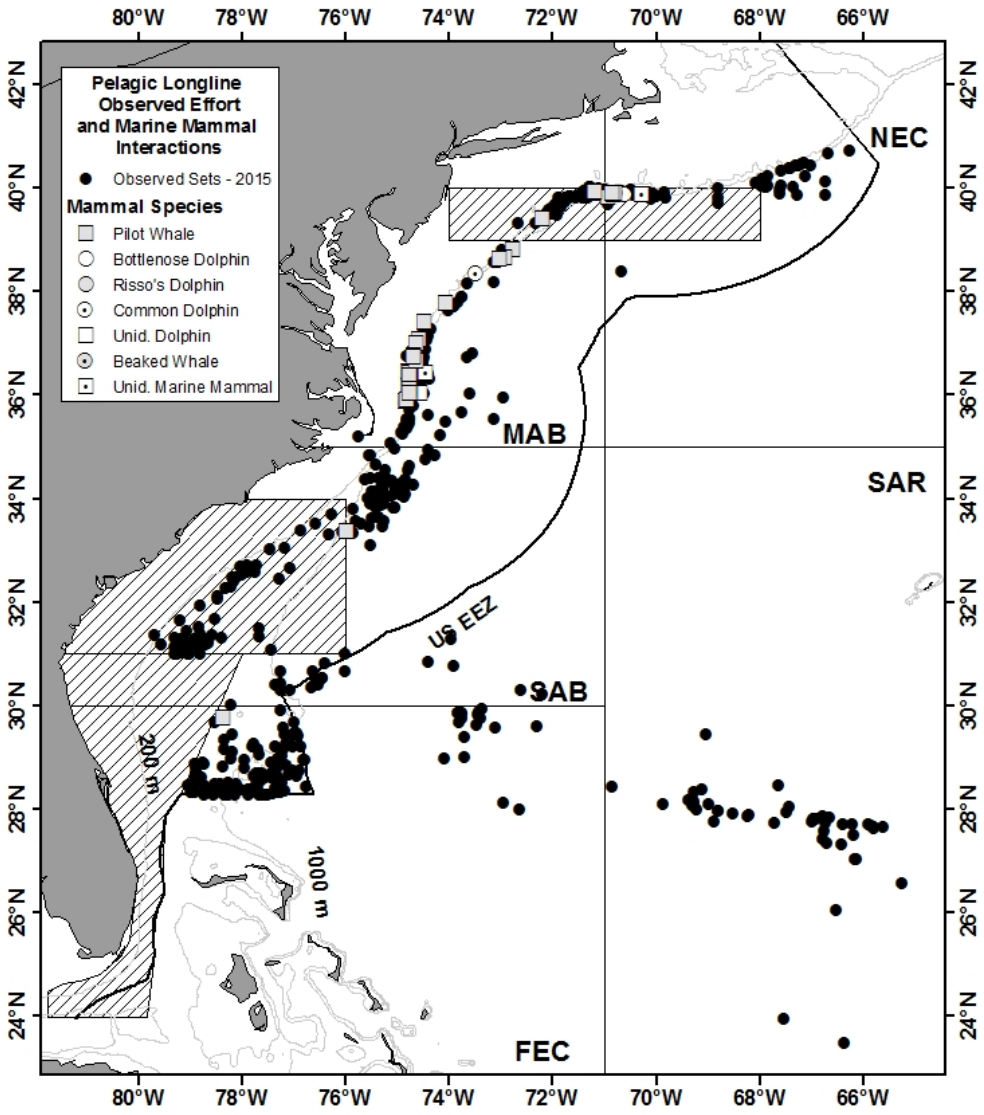


Figure 41. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2011. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.

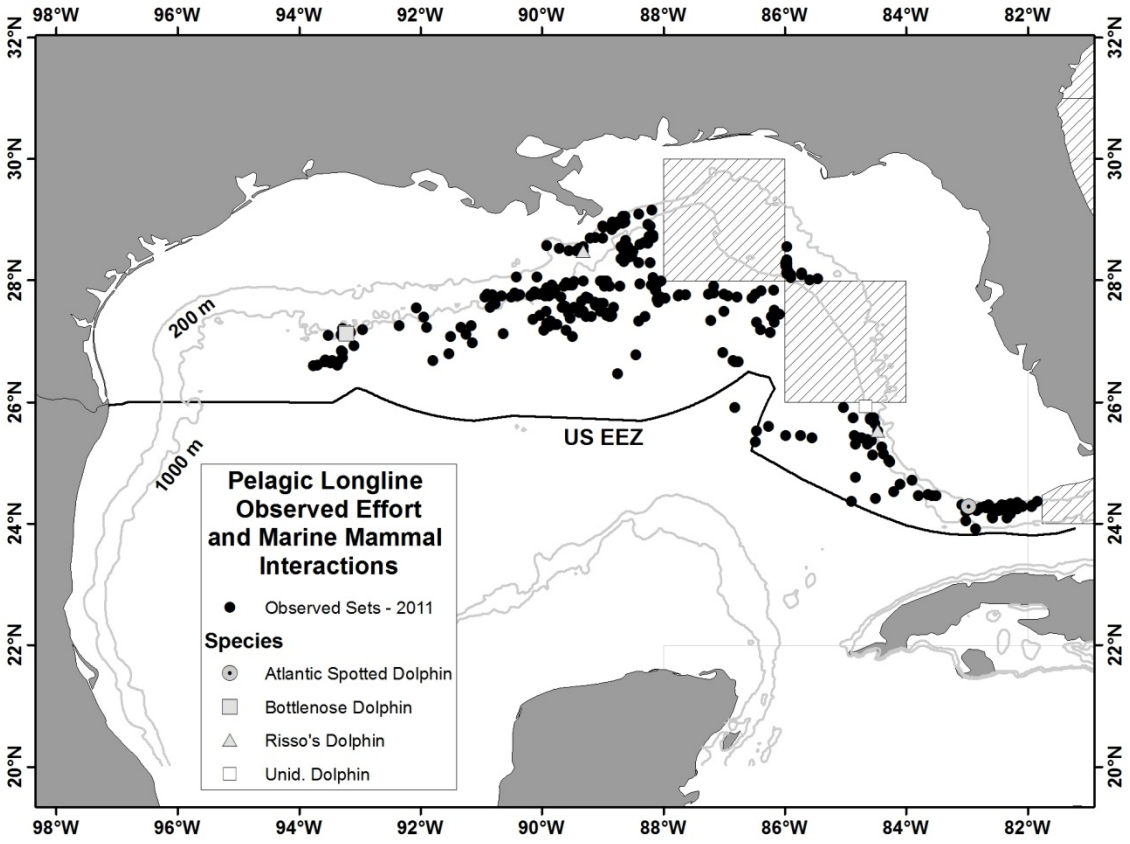


Figure 42. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2012. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.

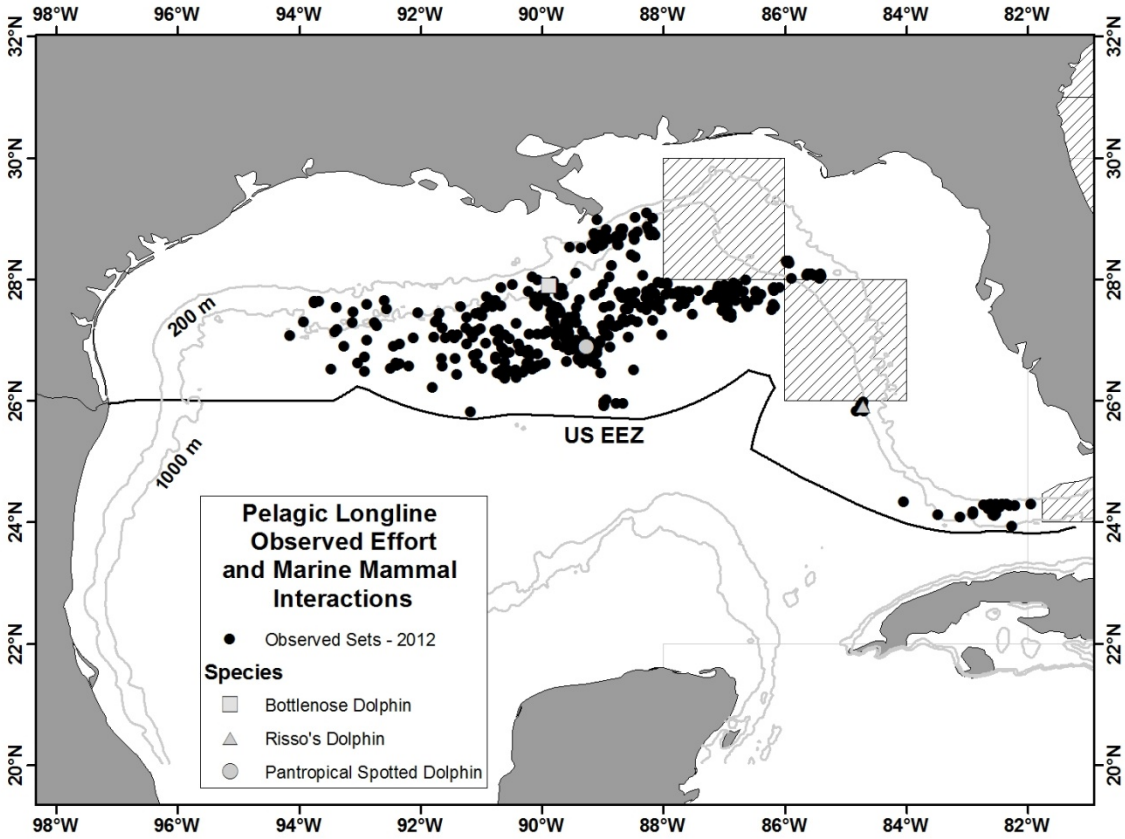


Figure 43. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2013. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.

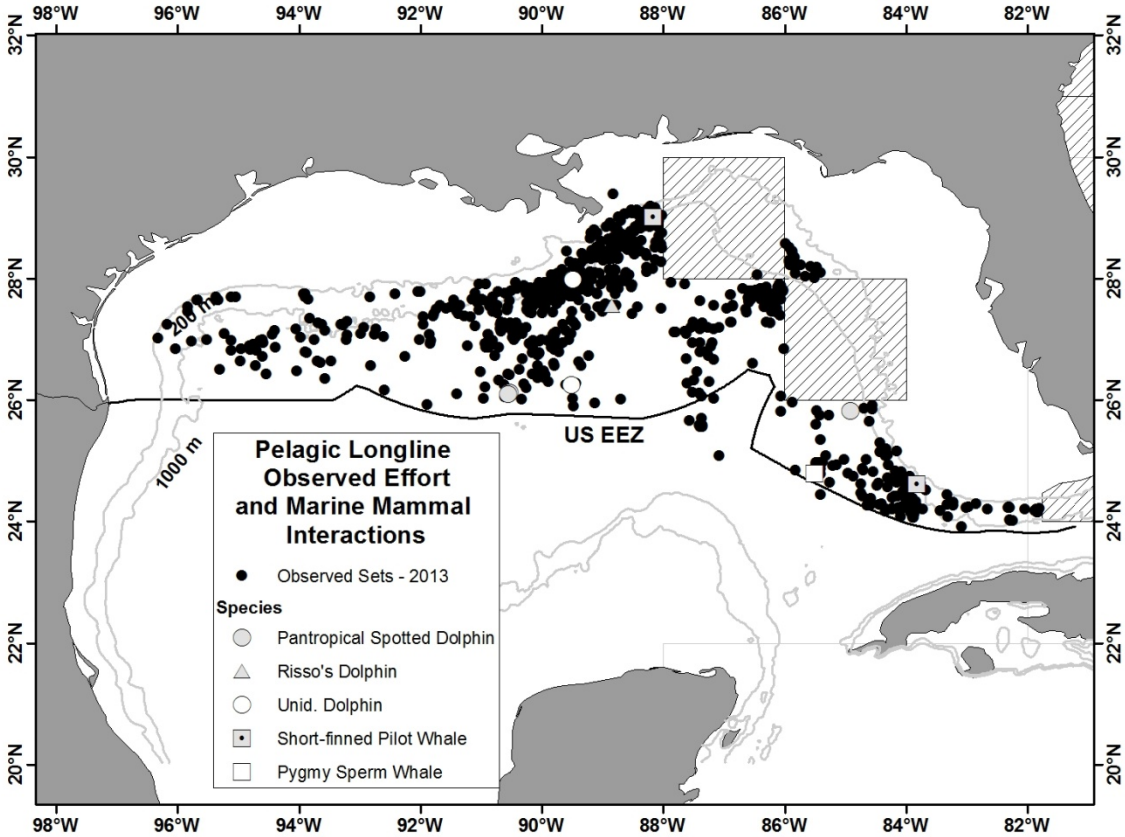


Figure 44. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2014. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.

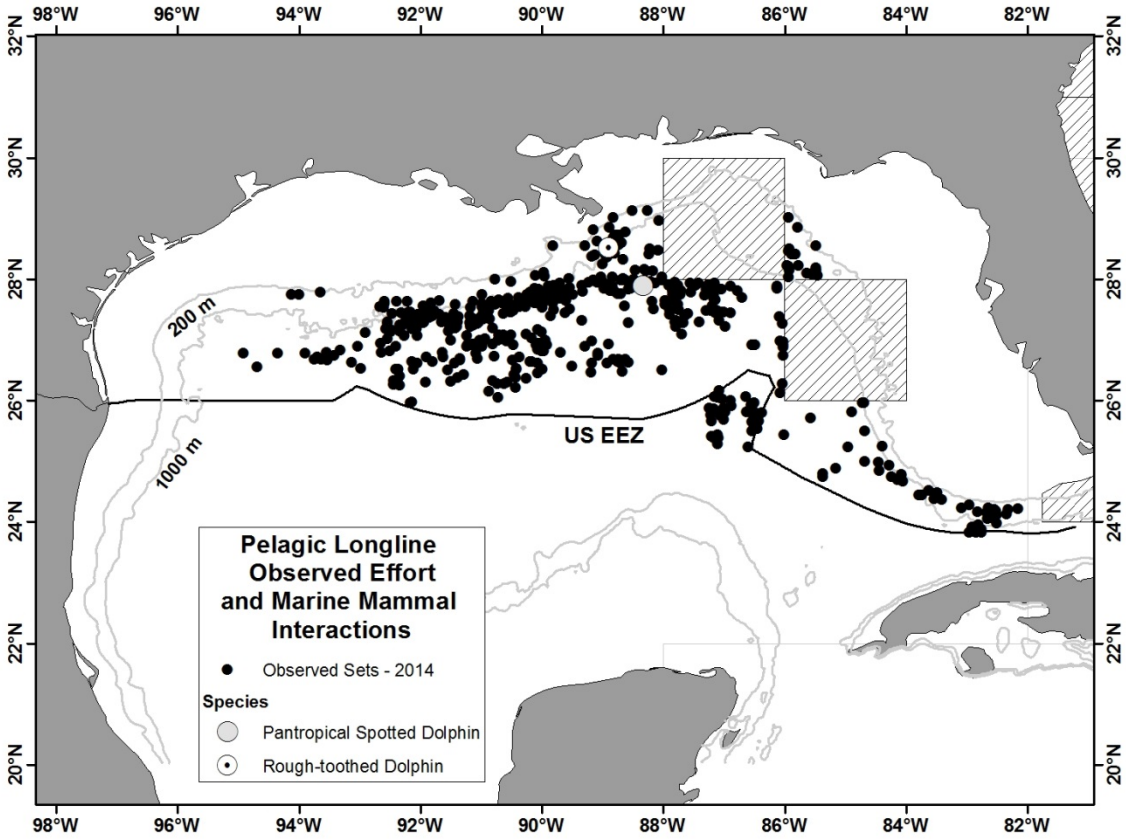
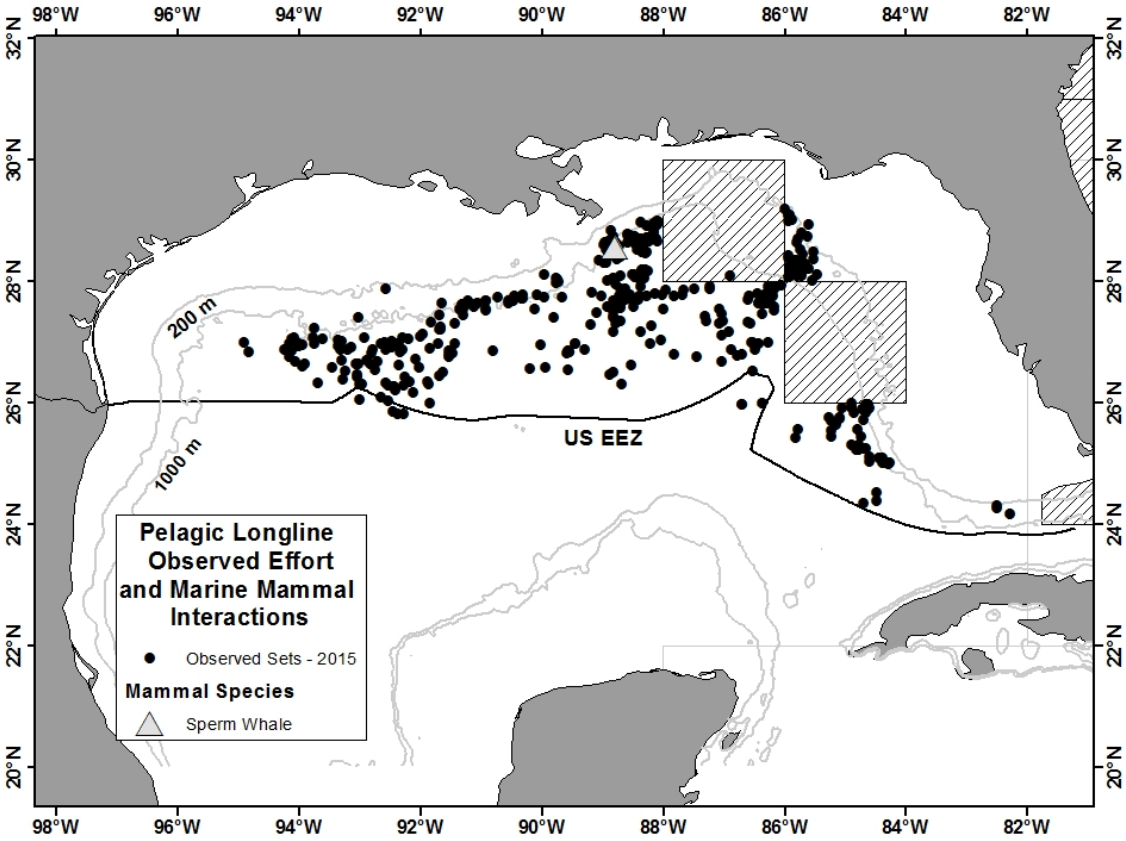


Figure 45. Observed sets in the Pelagic longline fishery in the Gulf of Mexico during 2015. Closed areas in the DeSoto canyon instituted in 2001 are shown as hatched areas.



Appendix IV:
Surveys and Abundance Estimates

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	NEC/SEC	One team data analyzed by DISTANCE.	N	NMFS 1991

					waters				
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
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 truncated sighting curve	N	Kingsley and Reeves 1998
14	1998	Jul-Aug	ship	4,163	south of Maryland	SEC	One team data analyzed by DISTANCE.	N	Mullin and Fulling 2003
15	1998	Aug-Sep	plane (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,	Y	

							2007, and 2008 to calculate pooled g(0)'s and year-species specific 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
25	2007	Jul-Aug	plane	46,804	Canadian waters from	DFO	uncorrected	N	Lawson and

					Nova Scotia to Newfoundland		counts		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
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 shallower than the 100-m depth contour, through the US and	NEC	Two-independent teams, both using naked eye in the same plane. Analyzed using DISTANCE, the independent observer option assuming point independence	Y	Palka 2012

					Canadian Gulf of Maine and up to and including the lower Bay of Fundy)				
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	
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		
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		

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**
Fin Whale	Western North Atlantic	1995	2,200	0.24	12	Palka 1996
		1999	2,814	0.21	18	Palka 2006
		2002	2,933	0.49	18	Palka 2006
		2004	1,925	0.55	22	Palka 2006
		2006	2,269	0.37	23	Palka 2005
		2007	3,522	0.27	25	Lawson and Gosselin 2009
		2011	1,595	0.33	40+41	Palka 2012
		2011	23	0.87	42	
		2011	**1,618**	**0.33**	**40+41+42**	Estimate summed from north and south surveys
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
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
Sperm Whale	North Atlantic	1982	219	0.36	1	CETAP 1982
		1990	338	0.31	2	
		1991	736	0.33	7	Waring *et al.*1992:1998
		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
Kogia spp.	Western North	1998	115	0.61	16	

	Atlantic	1998	580	0.57	14	Mullin and Fulling 2003
		2004	358	0.44	22	Palka 2006
		2004	37	0.75	21	Garrison *et al.* 2010
		2004	395	0.4	21+22	Estimate summed from north and south surveys
		2011	1,783	0.62	40+41	Palka 2012
		2011	2,002	0.69	42	
		2011	**3,785**	**0.47**	**40+41+42**	Estimate summed from north and south surveys
Beaked Whales	Western North Atlantic	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	
		1991	612	0.73	5	
		1993	330	0.66	9	
		1994	99	0.64	10	
		1995	1,519	0.69	12	Palka 1996
		1998	2,600	0.4	16	
		1998	541	0.55	14	Mullin and Fulling 2003
		2004	2,839	0.78	22	Palka 2006
		2004	674	0.36	21	Garrison *et al.* 2010
		2004	3,513	0.63	21+22	Estimate summed from north and south surveys
		2006	922	1.47	23	
		2011	5,500	0.67	40+41	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*; Palka 2012)
		2011	1,592	0.67	42	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*)

		2011	**7,092**	**0.54**	**40+41+42**	2011 estimates are for *Mesoplodon* spp. beaked whales alone (not including *Ziphias*); Estimate summed from north and south surveys
Cuvier's Beaked Whale	Western North Atlantic	2011	4,962	0.37	40+41	Palka 2012
		2011	1,570	0.65	42	
		2011	**6,532**	**0.32**	**40+41+42**	Estimate summed from north and south surveys
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
		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
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	
		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**
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
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	
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
		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
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
		2011	17,917	0.42	42	
		2011	**44,715**	**0.43**	**40+41+42**	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**

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**
Rough-toothed Dolphin	Western North Atlantic	2011	0	0	40+41	Palka 2012
		2011	271	1	42	
		2011	**271**	**1**	**40+41+42**	**Estimate summed from north and south surveys**
Bottlenose Dolphin	Western North Atlantic Offshore	1998	16,689	0.32	16	
		1998	13,085	0.4	14	Mullin and Fulling 2003
		2002	26,849	0.19	20	
		2002	5,100	0.41	18	Palka 2006
		2004	9,786	0.56	22	Palka 2006
		2004	44,953	0.26	21	Garrison *et al.* 2010
		2006	2,989	1.11	23	
		2011	26,766	0.52	40+41	Palka 2012
		2011	50,766	0.55	42	
		2011	77,532	0.4	**40+41+42**	**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	
		2007	4,862	0.31	25	Lawson and Gosselin 2009
		2011	**79,883**	**0.32**	**40+41**	Palka 2012
Harbor Seal	Western North Atlantic	2001	99,340	0.097	27	Gilbert *et al.* 2005
						Waring *et al.* 2015
		2012	70,142	0.29	43	
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
			208,720	0.14		
			216,490	0.11		
		2004	223,220	0.08	36	Sable Island
				95% CI		
				263,000-		
		2012	331,000	458,000		DFO 2013
Bryde's Whale	Northern Gulf of Mexico	1991-1994	35	1.1	30	Hansen *et al.* 1995
		1996-2001	40	0.61	33	Mullin and Fulling 2004
		2003-2004	15	1.98	35	
		2009	**33**	**1.07**	**38**	
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
		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**	
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	
		2009	**129**	**1**	**38**	
Atlantic Spotted	Northern Gulf of	1991-1994 oceanic	3,213	0.44	30	Hansen *et al.* 1995

Dolphin	Mexico	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	
		2009	624	0.99	0.05	
Bottlenose Dolphin	Northern Gulf of Mexico Oceanic					Mullin and Fulling 2004
		1996-2001	2,239	0.41	33	
		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
		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
		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
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
2015	177	0.28					55	0.54							1021	0.25	119	0.34

Note: this table only includes observed bycatch. The List of Fisheries has a complete list of marine mammal species interactions with this fishery.

[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, Unidentified		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
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
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	33	1.16			6.1-13.2	0.32-0.22	0-14.3	0.32	0.8-18.2	0.23					30	0.55					48	0.52	15	1.04		

Note: this table only includes observed bycatch. The List of Fisheries has a complete list of marine mammal species interactions with this fishery.

[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.	30	0.3	0	0	21	0.2	0	0	0	0	0	0	49	1.1	0	0

						34						7										
2002	0	0	0	0	70	0.32	26	0.29	0	0	22	0.26	0	0	0	0	0	0	0	0	0	0
2003	*	*	0	0	216	0.27	26	0.29	0	0	20	0.26	0	0	0	0	0	0	0	0	0	0
2004	0	0	0	0	200	0.3	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	25	0.44	11	0.63	19	0.45			0	0
2015	0	0			15	0.52	22	0.45	0	0			0	0	0	0	23	0.46	0	0	0	0

Note: this table only includes observed bycatch. The List of Fisheries has a complete list of marine mammal species interactions with this fishery.

[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	46	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	7.4	1.0	0	0

Note: this table only includes observed bycatch. The List of Fisheries has a complete list of marine mammal species interactions with this fishery.

[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															2[a]	na	0	0

Note: this table only includes observed bycatch. The List of Fisheries has a complete list of marine mammal species interactions with this fishery.

[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							2[a]	na		

Note: this table only includes observed bycatch. The List of Fisheries has a complete list of marine mammal species interactions with this fishery.

[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 - 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
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	0	0	233	0.24	0	0
2015	0	0	0	0	9.05	1	8.4	0.71	0	0			200	0.24		

Note: this table only includes observed bycatch. The List of Fisheries has a complete list of marine mammal species interactions with this fishery.

[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.

na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

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. The List of Fisheries has a complete list of marine mammal species interactions with this fishery.

[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		
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	33	1.16	3.7	0.49	177	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
2012	16	1	0	0	0	0	0	0			61.8	0.68
2013	0	0	0	0	0	0	27	0.95			0	0
2014	25	0.66	0	0	0	0	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

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		
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		

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	46	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

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	

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

Short-finned Pilot Whale, Western North Atlantic Stock

	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

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
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

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
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	2[a]	na	0	0	474	0.17	2[a]	na

Note: this table only includes observed bycatch. [a] Unextrapolated mortalities

na=not applicable; unk= observer coverage was absent or too low to detect bycatch, or no estimate generated; tbd= to be determined

Gray Seal

	Herring Purse Seine	Mid-Atlantic Bottom	Mid-Atlantic Gillnet	Mid-Atlantic Midwater	Northeast Bottom Trawl	NE Sink Gilllnet	Northeast Midwater

			Trawl				Trawl						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
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

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
2015	0	0	0	0	119	0.34

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 2017

Species	Stock	Updated
Sei whale	Nova Scotia Stock	2016
Blue whale	Western North Atlantic	2010
Sperm whale	North Atlantic	2014
Killer whale	Western North Atlantic	2014
Pygmy killer whale	Western North Atlantic	2007
False killer whale	Western North Atlantic	2014
Northern bottlenose whale	Western North Atlantic	2014
Sowerby's beaked whale	Western North Atlantic	2014
Cuvier's beaked whale	Western North Atlantic	2013
Blainville's beaked whale	Western North Atlantic	2013
Gervais' beaked whale	Western North Atlantic	2013
True's beaked whale	Western North Atlantic	2013
Melon-headed whale	Western North Atlantic	2007
Long-finned pilot whale	Western North Atlantic	2016
Short-finned pilot whale	Western North Atlantic	2016
White-beaked dolphin	Western North Atlantic	2007
Atlantic spotted dolphin	Western North Atlantic	2013
Pantropical spotted dolphin	Western North Atlantic	2013
Striped dolphin	Western North Atlantic	2013
Fraser's dolphin	Western North Atlantic	2007
Rough-toothed dolphin	Western North Atlantic	2013
Clymene dolphin	Western North Atlantic	2013
Spinner dolphin	Western North Atlantic	2013
Common bottlenose dolphin	Northern South Carolina Estuarine System	2015

Common bottlenose dolphin	Charleston Estuarine System	2015
Common bottlenose dolphin	Northern GA/ 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	2007
Cuvier's beaked whale	Gulf of Mexico Oceanic	2012
Blainville's beaked whale	Gulf of Mexico Oceanic	2012
Gervais' beaked whale	Gulf of Mexico Oceanic	2012
Common bottlenose dolphin	Gulf of Mexico Oceanic	2014
Common bottlenose dolphin	Gulf of Mexico, Continental shelf	2015
Common bottlenose dolphin	Gulf of Mexico, eastern coastal	2015
Common bottlenose dolphin	Gulf of Mexico, northern coastal	2015
Common bottlenose dolphin	Gulf of Mexico, western coastal	2015
Common bottlenose dolphin	Gulf of Mexico, Oceanic	2015
Common bottlenose dolphin	Gulf of Mexico, bay, sound and estuary (27 stocks)	2016
Common bottlenose dolphin	St. Joseph Bay	2015
Common bottlenose dolphin	Choctawhatchee Bay	2015
Atlantic spotted dolphin	Gulf of Mexico	2015
Pantropical spotted dolphin	Gulf of Mexico	2015
Rough-toothed dolphin	Gulf of Mexico (Outer continental shelf and Oceanic)	2016
Clymene dolphin	Gulf of Mexico Oceanic	2012
Fraser's dolphin	Gulf of Mexico Oceanic	2012
Killer whale	Gulf of Mexico Oceanic	2012

False killer whale	Gulf of Mexico Oceanic	2012
Pygmy killer whale	Gulf of Mexico Oceanic	2012
Dwarf sperm whale	Gulf of Mexico Oceanic	2012
Pygmy sperm whale	Gulf of Mexico Oceanic	2012
Melon-headed whale	Gulf of Mexico Oceanic	2012
Risso's dolphin	Gulf of Mexico	2015
Pilot whale, short-finned	Gulf of Mexico	2015
Sperm whale	Gulf of Mexico	2015
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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